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Managing Small Stream Networks for Improved Water Quality, Catchment Biodiversity and Ecosystem Services Protection

Authors: Mary Kelly-Quinn, Michael Bruen, Mike Bowes, Jens Carlsson, Edward Cox, Angela Gurnell, Sinéad Hogan, Ann-Marie Kelly, Marcin Penk, John O'Sullivan and Jeremy Piggott

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Rialtas na hÉireann
Government of Ireland

Environmental Protection Agency

The EPA is responsible for protecting and improving the environment as a valuable asset for the people of Ireland. We are committed to protecting people and the environment from the harmful effects of radiation and pollution.

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2. Office of Environmental Enforcement
3. Office of Evidence and Assessment
4. Office of Radiation Protection and Environmental Monitoring
5. Office of Communications and Corporate Services

The EPA is assisted by advisory committees who meet regularly to discuss issues of concern and provide advice to the Board.

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Lead organisation: University College Dublin

Identifying pressures

The SSNet project aims to advance knowledge on the role of small streams (first and second order) in water quality, biodiversity and ecosystem services protection. It also aims to inform policy, measures and management options. In doing so, the project supports the achievement of the Water Framework Directive objectives and other regulatory targets.

Informing policy

The research is relevant for the Water Framework Directive and other policies relating to biodiversity, such as national biodiversity plans, the EU Biodiversity Strategy for 2030, the Habitats Directive, the Birds Directive, the UN Convention on Biological Diversity and the new Nature Restoration Law. Elements of the research are also relevant to climate adaptation and mitigation, as well as agendas such as the UN Sustainable Development Goals.

Developing solutions

SSNet is the first large-scale research project in Ireland on first- and second-order streams to have undertaken investigations spanning hydrochemistry and multiple ecological elements, as well as experimental work, giving insights into the likely impact of climate change stressors.

It recommended that more widespread monitoring of the water quality of small streams should be undertaken to protect not only small stream biodiversity but also water quality further downstream. Here, there is great potential for citizen science to contribute. Although small streams may have relatively low levels of biodiversity at site level, compared with some mid-order rivers, their communities are more heterogeneous across and within tributaries, and they are thus important in terms of their collective or regional biodiversity. Therefore, assessment and protection of small stream biodiversity should take a network perspective. Small streams originating in areas with high regional biodiversity should be identified and given priority for monitoring and protection measures.

EPA RESEARCH PROGRAMME 2021–2030

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(2017-W-LS-14)

EPA Research Report

Prepared for the Environmental Protection Agency

by

University College Dublin

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This report is based on research carried out/data from 2016 to 2020. More recent data may have become available since the research was completed.

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

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Executive Summary

The overall goal of this project was to refocus attention on the small stream network in terms of management and policy. The Small Stream Network (SSNet) project is the first large-scale research project in Ireland on first- and second-order streams that undertook investigations spanning hydrochemistry and multiple ecological elements, as well as experimental work, giving insights into the likely impact of climate change stressors. Furthermore, it provided an opportunity to test emerging DNA-based tools. Twenty-seven potential headwater catchment types were defined based on geology, soil drainage and physiography. Sites featuring 13 of these types, representing 75% of the total small stream network, were selected for the SSNet investigations.

Significant water quality problems were detected based on both existing data (from EPA monitoring) and newly collected data. For example, a substantial portion (approximately 57%) of the EPA small stream sites exceeded one or more of the phosphorus and nitrogen nutrient thresholds, 9% exceeded all three thresholds, 30% exceeded two thresholds and 19% exceeded one threshold. The more intensive sampling highlighted rainfall-driven inputs of phosphorus and nitrogen (diffuse pollution), with peak concentrations occurring during the winter months at most sites. Coupled with rain-driven inputs, the most nutrient-impaired sites, Tolka and Clonshanbo, exhibited the highest peak concentrations of phosphorus and total ammonia nitrogen during the summer months, indicating the presence of point discharges, potentially from domestic wastewater or septic tanks. The combination of high nutrient concentrations and flows, even for relatively short durations, can provide a significant proportion of the total nutrient load. The nutrient spiralling experiments detected variability in uptake lengths, highlighting a need for further research. This is especially important, as nutrient retention capacity in headwaters is compromised by excess nutrient loading due to potential nutrient saturation and reduced hyporheic water exchange, especially during high flows.

The hydromorphological study derived a revised river type classification that provides a basis for defining "reference" physical habitat assemblages and judging the degree of degradation in the physical habitat condition of all river and stream geomorphological types in Ireland. The biological sampling, in combination with previous data, provides a valuable baseline dataset for headwater streams in Ireland. The findings draw attention to the challenge for freshwater biodiversity protection, as few species are common to most streams. In fact, only 23 macroinvertebrates (16%) were found at more than 50% of sites, and 80 taxa had fewer than 10 site records, of which 38 (26%) occurred at a single site. Patchy species distribution was also a feature of the macrophytes and phytobenthos datasets. This was reinforced in the study on tributaries of two subcatchments (Dargle and Ballinagee) where *c.*20–29% of the taxa occurred at a single site. A combination of geology and physiography (capturing altitude and gradient/flow and associated substrate types) best describes the factors structuring freshwater communities in small streams. Overall, the results highlight the need to consider networks of protected sites for effective biodiversity protection. Fish studies confirmed the presence of brown trout and sea trout in some cases in small coastal streams, highlighting their vital, but unrecognised, role for sea trout spawning and nursery.

The Exstream field experiment investigated the influence of variable flow, carbon dioxide enrichment, fine sediment pulses and lack of shading – individually and in combination – on stream communities and key functions. It highlighted siltation as a key stressor on small stream functioning, impacting different organism groups and the whole ecosystem. Although flow variability was a lesser influence, it still negatively affected the invertebrate community and moderated the effects of other stressors, particularly during low-flow episodes.

A catchment modelling methodology was demonstrated on six catchments that can help identify areas at the highest risk of export of nutrients or sediment and that can be applied to small catchments. While we have shown the model's use for identifying hotspots, further uses, for instance to reliably estimate specific numerical values for loads or concentrations in small headwater catchments, would require model calibration, regardless of the model used.

There is great potential to engage citizen scientists in the biomonitoring of small streams, which will not only fill data gaps but also raise awareness of water quality issues at the local level. In summary, protection of the water quality, biodiversity and ecosystem services of the small stream network requires a combined top-down (policy and management) and bottom-up (community and individual) effort.

1 Introduction

Rivers are network systems dominated by narrow channels, which are generally referred to as small streams. There is, however, no universally accepted definition of what constitutes a small stream, and the term is often used interchangeably with headwater (Riley *et al*., 2018). Most small streams lie in headwater reaches, but they also include small lowland streams and short coastal streams (Ovenden and Gregory, 1980; Moore and Richardson, 2003). Small streams, including headwaters, are typically categorised on the basis of stream order (generally zero to second order, e.g. Meyer *et al*., 2007a; Clarke *et al*., 2008; Barmuta *et al*., 2009), distance from source (up to 2.5 km, e.g. Furze, 2000), catchment area (1–10 km^2, e.g. Gomi *et al*., 2002; MacDonald and Coe, 2007) and stream width (typically less than 2.5 m, e.g. Anon., 2012; Downing *et al*., 2012). They have also been classified on the basis of flow regime into perennial, intermittent and ephemeral streams, with the last two most often referred to as temporary streams. Regardless of the categorisation, small streams constitute a large proportion of the river network; global figures are as high as 80% of the total length (Downing *et al*., 2012) and a similar overall figure has been given for Europe (Kristensen and Globevnik, 2014). In Ireland, 75% of the river network comprises first- (51%) and second-order (24%) streams (Kelly-Quinn *et al*., 2020). Although less common, some streams in Ireland are intermittent, flowing for only part of the time. Headwater flows in Ireland arise from several different sources: seepage from soils (including peatlands), subsoils and aquifers, springs and lake outlets (Callanan *et al*., 2008a).

The importance of small streams has been highlighted, although the number of papers is relatively small compared with what has been published on larger rivers. Small streams act as the “capillaries” of the landscape, capturing and moving water; for example, 70–90% of a river’s flow is estimated to come from the headwaters (e.g. Alexander *et al*., 2007). They play a critically important role in flow regulation through both water storage/groundwater recharge and flood amelioration, although there is a paucity of quantitative estimates. Retention and cycling of nutrients in small streams provide an important ecosystem service by transforming and regulating the downstream delivery of nutrients (Withers and Jarvie, 2008; Riley *et al*., 2018; Ferreira *et al*., 2022). Their contribution in this regard is often disproportionate to their size and can significantly influence downstream water quality and ecological integrity (Gomi *et al*., 2002; Alexander *et al*., 2007; Dodds and Oakes, 2008; Riley *et al*., 2018). Here, again, quantitative data are scarce for Ireland and in many other countries. The aforementioned services are among the 27 ecosystem services provided by small streams (Ferreira *et al*., 2022), which also include regulation of climate, erosion and flooding, and cultural services and the biodiversity that underpins the various services.

Meyer *et al*. (2007b) categorised the biodiversity of headwater streams, separating species into four categories: (i) species that are unique to this part of the river network; (ii) species that occur there but also in larger rivers; (iii) species that move into headwaters seasonally; and (iv) species that migrate there to complete particular life history stages (e.g. salmonids for spawning). They also proposed including species that live near these streams in semi-aquatic or riparian habitats. While any one stream may support few species, it is the combined total from the network that makes them important, and this emphasises the need to consider the biodiversity potential of each headwater branch, and, ultimately, the protection of key branches that make a significant contribution to either regional or catchment biodiversity, provide habitat for rare or protected species (e.g. pearl mussel) and contribute to the ecological integrity of the entire river network (Furze, 2000; Heino, 2005; Clarke *et al*., 2008; Finn, 2011). The most extensive study on headwaters in Ireland was carried out by Callanan *et al*. (2012), although it was based on only macroinvertebrates. Apart from limited data from the RIVTYPE (Characterisation of Reference Conditions and Testing of Typology of Rivers; Kelly-Quinn *et al*., 2005) and FORWATER (Forestry & Surface Water Acidification; Kelly-Quinn *et al*., 2008) projects, and those held by the EPA, the phytobenthos assemblages of small streams have not been adequately described, but are recognised for their important contributions in terms of nutrient assimilation and oxygen budgets.

Similarly, apart from Weekes *et al*. (2014), there are few studies on the macrophytes of small streams. Other sources of information on small streams in Ireland are more disparate and are part of studies with varying objectives. Overall, we have a very patchy knowledge of the biodiversity of small streams. It is generally hypothesised that the physical habitat and hydrological diversity contributes to the ecological autonomy of the network branches (Lowe and Likens, 2005), and thus their biodiversity importance, although few studies have demonstrated the role of these factors at network scale. Here, again, this is a knowledge gap that needs to be addressed in Ireland. The role of headwaters in supporting fish populations is well known, particularly in relation to salmonids. However, small coastal streams in Ireland, and elsewhere in Europe, have received little, if any, attention to date (Whelan, 2014). Ranging from several hundred metres to several kilometres in length, and generally no larger than first- or second-order systems, these are common along the entire seaboard of Ireland. They are likely to provide habitat for salmonids and are potentially a major player in the recruitment and sustainability of coastal sea trout fisheries (McCully and Whelan, 2013).

The small stream network is vulnerable to anthropogenic diffuse pollution from agriculture and forestry activities (e.g. Clarke *et al*., 2015; Mellander *et al*., 2015; Riley *et al*., 2018; Ferreira *et al*., 2022) and point sources of pollution from wastewater treatment plants due to their high connectivity with adjacent land, large contributing catchment relative to their size, low dilution capacity, and, in many cases, short water residence times, particularly where land has been drained (e.g. Snell *et al*., 2014; Riley *et al*., 2018). These discharges can compromise the capacity of headwater streams to assimilate and store nutrients and organic matter, with resulting higher export to downstream reaches (Alexander *et al*., 2007). Physical habitat or hydromorphological alterations due to drainage and channel modification pose problems at local and catchment levels (Bradley *et al*., 2015; EPA, 2022), affecting habitat heterogeneity and their biodiversity potential, with increased flow rates, reducing water and sediment residence times (Alexander *et al*., 2007). High flows can also result in increased streambank erosion and delivery of sediments from soil disturbance associated with agriculture and forestry. Furthermore, Ireland has a dense road network with numerous culverts on small streams (Kelly-Quinn *et al*., 2022a), which can lead to inputs of heavy metal and polycyclic aromatic hydrocarbon contaminants (Maltby *et al*., 1995). The Small Stream Network (SSNet) research project addressed many of these highlighted knowledge gaps, and provides a science-informed knowledge base for the management of the small stream network for improved local and downstream water quality and protection of biodiversity and ecosystem services.

1.1 Policy Relevance

Addressing water quality issues in the small stream network is of direct relevance to efforts to address restoration of water quality, as required by the Water Framework Directive (WFD; EU, 2000). Other policies, particularly in relation to biodiversity, include national biodiversity plans, the EU Biodiversity Strategy for 2030 and associated Nature Restoration Law, the Habitats Directive, the Birds Directive, the United Nations (UN) Convention on Biological Diversity (including Aichi targets), the Post-2020 Global Biodiversity Framework, the Common Agricultural Policy and associated schemes. Elements of the proposed research address one of the cross-cutting themes (land and water management, including soil systems) for climate adaptation and mitigation, as well as agendas such as the UN Sustainable Development Goals.

1.2 Aims and Objectives

The overall objective of SSNet was to advance knowledge on the role of small streams in water quality, biodiversity and ecosystem services protection, to inform policy, measures and management options to meet the WFD objectives and other regulatory targets.

The specific objectives were as follows:

- producing for stakeholders a synthesis of the current knowledge on the importance of small streams (published as a separate two-page fact sheet (https://www.catchments.ie/the-importance-of-the-small-stream-network-in-ireland/));
- investigating the hydrochemical characteristics of small streams in Ireland (Chapter 3);

- building an integrated understanding of headwater stream hydromorphology across Ireland (Chapter 4);
- describing the biodiversity of small streams at varying scales (Chapter 5);
- investigating hydrological influences on biodiversity and ecosystem function in small streams (Chapter 6);
- modelling the intervention required in the small stream network to impact on nutrient and sediment export (Chapter 7);
- exploring options for increased engagement of citizen science in monitoring the physical and ecological health of small streams (Chapter 8);
- making recommendations for the management of the small stream network (Chapter 8).

In SSNet, we used first and second order to define small streams. These are hereafter referred to as small streams or headwaters. A central task before the research investigations began was identification of the possible types of small stream settings based on catchment characteristics (Chapter 2).

2 Defining a Typology for Study Site Selection (SSNet Types)

Potential headwater catchment types were defined based on three physical descriptors: geology (GSI, 2006), soil drainage (EPA, 2019) and physiography (GSI, 2018). In terms of geology, three main categories were used: igneous/metamorphic (igneous: "I"), sedimentary non-calcareous (sedimentary: "S") and sedimentary calcareous (pure and impure; hereafter referred to as limestone: "L"). Sands and gravels were excluded since they occupy a very small area of the country. Soils were also divided into three major categories based on drainage characteristics: well drained ("W"), poorly drained ("Po") and peat ("Pe"). In terms of physiography, we, again, used three categories: hill to mountain ("M"), undulating to hill ("H") and flat to undulating ("P") – for further details see Cox *et al*. (2022). GLX represents sites on limestone with substantial groundwater input. In total, 95.97% of first- and second-order streams were classified based on the 27 SSNet catchment types (Figure 2.1; Cox *et al*., 2022). Thirteen of these types, representing 75% of the total first- and second-order stream network, were selected for the SSNet investigations (Table 2.1).

Four minimally impacted stream reaches (reference sites) were sought on first- and second-order streams in each of the SSNet catchment types. A desk study of candidate sites to determine possible human impacts included, where available, ecological water quality (Q-value scores) assigned to streams by the EPA, the EPA's Significant Pressures dataset (EPA, 2018) and the location of river channels that are part of the Irish Office of Public Works' arterial drainage scheme (OPW, 2004). However, there are few biological monitoring points on Irish first- and second-order streams. Furthermore, while the EPA's Significant Pressures dataset does indicate whether a WFD waterbody is actually experiencing a significant pressure, the spatial resolution was typically

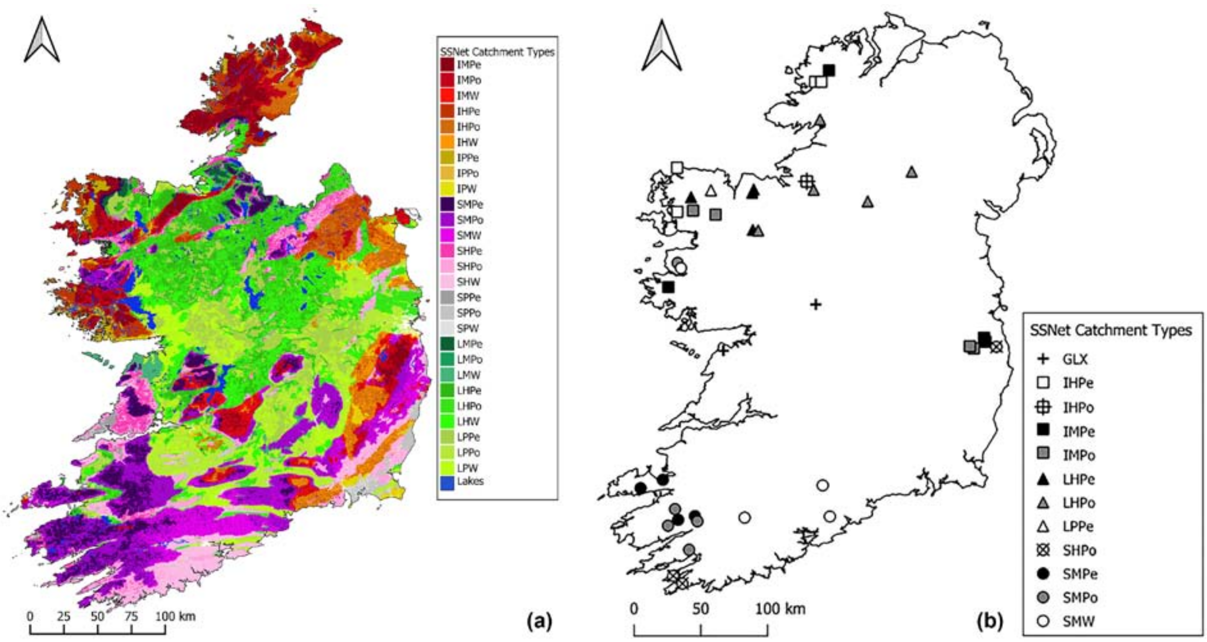


Figure 2.1. (a) Distribution of the SSNet types across Ireland; and (b) distribution of the minimally impacted sites identified from 12 of the 13 SSNet site types selected for the SSNet investigations, including an additional type, GLX, representing sites on limestone with substantial groundwater input. See Table 2.1 for explanation of the catchment type codes.

Table 2.1. The 13 SSNet types selected for the various research elements, ranked in order of channel length, site codes, physical descriptors, and the proportional representation of each type in the first- and second-order stream network

Prevalence by length	SSNet catchment type code	Geology	Physiography	Soil type	Percentage of total first- and second-order streams	Cumulative percentage of first- and second-order streams
1	SMPo	Non-calcareous sedimentary	Hill to mountain	Poorly drained	13.1	13.1
2	LHPo	Calcareous sedimentary	Undulating to hill	Poorly drained	7.3	20.3
3	IMPe	Igneous and metamorphic	Hill to mountain	Peat	7.1	27.4
4	LPPo	Calcareous sedimentary	Flat to undulating	Poorly drained	6.9	34.3
5	IMPo	Igneous and metamorphic	Hill to mountain	Poorly drained	5.7	40.0
6	LHPe	Calcareous sedimentary	Undulating to hill	Peat	5.6	45.6
7	IHPo	Igneous and metamorphic	Undulating to hill	Poorly drained	5.5	51.1
8	SHPo	Non-calcareous sedimentary	Undulating to hill	Poorly drained	5.5	56.6
9	SMW	Non-calcareous sedimentary	Hill to mountain	Well drained	5.2	61.8
10	LPPe	Calcareous sedimentary	Flat to undulating	Peat	5.1	66.9
11	SMPe	Non-calcareous sedimentary	Hill to mountain	Peat	4.9	71.8
12	IHPe	Igneous and metamorphic	Undulating to hill	Peat	3.3	75.1
13	GLX	Karstic geology with significant conduit groundwater flow paths	n/a	n/a	Approximately 20% of land cover	n/a

n/a, not applicable.

insufficient to aid site selection, with first- and second-order streams often located above the pressure that had been assigned to them. Thus, candidate streams were frequently located using Google Earth aerial imagery followed by a field visit to confirm the level of human modifications or land use pressures. This resulted in 42 study sites, details of which are given in Appendix 1.

3 Hydrochemical Characteristics of the Small Stream Network

3.1 Introduction

This chapter outlines the hydrochemical characteristics of small streams in Ireland based on monitoring data provided by the EPA and a desk-based characterisation of the sources of nutrient impairment by evaluating and analysing relevant historical data collected on some 199 small stream sites, including flow and rainfall data, and potential mitigation measures. It also profiles a number of sites ($n=73$) across the selected SSNet types outlined in Chapter 2 based on one-off chemical analysis (sampled summer of 2019). Higher frequency sampling (samples every 2 weeks) for nutrient analysis was carried out at a subset of sites ($n=7$), and rainfall event sampling at three of the seven sites. The analysis of fortnightly samples was combined with flow data for the generation of concentration–flow relationships to identify potential nutrient sources affecting the study sites and increase knowledge of nutrient behaviour, transport and nutrient sources under different flow regimes. Nutrient spiralling experiments were also undertaken to assess nitrogen and phosphorus uptake patterns during the summer months.

3.2 Chemical Characteristics of the Small Stream Sites in the EPA Dataset

3.2.1 Approach

EPA-monitored data from 2007 to 2017 were initially collated for 252 study sites. The number of study sites was subsequently reduced to 199 (referred to as EPA sites and shown in Figure 3.1) following the exclusion of sites with 10 or fewer sampling dates and sites that comprised a mix of SSNet types within 2 km of the monitored location.

Nutrient conditions at the EPA sites were analysed to identify the level of compliances and exceedances. In the case of molybdate-reactive phosphorus (MRP) and total ammonia nitrogen (TAN), the thresholds set in S.I. 272 (Government of Ireland, 2009) (later amended in S.I. 77; Government of Ireland, 2019) were used. There is no WFD threshold for total oxidised nitrogen (TON), and therefore the Local Authority Water Programme (LAWPRO) surrogate standards, i.e. a mean of < 1.8 mg N/L for high status and a mean of > 1.8 mg N/L and < 3.0 mg N/L for good status, were used. Further details of the analyses are given in Hogan *et al*. (2023).

3.2.2 Key results: nutrient impairment

In terms of MRP, 79 (39%) and 25 (12%) sites met the criteria for high (i.e. mean ≤ 0.025 mg P/L) and good (i.e. 0.025 mg P/L ≤ mean ≤ 0.035 mg P/L (not including high status)) status, respectively, meaning that approximately 49% of sites exceeded the threshold for

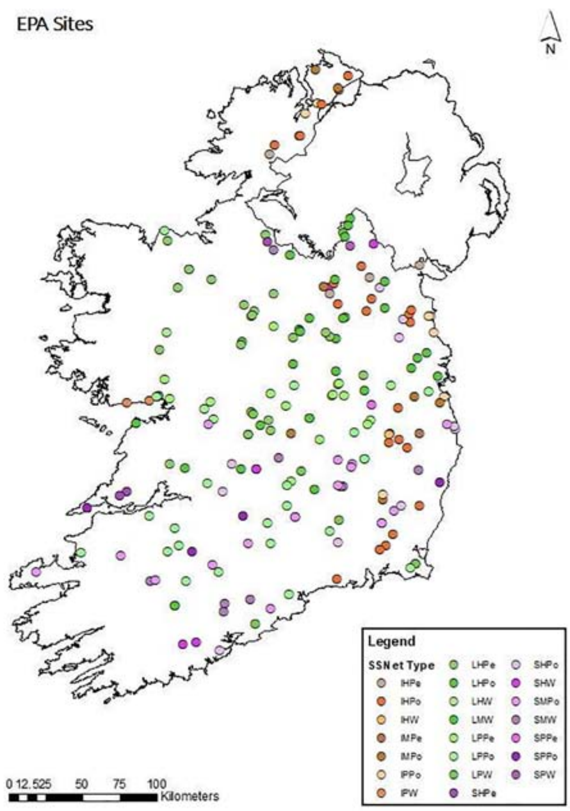


Figure 3.1. Location of 199 EPA extensive small stream sites. Colours pertain to SSNet types (Chapter 2): orange = igneous types; green = limestone types; and pink = sedimentary types.

good status, with values ranging from 0.0355 mg P/L to 1.5 mg P/L (Figure 3.2). For TAN, 39% of sites had high status (i.e. ≤0.040 mg N/L), 20% had good status (i.e. 0.040 mg N/L ≤ mean ≤0.065 mg N/L (not including high status)) and 41% exceeded the good status threshold (Figure 3.3), with mean values ranging from 0.0658 mg N/L to as high as 4.2763 mg N/L. Mean values for TON at 50% of sites were shown to be at or below the LAWPRO surrogate standard of 1.8 mg N/L for high status, with mean TON values at a further 17% of sites being greater than 1.8 mg N/L but less than 3.0 mg N/L, and 26% of sites having mean TON values greater than 3.0 mg N/L (Figure 3.4), where values ranged from 3.265 mg N/L to 7.9 mg N/L.

In terms of the combination of all three nutrients (MRP, TON and TAN) for the 11-year period (2007–2017), 18% of study sites were shown to comply with the thresholds for high status for MRP and TAN, and were below the mean of 1.8 mg N/L for TON. Only 1% of the remaining sites complied with good status for MRP and

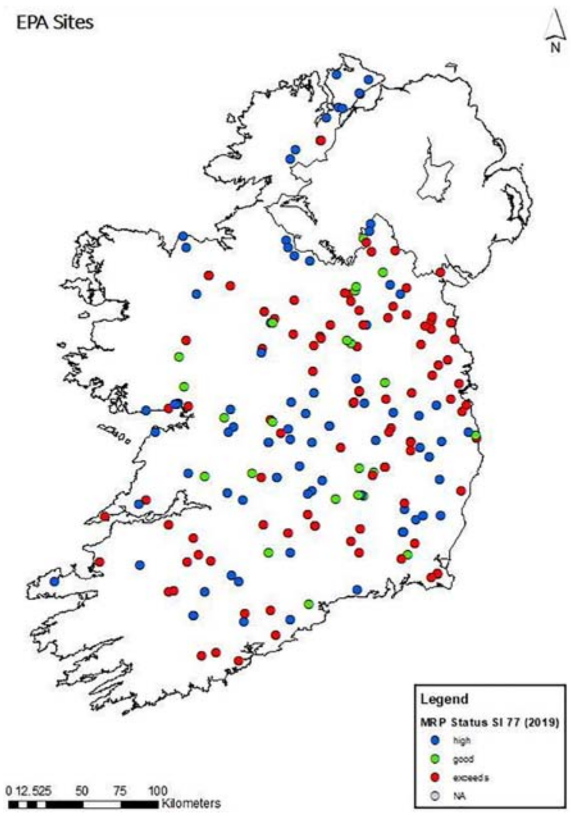


Figure 3.2. MRP status for the 199 EPA extensive small stream sites. Markers relate to S.I. 77 (Government of Ireland, 2019) high status (blue dots), good status (green dots) and exceeding compliance threshold (red dots); sites with no MRP data have a grey dot.

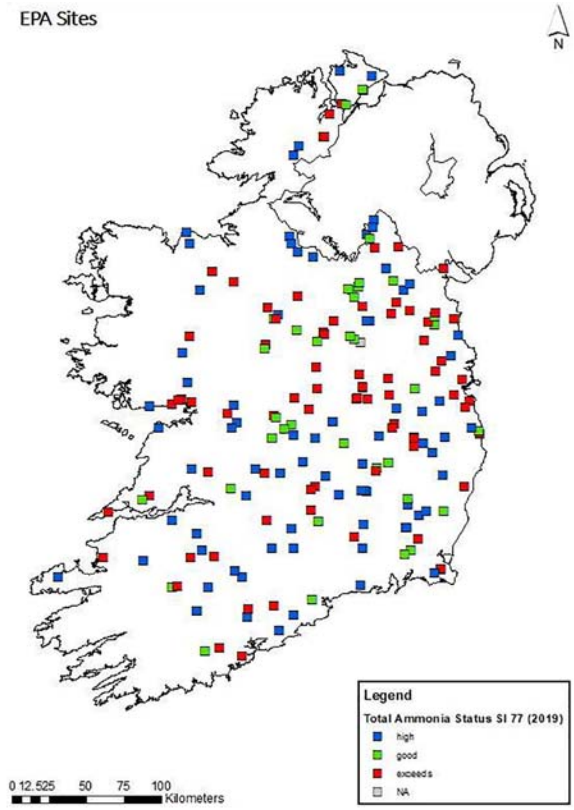


Figure 3.3. TAN status for the 199 EPA extensive small stream sites. Markers relate to S.I. 77 (Government of Ireland, 2019) high status (blue squares), good status (green squares) and exceeding compliance threshold (red squares); sites with no TAN data have a grey square.

TAN, and fell between the means of 1.8 mg N/L and 3.0 mg N/L for TON. Nine per cent of sites exceeded all three thresholds. A substantial portion (approximately 57%; two sites lacked TON data) of the 199 monitored EPA small stream sites exceeded one or more of the nutrient compliance thresholds; 9% of sites exceeded all three thresholds, 30% exceeded two thresholds and 19% exceeded one of the thresholds. These sites were then categorised according to the level of non-compliance (see Table 3.1). The SSNet catchment types that displayed the highest percentage of non-compliance were limestone types, at 65%, followed by igneous and sedimentary types, at 58% and 42%, respectively.

Of the 114 sites that had mean values exceeding the aforementioned nutrient thresholds, 82% ($n=94$) had Q-values less than Q4 and none was rated Q5 or Q4–5. The 20 Q4 sites had exceedances in two or more nutrients, mainly MRP and/or TON and TAN. While there was considerable variation in the range

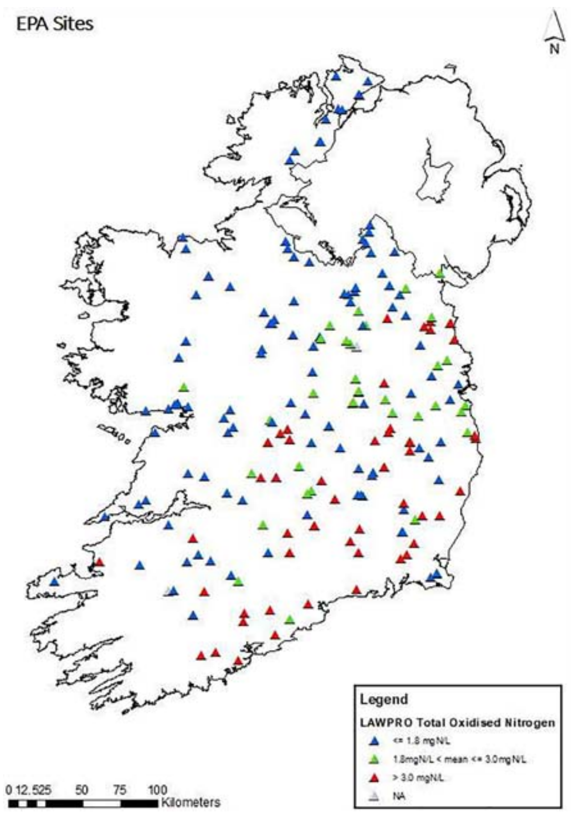


Figure 3.4. TON status for the EPA extensive small stream sites. Markers relate to LAWPRO high status (blue triangles), good status (green triangles) and exceeding threshold (red triangles); sites with no TON data have a grey triangle.

of these nutrient concentrations, the mean MRP across all Q4 sites was lower than other Q-value site groupings. Fifty-eight per cent (*n* = 47) of the sites (*n* = 81) that were nutrient compliant and that had a Q-value rating were rated Q4 or Q4–5.

3.3 Nutrient Characteristics from One-off Sampling

In summer 2019, one-off chemical analyses were conducted on 73 small streams around Ireland comprising potentially impacted and non-impacted sites, across the 12 most dominant SSNet types in the study sites (see Figure 3.5). The determinands considered were the nutrients total reactive phosphorus (TRP) (aka MRP), TAN and TON, and alkalinity, following standard methods (APHA, 1999). The other physicochemical determinands were measured in the field using a Hach Pocket Pro pH and conductivity probe for pH and conductivity and a Hach dissolved oxygen (DO) meter with optical sensor for DO percentage.

In terms of TRP, 34% of sites (25 of 73, including six that were potentially non-impacted) exceeded the good status threshold at the time of sampling, 10% were between the high and good status thresholds and 56% had values lower than the high status threshold. There was more variability in TRP in the potentially impacted sites, with values ranging from 0.003 mg P/L to 1.874 mg P/L, and the highest variability typically observed in the poorly drained SSNet catchment types (IHPo, IMPo, LHPo, SHPo and SMPo). There was less variability in the potentially non-impacted sites, with values ranging from below the detection limits to 0.173 mg P/L. For TAN, 5% of the sites exceeded the good status threshold and 5% were within it, with the remaining 90% having values less than the high status value. Similarly to TRP, TAN was more variable in the potentially impacted sites, with values ranging between 0.010 mg N/L and 0.508 mg N/L, with the greatest

Table 3.1. Level of non-compliance and the number of sites in each level with thresholds as defined in section 3.2

Level	Description	Nutrient	No. of sites	Total
1	Exceeds the threshold for only one nutrient	MRP > 0.035 mg P/L (S.I. 77)	13	37
		TAN > 0.065 mg N/L (S.I. 77)	11	
		TON > 3.0 mg N/L (LAWPRO)	13	
2	Exceeds the thresholds for two nutrients	MRP and TAN	43	59
		MRP and TON	15	
		TAN and TON	1	
3	Exceeds the thresholds for three nutrients	MRP, TON and TAN	18	18

Thresholds taken from S.I. 77 (Government of Ireland, 2019) for good status for MRP and TAN. For TON the value quoted is the LAWPRO upper guidance threshold of 3.0 mg N/L.

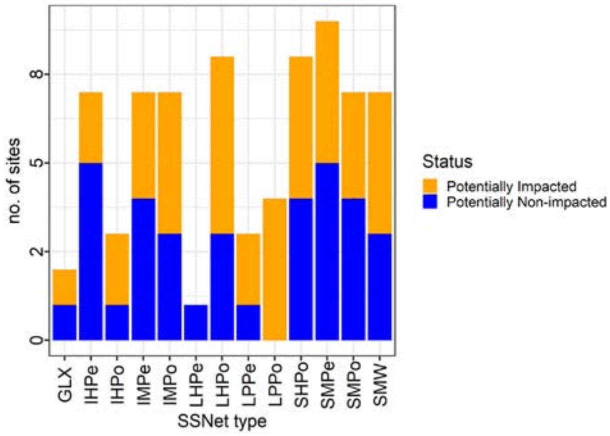


Figure 3.5. The number of potentially impacted (orange) and non-impacted (blue) one-off sampling sites across the 12 dominant SSNet types and a groundwater type (GLX).

variability observed in the SMW SSNet type. The potentially non-impacted sites had a narrower range of TAN values, between 0.001 mg N/L and 0.042 mg N/L. The results for TON were similar to those for TAN, with 10% of sites having values greater than 3.0 mg N/L, 7% having values between 1.8 mg N/L and 3.0 mg N/L, and 83% of sites having measured TON values of less than 1.8 mg N/L. Similarly to TRP and TAN, TON varied considerably in the potentially impacted sites, ranging between 0.00 mg N/L and 9.70 mg N/L, particularly in the LHPo and LPPo catchment types, with values, again, being less variable in the potentially non-impacted sites (ranging from 0.00 mg N/L to 2.94 mg N/L).

3.4 Higher Frequency Hydrochemical Sampling

3.4.1 Approach

In February 2020, a subset of 7 of the 199 EPA-monitored sites and the 73 snap-shot study sites were selected for intensive study (hereafter referred to as intensive sites). This investigation involved fortnightly water sampling from June 2020 to May 2021 at each site. Analyses for total phosphorus (TP), total dissolved phosphorus, soluble reactive phosphorus (SRP), TAN, nitrite, nitrate, the ions sulfate, chloride (Cl), sodium, potassium, calcium and magnesium, alkalinity, pH, DO, conductivity, total carbon (TC) and dissolved inorganic carbon were conducted using standard methods (APHA, 1999). The sites chosen represented SSNet types covering 45% of the total SSNet stream length in Ireland. The first two sites, Annalecka (53.063627, –6.409784; 412 m above sea level (m.a.s.l.) and Glashaboy (53.055556, –6.411111; 393 m.a.s.l.) in the Wicklow mountains, were from IMPo and IMPe SSNet types, respectively. The catchment geology for the two sites is described as Siluro-Devonian granitic rocks and appinite (GSI, 2021). They differ in their soil classification, however, with Annalecka being characterised by podzols (peaty), lithosols, peats and outcropping rock, and the Glashaboy catchment comprising significant areas of blanket peats. A further three sites, Vartry (53.112030, –6.189451), Newtown Mount Kennedy (53.106400, –6.134806) and Newcastle (53.071310, –6.074836), also in Wicklow, were located on sedimentary geology (Cambrian greywacke, slate, quartzite). While the Vartry and Newtown Mount Kennedy sites reflect hills to mountain physiography, with elevations of 116.22 m.a.s.l. and 241.25 m.a.s.l., respectively, they differ in their soil type. The Newtown Mount Kennedy site is categorised as having shallow acid brown earths/brown podzolics, lithosols and regosols, and is therefore well drained, while the Vartry site is poorly drained, being characterised by peaty gleys, surface water gleys and groundwater gleys. The Newcastle site is also poorly drained, with soils of surface water gleys and groundwater gleys. It is located at approximately 36 m.a.s.l. The final two sites, one in Meath and one in Kildare, are on limestone. The Tolka site (53.448025, –6.510386), with an elevation of 95 m.a.s.l., drains Visean limestone and calcareous shale, with poorly draining surface water gleys, and groundwater gleys with limestones. The Clonshanbo site (53.348031, –6.710533), with an elevation of 69 m.a.s.l., is on Tournaisian limestone and has poorly draining soil (surface water gleys and groundwater gleys with limestones). The suitability of all seven sites for installing automatic water samplers and for establishing flow records was confirmed prior to their selection.

3.4.2 Key results

A range of nutrient conditions existed across the seven intensive sites, highlighting different nutrient pressures (Table 3.2). At five sites mean TRP concentration was lower than the good status threshold (at four of which mean TRP concentration was lower than the high status threshold), while two sites (Newcastle and

Table 3.2. Intensive study sites in terms of SSNet type, significant pressure, land use and nutrient annual mean and maximum values recorded during the 2020/2021 sampling programme

	River						
	ANNA	CLON	GLASH	NEWC	NTMK	TOLK	VART
SSNet type	IMPo	LPPo	IMPe	SHPo	SMW	LHPo	SMPo
Pressure	Forestry	Agriculture	Forestry	Agriculture	Domestic wastewater	Agriculture	n/a
Corine land use	Coniferous forest	Pasture	Coniferous forest	Non-irrigated arable land	Pasture	Pastures/non-irrigated arable land	Natural pasture
TRP (mg P/L)	0.017	0.034	0.007	0.044	0.012	0.126	0.004
Range (mg P/L)	0.002–0.039	0.016–0.073	0.000–0.030	0.000–0.286	0.002–0.047	0.071–0.245	0.000–0.020
TAN (mg N/L)	0.069	0.022	0.013	0.023	0.014	0.115	0.010
Range (mg N/L)	0.003–0.782	0.000–0.157	0.000–0.050	0.000–0.271	0.000–0.155	0.000–0.245	0.000–0.111
TON (mg N/L)	0.032	2.5	0.3	3.3	2.9	1.1	0.9
Range (mg N/L)	0.0–0.6	0.95–5.4	0.03–0.6	2.2–6.8	1.9–3.6	0.2–1.8	0.0–1.6

ANNA, Annalecka; Corine, Coordination of Information on the Environment; CLON, Clonshanbo; GLASH, Glashaboy; n/a, not applicable; NEWC, Newcastle; NTMK, Newtown Mount Kennedy; TOLK, Tolka; VART, Vartry.

Tolka) exceeded the good status threshold. Although mean concentrations were low at five sites, all but the Glashaboy and Vartry sites exhibited peaks in TRP on a number of dates. In contrast, the Tolka site exceeded the good status threshold on all dates. Similar threshold compliance results for TAN also applied across the seven sites, with five sites having a mean value lower than the good status threshold, and four of these having a mean TAN value lower than the high status threshold. Again, two sites (Tolka and Annalecka) exceeded the good status threshold. At four sites the mean values for TON were less than 1.8 mg N/L, two sites had values between 1.8 mg N/L and 3.0 mg N/L, while only one site exceeded 3.0 mg N/L. At most sites ($n=6$), maximum values for both TRP and TAN were observed during high-rainfall events (cumulative rainfall totals from 14.3 mm to 54.1 mm on sampling day plus day prior to sampling). The maximum nutrient values recorded during these high-rainfall events exceeded the mean values by a factor of 10 in some cases, indicating significant rain-driven nutrient inputs.

3.5 Nutrient Retention Capacity: Insights from Nutrient Spiralling Experiments

3.5.1 Introduction

As nutrients are transported downstream along a river channel, they can be bioaccumulated by aquatic biota, retained due to chemical reactions (such as precipitation and sedimentation) or adsorbed to riverine sediments. These biogeochemical processes slow the nutrients' transportation downstream. This process of repeated cycling of nutrients between their inorganic and organic forms is termed nutrient spiralling (Newbold *et al*., 2007). In streams, this is combined with downstream transport by advection and spreading in all directions by dispersion due to turbulence. In this project, we were particularly interested in the physical characteristics of uptake of inorganic nutrients, especially phosphorus, by the benthic biota as the water moves downstream. In July 2022, nutrient uptake experiments were conducted in stream reaches of four of the seven intensive sites, namely Vartry, Newtown Mount Kennedy, Newcastle and Glashaboy. The four sites represent both a range of SSNet types (Table 3.2) and a variety of land use types based on the Coordination of Information on the Environment (Corine) land use database (EPA, 2022), including broadleaf forest, coniferous forest, grassland pasture, non-irrigated arable land and discontinuous urban fabric. The sites also reflect a range of background nutrient conditions from impacted to potentially non-impacted.

3.5.2 Approach

The experimental and data analysis methods adopted in the experiments followed those of Baker and Webster (2017). Background data relating to discharge and wetted width, together with ambient mean concentrations for Cl, SRP and TAN, were collected for all sites. Two sondes that measured real-time conductivity were deployed at locations halfway along the measured reach and at the downstream extent of the reach (T2 and T4, respectively, in Figure 3.6). A solution containing both a conservative tracer salt (sodium chloride) and two reactive non-conservative tracer salts (ammonium chloride and potassium dihydrogen phosphate) was injected at the upstream extent of the reach. The constant injection rate produced a rise in concentrations downstream of the injection point, forming a temporary concentration plateau. Water samples were collected at selected monitoring points along the test reaches at specific time intervals (based on the time it took to reach the plateau). The water samples were then analysed for SRP, TAN and Cl using standard methods.

Field measurements taken during each experiment were subsequently used to determine the discharge Q using a dilution gauging method based on the released solution concentration of chlorine, C_r, background concentration, C_b, plateau concentrations, C_p, and the constant pump drip rate (estimated as the total volume injected/time), Q_R, according to equation 3.1:

$$Q = \frac{(C_r - C_b) \times Q_R}{C_p - C_b} \tag{3.1}$$

A key quantity is the mass transfer coefficient, v_f, which describes how the rate of nutrient uptake by the benthos depends on its concentration in the water. It has dimensions of L/T (analogous to a velocity), and a linear relationship is usually assumed between the nutrient mass taken up per unit area of stream bed per unit time and its concentration in the stream water. Once v_f has been determined for a given situation, the nutrient uptake per unit area of stream bed per unit time, U (with dimensions of mg/m^2/s), can be calculated for any nutrient concentration in the stream, as:

$$U = v_f C \tag{3.2}$$

where C is the concentration of the solute in the stream; here we estimated it using the measured background concentration (Table 3.3).

The average distance travelled by a nutrient molecule in dissolved inorganic form in a stream is called the "uptake length" (S_w) and this usually dominates the total spiralling length in streams. If the stream depth (h) and velocity (u) are known, S_w can also be calculated from v_f using:

$$S_w = \frac{u\,h}{v_f} \tag{3.3}$$

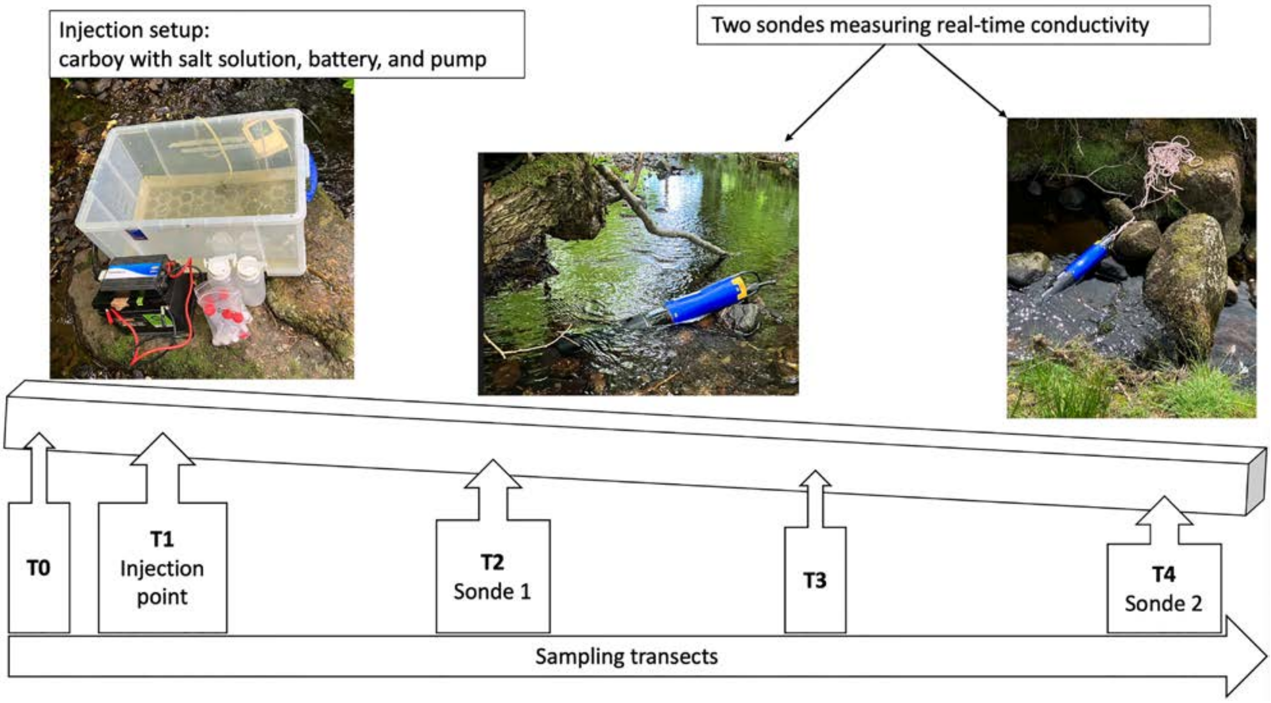


Figure 3.6. Experimental set-up for the spiralling experiments, including injection point, sonde locations and transect sections along a stream transect. This set-up was used at all four study sites.

Table 3.3. Results from the nutrient spiralling experiment

Parameter	Unit	Newtown Mount Kennedy	Glashaboy
Average stream width	m	3	3.25
Average stream depth	m	0.23	0.02
Average velocity	m/s	0.022	0.079
Average discharge	L/s	15.38	5.11
Background TAN	mg N/L	0.014	0.015
Background MRP	mg P/L	0.045	0.031
TAN			
k_w	per m	0.00162	0.00658
S_w	m	616	152
v_f	m/s	8.326×10^{-6}	1.035×10^{-5}
U	$mg/m^2/s$	0.000117	0.00015
Uptake per m^2 per day	$mg/m^2/day$	10.1	13.0
Stream input per day	mg/day	18,606	6403
MRP			
k_w	per m	0.00180	0.00259
S_w	m	554	386
v_f	m/s	9.252×10^{-6}	4.071×10^{-6}
U	$mg/m^2/s$	0.000413	0.000128
Uptake per m^2 per day	$mg/m^2/day$	35.7	11.1
Stream input per day	mg/day	59,361	13,877

Note that:

$$S_w = \frac{u}{k_c} = \frac{1}{k_w} \tag{3.4}$$

where k_c is a first-order uptake rate coefficient, given by:

$$k_c = \frac{v_f}{h} \tag{3.5}$$

where k_w is the inverse of S_w, and k_w is therefore also equal to (k_c/u).

These quantities can be determined for a stream by introducing nutrients into the stream at a point and measuring how their concentration changes at locations downstream (described above). Introducing a constant concentration of the nutrient for a fixed duration generates a temporary steady-state plateau of the nutrient concentration. The choice of duration depends on the stream characteristics and discharge, but should be sufficiently long to produce the desired "plateau". The relationship between distance downstream (x) and the nutrient concentrations of the plateau (C_p) at each downstream measuring point is described by equation 3.6 (see Stream Solute Workshop, 1990):

$$C_p(x) - C_b = (C_0 - C_b)\exp(-k_w\, x) \tag{3.6}$$

where $C_p(x)$ is the concentration of the plateau at a distance, x, downstream of the base point, C_b is the background concentration in the stream, and C_0 is the constant concentration achieved at the base point.

Equation 3.6 implies that the logarithm of ($C_p(x)-C_b$) should plot as a straight line with respect to the distance (x) downstream from the base point, and the slope of this line is $-k_w$. Thus, k_w can be determined from a graph of the experimental results, and, from this, the other quantities (described above), including nutrient uptake (v_f) and uptake length (S_w), can be calculated using average values for the stream depth (h) and velocity (u).

3.5.3 Key results

Although the project's experiments were conducted on four streams, we considered that the results for

Newcastle should be treated with caution, as it had a much higher background phosphorus concentration and a much greater variability in conductivity than the other streams. In contrast, Vartry had much lower nutrient concentrations than the other streams, and considerably less nutrients were therefore injected during the experiment. Thus, the chemical analysis results from Vartry were more sensitive to variability because of the much lower concentrations involved. Therefore, here we report the results for the other two experiments only, i.e. those conducted on the Newtown Mount Kennedy and the Glashaboy streams (Table 3.3). The Cl (conservative) plateau for the experiment in Glashaboy is shown in Figure 3.7 and the MRP plateaux for points T2 and T4 are shown in Figure 3.8, which shows a decline in the plateau concentration, i.e. uptake of MRP, for the downstream points, typical for a non-conservative substance.

For TAN, the uptake length (S_w) varied between 152 m for the Glashaboy stream and 616 m for the Newtown Mount Kennedy stream, mainly due to the latter's greater discharge and/or its greater depth, both of which would tend to increase S_w. The greater depth and flow velocity reduces the ability of the biota (and sediment) to retain phosphorus and nitrogen, and so the spiral length is longer. The TAN uptakes, assuming the measured background concentration, were similar for both streams, with values of 10.1 mg/m^2/day and 13.0 mg/m^2/day, respectively.

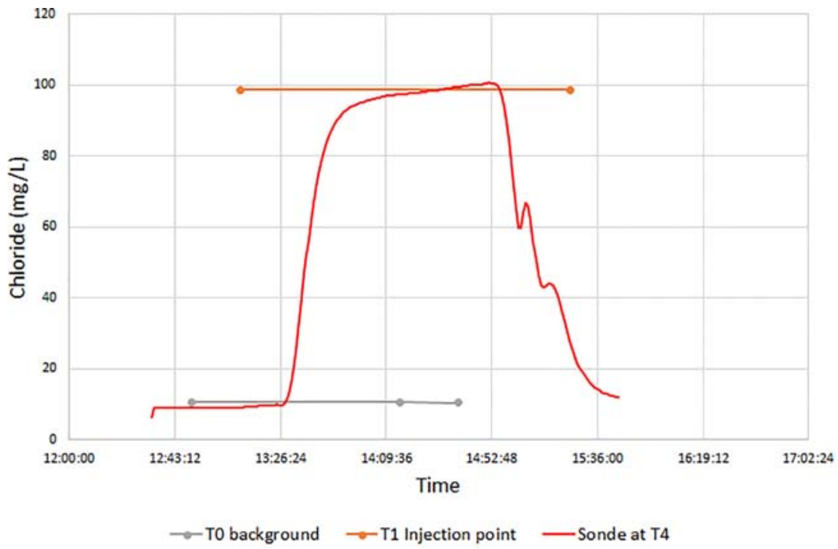


Figure 3.7. Cl plateau for Glashaboy.

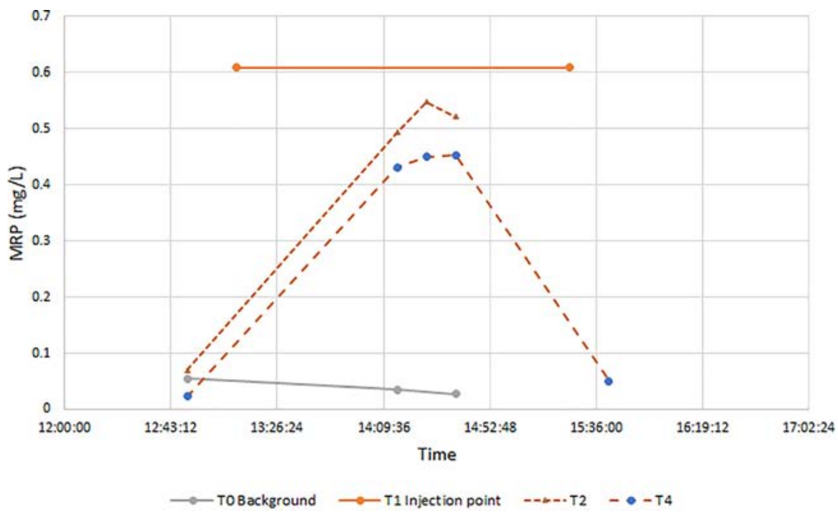


Figure 3.8. MRP plateaux for Glashaboy.

For MRP, the uptake length varied from 386 m for the Glashaboy stream to 544 m for the Newtown Mount Kennedy stream, again being strongly influenced by the greater discharge. However, in contrast to TAN, the MRP uptake rates (assuming background concentrations) were quite different, 11.1 mg/m^2/day for the Glashaboy stream and 35.7 mg/m^2/day for the Newtown Mount Kennedy stream. The higher background MRP concentration in the latter contributed to this difference. For general comparison, the total amount of each nutrient entering the test section per day in each stream is also given in Table 3.3.

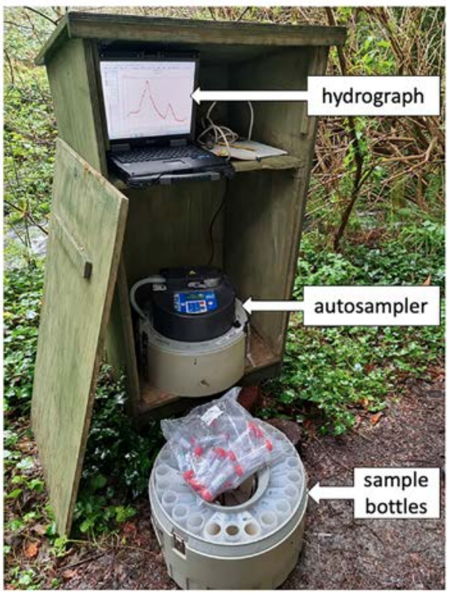
3.6 Nutrient Input Characteristics During Storm Events

3.6.1 Approach

Three sites (Newcastle, Vartry and Newtown Mount Kennedy), again selected from the intensive sites, and described in section 3.4, were used to investigate nutrient conditions and behaviour during high-flow, rainfall-driven events. The sites were instrumented with Teledyne ISCO 6712 portable autosamplers fitted with ISCO 750 area velocity (AV) flow modules and probes, which measured flow and water level at 1-min intervals (Figure 3.9a). The AV probe was calibrated in a laboratory flume and then mounted on a concrete slab to be secured into the riverbed (Figure 3.9b). When rainfall events were forecast, a “firing schedule” was pre-programmed in the autosamplers to extract water samples at specified time intervals (varying from regular 20-min intervals for the early part of the storm, on the rising hydrograph limb, to 90-min intervals, on the receding hydrograph limb), coinciding with rainfall episodes of between 24 and 36-hours’ duration. Four events were captured at the Newcastle and Newtown Mount Kennedy sites, and three at the Vartry site. In each instance, eight samples were collected over the storm duration (Figure 3.11), and analysis of these samples was subsequently undertaken using standard methods (APHA, 1999) to determine TP, SRP, TAN, TON and TC concentrations.

3.6.2 Key results

The key results from the event sampling (September 2022 to April 2023), including event date, event mean values and cumulative rainfall (calculated by adding rainfall on day of sampling and previous day), are summarised in Table 3.4. To facilitate comparison, nutrient means, 75th percentile values and maximum values measured during the fortnightly sampling in 2020/2021 are also included.



(a)

(b)

Figure 3.9. Instrumentation at the Newcastle site, including (a) a laptop showing the hydrograph, the autosampler with AV module and sample bottles containing sample water captured during an event; and (b) the AV probe in the centre of the channel being secured to the riverbed.

Table 3.4. Sampling results for three sites from heavy rainfall events captured over winter 2022/2023

Site		Rainfall (mm)	TP (mg P/L)	SRP (mg P/L)	TAN (mg N/L)	TON (mg N/L)
Newtown Mount Kennedy	Mean 2020/2021*		0.027	0.007	0.014	2.6
	75th percentile 2020/2021*		0.032	0.011	0.010	3.2
	Max. 2020/2021*		0.072	0.027	0.155	3.6
	Event 1: 22/9/22	11.7–13.5	0.055	0.015	0.032	1.6
	Event 2: 19/10/22	45.1	0.125	0.036	0.064	1.4
	Event 3: 15/3/23	±18	0.041	0.005	0.022	1.9
	Event 4: 1/4/23	12	0.062	0.008	0.087	0.6
Newcastle	Mean 2020/2021*		0.053	0.033	0.044	3.18
	75th percentile 2020/2021*		0.055	0.030	0.038	3.85
	Max. 2020/2021*		0.324	0.195	0.286	4.59
	Event 1: 22/9/22	11.7–13.5	0.057	0.009	0.024	2.8
	Event 2: 19/10/22	45.1	0.192	0.104	0.094	2.2
	Event 3: 15/3/23	±18	0.106	0.057	0.084	2.2
	Event 4: 1/4/23	12	0.090	0.038	0.057	2.7
Vartry	Mean 2020/2021*		0.010	0.002	0.010	0.94
	75th percentile 2020/2021*		0.014	0.003	0.007	1.18
	Max. 2020/2021*		0.030	0.009	0.111	1.56
	Event 1: 19/10/22	52.1	0.045	0.001	0.055	0.7
	Event 2: 15/3/23	±20	0.024	0.000	0.012	0.9
	Event 3: 2/4/23	9	0.009	0.001	0.003	0.8

Mean values for the nutrients TP, SRP, TAN and TON were calculated from concentrations of eight samples captured over a period of 24–36 hours. The cumulative rainfall is calculated from the actual rain on the day of sampling and the previous day.

***Values calculated during the fortnightly sampling on intensive sites in 2020/2021 (section 3.4).**

Max., maximum.

At the Newtown Mount Kennedy site, the SRP concentrations during events 1 and 2, which ranged from 0.002 mg P/L (event 1) to 0.048 mg P/L (event 2), were almost always above the fortnightly sampling (2020/2021) mean of 0.007 mg P/L, and the majority of concentrations (from six of eight samples) captured during event 2 were generally above the good status threshold. In contrast, events 3 and 4 had relatively low SRP concentrations (ranging from 0.000 mg P/L to 0.008 mg P/L). TP concentrations were consistently higher than those from SRP, sometimes by a factor of 10, which is likely to reflect the role of increased particulate mobilisation during rainfall events. The TAN concentrations ranged from 0.016 mg N/L (event 3) to 0.112 mg N/L (event 4) across all four events at Newtown Mount Kennedy. Values here were above the fortnightly sampling mean of 0.014 mg N/L, and many of the concentrations captured during event 2 (from six of eight samples) and event 4 (all eight samples) were above the good status threshold for TAN. TON values, ranging from 0.3 mg N/L (event 4) to 2.3 mg N/L (event 3), were all below the 2020/2021 mean of 2.6 mg N/L.

At the Newcastle site, events 2, 3 and 4 had concentrations for SRP that were mostly higher (seven of eight samples for event 2; all samples for event 3; six of eight samples for event 4) than the 2020/2021 mean value (0.033 mg P/L) and were also above the good status threshold. Similarly to the Newtown Mount Kennedy site, the dominant phosphorus input for all events was from TP, and in some cases the values were three times that of TRP, again suggesting increased particulate mobilisation. TON concentrations across all events were generally less than the 2020/2021 mean value. For the Vartry site, only a small number of values (3 of 24 samples across the three events) were above the 2020/2021 mean value for SRP (0.002 mg P/L). As with the previous two sites, the dominant phosphorus was from TP, with values in excess of 10 times that of SRP. During events 1 and 2, all TAN values were greater than the 2020/2021 mean (ranging from 0.011 mg N/L to 0.078 mg N/L) and half of the values during event 1 exceeded the good status threshold. Again, and similarly to the other sites, TON was always below the 2020/2021 mean value.

Generally, the event mean values for the nutrients TP, SRP and TAN from each captured event at the three sites almost always exceeded the 2020/2021 mean concentrations from the fortnightly monitoring (see Table 3.4). The fortnightly sampling, however, did not capture many high-water events. In contrast, all mean values for TON from the storm events were less than the 2020/2021 mean. There appears to be a dilution of TON during the course of the storm response for the Newtown Mount Kennedy site (Figure 3.10a), but, more generally, and based on a comparison of TON from 2021/2022 with the storm event values, there appears to be some dilution of TON in the storm response (albeit variable) in all three catchments. The events with the highest recorded rainfall for each site, i.e. 45.1 mm recorded at the Newtown Mount Kennedy forest laboratory rain gauge (close to the Newtown Mount Kennedy and Newcastle sites – GPS 53.09778, –6.11361; event 2) on 19 October 2022 and 52.1 mm recorded at the Roundwood Filter Beds (close to the Vartry site – GPS 53.048, –6.189; event 1) on the same date, had some of the highest mean values for TP, SRP and TAN. The data in Table 3.4, again, indicate the presence of significant rain-driven nutrient inputs at all sites. For some events, the highest concentrations of TP and SRP occurred on the rising limb of the hydrograph (Figure 3.10a and b and Figure 3.11a and b), suggesting rapidly mobilised sources of all nutrients. This indicates proximal source(s), which could be within-channel mobilisation of retained nutrients and/or diffuse sources close to the river margins. However, for other

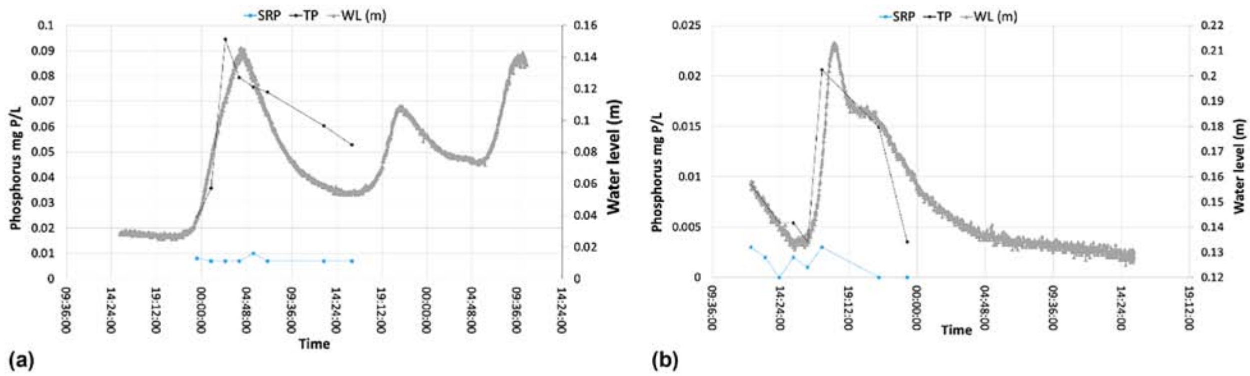


Figure 3.10. Event 4 water level and TP and SRP dynamics at the (a) Newtown Mount Kennedy and (b) Vartry sites. Peak concentrations for TP and SRP occurred on the rising limb of both hydrographs, i.e. at 02:16 hours at Newtown Mount Kennedy and 16:48 hours at Vartry.

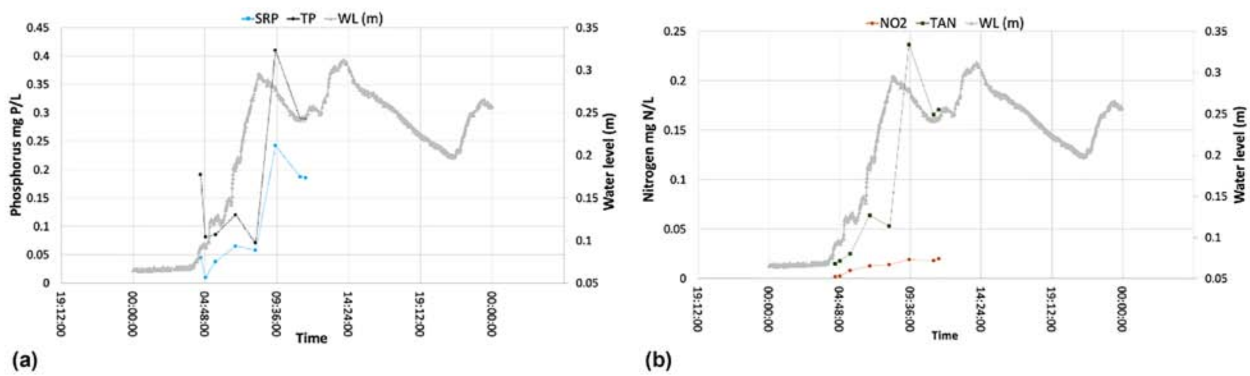


Figure 3.11. Event 2 water level and (a) TP and SRP dynamics; and (b) NO_2 and TAN dynamics at the Newcastle stream. Peak concentrations for TP, SRP and TAN occurred on the falling limb just after the peak on the hydrograph, i.e. before 08:28 hours.

events, the peak concentrations occurred on the falling limb (Figure 3.11a and b), suggesting that the major nutrient sources are distal from the monitoring site and take time to be transported. This pattern, however, differed across sites and events, with peaks in nutrient concentration being observed at different points on the hydrograph during separate events at each site. This suggests that nutrient behaviour at the three sites is governed by a complex dynamic response to rainfall, with data suggesting that other factors, in addition to contemporaneous rainfall, play a role.

3.7 Sources of Nutrient Impairment and Interventions Needed

From the results of the levels of impact analysis, described in section 3.2, each site was assigned a numerical score depending on the severity of each type of exceedance, and these were summed for each site to give an overall score for the site, related to overall nutrient impairment. These were mapped as cumulative levels of impact for non-compliant sites (the full method is described in Hogan *et al*., 2023). There were eight “blackspot” sites with combined level of impact scores between 15 and 18; 15 level 2 sites (with combined level of impact scores between 13 and 15); 44 level 3 sites (with combined level of impact scores between 12 and 13); 33 level 4 sites (with combined level of impact scores between 10 and 12); and 14 level 5 sites (with combined level of impact scores between 8 and 10). The SSNet types IHPo, IPPo, LPPo, LPW and SMW all had “blackspot” sites. The higher scores (or “blackspot” sites) suggest locations where mitigation might be prioritised.

Many of the sites of higher impact scores are associated with the IHPo and LPPo SSNet types, which are quite dispersed and fragmented, rather than being concentrated in any single geographical area. Table 3.5 summarises the results from the level of impact assessment, identifies potential sources of nutrient inputs and suggests mitigation strategies that may be beneficial to improve the nutrient conditions. Most sites were characterised by multiple and different nutrient inputs. Based on analyses conducted by Hogan *et al*. (2023) that (i) evaluated seasonal nutrient trends over time and proximity to wastewater treatment plants of these systems, and (ii) examined temporal correlation analyses, investigating the relationship between nutrient concentrations and a range of hydrometeorological variables, specifically flow, water level and rainfall conditions, these sources were then recognised as being either very likely, such as those sites downstream from urban wastewater treatment outfalls, or likely. The likely sources were then defined as coincident, i.e. similar types of nutrient input sources are suggested by both the seasonal patterns observed from time series graphs and the temporal rainfall and/or flow correlation data, or mixed, i.e. different types of nutrient input sources are suggested by the seasonal patterns and the shorter term temporal correlations with flow and precipitation.

Table 3.5. Results of the level of impact assessment: possible sources of nutrients, the analyses that highlight the potential sources and potential mitigation measures

Priority	No. of sites	Possible sources of nutrients		Rationale for nutrient sources	Potential mitigation measures
1	8	Very likely	UWWT ($n=7$)	Proximity to UWWT outfalls –ve temporal correlations	Ensure compliance with discharge licence and capacity
		Likely	Coincident diffuse ($n=1$)	+ve temporal correlations	Farmyard and DWW checks/buffer zones/fertiliser management
2	20	Very likely	UWWT ($n=9$)	Proximity to UWWT outfalls –ve temporal analysis	Ensure compliance with discharge licence and capacity
		Likely	Coincident point ($n=3$) or diffuse ($n=3$)	Either –ve or +ve temporal correlations Similar seasonal trends	Farmyard and DWW checks/buffer zones/fertiliser management
			Mixed ($n=5$)	Both +ve and –ve temporal correlations Mixed seasonal trends	
3	25	Very likely	UWWT ($n=10$)	Proximity to UWWT outfalls	Ensure compliance with discharge licence and capacity
		Likely	Coincident point ($n=2$) or diffuse ($n=2$)	Either –ve or +ve temporal correlations Similar seasonal trends	Farmyard and DWW checks/buffer zones/fertiliser management
			Mixed ($n=11$)	Both +ve and –ve temporal correlations Mixed seasonal trends	
4	45	Very likely	UWWT ($n=9$)	Proximity to UWWT outfalls	Ensure compliance with discharge licence and capacity
		Likely	Coincident point ($n=4$) or diffuse ($n=6$)	Either –ve or +ve temporal correlations Similar seasonal trends	Farmyard and DWW checks/buffer zones/fertiliser management
			Mixed ($n=26$)	Both +ve and –ve temporal correlations Mixed seasonal trends	
5	16	Very likely	UWWT ($n=1$)	Proximity to UWWT outfalls	Ensure compliance with discharge licence and capacity
		Likely	Coincident point ($n=1$) or diffuse ($n=3$)	Either –ve or +ve temporal correlations Similar seasonal trends	Farmyard and DWW checks/buffer zones/fertiliser management
			Mixed ($n=11$)	Both +ve and –ve temporal correlations Mixed seasonal trends	

Priority sites are listed in order of those most heavily impacted. Potential mitigation measures are not intended to be comprehensive.

–ve, negative; +ve, positive; DWW, domestic wastewater; UWWT, urban wastewater treatment.

Table first published in *Hydrobiologia*, 850, 3293–3311, 2023, by Springer Nature.

4 Hydromorphology

4.1 Introduction

The sedimentary structure, morphology and dynamics (hydromorphology) of stream and river reaches create a range of habitats for aquatic organisms at various stages in their life cycle. The interaction between hydromorphology and biology has the capacity to structure both aquatic communities and their functioning (Beisel *et al*., 1998, 2000; Wolter *et al*., 2016). To date, these interactions are poorly understood. The overall aim of the research was to devise a top-down hierarchical approach for characterising the hydrogeomorphology of Irish headwater streams. The hypothesis was that the very local site-scale (*c*.100-m channel length) physical properties of small streams reflect the physical character of the extended reaches (*c*.1 to 5-km channel length) in which they are located, which, in turn, reflect the broad physical characteristics of their headwater catchments. Four specific research objectives were addressed. The first three focused on delivering information and analysis at three nested spatial scales: catchment, reach and site. The fourth was to establish the degree to which the outcomes of the analyses at the site scale could be linked to those at the reach and catchment scales, to provide a hierarchical hydrogeomorphological characterisation of Irish headwater streams.

4.2 Approach

Minimally impacted headwater stream reaches were selected, as outlined in Chapter 2 and described in more detail in Cox *et al*. (2022). For this research element it was essential that the sites had, as far as possible, near-natural hydromorphology in terms of their morphology and the physical processes controlling the morphology. Due to drainage impacts in many lowland areas, the final list of 42 stream reaches (two sites per reach = 84 sites) were located in upland areas on igneous and non-calcareous sedimentary geology, but should be representative of Ireland's headwater streams within these constraints.

Using a top-down approach, data on catchment, reach- and site-scale stream physical properties were collected for 42 Irish headwater streams. Summary catchment properties (rock type, topography, soil type, hydrology) were extracted from secondary sources. Reach-scale physical controls on stream hydrogeomorphology (planform, gradient, degree of confinement, bed material) were also assembled, mainly from secondary sources. Reach- and site-scale data were collected using a pre-existing survey method (Modular River Physical Survey (MoRPh5); Gurnell *et al*., 2019, 2020). Two MoRPh5 surveys were undertaken on each stream, where each MoRPh5 survey covered five 10-m contiguous channel lengths. These two surveys provided physical habitat information for two adjacent 50-m-long study sites, allowing us to investigate (i) the influence of local bed material variability on the reach-scale classification of river type and (ii) the degree to which the physical habitat assemblage at each site was sensitive to reach- or site-scale controlling factors. The surveys included the abundance of channel surface bed material patches (bedrock, boulder, cobble, gravel-pebble, sand, silt, clay, organic material, peat). They also recorded abundance of physical bed features (exposed bedrock, vegetated and unvegetated rocks (boulders), vegetated and unvegetated mid-channel bars, islands, cascades) and a count of other physical features (pools, riffles, steps, waterfalls), and bank face/channel margin features (unvegetated and vegetated sidebars, berms, benches, stable and eroding cliffs, marginal backwaters and bank toe deposits). The aim of the analysis was to assess whether the abundance and mosaic of physical habitats appeared to change according to the hydrogeomorphological river type across the 84 study sites. A second detailed analysis within two headwater catchments (Vartry and Ballinagee – 42 physical survey sites across three tributaries in each catchment) investigated the length of reach within which a single river type might persist, and validated the associations established between river type and physical habitat assemblage that were found for steep headwater streams during the national-scale analysis.

4.2.1 River type classification

We developed a hydrogeomorphological river type classification using information from extended reaches of stream (typically > 1 km in length) enclosing the paired sites where MoRPh5 field surveys were undertaken. Eight indicators were used to assign reaches to a river type (Gurnell *et al*., 2020): A1 – braiding index, A2 – sinuosity, A3 – anabranching index, A4 – valley confinement, A5 – valley gradient, A6 – bedrock reaches, A7 – coarsest bed material type, and A8 – average bed material size. Details of how each indicator was estimated are given in Cox *et al*. (2022). The values estimated for the eight indicators were used to refine the river type classification of Gurnell *et al*. (2020) for application to Irish headwater streams.

4.3 Key Results

Six river types for steeper streams were defined using the calibre of their bed material: A (A6 = bedrock rivers), B (A7 = bedrock, A8 = boulder), C (A7 = bedrock, A8 = cobble), D (A7 = boulder, A8 = cobble), E (A7 = bedrock, A8 = gravel-pebble) and F (A7 = boulder, A8 = gravel-pebble). These new river types for steeper streams were applicable to 66 of the 84 sites (A – 13 sites, B – 2 sites, C – 15 sites, D – 18 sites, E – 7 sites, F – 11 sites), giving sufficient replication to support statistical analysis for all but type B. The remaining 18 sites with finer bed material were assigned to three classes (H – 13 sites, J – 4 sites, M – 1 site), in accordance with the original classification (Gurnell *et al*., 2020). The six types were integrated into the previous classification proposed

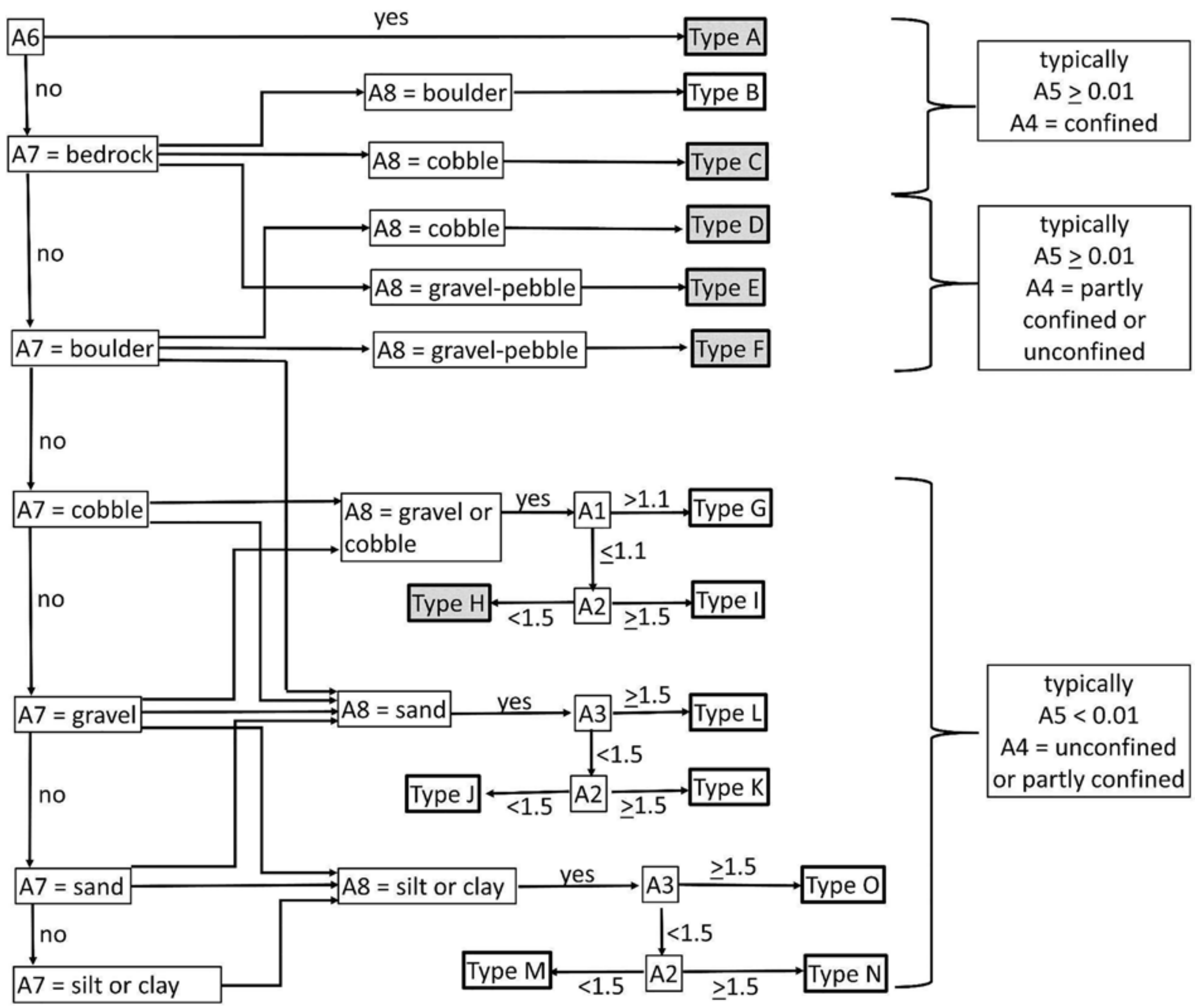


Figure 4.1. Flow diagram used to assign study sites to river types based on the values of indicators A1 to A8. River types in bold boxes are those identified across the study headwater streams.

by Gurnell *et al*. (2020); the revised flow diagram underlying the classification of hydrogeomorphological types is shown in Figure 4.1.

The calculated indicators (A1 to A8) revealed that all 42 stream reaches were single thread (the braiding index (A1) and the anabranching index (A3) were both 1 in all cases), and only one reach had a sinuosity index (A2) that marginally exceeded 1.5. However, valley confinement (A4), valley gradient (A5) and the three bed material indicators all varied widely.

Notable changes in the relative abundance of bedrock to sand and organic material were detected across the stream beds of sites from river types A to M. (Figure 4.2). Certain bed features and bank face/margin features also show notable changes as the river type changes (Figures 4.3 and 4.4). The relative abundance of exposed bedrock, vegetated rocks and cascades (Figure 4.3a,c,g) gradually decreases, to be replaced by unvegetated rocks and steps (Figure 4.3b,h) and then by riffles (Figure 4.3i), across the river types A to M. Among the bank face/margin features recorded (Figure 4.4), the abundance of marginal backwaters (Figure 4.4j) gradually declines as unvegetated and vegetated sidebars (Figure 4.4a,b,c), berms and benches (Figure 4.4d,e,f) increase in abundance, and, in turn, the abundance of eroding cliffs progressively increases (Figure 4.4h) from river types A to M.

The hydrogeomorphological river type classification and the suite of physical features or habitats associated with each type were tested in two headwater catchments. Continuous field surveys were conducted along six first-order tributaries, ranging from 1300 to 1900 m in length, three in each catchment, and were combined with secondary data so that river types could be assigned to all 93 100-m lengths of stream that were investigated. Adjacent 100-m reaches were found to be either the same river type or an adjacent river type in every case, with small differences between adjacent reaches most probably reflecting slight changes in the local stream gradient or confinement that could alter local stream power (for the same discharge) within a river reach. The analysis supported the use of 1-km stream lengths in such steep headwaters as a reliable basis for determining "homogeneous" river types. Furthermore, detailed MoRPh5 surveys were available for 42 of these 100-m lengths of stream, allowing associations between river type and their physical habitat assemblages to be investigated. The 42 comprised 10 type A, 10 type C,

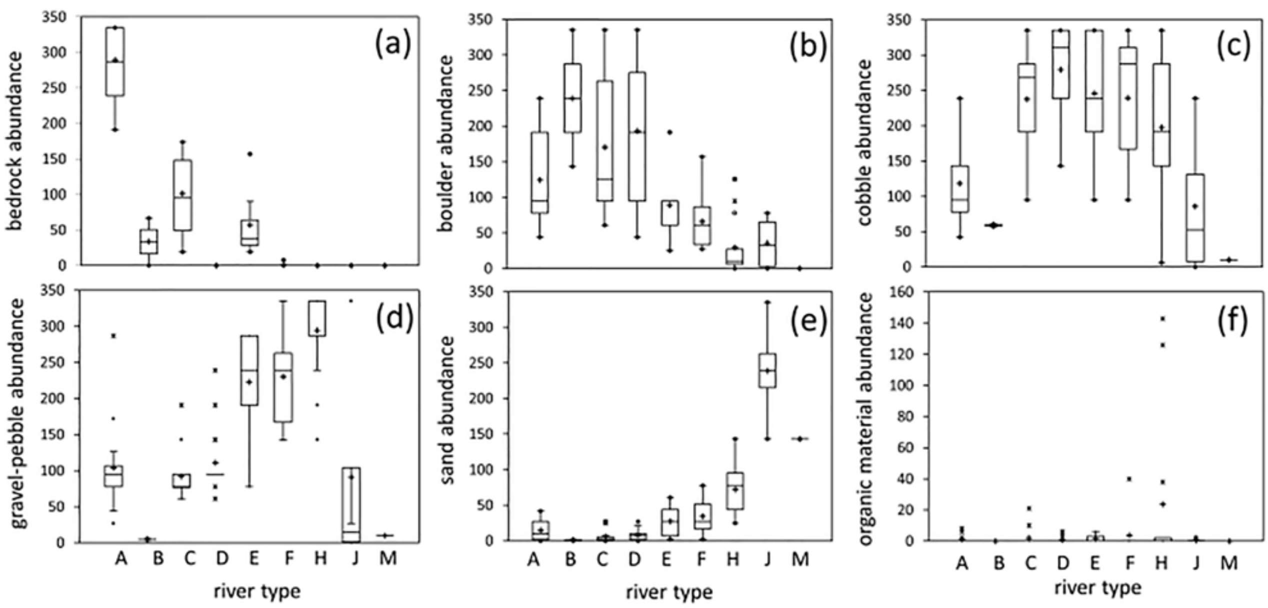


Figure 4.2. Box and whisker plots of the abundance of different bed material patches observed in association with sites assigned to different river types (note that different scales are used on the vertical axes to highlight contrasts between river type classes according to each of the characteristics that are being plotted). Figure first published in *Hydrobiologia*, 850, 3391–3418, 2022, by Springer Nature.

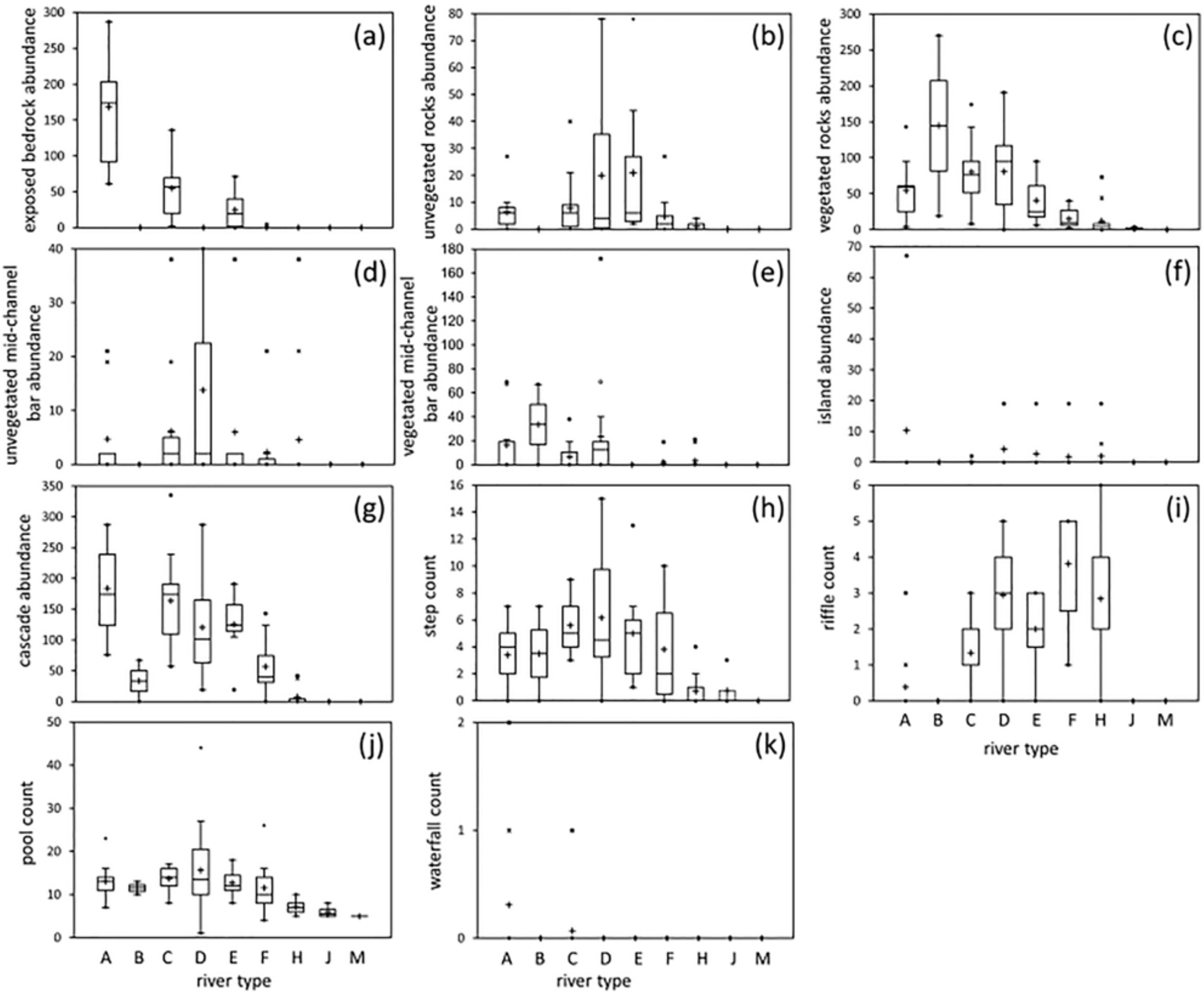


Figure 4.3. Box and whisker plots of the abundance of different bed physical habitats observed in association with sites assigned to different river types (note that different scales are used on the vertical axes to highlight contrasts between river type classes according to each of the characteristics that are being plotted). Figure first published in *Hydrobiologia*, 850, 3391–3418, 2022, by Springer Nature.

6 type D, 4 type E, 10 type F and 2 type H streams. Analysis once again identified clear gradients in the relative abundances of different bed material patches, channel bed and bank face/margin physical features as the river type changed, validating, at this local scale, the river types and their associated distinct physical habitat assemblages as established for headwater streams across Ireland.

In Table 4.1, the characteristics of the different geomorphological river types that were represented by at least four study sites are summarised. Table 4.1 shows the reach-scale characteristics, key indicator habitats and habitat assemblages typically associated with each river type.

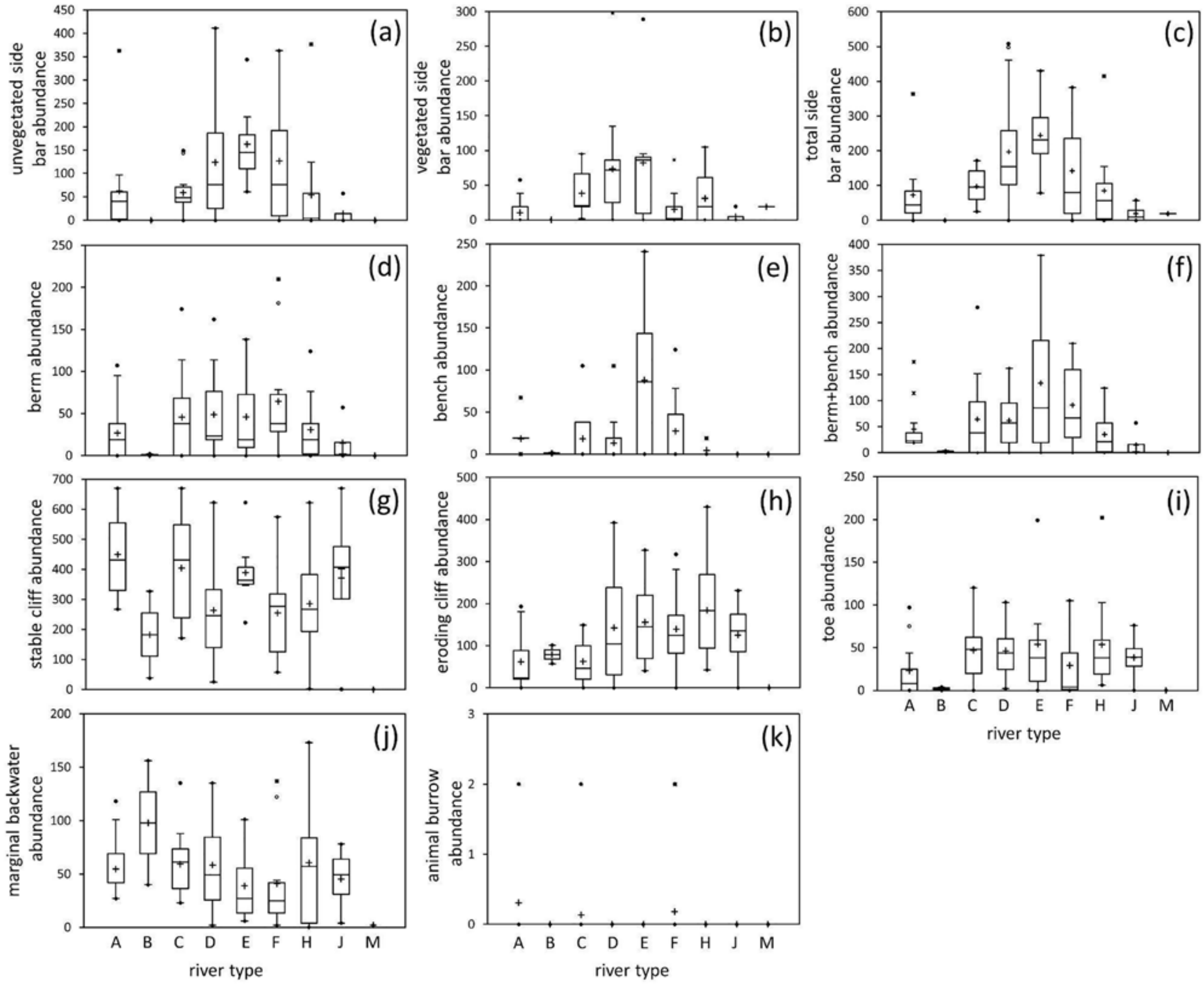


Figure 4.4. Box and whisker plots of the abundance of different bank face/margin physical habitats observed in association with sites assigned to different river types (note that different scales are used on the vertical axes to highlight contrasts between river type classes according to each of the characteristics that are being plotted). Figure first published in *Hydrobiologia*, 850, 3391–3418, 2022, by Springer Nature.

Table 4.1. Reach-scale characteristics, key indicator habitats and characteristic habitat assemblages of river types A, C, D, E, F, H and J

River type, reach-scale characteristics	Key indicator habitats	Characteristic habitat assemblage	Image
A: Steep bedrock rivers, typically confined	Extensive exposed bedrock and boulders forming steep cascades. Steep to vertical bedrock banks	Bed material is typically bedrock and boulder with some cobble and little finer material. Large areas of cascade typically often interspersed with step-pool sequences. Few vegetated or unvegetated rocks and mid-channel bars. Marginal habitats are simple with some backwaters but few sidebars, berms or benches	
C: Steep bedrock-boulder rivers, typically confined	Frequent sections of exposed bedrock. Widespread boulder and bedrock cascade sections, and stable bedrock and boulder cliffs	Bed material dominated by boulder and cobble with some gravel-pebble and little fine material. Cascades and steps separated by pools are typical. Large stable boulders common. Occasional boulder and cobble mid-channel bars	
D: Steep boulder-cobble rivers, typically confined or partly confined	Predominantly boulder and cobble bed with occasional bedrock exposure. Frequent steps and unvegetated mid-channel bars	Bed material typically cobble and boulders with occasional finer material. Bedrock is rare. Cascades and steps common but less frequent than for type C. Riffles, small pools, stable vegetated boulders and vegetated bars comprising boulders and large cobbles are frequent	
E: Moderately steep gravel-pebble rivers, typically partly confined	Predominantly gravel-pebble bed with some boulders and occasional bedrock exposure. Frequent sidebars, berms and benches	Bed material typically gravel-pebble interspersed with cobbles and also small amounts of bedrock, boulder and sand. Step-pools are common. Cobble-boulder cascades, riffles, sidebars, berms, benches and both stable and eroding cliffs are frequent	
F: Moderately steep gravel-pebble rivers, typically partly confined or unconfined	Predominantly gravel-pebble bed with some cobbles. Frequent riffles, sidebars, berms and benches	Bed material typically gravel-pebble interspersed with cobbles. Occasional boulders and sand in low-velocity areas. Pools, riffles, steps, sidebars, berms and benches are common, as are vertical sections of stable and eroding cliff	
H: Moderate gradient straight-sinuous gravel-pebble rivers, typically partly confined or unconfined	Predominantly gravel-pebble bed with some cobbles. Frequent riffles and eroding cliffs	Bed material dominated by gravel-pebble and interspersed with cobbles and sand. Riffles, pools and sections of eroding cliff are frequent. Occasional sidebars, berms and benches	
J: Moderate gradient straight-sinuous sand-bed rivers, typically partly confined or unconfined	Predominantly sand bed with few indicator physical habitats	Bed material primarily sand with some cobbles and gravel-pebble. Long sections of pool or slow-flowing water (glide)	

5 Aquatic Biota

5.1 Introduction

This chapter presents the results of investigations on the macroinvertebrates, phytobenthos, macrophytes and fish of small streams. The objectives in relation to the macroinvertebrates, phytobenthos and macrophytes were two-fold: first, to establish a baseline reference database on the biodiversity of the small stream network based on the sites studied and supplemented with available data (e.g. Callanan *et al*., 2014; Weekes *et al*., 2014); and, second, to explore site types based on three biological elements and determine the physical or chemical factors driving this community structure. Biological sampling took place within the 100-m reaches identified for hydromorphological surveying during site selection (see Chapter 3). The work on macroinvertebrates, phytobenthos and macrophytes involved sampling sites across the 13 selected SSNet types (described in section 4.1), hereafter referred to as the "extensive sites". A second element was based on sampling multiple sites on three branches of the small stream network in three catchments (section 4.2), referred to as "grouped sites". The final element investigated fish in a number of small coastal streams.

5.2 Macroinvertebrates: Extensive Sites

5.2.1 Approach

Multi-habitat kick samples (20 seconds) were collected in spring 2021 at 42 sites using a standard Freshwater Biological Association kick net (250-mm wide, 1-mm mesh), with the time spent in the mesohabitat proportional to their representation in the stream site, according to Feeley *et al*. (2012a). This included checking cobbles for attached macroinvertebrates. One kick sample was taken in each of the three central 10-m modules of a 100-m reach, which had been previously surveyed using the MoRPh5 approach (as described in Chapter 4). The samples were fully sorted, and the macroinvertebrates identified to species level where possible, with the exception of the Chironomidae, Oligochaeta, Simuliidae and some dipteran larvae, which were not identified further.

5.2.2 Key results

Across the 42 sites, a total of 144 taxa were recorded. As expected, this figure was dominated by insect larvae, with Trichoptera the most species-rich group, followed by Coleoptera and Diptera (Figure 5.1). This is very much a minimum estimate, as *Ecdyonurus* was left at genus level (three species were distinguished but not at all sites) and Chironomidae and many of the other Diptera were not identified further. The seven taxa listed as others are Turbellaria, *Sialis*, *Velia*, Collembola, Acaria, Nematomorpha and two families of the Odonata.

The taxa had a patchy distribution, with only Chironomidae common to all sites (Figure 5.2). Twenty-three (16%) were found at more than 50% of sites and 80 taxa were recoded at fewer than 10 sites, of which 38 (26%) occurred at a single site. Taxa that occurred at more than 70% of sites included Simuliidae, *Siphlonoperla torrentium*, *Leuctra inermis*, *Isoperla grammatica*, *Baetis rhodani/atlanticus*, *Rhyacophila dorsalis*, *Dicranota*, *Plectrocnemia conspersa*, *Gammarus duebeni* and *Ecdyonurus* spp.

Taxon richness was variable across the SSNet types, ranging from an average of 25 in the igneous/metamorphic mountain sites on peat (IHPe) to 47 at the single site in the limestone plains (LPPe) (Figure 5.3).

The non-metric multi-dimensional scaling (nMDS) plot in Figure 5.4 visualises dissimilarity in community structure between the 42 extensive sites. Sites within the same SSNet type (primarily in terms of

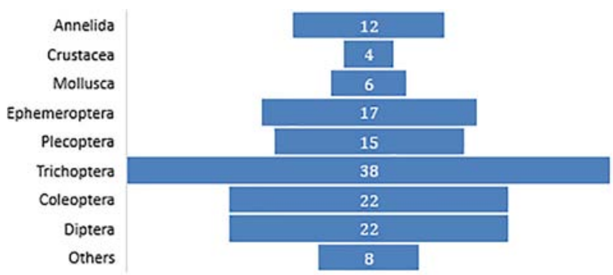


Figure 5.1. Number of taxa recorded in broad taxonomic groups across all 42 sites.

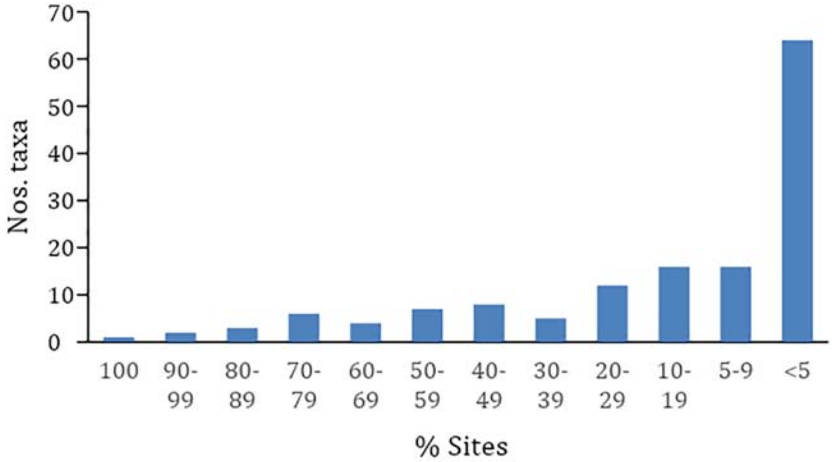


Figure 5.2. Percentage of site occurrences of the 144 macroinvertebrate taxa.

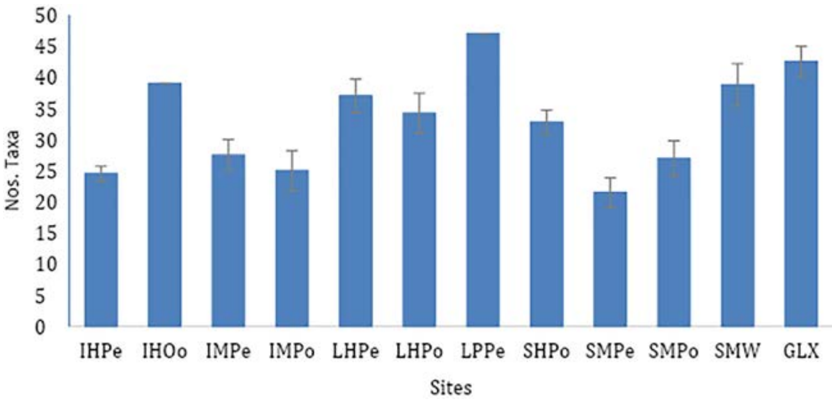


Figure 5.3. Mean (±standard error) macroinvertebrate taxon richness across the SSNet types.

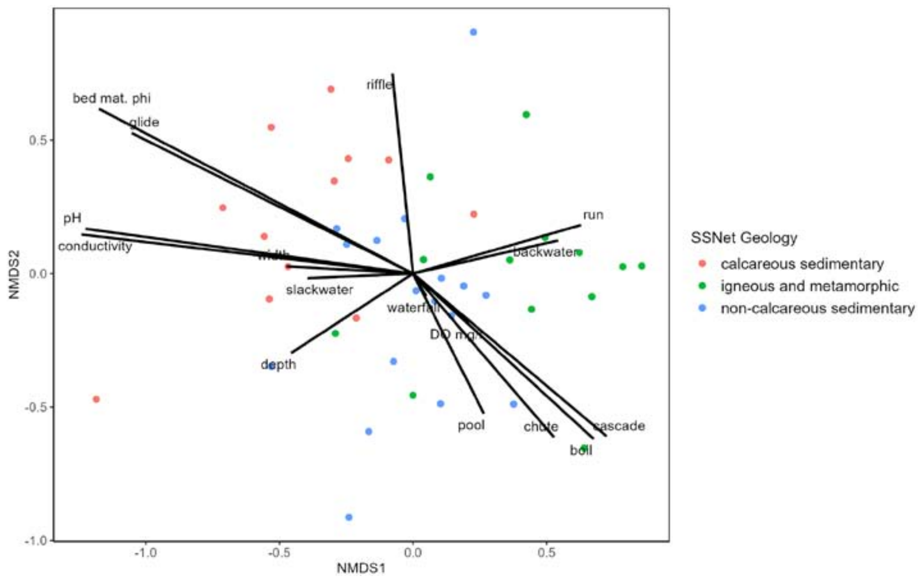


Figure 5.4. nMDS plot of macroinvertebrate community dissimilarity across the extensive sites, with a subset of the environmental variables represented as vectors. The lines are gradients representing the direction of that variable's steepest increase with respect to the ordination configuration. Their length represents the degree to which each dependent variable is explained by the ordination scores.

geology) tended, as expected, to have more similar macroinvertebrate communities. Similarly, sites from adjacent SSNet types (primarily those within the same or a similar geological class) tended to be located closer together on the ordination plot. For example, sites with calcareous sedimentary geology (the reddish sites) are largely clustered together to the left of the centre of the plot, while igneous/metamorphic sites (green) are to the right, with the non-calcareous sites between these two extremes. This is supported by the calculations of the "envfit" function from the "vegan" R package showing the correlation between the environmental variables and the ordination plot points. A clear gradient of increasing pH and conductivity is seen as we move from right to left of the plot (Figure 5.4). This is mirrored by a similarly significant gradient in decreasing particle size (phi units increase from right to left).

Sites with similar geology and physiography tend to have more similar macroinvertebrate communities. The pattern is less clear for soil drainage type. These observations are supported by the permutational multivariate analysis of variance (PERMANOVA) (Anderson *et al.*, 2008), which showed community structure to be significantly different within all the physical descriptor classes, with the exception of soil drainage type (Table 5.1). However, the subsequent pairwise tests detected significant differences in only community structure between the three geology classes.

Cluster analysis of the community structure data was performed using a modified version of the divisive hierarchical clustering approach known as two-way indicator species analysis (TWINSPAN) (Roleček *et al.*, 2009) to preferentially divide clusters with high compositional heterogeneity (see Figure 5.5). TWINSPAN group 1 has a similar average pH to the sites in group 3, but a higher average conductivity and a substantially lower average valley gradient (Table 5.2). Unsurprisingly, given the relationship between gradient and stream power, this lower valley gradient was also accompanied by a lower average particle size. In comparison with group 1, groups 2 and 3 tended to be steeper with a larger average bed material size. They can also be differentiated by their average river width and pH, as well as a modest difference in average conductivity. TWINSPAN groups 5 and 6 have notably lower pH than groups 1 to 4, although conductivity is comparable to groups 2 and 3. On average, sites in group 5 tend to be wider and steeper than those in group 6, with a greater average alluvial particle size. Group 4 consists of only one site, a low-gradient, groundwater-fed site, fundamentally distinct in character from the other extensive sites. Site GLX_kil, on the extreme left of Figure 5.5, was physically distinct from the others, in that, although it was technically a first-order stream, it had a much higher volume of flow than the other sites, with a deep trapezoidal "canal-like" channel, and was almost entirely fed by groundwater. There was a significant difference ($F=4.15$, degrees of freedom (df) 2, $p=0.05$) between the six groups. Pairwise comparisons highlighted a significant difference between group 1 and groups 2, 3, 5 and 6. Group 6 was also significantly different from groups 2 and 3, and group 5 differed significantly from group 3. Overall, this gives three main types, as illustrated in Figure 5.5. However, group 4 (high conductivity, limestone) is represented by a single site and group 6 has only four sites.

Table 5.1. Results of PERMANOVA (based on a Bray–Curtis matrix), assessing whether community structure varies significantly within each of the physical descriptor classes, and whether the degree of dispersion (variance) varies significantly within each physical descriptor class

	PERMANOVA of community structure				ANOVA of dispersion	
Physical descriptor class	df	Sum of squares	Pseudo-*F*	*p*-Value	*F*-value	*p*-Value
Geology	2	1.779	3.614	0.0001	0.443	0.645
Physiography	3	1.154	1.429	0.044	11.393	0.00002
Soil	3	1.106	1.364	0.078	5.439	0.003
SSNet type	11	4.571	1.831	0.0001	1.580	0.156
Morphological river type	7	2.576636	1.422	0.016	1.505	0.199

$p<0.05$ significant.

ANOVA, analysis of variance; df, degrees of freedom.

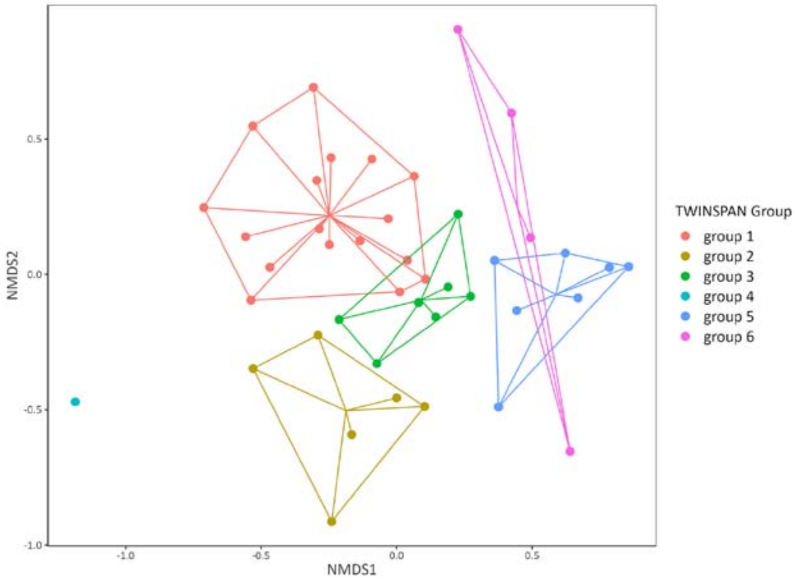


Figure 5.5. Results of TWINSPAN superimposed on nMDS plots visualising the dissimilarity of the different macroinvertebrate communities, where each point represents a site. The sites are colour-coded according to TWINSPAN group, and each group is bounded by a hull. Lines radiate from the centroid of each group to their respective sites.

Table 5.2. Physical and chemical properties of the sites within the TWINSPAN groups

TWINSPAN group	Average pH	Average conductivity (µS/cm)	Average water width (m)	Average bed material particle size (phi)	Average bed material particle size (class)	Average valley gradient	No. of sites in TWINSPAN group
1	7.24	152.08	2.73	–4.68	Gravel-pebble	0.04	17
2	7.10	62.60	1.31	–6.23	Cobble	0.11	6
3	7.22	81.33	2.37	–6.47	Cobble	0.13	7
4	7.52	705.00	4.76	4.09	Sand	0.00	1
5	6.33	56.47	2.19	–6.64	Cobble	0.12	7
6	6.26	59.65	1.55	–5.43	Gravel-pebble	0.03	4

The indicator analysis highlighted that, although there were some significant associations of particular species with the TWINSPAN groups, community structure rather than the abundance of a few indicator species is more likely to distinguish the groups.

5.3 Macroinvertebrates: Grouped Sites

5.3.1 Approach

The six headwater streams selected in the Dargle and Ballinagee catchments for the hydromorphological study in Chapter 4 were also sampled for macroinvertebrates in spring 2021. Seven sites on three tributaries in each catchment were sampled at seven points along the length of each stream. The aim was to investigate whether there were substantial differences in the macroinvertebrate communities of different branches of the small stream network within catchments. This was prompted by literature that highlights heterogeneity in the physical habitat of small streams that would potentially lead to heterogeneity in species occurrences and community structure within and between branches. The results will feed into recommendations for biodiversity management in small stream networks.

5.3.2 Key results

The communities in the six headwater streams were again dominated by insects, in particular Plecoptera,

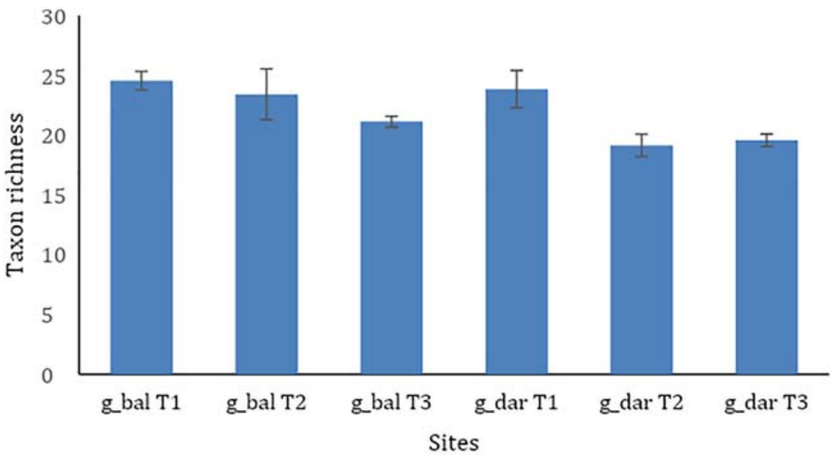


Figure 5.6. Mean (±standard error) taxon richness in the six headwater streams. g_bal, Ballianagee sites; g_dar, Dargle sites; T, tributary.

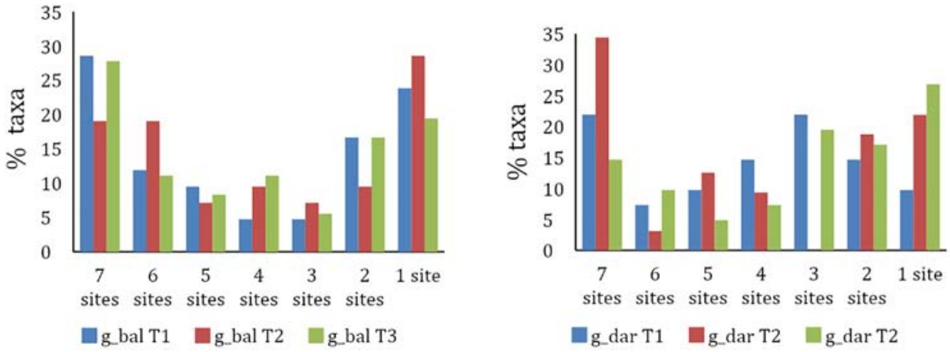


Figure 5.7. Percentage of macroinvertebrate taxa occurring at sites 1 to 7 in the Ballinagee (left) and Dargle (right) stream branches.

which constituted over 40% of the abundance values at all but 13 of the 42 sampling points. Across the six tributaries, a total of 67 taxa were recorded, but each tributary had, on average, only between 20 and 25 taxa present (Figure 5.6). Furthermore, within most tributaries, *c.*20–29% of the taxa occurred at a single site only (Figure 5.7).

The two catchments were distinct in terms of community structure, and one tributary in each catchment differed from the other two, as illustrated in Figure 5.8, with sites within tributaries most similar to each other. The PERMANOVA indicated significant differences between catchments ($F = 3.46$, df 1, $p < 0.05$) and tributaries ($F = 7.19$, df 1, $p < 0.05$). The macroinvertebrate communities of the study sites appear to be structured by different environmental factors depending on the spatial scale. At the tributary scale, there is a strong water chemistry gradient – with differences in pH and conductivity strongly correlated with differences in tributary macroinvertebrate

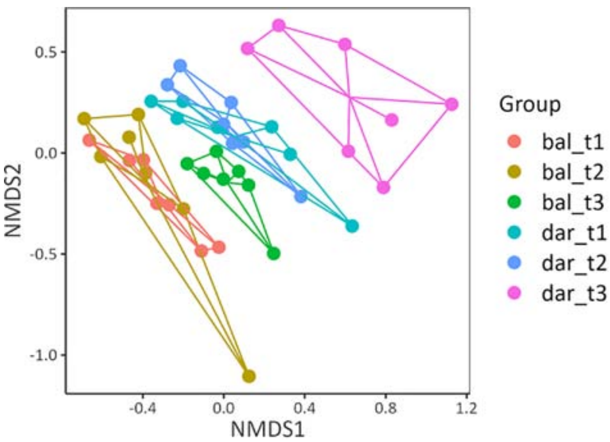


Figure 5.8. nMDS spider–hull plots visualising the dissimilarity of the different macroinvertebrate communities, where each point represents a site and the points are colour-coded by tributary. The points from each individual tributary are bound by a hull, and the lines radiating from a single point show their location with respect to the centroid of the group.

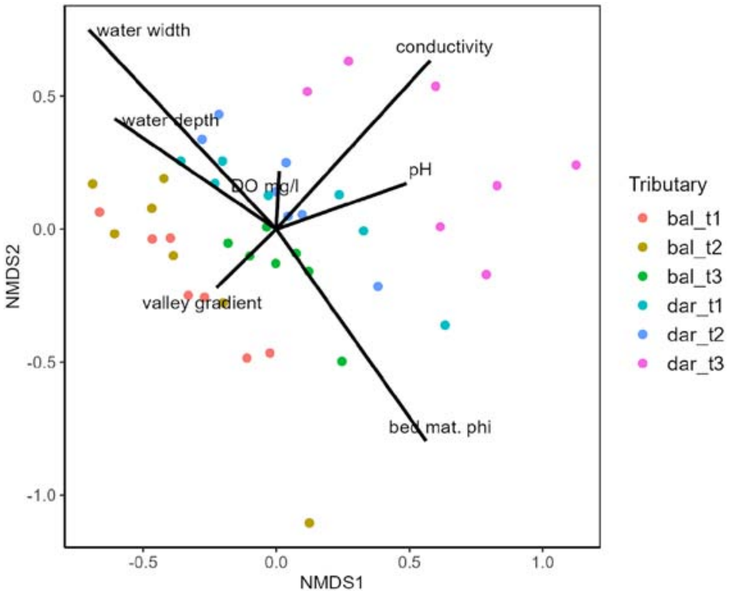


Figure 5.9. nMDS plot of macroinvertebrate community dissimilarity across the grouped sites, with a subset of the environmental variables represented as vectors. The lines are gradients representing the direction of that variable's steepest increase with respect to the ordination configuration. Their length represents the degree to which each dependent variable is explained by the ordination scores.

community. At the site scale, there is a strong hydromorphological gradient – with differences in stream width and average bed material particle size strongly correlated with the ordination configuration of the sites within the individual tributaries (Figure 5.9).

5.4 Macroinvertebrates: Implications of Seasonal Variation for Biomonitoring Small Streams

5.4.1 Introduction and approach

The macroinvertebrate community composition of small streams can change substantially over the summer months due, in part, to the emergence of adults and appearance of other taxa such as Coleoptera. Callanan *et al*. (2008b) highlighted that too few taxa may be present in the summer in some small streams to reliably determine the ecological status of the stream using the available indices. A similar but smaller scale exercise was conducted in the headwaters of the Dargle catchment, where the influence of season on metric calculations was investigated for high-gradient upland sites. The additional sampling in summer (August 2021) also provided insights to inform future monitoring of small stream biodiversity. Sampling took place at sites S1, S3, S5 and S7 in each of the three tributaries (T1, T2 and T3) studied in section 5.2, and using the same method.

5.4.2 Key results

As expected, taxon richness in all three tributaries was higher in spring (Figure 5.10). Mean values were statistically different for all tributaries (Kruskal–Wallis: T1: $H(2) = 5.33$, $p = 0.021$; T2: $H(2) = 4.80$, $p = 0.028$; T3: $H(2) = 5.74$, $p = 0.017$). This is supported by the results

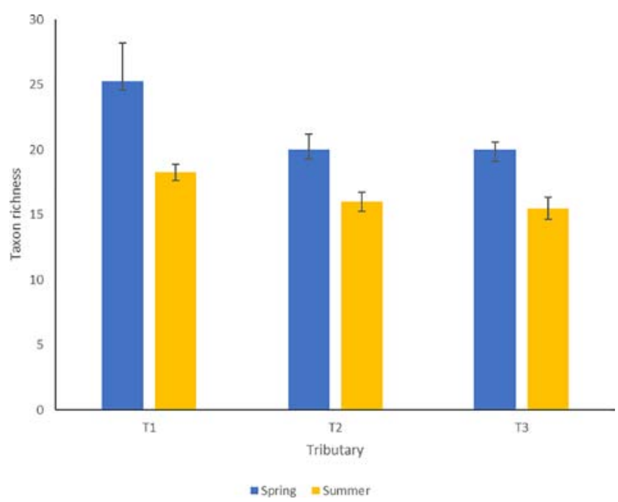


Figure 5.10. Mean (±standard error) taxon richness in the Dargle tributaries in spring and summer.

of a PERMANOVA and is illustrated in the nMDS plot in Figure 5.11.

Between 33% and 50% of the taxa were common to both seasons, while up to 44% of taxa were unique to spring (Table 5.3). The unique species were mainly Plecoptera, most of which would have emerged before the summer. Fewer species were unique to summer (13–24%); these included beetle species that occurred at some but not all sites (e.g. *Hydroporus*, *Oreodytes*, *Elmis, Limnius, Oulimnius, Heloporus, Acanaena, Laccobius, Odeles*), as well as *Baetis vernus* and two trichopteran species (*Polycentropus kingi* and *Rhyacophila munda*).

In terms of implications for water quality metrics, the Q-value scores were reduced from Q5 to Q4, or to as low as Q3–4, in summer due to the emergence of most Group A (pollution-sensitive) taxa. The Biological Monitoring Working Party (BMWP) and Average Score Per Taxon (ASPT) scores were also impacted. The only metric that indicated unimpacted water quality in summer was the small stream impact score, as all values were >7.25, indicating no pollution risk.

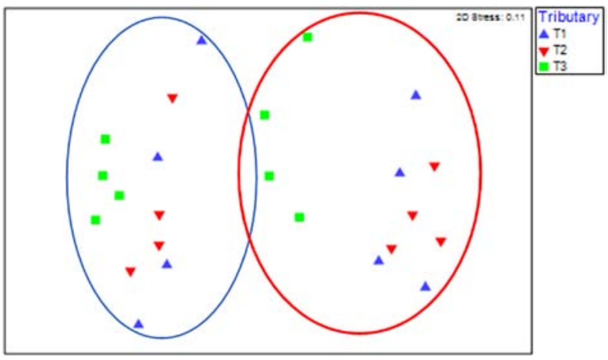


Figure 5.11. nMDS plot of the community structure in the Dargle catchment during the spring (red circle) and summer (blue circle) seasons.

5.5 Phytobenthos

5.5.1 Approach

Benthic diatoms were surveyed according to European Standard 15708:2009 (CEN, 2009) in September 2021. Diatoms were scrubbed from approximately five cobbles (more if the sample was too dilute) within a central 20-m stretch of the 100-m MultiMoRPh reach. The bulk sample from each site was retained in a plastic tube and preserved in ethanol. Pre-treatment, identification and enumeration of the benthic diatoms was performed according to European Standard 14407:2014 (CEN, 2014a) by Lydia King.

5.5.2 Key results

A total of 200 taxa (some may represent species aggregates) were recorded, seven of which were at genus level but were included in the total taxonomy richness count, as they may represent different species from the others within the same genus. Apart from one GLX site (Killegan River), which had just eight taxa, and the single IHPo site (11 taxa), the other SSNet types had mean richness values between 18 and 24.

The results of the PERMANOVA show that community structure was significantly different between the geological and SSNet type descriptor classes, but not with respect to physiography, soil or morphological river type (Table 5.4). It should be noted that the significant difference in diatom community structure between geological classes should be interpreted with caution, as the degree of dispersion (variance) is significantly different between the geological types and so may be driving the observed dissimilarity in community structure.

Although a range of environmental variables appear to structure the phytobenthos communities, the strongest explanatory variable that shows the

Table 5.3. Percentage of macroinvertebrates common to spring and summer samples, percentage unique to each season and richness values for each season

Tributary	Common to spring and summer (%)	Unique to spring (%)	Unique to summer (%)	Richness spring	Richness summer	Total richness
T1	50	36.4	13.6	38	28	44
T2	41.5	34.1	24.4	31	27	41
T3	33.3	43.8	22.9	37	27	48

Table 5.4. Results of PERMANOVA assessing whether diatom community structure varies significantly within each of the physical descriptor classes, and whether the degree of dispersion (variance) varies significantly with each physical descriptor class

	PERMANOVA of community structure				ANOVA of dispersion	
Physical descriptor class	df	Sum of squares	Pseudo-*F*	*p*-Value	*F*-value	*p*-Value
Geology	2	2.611	5.209	1.00×10^{-4}	6.937	0.003
Physiography	3	1.003	1.116	0.312	4.010	0.013
Soil	3	1.321	1.512	0.052	2.309	0.092
SSNet type	11	5.403	2.111	1.00×10^{-4}	2.038	0.060
Morphological river type	7	2.304	1.1105	0.260	1.510	0.197

$p < 0.05$ significant.
ANOVA, analysis of variance.

greatest correlation is pH, with a clear east–west axis (Figure 5.12).

5.6 Macrophytes

5.6.1 Approach

Macrophyte investigations took place at the majority of sites in August to early October 2020 and at a few remaining sites in August 2021. Sampling followed European Standard 14184:2014 to include surveying of all vascular plants, bryophytes, charophytes and macroalgae (CEN, 2014b). A central 60-m stretch within the 100-m MultiMoRPh reach was surveyed. The percentage cover of those plants in the channel (i.e. at or below the normal water level) was estimated. Vascular plants were identified in the field where possible, and voucher samples retained for laboratory identification where this was not possible. Macroalgal samples were kept chilled and identified in the laboratory within 6 days of collection. The majority of bryophytes were dried for later identification. Identifications of the bryophytes were confirmed by Nick Hodgetts, Botanical Services, UK.

5.6.2 Key results

A total of 184 macrophytes were recorded from the 42 sites, but all SSNet types recorded average taxon richness values less than 20 (Figure 5.13). The sites on igneous/metamorphic geology had lower richness values than the other site types. Bryophytes accounted for 94 of the total richness count and dominated the communities at all but the limestone

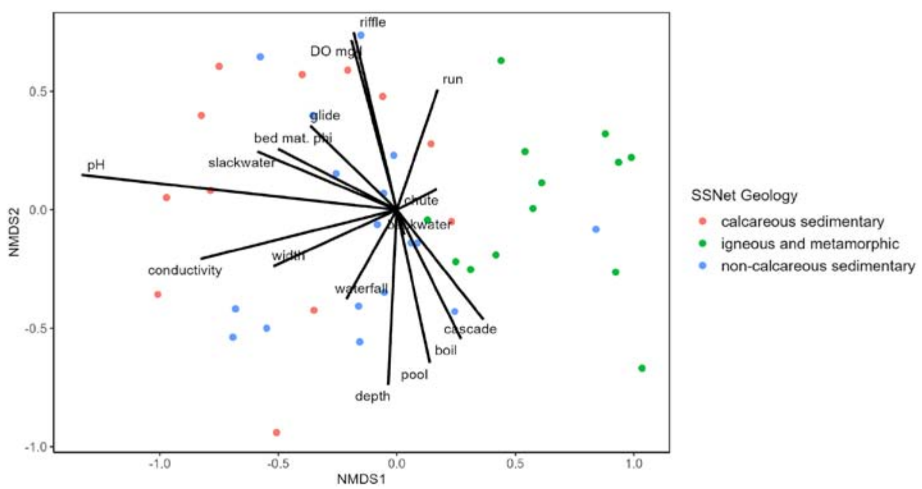


Figure 5.12. nMDS plot of diatom community dissimilarity across the extensive sites, with subset of the environmental variables represented as vectors. The lines are gradients representing the direction of that variable's steepest increase with respect to the ordination configuration. Their length represents the degree to which each dependent variable is explained by the ordination scores.

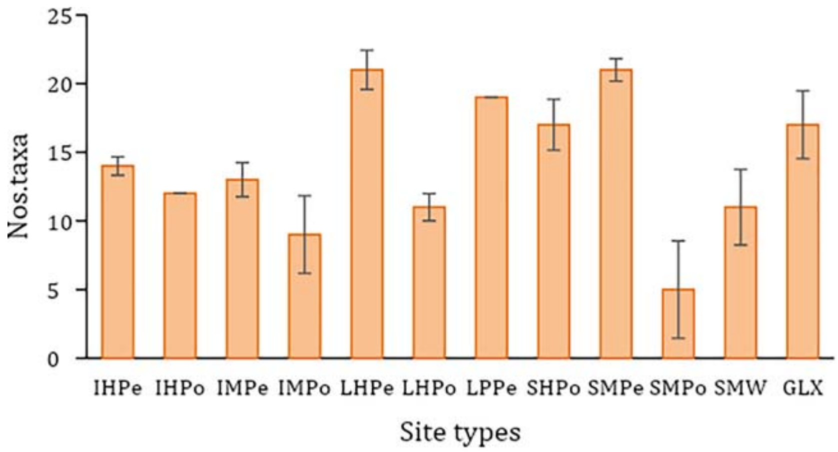


Figure 5.13. Mean (±standard error) macrophyte taxon richness across the SSNet types.

sites (Figure 5.14). Macroalgae (27 taxa) were the least diverse, with a mean richness value of less than 5 across all sites. Among the most commonly occurring species were the liverwort *Scapania undulata*, which occurred at 30 sites, followed by *Racomitrium aciculare* (moss – 22 sites), *Chiloscyphus polyanthos* (liverwort – 20 sites), *Juncus articulatus* (rush – 19 sites) and *Pellia* (liverwort –19 sites). Eighty-four taxa (7 macroalgae; 51 bryophytes; 26 vascular plants) had single-site occurrences.

Three species with low (≤4) species trophic rank scores were recorded, but with low cover values. These were *Cladophora* (one site in GLX, IMPe , IHPe and SMW), *Brachythecium rutabulum* (one SHPo site) and *Apium nodiflorum* (two GLX sites, one IHPe site and one SHPo site).

With the exception of soil drainage type, there were significant differences in community structure between site groups based on all physical descriptors (Table 5.5). This should be interpreted with caution for physiography, where there was a significant difference in the degree of dispersion between the groups.

The macrophyte communities are structured by a pH/conductivity gradient that aligns with a progression from igneous/metamorphic to calcareous geology. There is an opposite gradient of increasing percentage of high-energy biotopes (chute, boil, cascade) from left to right (Figure 5.15).

5.7 Salmonid Populations of Small Coastal Streams

5.7.1 Introduction

The fish study was a pilot exercise focused on the understudied, but numerous, small coastal streams that are considered potentially important as sea trout spawning and nursery habitat. The aim was to survey a minimum of five small east and five small west coast streams to assess the presence of sea trout (*Salmo trutta*) and wider fish diversity.

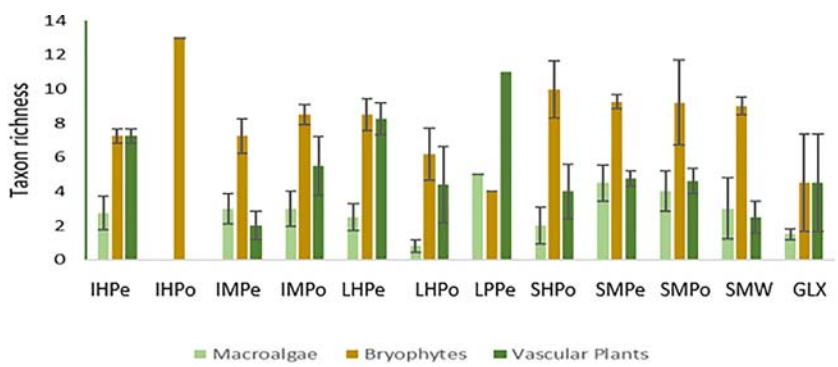


Figure 5.14. Mean (±standard error) number of macroalgae, bryophyte and vascular plant taxa across the SSNet types.

Table 5.5. Results of PERMANOVA assessing whether community structure varies significantly within each of the physical descriptor classes, and whether the degree of dispersion (variance) varies significantly with each physical descriptor class

	PERMANOVA of community structure				ANOVA of dispersion	
Physical descriptor class	df	Sum of squares	Pseudo-*F*	*p*-Value	*F*-value	*p*-Value
Geology	2	1.779	3.614	0.0001	0.443	0.645
Physiography	3	1.154	1.429	0.044	11.393	0.00002
Soil	3	1.106	1.364	0.078	5.439	0.003
SSNet type	11	4.571	1.831	0.0001	1.580	0.156
Morphological river type	7	2.574	1.422	0.016	1.505	0.199

$p < 0.05$ significant.
ANOVA, analysis of variance.

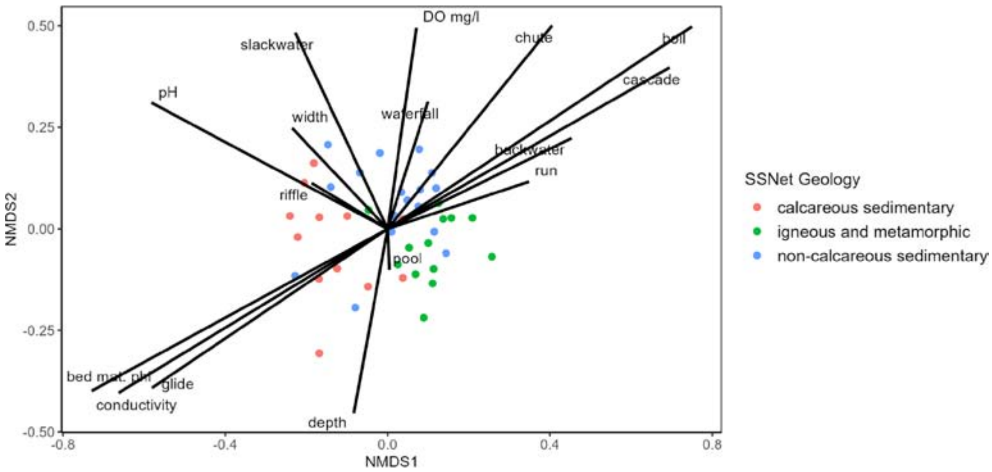


Figure 5.15. nMDS plot of macrophyte community dissimilarity across the extensive sites, with a subset of the environmental variables represented as vectors. The lines are gradients representing the direction of that variable's steepest increase with respect to the ordination configuration. Their length represents the degree to which each dependent variable is explained by the ordination scores.

5.7.2 Approach

A total of 20 candidate sites were selected based on access from road, distance from the sea and size of stream. Site visits in 2020 (11 August to 8 September) confirmed whether the sites were suitable for electrofishing. Eight of the 20 sites were unsuitable due to their small size. The remaining 12 sites (Table 5.6 and Figure 5.16) were sampled, with most sites subjected to three-pass standard electrofishing (eight sites), in accordance with Bohlin *et al*. (1984). However, in a few cases, the density of brown trout was very low or brown trout was absent (four sites), not warranting further electrofishing efforts, and such sites were subsequently subjected to only single-pass electrofishing. All caught brown trout were sedated (clove oil), measured (fork length to the closest millimetre) and weighed (to the closest 0.1 g).

5.7.3 Key results

All six east coast sites supported brown trout and three streams had sea trout (as judged by the silvery colour). Note that the colouration of brown trout is very plastic, and silvery sea trout can rapidly change colouration and become indistinguishable from brown trout. European eels (*Anguilla anguilla*) were observed at all east coast sites, while three-spined stickleback (*Gasterosteus aculeatus*) and flounder (*Platichthys flesus*) were found in many of the streams (Table 5.6). Density estimates of brown trout varied from 18 to 54 brown trout per 100 m^2 when undertaking three-pass electrofishing (five out of the six sites). The average density of brown trout across all five east coast streams that were subjected to three-pass electrofishing was 40.7 (standard deviation (SD) = 14.7) fish per 100 m^2 (Table 5.5). Two of the east

Table 5.6. Fish species caught by electrofishing at the 12 sites, including numbers caught and density figures, where estimated

Stream name	Site code	Brown trout	Density (fish/100 m²)	Other fish species observed/comments
Shanganagh River	E1	17	n/a	Eel, flounder
Three Trout Stream	E2	55	14.7	Eel, three-spined stickleback, flounder
Newcastle Lower	E3	63	41.8	Sea trout (5), eel, three-spined stickleback, flounder
Leamore Upper	E4	77	41.4	Eel, flounder
Rathnew stream	E5	77	46.4	Sea trout (1), eel, three-spined stickleback
Potters River	E6	62	24.5	Sea trout (8), eel, three-spined stickleback, flounder
Aughness	W1	1	n/a	Sea trout (1), eel, three-spined stickleback, fished 100 m
n/a	W2	n/a	n/a	No fishing. Stream had no running water
n/a	W3	n/a	n/a	Stream overgrown and no running water
Doolough	W4	n/a	n/a	Stream overgrown and no running water
Tullaghanduff	W5	0	n/a	Eel, three-spined stickleback, fished 100 m
n/a	W6	n/a	n/a	Stream overgrown and no running water
n/a	W7	n/a	n/a	Stream overgrown and no running water
n/a	W8	n/a	n/a	Stream overgrown and no running water
n/a	W9	42	26.9	Sea trout (1), eel, flounder
n/a	W10	n/a	n/a	Refused access to land due to COVID-19
Aghoos	W11	40	36.2	Eel
n/a	W12	0	n/a	Eel, three-spined stickleback, fished 80 m
Carrowaglogh/Portacloy	W13	54	27.5	Eel, three-spined stickleback, flounder
n/a	W14	n/a	n/a	Waterfall outlet. No fishing

n/a, not applicable.

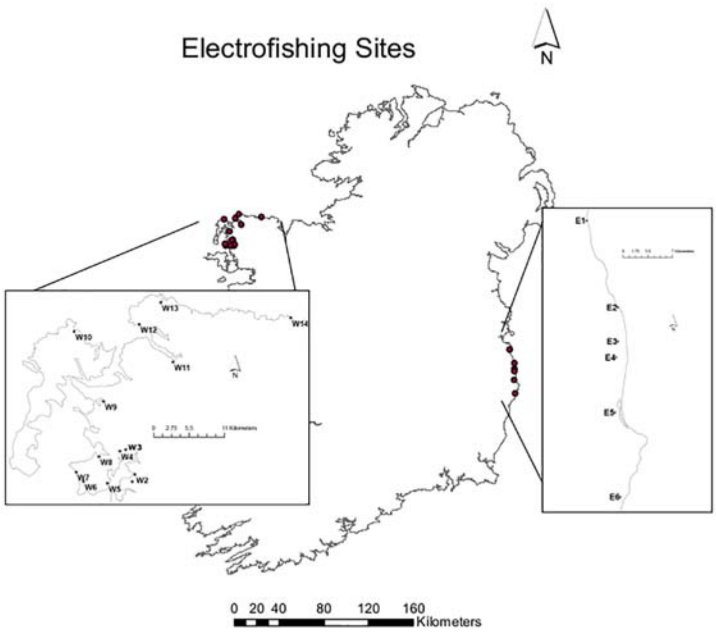


Figure 5.16. Electrofished sites on the east and west coast of Ireland.

coast streams sampled have been classified as poor quality by the EPA – the Shanganagh River (site code E1) and the Rathnew stream (site code E5). However, despite the poor quality, both supported brown trout, and sea trout were encountered in the Rathnew stream.

Four of the west coast streams had brown trout and two of these streams supported sea trout (based on the colouration). Similarly to the east coast streams, European eel were detected in all sampled streams, while three-spined stickleback and flounder were observed in several streams (Table 5.6). The number of brown trout per 100 m^2 ranged from 29 to 37 at those sites where three-pass electrofishing was carried out. The average density of brown trout across all three west coast streams that were subjected to three-pass electrofishing was 32.2 (SD = 4.1) brown trout per 100 m^2, with at least two cohorts found at each site. In total, 14 sea trout were caught in the east coast streams, compared with only 2 sea trout in the west coast streams.

Most sites supported predominantly two cohorts, fish of the year and 1+ fish, but small numbers of larger fish were also present in some of the east coast sites. For example, site E6 (Potters River on the east coast) held some larger fish up to 38 cm in length.

5.8 A Role for Environmental DNA in Aquatic Biota Monitoring?

5.8.1 Introduction

The objective for this research task was to assess how well environmental DNA (eDNA)-based methods could detect macroinvertebrate and phytobenthos biodiversity, compared with traditional morphological identification methods. To this end, we sampled water and diatom scrapings that were subsequently subjected to high-throughput sequencing-based metabarcoding.

5.8.2 Approach

The macroinvertebrates that the lists generated from morphological identifications on samples collected from three sites on tributaries of the River Dargle (grouped sites) were compared with eDNA extracted from water samples. The phytobenthos (diatoms) results from morphological identifications were compared with genetic analysis on phytobenthos scrapings from 40 extensive sites.

Sampling and filtering

Water samples (biological samples) from the two target streams were collected in 2-L sterile bottles in May and June 2021. Each stream was sampled in the lower and upper reaches, as well as in the larger tributaries. Each sample consisted of three biological replicates and one “cooler blank” (a sample bottle filled with distilled water that was opened at the time of sampling and kept together with the biological samples). The biological replicates were sampled at three equidistant points in a transect across the stream. Samples were kept on ice until arrival at the laboratory and all samples were filtered on the day that they were sampled through individual Cytiva Whatman Binder-free Glass Microfiber Filters, grade 934-AH circles (1.5 μm). The filters were kept at –20°C until further processing. The phytobenthos samples were collected as outlined in section 5.4.1 and were stored in molecular-grade ethanol (99.8%).

DNA extraction, amplification and library preparation

The filters from water samples were cut into halves (one half for analysis and the other half for archival storage) and shredded to increase surface area for eDNA extraction using a Qiagen QIAshredder and Qiagen DNeasy Blood & Tissue Kit. A total volume of 5 mL of diatom scrapings was spun down in a 50-mL Falcon tube at 20,817 g for 10 minutes to form a pellet that was then subjected to DNA extraction using a modified chloroform/isoamyl alcohol protocol (Petit *et al*., 1999). Polymerase chain reaction (PCR) preparations were carried out in a dedicated low-copy DNA laboratory. Each extraction effort included an extraction blank (extraction process without input of eDNA filter), which was subsequently processed as a normal eDNA sample. To multiplex the water samples, a combination of uniquely tagged (mlCOIintF and jgHCO2198) primers was used to amplify an approximately 313-bp section of the mitochondrial cytochrome oxidase I gene (*COI*) (Leray *et al*., 2013). Similarly to multiplex diatom samples, a combination of uniquely tagged forward and reverse primers was used to amplify an approximately 312-bp section of ribulose bisphosphate carboxylase large subunit plastid gene (*rbcL*) (Vasselon *et al*., 2017).

Extracted water samples were subject to PCR using 2 × 20-μL reactions to increase volume, and have a larger volume to pool, to mitigate potentially low amplicon concentrations. Diatom DNA was amplified using a “nested PCR” using two rounds of PCR (the first round to amplify and a second round to attach the individual tags). The second PCR was made up to 30-μL reactions to increase the amount of amplified DNA. Each library, for both water and diatom samples, was pooled, quantified and, finally, combined in equimolar amounts. A PCR blank was included in each library. In total, two libraries were prepared for commercial sequencing using a paired-end 250-bp NovaSeq 6000 Illumina system from Novogene Europe.

Bioinformatics

The raw reads were sorted into original samples using Cutadapt (Martin, 2011). Denoising the reads into amplicon sequence variants (ASVs) was carried out using the DADA2 pipeline (Callahan *et al.*, 2016). Taxonomic assignment was performed with the Ribosomal Development Project's naive Bayesian Classifier (RDP Classifier v2.0.3; Wang *et al.*, 2007), using a broad-scale *COI* database that also includes non-eukaryotic outgroups (Porter and Hajibabaei, 2018), and, for the *rbcL*, the RbcL Diat. barcode Reference Set (https://doi.org/10.5281/zenodo.4741478) containing 1432 taxa was used. Of the resulting classifications, those with a bootstrap confidence (BSC) value of < 0.6 were changed to NA in the *COI* and *rbcL* datasets, respectively, with the highest classification above the threshold retained for the species-level notation in the taxonomy table (e.g. family assignment > 0.6 BSC; genus assignment < 0.6 BSC; species-level notation would be "*familyname_sp.*" for the *COI* data and "*Species*" for the *rbcL* data). The threshold of 0.6 was chosen as an appropriate trade-off between minimising the risk of false positives that could arise from not having a reference sequence in the database for the query sequence (i.e. missing references for a species) and improving the number of accurate species-level assignments (Gold *et al.*, 2021). Control correction to remove potential contaminant ASVs was carried out using field and laboratory controls (field: cooler and filter blanks; laboratory: extraction blanks and no-template controls) and a relative abundance threshold of 10% (see, for example, Antich *et al.*, 2021). All ASVs classified as "unassigned" were removed from the data. Further species verification was performed on both the *COI* and the *rbcL* datasets P (without using the 0.6 threshold), using a local blastn database consisting of the complete nucleotide (nt) database (downloaded 9 November 2022). The per cent identity was set to be at least 95%. A total of five alignments were reported to allow for detection of equal per cent identity and e-value (expected value) across species.

5.8.3 Key results

Sequencing and bioinformatics

Across all water samples and diatom sampling efforts, a total of 25,057,344 and 20,876,974 raw reads were obtained, respectively, from sequencing. Of these, 94.55% and 94.93%, respectively, passed filtering and were sorted into original samples and paired into the sequenced samples in accordance with individual barcodes. The RDP Classifier assigned 59,012 reads to species in the *COI* data, while 3,346,156 reads were assigned to taxa level in the *rbcL* dataset. The RDP analyses detected a total of 81 species based on the *COI* data, while 106 taxa were detected in the *rbcL* data. Furthermore, blastn-based analyses detected 67 species in the *COI* data, while 159 species were detected in the *rbcL* data. The three kick samples analysed from the River Dargle returned between 26 and 29 detected macroinvertebrate species, while eDNA analyses of water samples showed 9 and 10 invertebrate species (Figure 5.17a–c). Similarly,

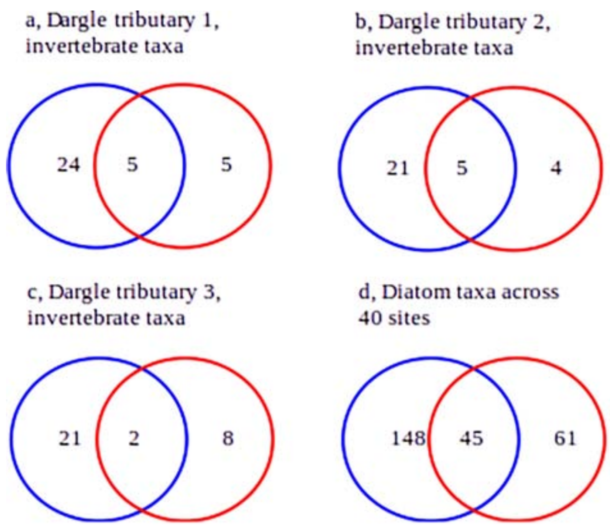


Figure 5.17. Venn diagrams showing invertebrate taxa detected in kick samples or visual diatom identification (blue) and through eDNA of water samples and genetic analyses of scraping (red) samples, with taxa detected in both analyses in the overlapping area. Kick samples/water eDNA samples (a–c) and diatoms visual/genetics (d).

across the 40 sites analysed for diatom diversity, a total of 193 taxa were detected using morphological methods, while 106 taxa were detected using genetics (Figure 5.17). In the Venn diagrams we have the taxa unique to the kick samples/visual diatom identification, those unique to the genetic samples (water and scrapings) and those taxa detected using both methods. It is noteworthy that analyses using traditional morphological methods always detected more taxa. This could be due to there being no truly “universal primers” that can amplify all taxa or to not all organisms shedding tissues or DNA to the same extent, but, it is perhaps more likely that not all taxa have reference sequences in the databases, and this is perhaps more prominent in the diatom database that contains only some 20% of the available taxa (B. Kennedy, EPA, 26 April 2023, personal communication). Furthermore, there might be taxa in Irish waters that are not represented in any genetic repository. In conclusion, the analyses demonstrate that genetic methods are capable of detecting taxa, but the taxa are not necessarily the same as those detected by traditional methods. Further development of the genetic databases should increase the number of taxa detected. In particular, generating DNA reference sequences for Irish specimens would greatly improve the detection capability. As these new sequences are added to the repositories, the data generated here can be reanalysed with improved detection of taxa (as many sequences generated in this project do not have a match in the existing genetic repositories).

6 Hydrological Influences on Biodiversity and Ecosystem Function in Small Streams

6.1 Introduction

Climate change will add to the suite of stressors potentially impacting the ecology and functioning of surface waters, including small streams (Palmer *et al.*, 2009; O'Briain, 2019). These include changes in flow regime, potentially elevating fine sediment inputs, and increases in CO_2 concentrations. This chapter reports on the results of a large-scale mesocosm experiment investigating the influence of variable flow, elevated CO_2 enrichment, fine sediment pulses and lack of shading – individually and in combination – on the biodiversity of stream communities and key functions. This experiment was carried out in collaboration with the Biodiversa Land2Sea project (https://land2sea.ucd.ie/) to maximise output from a labour-intensive and costly experiment.

6.2 Experimental Approach

A field experiment was carried out from May to July 2022 at Annamoe, County Wicklow. The experimental system consisted of 128 circular channels, 25 cm in diameter and 9 cm deep, with a central opening 5 cm in diameter and 1 cm lower than the outer edge (Figure 6.1). The channels were continuously gravity fed with water from eight header tanks at the average discharge rate of 1.9 L/min. Water entered each channel through a jet that directed the water around the outer perimeter, generating an average current velocity of 9.4 cm/s, and left the channel over the central opening. The header tanks were supplied with water by two electric pumps from the nearby stream, fitted with a protective 4-mm intake mesh (Figure 6.1). The source stream was of low alkalinity

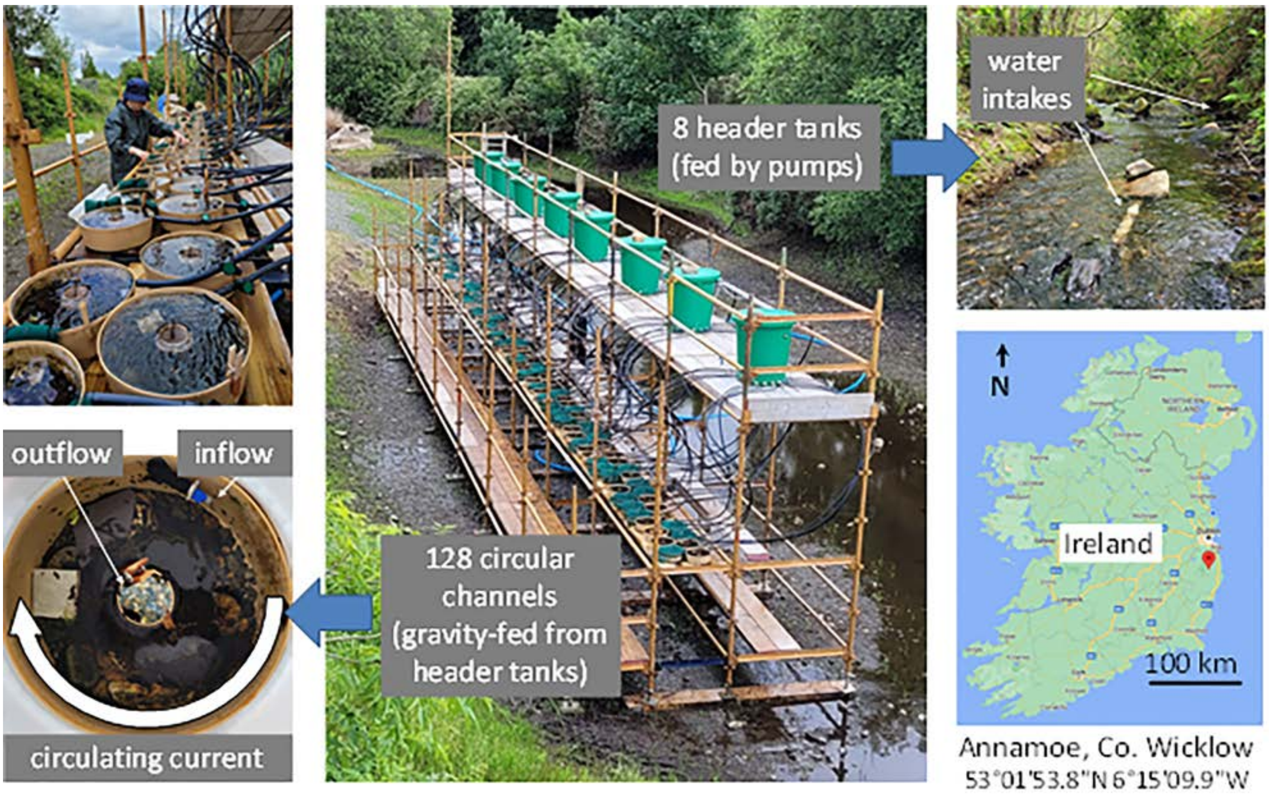


Figure 6.1. The experimental stream set-up in Annamoe, County Wicklow.

(8.1 mg $CaCO_3$/L) and low nutrient concentration (SRP: 2.3 µg/L; total ammonia as N: 0.01 mg/L; TON: 0.69 mg/L).

Each channel was filled with 500 mL of gravel-pebbles screened to 2–15 mm, 20 mL of sand screened to 0.5–2 mm and 14 stones approximately 3–5 cm across collected from the nearby river channel. Each channel was also supplied with 5 g of moss by wet weight, and five assorted leaves (birch, willow, alder), as habitat enrichment and food subsidy for invertebrates. The continuous water supply allowed natural drift and pre-colonisation of stream biota, including algae and invertebrates, for 33 days prior to experimental manipulations (27 May to 28 June). This natural colonisation was supplemented by kick samples from the source stream and the larger Avonmore River at the end of the pre-colonisation period. The invertebrates were allowed to settle down for 1 day before the manipulative period commenced.

The manipulative phase took place over 24 days (29 June to 22 July). The experimental treatments consisted of four pressures with two levels each (perturbed vs control) in full-factorial design, replicated eight times (Figure 6.2). The variable flow perturbation oscillated between 0.130 and 0.046 m/s every 5 days (two full high–low cycles), compared with stable flow of 0.094 m/s in the control treatment. Sediment perturbation was implemented in two pulses of 100 mL, coinciding with each onset of high flow to mimic sediment run-off after rainfall. Sediment was sourced from the bank of the source stream and screened to < 0.5 mm with a relatively uniform particle size distribution. It had low phosphorus (0.464 mg/g of dry weight), organic matter (6.7% of dry weight) and carbonate content (3% of dry weight). Background (control) treatment had negligible fine sediment cover. CO_2 perturbation was applied continuously as a 2–3× enrichment over the background (control) levels, consistent with the Intergovernmental Panel on Climate Change Fifth Assessment Report Representative Concentration Pathway 6.0 “moderate–high emissions” scenario. This enrichment led the pH to drop by 0.3–0.5 from the background pH of 6.7–7.0 over the experimental time. Light perturbation was applied continuously as an absence of shading, whereby control treatment was provided with a 40% shade cloth, consistent with moderate riparian shading (Figure 6.2). The experimental installation was split into two blocks to better account for any environmental gradients, and the treatments were randomly allocated to channels within the blocks.

The response variables were the abundance and structure of benthic algae (main primary producers) and invertebrate communities (main consumers). Whole-channel total respiration and net productivity were also measured as cumulative measures of whole-ecosystem processes. Benthic algal biomass was approximated as chlorophyll-a concentration on unglazed tiles, which were preconditioned face down in the source stream for a week. This was measured *in situ* every 5 days, at the end of each variable flow phase, using reflectance (BenthoTorch, bbe Moldaenke GmbH) separately for cyanobacteria, green algae, diatoms and their total. Benthic invertebrates were sampled from the entirety of the experimental channels at the end of the experiment, after the whole-channel respiration and productivity measurements were made. Whole-channel total respiration and net

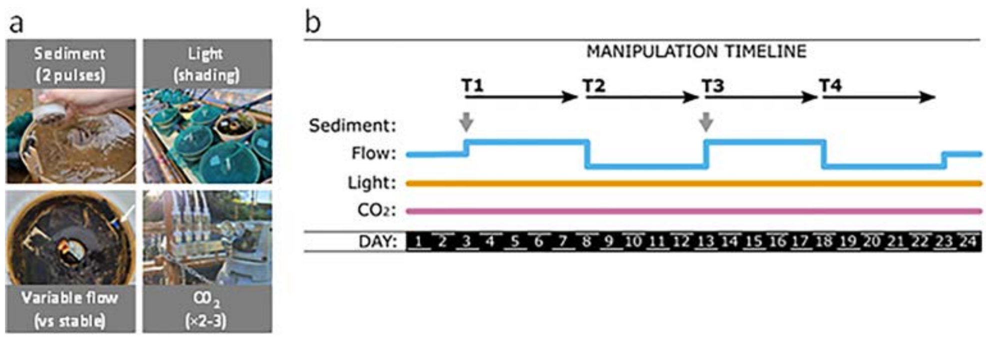


Figure 6.2. Experimental manipulations (a) and their timeline (b). Under variable flow treatment, water current velocity oscillated between higher and lower than the baseline every 5 days. Sediment was applied in two pulses coinciding with the onset of high flow. Light (lack of shading) and CO_2 enrichment were applied continuously.

productivity (simultaneously on separate 64 channels; $n = 4$) were approximated from dissolved O_2 changes at the end of experiment with the flow switched off. Cling film was placed on the water surface to minimise oxygen exchange with the atmosphere. Opaque blue plastic covered the respiration channels to shade them, while transparent plastic covered the productivity channels as a rain guard. During the whole-channel measurements, water temperature was recorded every 15 minutes on loggers (HOBO UA-001-08, Onset) in a subset of eight channels equally distributed along the experimental set-up to quantify any temperature gradients. In addition, light intensity was recorded every 15 minutes using a centrally positioned logger (HOBO MX2202, Onset).

6.3 Key Results

6.3.1 Algae (BenthoTorch)

Sediment was the most pervasive influence on benthic algae. Algal concentration typically decreased after both of the sediment pulses, but subsequently recovered. Moreover, at the end of the experiment, sediment increased algal concentration ($p < 0.001$) by 53%, similarly for all algal groups. Flow variability by itself affected only diatoms and only temporarily during the first slow-flow phase ($p = 0.017$), decreasing their concentration by 23% in the slow-flow phase. However, it was frequently involved in significant interactions, masking the effects of other stressors in the slow-flow phase. While enriched CO_2 increased algal concentration at stable flow during the first slow-flow phase, at variable flow it was similarly low regardless of CO_2 enrichment ($p = 0.014$). This effect was conveyed mostly through diatoms ($p = 0.020$) and cyanobacteria ($p = 0.004$), but not green algae ($p > 0.05$). Flow interacted similarly with light on green algae in the first slow-flow phase ($p = 0.007$) and with sediment cyanobacteria in the second slow-flow phase ($p = 0.012$), in both cases masking the effects of those other stressors.

Light changed the relative abundance of the different algal groups. Without shading, cyanobacteria doubled, on average, across the experimental timeline, up to tripling at any one time point ($p < 0.001$). In parallel, green algae concentration decreased by 15%, on average, across the whole experiment, possibly outcompeted by cyanobacteria. Average recorded diatom and total algal concentration both increased throughout the experiment, but this was not statistically significant ($p > 0.05$). Light also interacted with sediment. While added sediment increased algal concentration at shaded conditions during the second high-flow phase, at unshaded conditions it was similarly high regardless of CO_2 enrichment ($p = 0.009$). CO_2 enrichment increased the concentration of diatoms ($p = 0.005$, by 31%) and total algae ($p = 0.019$, by 32%) at the second of four recording intervals. A three-way interaction was also noted at the end of the experiment ($p = 0.029$), whereby the enhancing effect of siltation on diatom biomass was more readily observed when shading was removed or when CO_2 was enriched, but not in the other combinations. Under shading and baseline CO_2, diatom biomass was similarly low, whereas, without shading and under CO_2 enrichment, diatom biomass was similarly high, regardless of siltation.

6.3.2 Macroinvertebrate results

The responses of 26 community-level metrics to the experimental manipulations were computed. Total macroinvertebrate abundances were not significantly affected by any stressor; however, flow variability was almost significant ($p > 0.0506$; Figure 6.3). Sediment pulses had the most pervasive impact, with 16 community-level metrics significantly affected. Sediment pulses decreased total richness as well as decreasing the abundance and richness of Ephemeroptera, Plecoptera, Trichoptera (EPT) taxa. However, the proportion of dipteran taxa increased (Figure 6.3).

Variability of flow velocity decreased abundances of EPT taxa and Ephemeroptera ($p = 0.004$ and $p = 0.034$, respectively). This decrease in Ephemeroptera abundance can be attributed to reduced abundance of *Baetis rhodani/atlanticus* ($p = 0.036$). CO_2 enrichment reduced abundance of Ephemeroptera ($p = 0.047$); however, similarly to flow-velocity variability, this can be attributed to reduced abundance of *B. rhodani/atlanticus* ($p = 0.012$) (Figure 6.3).

Shading (or lack of shading) did not have any significant main effects. However, two-way interactions were observed between shading and CO_2 and shading and flow-velocity variability. An antagonistic interactive effect was identified between shading and CO_2 on insect richness. At ambient CO_2 concentrations,

	Sediment	CO_2	Flow
Total Richness	** ↘ [0.074]		
Margalef index	* ↘ [0.058]		
 EPT	*** ↘ (A) [0.178] ** ↘ (R) [0.082]		** ↘ (A) [0.071]
 Ephemeroptera	*** ↘ (A) [0.19] * ↘ (R) [0.054]	* ↘ (A) [0.035]	* ↘ (A) [0.040]
 Trichoptera	* ↘ (R) [0.048]		
 Diptera	* ↗ (Prop.) [0.047]		
 *Baetis agg.*	*** ↘ (A) [0.106]	* ↘ (A) [0.056]	* ↘ (A) [0.039]
 *Seratella ignita*	*** ↘ (A) [0.109]		
 *Gammarus duebeni*	* ↘ (A) [0.054]		** ↘ (A) [0.072]
 *Limnius volckmari*		* ↘ (A)	

Figure 6.3. Summary of significant ($p < 0.05$) main effects of stressors (sediment, CO_2, flow variability) on community-level metrics and common taxa. No significant main effects of shading were observed and so are not included. Significance levels: *$p < 0.05$, **$p < 0.01$, ***$p < 0.001$. Arrows indicate either an increase or decrease, and the size of the arrow corresponds to partial eta-squared scores. Values in square brackets are partial eta-squared scores. A, abundance; Prop., proportion; R, richness.

shaded channels had a reduced insect richness. However, at enriched CO_2 concentrations, shading increased insect richness. A similar antagonistic effect was observed between shading and flow velocity on dipteran richness, i.e. at stable flow velocity the shaded channels had reduced dipteran richness, while at variable flow velocities the shaded channels had an increased dipteran richness.

A significant two-way antagonistic interaction also occurred between CO_2 and sediment. Channels with ambient sediment levels had reduced coleopteran abundance and richness when enriched with CO_2. However, channels that received sediment pulses had increased abundance and richness when enriched with CO_2. In addition, the abundance and richness of coleopterans converged in response to CO_2 enrichment in channels that did and did not receive sediment pulses.

Multivariate analyses (PERMANOVA) determined that benthic community composition was affected by sediment and flow velocity variability ($p < 0.05$) as main effects only. A follow-up univariate analysis showed that four of seven of the most common taxa were significantly affected by stressor main effects and/or interactions between three of four stressors ($p < 0.05$). In addition, CO_2 enrichment also reduced the abundance of *Limnius volckmari* ($p = 0.041$). Increased variability in flow velocity reduced the abundance of *G. duebeni* ($p = 0.004$). Sediment pulses negatively impacted abundances of *B. rhodani* ($p < 0.001$) and *Seratella ignita* ($p < 0.001$). It also reduced the abundances of *G. duebeni* ($p = 0.013$) (Figure 6.3).

6.3.3 *Whole-channel total respiration and net productivity*

Sediment addition was a key influence on whole-channel respiration rate ($p = 0.006$), decreasing it by 17%. Neither flow nor CO_2 nor light was significant on its own. However, flow was significant in a two-way interaction with light ($p = 0.032$), whereby variable flow decreased respiration under unshaded but not under shaded conditions. Moreover, both flow and sediment were influential in a four-way interaction ($p = 0.025$), but there was insufficient statistical power under multiple comparisons to resolve it.

On average, across all channels, the whole-channel net productivity rate was negative, indicating net heterotrophy under the experimental conditions (i.e. excess of respiration over primary productivity). CO_2 enrichment was a key driver of net productivity ($p = 0.004$), decreasing heterotrophy by 76%. Sediment addition and lack of shading were also influential ($p = 0.046$ and $p = 0.049$, respectively), decreasing heterotrophy by 56% and 41%. Flow had no effect on its own. However, it was significant in a three-way interaction ($p = 0.032$), whereby the positive effect of CO_2 enrichment on productivity was more clearly manifested under stable flow and shaded conditions than under other combinations. Moreover, both flow and sediment were influential in a four-way interaction ($p = 0.043$), whereby the positive effect of CO_2 enrichment on productivity was more clearly manifested under stable flow, lack of siltation and shaded conditions than under other combinations.

6.4 Overall Findings

Among the examined stressors, sediment pulses were the most universally important perturbation. Algal concentration decreased after the sediment pulses, but subsequently recovered, and eventually increased at the end of the experiment. Sediment affected invertebrate communities, typically decreasing the richness and the abundance of the more sensitive insect taxa, and the abundance of the freshwater shrimp, while increasing the proportion of true flies. Sediment also decreased the whole-channel total respiration. Flow variability was a lesser influence. By itself, it decreased the abundance of the more sensitive insect taxa, and of freshwater shrimp. However, it was important in moderating the influence of other stressors on algae, invertebrates, whole-channel total respiration and net productivity. Repeated algal sampling revealed that this was particularly the case during the slow-flow phase of the variable flow perturbation. Increased light (absence of shading) changed algal composition, enhancing the abundance of cyanobacteria and decreasing green algae, potentially through competition. Its effect was manifested by increased whole-channel net productivity, and it also had interactive effects with other stressors on invertebrates. Finally, CO_2 enrichment increased algal abundance temporarily. It also reduced abundance of mayflies and the beetle *L. volckmari*, and richness of true flies. At the whole-channel level, CO_2 enrichment was a key driver of net productivity.

7 Modelling the Intervention Required in the Small Stream Network to Impact on Nutrient and Sediment Export

7.1 Introduction

Investigating the usefulness of numerical conceptual modelling for estimating the water quality of streams from diffuse sources is particularly challenging for small headwater catchments. Some of the major issues are (i) paucity of meteorological, flow and water chemistry data at scales suitable for calibration and validation in most small catchments, and (ii) the particular sensitivity at small spatial scales of water quality to isolated individual influences (e.g. point sources or specific activities) that may not be adequately quantified. One example of the latter is the influence of wastewater treatment plants or combined sewer overflows. While the locations of many of these are mapped, the discharges and water pollution delivered through them are not always fully quantified. The EPA already has tools to produce Pollution Impact Potential maps indicating areas with a high relative risk of mobilisation of specific pollutants (nitrogen and phosphorus) from specific local areas for use in targeting specific mitigation measures at a local (typically farm-scale) level. These are well used and appreciated by organisations such as LAWPRO and the Agricultural Sustainability Support and Advisory Programme. The EPA also has the Source Load Apportionment Model (Mockler and Bruen, 2018), developed for larger catchments. However, a different tool is required when the focus is on quantifying export from landscape units within small headwater catchments, as studied by this project.

Here we describe a methodology, based on conceptual hydrological modelling, for quantifying the export (loads and concentrations) of diffuse source contaminants from small catchments to assist in determining impacts (including of mitigation measures) on downstream waterbodies. A number of such models are available and were considered for demonstrating the approach. The practical need for (i) applicability in ungauged small catchments and (ii) a simple graphical display of simulation results for ease of use suggested the widely used Soil Water Assessment Tool (SWAT) model (Gassman *et al*., 2007). Since 2007, SWAT has improved its simulation of flow pathway partitioning aimed at supporting water quality modelling (nitrogen, phosphorus and sediment) (Kannan *et al*., 2007) and performed best in a model intercomparison test in Irish catchments (Nasr *et al*., 2007). Here, for demonstration purposes, the programme has been applied to six SSNet catchments to investigate if it can be used to indicate particular diffuse areas linked to the export of contaminants from the catchment to downstream waterbodies for further, more detailed, investigation to inform remediation approaches to achieve specified reductions from such small catchments.

The SWAT model requires the following data: (i) a digital terrain model (DTM) that is used to determine a drainage network of streams and channels, (ii) a geographic information system (GIS) map of soil types, (iii) a GIS map of land use and (iv) meteorological information. The soil types and land use information are used to determine model parameters. In this work, a 5-m resolution DTM, used in the DiffuseTools project and provided by the EPA, was used. The land use map was a 100-m resolution geoTiff derived from the 2018 Copernicus Corine map (https://land.copernicus.eu/pan-european/corine-land-cover), with the 44 land use classes reclassified for the SWAT model. The soil map was a 1300-m resolution geoTiff derived from a reclassification of the Soil Map of Ireland (http://gis.teagasc.ie/soils/downloads.php). The reclassifications of both the land use and soil maps were carried out and made available for general use by Basu (2021). A higher spatial resolution soil map (National Soils Hydrology map) is now available (see https://gis.epa.ie/), and land use information from the Land Parcel Identification System may also be available (see https://www.gov.ie/en/service/1eb4d-land-parcel-identification-system-lpis/), and it would be interesting further work to examine the differences this additional information would produce. The SWAT model also requires meteorological information, obtained in this case from Met Éireann.

7.2 Approach

The six test catchments used in this demonstration of the methodology were selected from a subset of the SSNet catchments in which there were no known outfalls from wastewater treatment plants or combined sewer overflows, as these are likely to be point sources of nutrients that would complicate the identification of areas of elevated diffuse nutrient inflows. The selection spanned a range of catchment areas and SSNet types. Two of the larger SSNet extensive catchments were selected (Rapemills and Killeen stream), with areas of approximately 54 and 50 km^2, respectively. Two of the smallest catchments (Askanagap and Owenboy), with areas of approximately 12 and 11 km^2, respectively, were also selected, as were two further catchments with areas between these extremes (Devlins and Pollanassa), i.e. areas of approximately 22 and 18 km^2, respectively (see Table 7.1 and Figure 7.1).

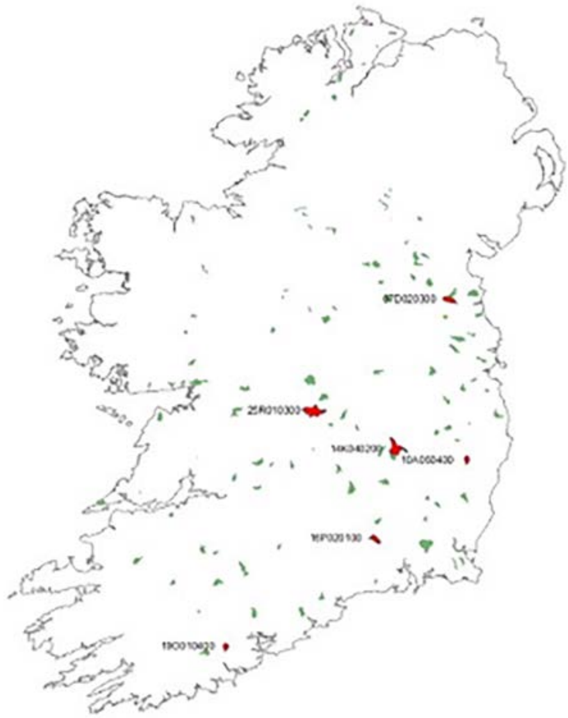


Figure 7.1. Locations of catchments for SWAT demonstration. Demonstration catchments are shown in red with EPA identifier, and other SSNet catchments without WTPs or known CSOs are shown in green.

7.3 Key Results

Note that in the following description of the results the soil types are identified by their Irish Soil System classification and then the percentage of the catchment area with that soil type. The latter is determined by SWAT from the 1300-m resolution soil raster map and may not be as accurate as a determination from a shapefile.

7.3.1 *Rapemills (area 53.6 km^2, SSNet typology LHPe)*

The Rapemills catchment delineated by SWAT has an area of 53.6 km^2. Its land use is mainly pasture (57%) and other agriculture (23%), but with some forest and wetlands. Most of the catchment has soil types fine loamy drift with limestones (49%) and peat (43%). The SWAT delineation divided the catchment into 56 channels in 14 subbasins (Figure 7.2). A total of 261 hydrological response units (HRUs) were generated. These consider different combinations of soils, land use and slope, differentiating between floodplain and upland areas of the catchment, and treat each differently by applying different model parameters for each HRU. The parameters are chosen from an extensive database linked to land use, soil type and slope characteristics. The SWAT output can be used to generate a map of the loads exported from the individual HRUs, and this can be used to identify "hotspots" with the highest amounts exported per unit area. For instance, the red areas in Figure 7.2 show the locations of the highest phosphorus export HRUs, contributing 20% of the total load, and together with the yellow areas show the areas generating the top 40% of the load. These are mainly grouped into four separate regions. The red areas are about 6% of the total catchment area and they generate the top 20% of the load. The combined red and yellow areas are 12% of the total catchment area and they generate the top 40% of the load. This information can be used to target

Table 7.1. Selected catchments representing different SSNet types and sizes

River name	Area (km^2)	SSNet type	EPA code
Rapemills	54	LHPe	25R010300
Killeen stream	50	LPPo	14K040200
Devlins	22	IHPo	07D020300
Pollanassa	18	SHPo	16P020100
Askanagap	12	SMW	10A060400
Owenboy	11	SHW	19O010400

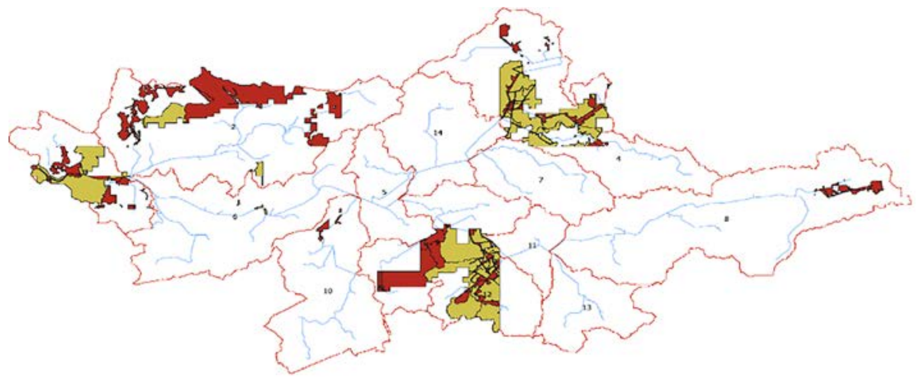


Figure 7.2. Rapemills SWAT delineated streams and subcatchments showing the HRUs delivering the highest 20% (red) and 40% (red + yellow) of loads in the Rapemills catchment.

the areas contributing most per unit area to the load for consideration for measures to reduce the loads they generate. Corresponding figures can be produced for any chosen percentages of the total load (and also for nitrogen and suspended sediment). This illustrates a methodology that can be applied generally to focus specific site investigations on hotspots. However, this project does not consider what specific measures may be appropriate (or their effectiveness), although the SWAT model can be used for this purpose.

7.3.2 Killeen stream (area 50.1 km², SSNet typology LPPo)

The same methodology can be applied to the other catchments. For instance, the Killeen stream catchment has an area of 50.1 km². Its land use is predominantly pasture (80%), with some other agriculture (16%). The main soil types are fine loamy drift with limestones (37%), fine loamy drift with siliceous stones (28%) and silty river alluvium (13%). The SWAT delineation divided the catchment into 47 channels and 11 subbasins (Figure 7.3). A total of 179 HRUs were defined. As with the Rapemills catchment, an examination of the loads exported from the individual HRUs indicates those with the highest amounts exported per unit area. The red areas in Figure 7.3 show the locations of those high-export HRUs, defined as contributing 20% of the total load, and together with the yellow areas show the locations contributing the top 40% of the load. As with the Rapemills catchment, only about 6% of the total catchment area generates the top 20% of the load and about 12% of the area generates the top 40% of the load. Note the "blockiness" of the red area in the north-eastern corner of the catchment. This is due to the combination of the small catchment size and the relatively lower resolution (1300 m) of the soil's raster map.

7.3.3 Devlins (area 21.8 km², SSNet typology IHPo)

The Devlins catchment has an area of 21.8 km². Its land use is predominantly pasture (93%), with a little other agriculture (5%) and a small amount of forestry. The main soil types are surface water gleys (51%), coarse loamy drift over shale bedrock (30%) and urban (8%). The SWAT delineation divided the catchment into 39 channels and 15 subbasins (Figure 7.4). A total of 128 HRUs were defined. The red areas in Figure 7.4 show the locations of those high-export

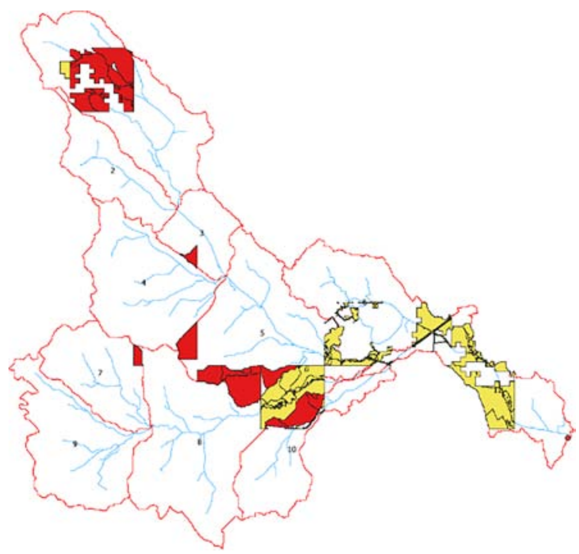


Figure 7.3. Killeen stream SWAT-delineated streams and subcatchments showing the HRUs delivering the highest 20% (red) and 40% (red + yellow) of loads in the Killeen stream catchment.

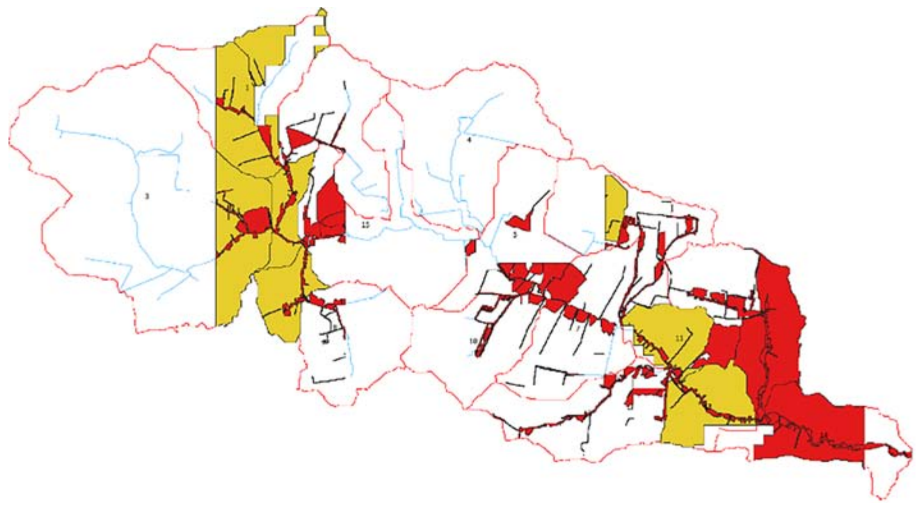


Figure 7.4. Devlins SWAT-delineated streams and subcatchments showing the HRUs delivering highest 20% (red) and 40% (red + yellow) of loads in the Devlins catchment.

HRUs contributing 20% of the total load, and together with the yellow areas show the locations producing the top 40% of the load. In this catchment, about 13% of the total area generates the top 20% of the load, and about 28% of the area generates the top 40% of the load. These hotspots are mostly in the downstream part of the catchment, with some more central. As with the Killeen stream catchment, note the "blockiness" of some of these areas due to the 1300-m resolution of the soil raster map. While the percentages of the catchment areas involved are higher than for the previous two catchments, the hotspot areas are grouped into two main areas, and this should facilitate field investigations to identify appropriate measures.

7.3.4 Pollanassa (area 17.6 km², SSNet typology SHPo)

The Pollanassa catchment has an area of 17.6 km². Its land use is predominantly pasture (91%), with some forestry (9%). The main soil types are fine loamy drift with siliceous stones (62%), luvisols and surface water gleys on drift with a mixture of limestone and siliceous stones (21%) and well-drained brown earths on drift with siliceous stones (16%). The SWAT delineation divided the catchment into 19 channels and 9 subbasins (Figure 7.5). A total of 64 HRUs were defined. The red areas in Figure 7.5 show the location of the single high-export HRU that by itself contributes 20% of the total load, and together with the yellow areas show the locations producing the top 40% of the load. Remarkably, most of the load is estimated to come from three relatively small and compact areas of the catchment. Less than 2% of the total catchment area generates the top 20% of the load, and about 5% of the area generates the top 40% of the load. This suggests that measures to reduce phosphorus export targeted at these areas would have great potential for reducing loads in the streams.

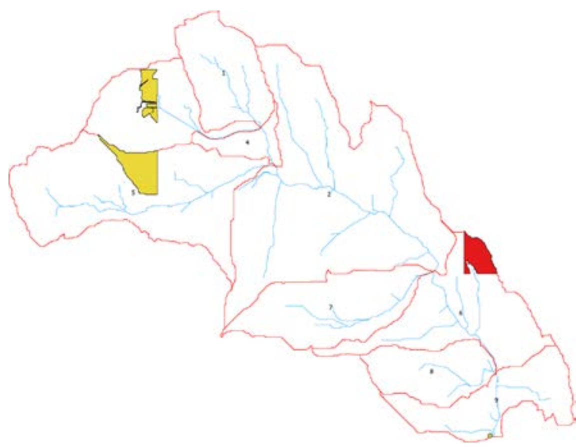


Figure 7.5. Pollanassa SWAT-delineated streams and subcatchments showing HRUs delivering the highest 20% (red) and 40% (red + yellow) of loads in the Pollanassa catchment.

7.3.5 Askanagap (area 12 km², SSNet typology SMW)

The Askanagap catchment has an area of 12 km². Its land use is predominantly pasture (52%), with some areas of rough grass and brush (28%) and forestry (19%). The main soil types are humic brown podzolics (48%), peat (38%) and some well-drained brown earths (8%). The SWAT delineation divided the catchment into 45 channels and 11 subbasins (Figure 7.6). A total of 203 HRUs were defined. The red areas in Figure 7.6 show the location of the high-export HRUs that contribute 20% of the total load, and together with the yellow areas show the locations producing the top 40% of the load. These are more widely distributed throughout the catchment than for Pollanassa, but are concentrated in three main areas. Approximately 9% of the total catchment area generates the top 20% of the load and about 16% of the area generates the top 40% of the load.

7.3.6 Owenboy (area 10.7 km², SSNet typology SHW)

The Owenboy catchment has an area of 10.7 km². Its land use is predominantly pasture (92%) with between 2% and 3% each of evergreen forest, mixed forest and rock. The soil type is predominantly coarse loamy drift with siliceous stones. The SWAT delineation divided the catchment into 52 channels and 14 subbasins (Figure 7.7). A total of 129 HRUs were defined. The red areas in Figure 7.7 show the locations of those high-export HRUs contributing 20% of the total load, and together with the yellow areas show the locations contributing the top 40% of the load. In this catchment, only about 3.5% of the total catchment area generates the top 20% of the load and about 7% of the area generates the top 40% of the load. However, note that, in contrast to most of the preceding catchments, the hotspots have a distinctive pattern, not seen in the previous catchments. They are more dispersed and are mainly in relatively narrow bands in the floodplains adjacent to the streams. The paucity of hotspots in the upland areas may be characteristic of well-drained catchments such as this, but more catchments would have to be analysed to confirm whether or not this is a more general pattern.

7.4 Concluding Comments

A modelling methodology has been demonstrated that can help identify the areas that are hotspots for phosphorus export to channels. These can reduce the locations to be investigated for implementing specific measures. The model used for the demonstration was SWAT, but other semi-distributed water quality

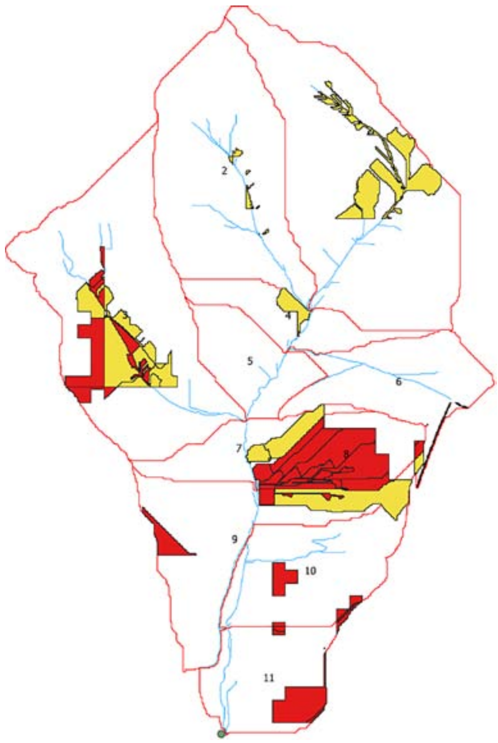


Figure 7.6. Askanagap SWAT-delineated streams and subcatchments showing the HRUs delivering highest 20% (red) and 40% (red + yellow) of loads in the Askanagap catchment.

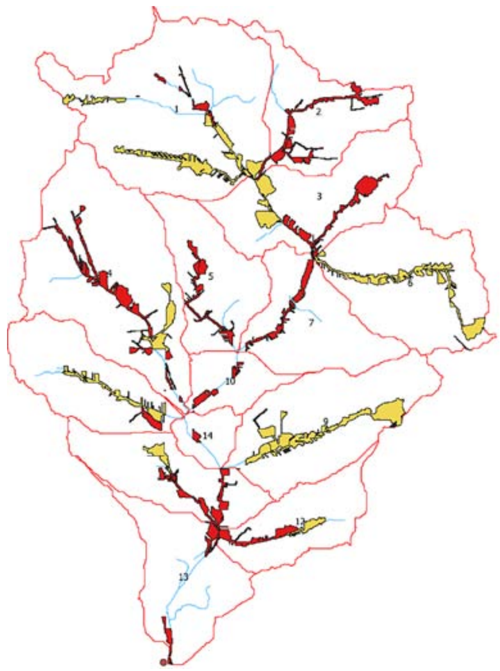


Figure 7.7. Owenboy SWAT-delineated streams and subcatchments showing the HRUs delivering highest 20% (red) and 40% (red + yellow) of loads in the Owenboy catchment.

catchment models could be expected to provide similar information. However, for use in small catchments, it is important that the model can operate without formal calibration, for instance by using readily available topography, soils and land use information, because the data needed for numerical calibration are rarely available for all such small catchments. While lack of calibration means that there is greater uncertainty about specific numerical estimates of loads, the identification of hotspot HRUs is likely to be more robust, as the relative comparisons of export to be expected from specific combinations of slopes, land use and soils are largely dependent on their mapping. For this demonstration with small catchments, the limitation of the resolution of these maps is apparent from the "blockiness" of some of the figures presented. It is apparent that quite a range of spatial patterns of export hotspots occur in the different SSNet catchment typologies, and this will have implications for the choice of suitable management measures. To illustrate the range of possibilities, Figure 7.8 shows the relationship between the cumulative percentage of catchment area and the percentage of cumulative load of soluble phosphorus they can produce for all six catchments. All the curves lie above the 1:1 line in the graph, indicating that, as might be expected, a smaller percentage of the area of a catchment may produce a larger percentage of the phosphorus load. However, there is a large range in the percentages involved. For instance, in the Pollanassa catchment, as little as 2% of the catchment can produce up to 20% of the load, while, for the Devlins catchment, the corresponding figure is 13% of the catchment area. In general, the more distant the cumulative area load curve is from the 1:1 line, the more concentrated are the hotspot areas. Conversely, the closer the curve is to the 1:1 line, the more distributed are the hotspot areas, implying greater complexity in managing the phosphorus loads. However, modelling results such as those demonstrated above can assist with this work, by showing potential locations of relative export hotspots. Note that we have not depended on model estimates of absolute amounts of exported phosphorus, as this would require considerable effort in data acquisition for model calibration and quantification of nutrient inputs, which may not be justified as a general procedure for all small catchments. The approach shown here is a relatively simple model-based spatial comparison of factors relating to risk of phosphorus export that can be implemented with much less effort to identify areas for further, more detailed, examination. A similar methodology can be applied to identify potential hotspots for the export of sediment and chemicals.

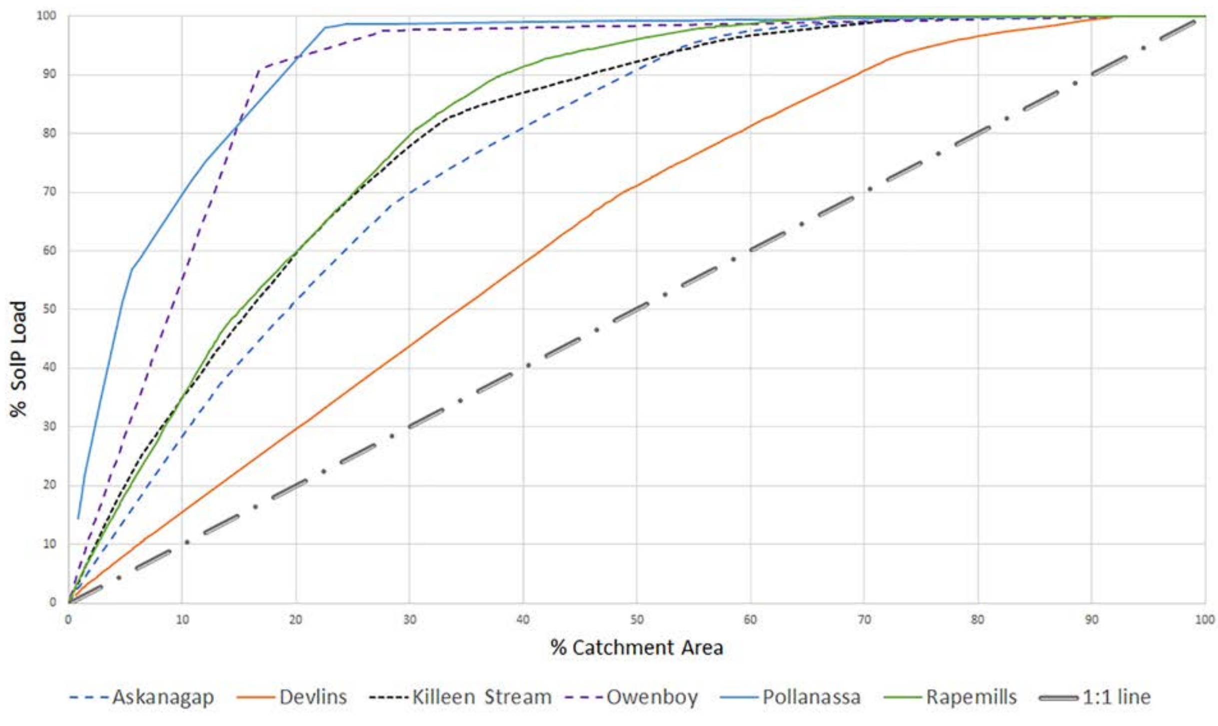


Figure 7.8. Cumulative area: load relationships. SolP, soluble phosphorus.

8 Discussion and Recommendations

The overall goal of this project was to refocus attention on the small stream network in terms of management and policy. SSNet is the first large-scale research project on first- and second-order streams to be conducted in Ireland and undertook investigations spanning hydrochemistry and multiple ecological elements, as well as experimental work, giving insights into the likely impact of climate change stressors. Furthermore, it provided an opportunity to test emerging DNA-based tools.

8.1 Hydrochemical Investigations

Significant water quality problems are evident in the small stream network based on both existing data (from EPA monitoring) and newly collected data. For example, a substantial portion (57%) of the 199 monitored EPA small stream sites exceeded one or more of the MRP, TAN and TON nutrient thresholds: 9% of sites exceeded all three thresholds, 30% exceeded two thresholds and 19% exceeded one threshold. Many of these sites are therefore impacted by multiple nutrient stressors, requiring a multi-stressor perspective to inform identification of dominant stressors and their effective mitigation. Sixty-five per cent of the SSNet catchment types that displayed the highest percentage of non-compliance were limestone types (at 65%), followed in turn by igneous and sedimentary geology types (at 58% and 42%, respectively). Elevated phosphorus levels appeared to be the most widespread problem, with concentrations above the good status threshold at 77% of the non-compliant sites (section 3.7) (Hogan *et al*., 2023). While the situation is somewhat better in terms of TAN (62%) and TON (41%), there was still a substantial number of sites exhibiting persistent unsatisfactory concentrations. Some sites recorded peaks in MRP concentrations during the summer, presumably as a result of point source effluent discharges during a period of low dilution. This is consistent with other studies that cite wastewater as the main contributor to non-compliance through chronic discharge inputs during the growing season (Bowes *et al*., 2012, 2014; Melland *et al*., 2012; Ockenden *et al*., 2014). Peak concentrations at other sites were confined to winter months, when heavy rainfall increases diffuse pollution inputs. Several studies have pointed out that nutrient losses from agricultural run-off are a major issue, and reducing those losses is necessary for water quality improvements (Alexander *et al*., 2007; Roberts *et al*., 2017; Buskirk *et al*., 2020).

The findings from the intensive sampling (fortnightly and events) mirrored that of the EPA-monitored sites, highlighting rainfall-driven inputs of phosphorus and nitrogen (diffuse pollution), with peak concentrations occurring during the winter months at most sites. This resulted in maximum nutrient values that exceeded mean values by a factor of 10 in some cases. The event with the highest rainfall (from 45.1 to 52.1 mm) resulted in the highest mean values for TP, SRP and TAN at all sites. Peak nutrient concentrations were temporally variable across rainfall events; some peak concentrations in SRP, for example, occurred on the rising limb, with others occurring on the falling limb at the same site. Similarly, patterns were observed for the other three nutrients discussed, TP, TAN and TON, indicating both the in-channel rapid mobilisation of nutrients and overland inputs that take time to arrive. This aligns with findings of other studies (e.g. Bowes *et al*., 2005). Coupled with rain-driven inputs, the most nutrient-impaired intensive sites, Tolka and Clonshanbo, exhibited the highest peak concentrations of phosphorus and total ammonia during the summer months, indicating the presence of point discharges that potentially originate from domestic wastewater or septic tanks.

Nutrient uptake/assimilation is one of the key ecosystem services provided by small streams, as it controls nutrient export to downstream reaches and influences water quality in the wider catchment (Mulholland *et al*., 2008; Ferreira *et al*., 2022). Although nutrient retention in small streams has been widely studied in many countries (e.g. Weigelhofer, 2017), it remains a poorly researched topic in Ireland. The experiments carried out in this project were very much a pilot exercise, and the results pose many questions for further investigations. For TAN, the uptake length (S_w) varied between 152 m for the Glashaboy stream and 616 m for the Newtown Mount

Kennedy stream, with the longer uptake length in the Newtown Mount Kennedy stream reflecting greater discharge and/or depth, both of which would tend to increase S_w, compromising efforts to make direct comparisons. The uptake lengths for MRP varied from 386 m for the Glashaboy stream to 544 m for the Newtown Mount Kennedy stream, again being strongly influenced by the greater discharge. However, in contrast to TAN, where uptake rates were quite similar (10.1 and 13.0 mg/m^2/day, respectively), the values for MRP varied from 11.1 mg/m^2/day for the Glashaboy stream to 35.7 mg/m^2/day for the Newtown Mount Kennedy stream. The higher background MRP concentration in the latter contributed to this difference. Variability in uptake rates among small streams with different nutrient conditions is not uncommon due to various biotic (bacteria, fungi, biofilms, macrophytes that facilitate nutrient uptake) and abiotic (channel size, surface area to channel volume ratio, flow conditions, etc.) conditions (Finkler *et al*., 2021). However, the literature generally highlights that nutrient retention capacity in headwaters is compromised by excess nutrient loading due to potential nutrient saturation and reduced hyporheic water exchange (e.g. Weigelhofer *et al*., 2018). Furthermore, higher velocities reduce the potential for nutrient uptake, as demonstrated in the nutrient spiralling experiments. This is a key point, as phosphorus inputs from agricultural land increase substantially during flood events, as reported in this project. Further research is needed to improve understanding of the multiple abiotic and biotic factors that influence the various uptake and retention processes in headwater streams (Weigelhofer *et al*., 2018), especially during high-flow conditions (e.g. Newcomer-Johnson *et al*., 2016), to better inform sustainable management of river systems. Developing linkages between the biotic factors and nutrient uptake metrics should be a particular focus. This can help restoration design to optimise the nutrient retention capacity of small streams (Ruzhong *et al*., 2020).

The high-frequency sampling during storm events at three catchments showed that peak concentrations of TP, MRP and TAN during intermittent storms can greatly exceed concentrations measured at fortnightly intervals. Because of the combination of higher concentrations and flows, even for relatively short durations, these events can provide a significant proportion of the total load. This suggests that even regular fortnightly sampling programmes are likely to underestimate loads. This is particularly important in small catchments with typically shorter storm durations, and has implications for reporting annual loads. In contrast, there is some indication that TON loads may be underestimated due to dilution during storm events, as the concentrations in higher flows may be less than those suggested by regular fortnightly sampling. However, more research is needed to assess whether this can be generalised for different catchment types, as multiple possible pathways are likely to be involved. These events also suggest that the dominant nutrient input to small streams, other than anthropogenic point discharges (e.g. wastewater), were diffuse sources, making mitigation more challenging.

8.2 Hydromorphology

The sedimentary structure, morphology and dynamics (hydromorphology) of stream and river reaches create a range of habitats for aquatic organisms at various stages in their life cycle. Hydromorphology is also a key element in the assessment of waterbody status and requires an understanding of the reference condition. The revised river type classification derived in this project provides a basis for defining “reference” physical habitat assemblages and judging the degree of degradation in the physical habitat condition of all river and stream geomorphological types (A to N; Figure 4.1) in the context of Ireland and other countries of similar landscape characteristics across northern and western Europe. Although the focus in this report has been on the six steepest river types (A to F), it is important to stress that both riparian and aquatic vegetation structure become increasingly important components of the habitat mosaic in the lower gradient stream types (G to N) and are major controls on channel morphology and its dynamics in the lowest gradient streams. Although vegetation was relatively invariant in the investigated steep streams, the river typing and physical habitat characterisation methods proposed herein incorporate fine detail on vegetation structure that can be extracted and incorporated into analysis of these lower gradient types.

Despite the sizable datasets used in our top-down “framework” approach in the present analysis, further independent testing should continue to further validate our outcomes on an increasingly large sample of sites. The previous river type classification is undergoing such testing as per the Biodiversity Metric 3.0 (Panks *et al*., 2022), and its River Condition Assessment (Gurnell *et al*., 2020) is applied by numerous professional river scientists across England. The English focus inevitably means that lower energy river types are the focus of this testing. Furthermore, of the river types tested in our analysis of Irish headwater streams (A, B, C, D, E, F, H, J and M), three were represented by very small numbers of sites (type B – 2, type J – 4, type M – 1), and, although types J and M have been robustly investigated in England, type B is a completely new river type that needs further investigation.

8.3 Ecology

The investigations on the macroinvertebrate fauna highlighted a number of key findings that have implications for both biodiversity protection and water quality monitoring. The 42 stream sites surveyed as part of the extensive survey recorded 144 taxa, a slightly higher per site number than that recorded by Callanan *et al*. (2014) (174 across 74 sites). While a high proportion of the taxa are common to both studies, each has a number of unique records. When the two datasets are combined, the total is just under 200 taxa. As noted previously, this figure would be much larger if the Chironomidae and other Diptera were identified to species level. Nevertheless, the biodiversity importance of the relatively small number of sites in this project is clear for the other groups. For example, 17 of the 18 stonefly species in Ireland were recorded and 17 of the 32 riverine Ephemeroptera species (19 by Callanan *et al*., 2012) were detected, as well as 38 of the 115 Trichoptera species inhabiting flowing water (Feeley *et al*., 2016, 2020). These streams are important sources of species that restore ecological health to streams, and especially in reaches further downstream once pollutants and other anthropogenic pressures have been removed.

The findings also draw attention to the challenge for freshwater biodiversity protection, as few species are common to most streams. In fact, only 23 (16%) macroinvertebrate species were found at more than 50% of sites, and 80 taxa had fewer than 10 site records, of which 38 (26%) occurred at a single site. When only insects are considered, 45% of the species occur at less than 5% of sites, similar to the figure of 46% reported by Kelly-Quinn *et al*. (2002) from 62 stream and river sites. The results for the phytobenthos and macrophytes also highlight low site occurrences of most species. The high heterogeneity in stream and site characteristics promotes heterogeneity in community composition and structure, and collectively contributes to high regional biodiversity (Finn *et al*., 2011). Kelly-Quinn *et al*. (2020) reported that total regional macroinvertebrate diversity (in terms of total species recorded) declined with increasing agricultural intensity, which may, in part, be caused by reduced habitat heterogeneity across sites due to siltation or hydromorphological degradation in combination with unsuitable hydrochemical conditions. This was also reinforced in the grouped site study on two subcatchments. Here, again, some of the tributaries differed in community structure, and *c*.20–29% of the taxa occurred at a single site only. The macroinvertebrate communities of the tributaries appear to be structured by different environmental factors depending on the spatial scale. At the tributary scale, there was a strong water chemistry gradient – with differences in pH and conductivity strongly correlated with differences in tributary macroinvertebrate community. At the site scale, there is a strong hydromorphological gradient – with differences in stream width and average bed material particle size strongly correlated with the ordination configuration of the sites within the individual tributaries. Overall, the results highlight the need to consider networks of protected sites for effective freshwater biodiversity protection.

The results for the summer macroinvertebrate data highlight a challenge for water quality rating in that the Q-value in its present form does not always return the expected value for a reference river. If ecological status is to be determined for small stream sites, modification of the Q-value may be required for assessments undertaken during the summer months.

The various analyses on potential stream types returned statistical differences in macroinvertebrate communities across the various SSNet types and types defined by geology and physiography. However, the pairwise test detected significant differences between site groups based on geology only. This may in part reflect the low replication in some of the 13 SSNet types. The analysis also highlights the importance of substrate, which is, in part, determined by geology and gradient. A relatively similar result was obtained for the phytobenthos and diatoms, where communities were significantly different between geology and physiography classes and SSNet types. Therefore, a combination of geology and physiography (capturing altitude and gradient/flow and associated substrate types) best describes the factors governing freshwater communities in small streams, a conclusion that was also reached by Callanan *et al*. (2012) in defining four stream groups.

The importance of small streams for salmonid production is well reported, especially in terms of providing spawning and nursery habitat (Kelly-Quinn *et al*., 1996). These authors emphasised that, although the numbers of trout in some upland stream sites can be relatively small, the overall numbers in the extensive small stream network can be quite substantial, and trout will exploit the habitat to the limit of their environmental tolerances. This further emphasises the importance of protecting the network of small streams. This project focused fish studies on understudied, but numerous, small coastal streams, and was very much a pilot exercise. All east coast streams and four of the six electrofished streams on the west coast harboured brown trout. Sea trout, identified from the silvery appearance, were detected in three east coast and two west coast streams. However, colouration is very plastic in brown trout, and returning sea-running trout can quickly change their appearance to be indistinguishable from resident brown trout. Hence, while the presence of silvery brown trout can be taken as evidence of sea trout presence, absence of silvery brown trout cannot be used as evidence of absence of sea trout. However, in general, the east coast streams tended to support more sea trout. East coast streams also tended to be larger when using stream width as a proxy for stream size (the average width of east coast streams was 3.7 m, while west coast streams were on average 2.2 m wide). In addition, several target streams in the west were too small or lacked running water, preventing meaningful surveys of fish fauna. It is likely that the production of brown trout is higher among the east coast streams than among west coast streams, as evidenced from the average densities of brown trout among the east and west coast streams (40.7 and 32.2 brown trout per 100 m^2, respectively). Most sites supported predominantly two cohorts, fish of the year and 1+ fish, but some east coast sites had some larger fish, and this most likely indicates a higher presence of resident brown trout than in the west coast streams. Furthermore, one narrow stream (W11 – 1.5 m) showed a skewed age distribution, with a high proportion of 0+ fish. This stream section is probably unsuitable for resident brown trout, but the high proportion of 0+ fish indicates that small coastal streams are likely to play a vital, but unrecognised, role in sea trout spawning and nursery.

Ensuring access to and from the sea is essential to maintain the potential contribution of small coastal streams to sea trout production. Flounder was detected in all east coast streams but only in two west coast streams, and three-spined stickleback were found at most sites (four east coast and four west coast streams). The life cycle of three-spined stickleback can be completed fully within fresh water and will not indicate the accessibility of habitats for anadromous species. The European eel, present in all fished sites, however, is catadromous and needs access to the sea to complete its life cycle. European eels are renowned for being able to pass obstacles in rivers and, hence, are probably more adapted than even sea trout to accessing upstream habitats. The most interesting species encountered in this study, in relation to stream accessibility, is the flounder (detected in all east coast and two west coast streams), as flounder spawn at sea and often migrate up into coastal streams. Thus, if flounder occur, obstacles for sea trout are likely to be present. This corresponds well to our findings that most streams supporting flounder also supported sea trout (four east coast and two west coast streams).

The Exstream field experiment highlighted siltation as a key stressor on small stream functioning, impacting different organism groups and the whole ecosystem. This is in line with previous findings, both in Ireland (García-Molinos and Donohue, 2009; Cocchiglia *et al*., 2012; Conroy *et al*., 2016) and internationally (Wood and Armitage, 1997; Kemp *et al*., 2011), with some studies going as far as to consider it a "master stressor" (Davis *et al*., 2018; Blöcher *et al*., 2020). Furthermore, fine excess sediments can impact nutrient uptake by restricting access to hyporheic water exchange (Weigelhofer *et al*., 2018). In light of this, continuous efforts are necessary to prevent land erosion and preserve structural integrity of riparian vegetated buffers, which can effectively intercept sediment (Feld *et al*., 2018). This is particularly important in the small stream network, where there is high land–water contact. Although flow variability was a lesser influence, it still negatively affected the invertebrate community and moderated the effects of other stressors, particularly during low-flow episodes. While short-term fluctuations are difficult to predict, seasonal and annual flow ranges are expected to become exacerbated by climate change (Gudmundsson *et al*., 2021), with the widest ranges predicted in smaller catchments (Murphy *et al*., 2023). At a catchment scale, river flow fluctuations can be magnified by channel modifications, such as arterial drainage, and loss of wetland storage in the headwaters, for example through peatland drainage. Conversely, the fluctuating amplitude and duration can be decreased by catchment storage and the slowing down of flow. This is for its own benefits, as well as moderating impact on other stressors. More broadly, hydromorphological modifications are a key stressor in Ireland, affecting about one-third of all monitored rivers (EPA, 2022). Both sediment and flow regime are likely contributing factors.

8.4 Modelling to Identify Areas of High Pollution Risk in Small Stream Catchments to Inform Mitigation

A catchment modelling methodology was demonstrated that can help identify general areas at the highest risk of nutrient or sediment export from headwater catchments, and be applied to small catchments. It can use any semi-distributed or distributed model for which parameters can be linked to mapped information, such as soil type, land use and topography. The initial use of a database for determining many of the model parameters is both a strength and a weakness of such models. Its strength is that the model can be easily applied without the need for discharge or water quality time series data for calibration. However, a weakness is uncertainty about the appropriateness of the parameters when the model is applied in a new region. While the methodology can be used with a number of different models, it was demonstrated here with the widely used SWAT model. This showed a wide range in behaviour for the different catchment types used, both in the distribution of the hotspot areas and in the percentage of the catchment area they represent. This has implications for how further, more detailed investigations can be undertaken and for the selection of appropriate measures. For instance, some catchments had hotspots grouped into a small number of individual semi-contiguous areas (particularly the Rapemills and Pollanassa catchments), for which measures that can reduce nutrient and sediment amounts mobilised, and/or intercept delivery to streams, may be considered. Others show a more distributed pattern. For instance, the hotspots in the Owenboy catchment in particular are distributed in ribbon-like patterns close to the channels. For such a pattern, interception measures would be more expensive to implement, as they would have to be more widely distributed. Other catchments fall between these two extremes in terms of the spatial distribution of hotspots. The percentage area of catchment hotspots associated with a given percentage of load also varies widely. While in all catchments the percentage area in the hotspots is always less than the percentage load exported from them, the individual catchments differ considerably in terms of the amounts. For instance, in catchments such as Pollanassa or Owenboy, less than 5% of the hotspot areas can deliver 20% of the phosphorus load, while, for Devlins, it is 13% of the area. However, there are issues with this type of approach. While we have shown the model's use for identifying hotspots, further uses, for instance to reliably estimate specific numerical values for loads or concentrations in small headwater catchments, would require model calibration, regardless of the model used. This requires high temporal resolution information on flows, nutrient concentrations and amounts, and timing of nutrients spread on land. It is accepted that it is impracticable to do this for all small catchments, but a campaign that produced such information for some catchments representing each of the major SSNet types would allow any numerical model being considered to be tested systematically for appropriateness for a wide range of Irish conditions. Teagasc's Agricultural Catchments Programme has done this for grassland catchments in lowland settings. Generating similar information from other catchment typologies for model calibration and validation would increase confidence in the more widespread use of the modelling approach.

The underrepresentation of first- and second-order streams in the EPA surveillance monitoring programme curtails identification of degraded streams and protection of biodiversity. This deficit is, to some extent, being addressed by LAWPRO scientists, but needs to be greatly extended. The extensive length of the small stream network limits what can be covered by the EPA and LAWPRO. There is great potential to fill the gaps by engaging citizen science in biomonitoring of small streams, which will not only generate data but will also raise awareness of water quality issues at the local level. Two citizen science indices are available: the Citizen Science Stream Index, based on six macroinvertebrate indicators, and the Small Stream Impact Score, based on a larger number of indicators in five macroinvertebrate groups. Several training workshops have been undertaken with volunteers to date and are ongoing. As noted by Kelly-Quinn *et al*. (2022b) in a paper prepared during the SSNet project, effective and sustained citizen science requires national coordination and the establishment of local hubs of trainers, as well as procedures to address data capture, validation, storage, analysis and communication. The authors produced a framework for establishing and operationalising citizen science projects that captures these and other essential elements. In summary, protection of the water quality, biodiversity and ecosystem services of the small stream network requires a combined top-down (policy and management) and bottom-up (community and individual) effort.

8.5 Recommendations

- Small streams lie outside the scope of the WFD but should be given higher visibility in river basin management planning.
- Diffuse agricultural and point source inputs are co-occurring pressures on the small stream network that can extend downstream. Water quality improvements require that wastewater entering these streams is properly treated; urban and domestic wastewater facilities are of the highest standard; soil erosion is minimised; and riparian buffer zones are established and retained, especially in areas of preferential flow, in addition to fertiliser and manure management measures.
- Research is required to determine the impact of other pollutants on small streams, in particular pesticides from agricultural and forestry operations.
- To cover the predominant SSNet catchment types highlighted in this report, the number of small stream sites in the EPA surveillance monitoring programme should be increased. At present, only *c.*10% of surveillance monitoring sites are on small streams. This increase should be in combination with extensions to the small stream investigations of LAWPRO. Small coastal streams should also be incorporated into water quality investigations. The Environment Agency in the UK is designing and testing monitoring and condition assessment methods for the small stream network.
- Further independent testing of the river classification derived from the 42 sites should continue to validate our outcomes on an increasingly larger sample of sites.
- Excess fine sediment poses a high threat to small streams, given their high land–water contact, and this is likely to be exacerbated by more intensive rainfall events due to climate change. Mitigation of this pressure needs to be more widely addressed in the small stream network.
- Standardised methods for monitoring deposited sediment need to be established together with metrics to measure deviation from reference deposited sediment levels.
- Research is required to provide further insights into the nutrient uptake capacity of small streams, factors influencing that capacity and how restoration efforts may optimise nutrient retention capacity.
- Although small streams may have relatively low biodiversity at site level, compared with some mid-order rivers, their communities are more heterogeneous across tributaries and within tributaries, as demonstrated by SSNet, and thus important in terms of their collective or regional biodiversity. Therefore, assessment of small stream biodiversity should take a network perspective.
- Small streams originating in areas with high regional biodiversity should be identified and given priority for monitoring and protection measures. One such area highlighted by Feeley and Kelly-Quinn (2012b) is the Slieve Bloom Mountains. These streams are strategically important as sources of fauna for ecological restoration of rivers in the midlands when pollution pressures have been mitigated.
- Consider how biodiversity metrics may be incorporated into water quality monitoring programmes. Biodiversity monitoring was the core theme of the Biodiversa 2022 call. Genetic methods, as demonstrated in this project, have the potential to address this challenge but will

require the population of genetic libraries for key biodiversity indicators.

- Due to the large numbers and diverse locations of small catchments and an increased monitoring requirement, there is potential to capitalise on the capacity of citizen science to contribute to monitoring water quality in small streams. This will not only contribute data, but will increase awareness in local communities. It will, however, require coordination and support, as outlined by Kelly-Quinn *et al*. (2022b).
- Further fisheries investigations on small streams are necessary to quantify their contribution to sea trout fisheries and their sensitivity to variable flow imposed by climate change.
- Collect the data needed for formal calibration of the small stream catchment modelling methodology presented here. This will require intensive sampling at a number of different catchment types to allow better generalisation nationally.
- Examine the benefits that might be expected from using improved resolution mapping and information about soils and land use, particularly for modelling small headwater catchments.

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Abbreviations

ASV	Amplicon sequence variant
AV	Area velocity
BSC	Bootstrap confidence
Corine	Coordination of Information on the Environment
df	Degrees of freedom
DO	Dissolved oxygen
DTM	Digital terrain model
eDNA	Environmental DNA
EPT	Ephemeroptera, Plecoptera, Trichoptera
GIS	Geographic information system
HRU	Hydrological response unit
LAWPRO	Local Authority Water Programme
m.a.s.l.	Metres above sea level
MoRPh5	Modular River Physical Survey
MRP	Molybdate-reactive phosphorus
nMDS	Non-metric multi-dimensional scaling
PCR	Polymerase chain reaction
PERMANOVA	Permutational multivariate analysis of variance
SD	Standard deviation
SRP	Soluble reactive phosphorus
SSNet	Small Stream Network
SWAT	Soil Water Assessment Tool
TAN	Total ammonia nitrogen
TC	Total carbon
TON	Total oxidised nitrogen
TP	Total phosphorus
TRP	Total reactive phosphorus
TWINSPAN	Two-way indicator species analysis
UN	United Nations
WFD	Water Framework Directive

Appendix 1 Characteristics of the 42 Extensive Study Sites (Included are the TWINSPAN Group Assigned)

Site code	Latitude	Longitude	River name	Wetted width (m)	Depth (m)	Flow mesohabitats						
						Cascade	Chute	Riffle	Run	Glide	Pool	Other
GLX_cah	53.099166	–9.236118	Caher River	5.72	0.25	3.3	6.3	3.3	4.3	7.3	68.0	7.3
GLX_kil	53.417422	–8.203321	Killegan River	4.76	1.23	0.0	0.0	0.0	0.0	85.3	10.7	4.0
IHPe_cre	54.028342	–9.790461	Creggan	0.96	0.18	3.3	4.3	23.3	7.0	7.7	48.0	6.3
IHPe_doi	54.916801	–8.218568	Doire Na Coradh	2.16	0.59	0.0	2.0	0.7	2.3	4.3	88.7	2.0
IHPe_t_gwe	54.91724	–8.148584	Gweebarra River	1.06	0.17	37.0	4.7	4.0	13.0	27.3	9.3	4.7
IHPe_t_por	54.325039	–9.807669	Portacloy Bay Inlet	1.95	0.19	10.0	2.7	11.3	12.3	35.7	19.0	9.0
IHPo_sto	54.246098	–8.311705	Stonepark Inlet	0.78	0.05	7.0	10.7	26.7	19.0	22.3	10.0	4.3
IMPe_avo	53.138489	–6.299809	Avonmore	1.87	0.16	12.3	15.0	11.3	22.3	23.0	9.0	7.0
IMPe_gwe	54.979173	–8.05726	Gweebarra River	2.82	0.40	24.0	4.3	18.3	3.7	22.3	17.0	10.3
IMPe_t_lif	53.16543	–6.316721	River Liffey	2.19	0.21	28.0	5.0	4.0	12.0	25.3	17.7	8.0
IMPe_t_owe	53.505152	–9.870121	Owenglin River	1.24	0.12	5.0	4.7	13.3	3.7	6.3	59.0	8.0
IMPo_balm	54.000238	–9.353411	Ballynagoraher River	1.53	0.15	25.3	4.0	12.3	25.0	9.0	13.0	11.3
IMPo_balw	53.09605	–6.431896	Ballinagee River	2.60	0.17	22.3	5.0	10.0	14.0	5.7	34.0	9.0
IMPo_let	54.0238	–9.611855	Altaconey	3.04	0.24	31.7	7.3	9.0	27.0	1.7	13.3	10.0
IMPo_t_lif	53.112109	–6.475987	Fraughan Brook	2.18	0.19	23.0	8.0	5.3	17.3	29.3	9.7	7.3
LHPe_eas	54.188718	–8.920181	Easky River	1.47	0.17	0.0	5.0	8.3	8.0	60.3	12.3	6.0
LHPe_spa	53.916928	–8.924909	Spaddagh River	2.29	0.14	1.0	2.3	16.7	15.7	48.3	10.7	5.3
LHPe_t_owe	54.129575	–9.637832	Owenmore River	1.09	0.12	5.7	4.7	28.0	15.3	19.7	20.7	6.0
LHPe_wee	54.16688	–8.936924	Gowlan River	2.06	0.16	0.0	0.0	1.0	2.3	65.0	26.7	5.0
LHPo_cas	54.191093	–8.232033	Cashel stream	1.88	0.13	8.3	4.3	10.7	10.7	34.0	27.3	4.7
LHPo_esh	54.308863	–7.103574	Scotstown stream	2.88	0.13	0.0	8.0	19.7	11.3	44.7	11.0	5.3
LHPo_swi	53.913115	–8.862602	Swinford River	0.91	0.21	0.0	0.0	0.0	0.0	76.3	16.7	7.0
LHPo_t_lei	54.663706	–8.160676	Leitrim Hill stream	2.37	0.09	13.7	4.7	1.3	8.3	38.3	28.0	5.7
LHPo_t_woo	54.111298	–7.61107	Woodford River	4.47	0.21	0.0	5.0	19.0	3.0	55.3	11.7	6.0
LPPe_duv	54.178522	–9.411892	Duvowen River	1.71	0.16	0.0	1.3	38.0	13.0	26.3	15.0	6.3
SHPo_kil	51.577642	–9.724637	Kileen South	1.29	0.13	25.7	4.0	0.0	8.0	26.0	28.3	8.0
SHPo_t_aug	51.575073	–9.738337	Aughaleigue More	1.42	0.15	12.0	11.7	1.3	12.3	14.3	40.0	8.3
SHPo_t_toor	51.52436	–9.636638	Toormore Bay Inlet	1.67	0.08	16.7	4.3	9.7	2.3	46.3	16.7	4.0
SHPo_var	53.117563	–6.179115	Vartry River	2.37	0.19	0.0	2.0	39.7	25.7	21.7	6.0	5.0
SMPe_cum	51.953931	–9.698415	Cummeralooderry stream	2.98	0.15	28.3	8.0	5.0	9.0	16.3	24.3	9.0
SMPe_owe	51.982176	–9.516751	Owengarriff River	4.54	0.15	50.3	7.0	4.3	6.3	4.7	17.7	9.7
SMPe_owg	52.160662	–10.118436	Owenladondrig River	2.59	0.22	9.3	7.0	0.0	11.7	27.7	36.3	8.0
SMPe_t_kno	52.221537	–9.876821	Knockglass Beg	1.29	0.11	20.3	3.3	0.0	8.0	11.3	48.0	9.0
SMPo_coo	51.753564	–9.570879	Glengarrif River	2.02	0.24	0.0	0.0	0.0	2.0	35.0	57.3	5.7
SMPo_t_bla	51.91213	–9.807395	River Blackwater	1.46	0.10	24.0	9.3	12.3	11.3	6.0	27.0	10.0
SMPo_t_car	53.683982	–9.773989	Carrowniskey River	1.70	0.30	18.0	7.0	6.7	9.7	2.3	48.0	8.3
SMPo_t_gad	52.025814	–9.735635	Gaddagh River	2.51	0.18	18.0	7.3	12.3	12.0	20.7	22.0	7.7
SMPo_t_owb	51.94625	–9.488487	Owbaun River	2.51	0.24	9.0	4.7	12.7	18.3	32.0	15.3	8.0
SMW_awb	51.976283	–8.972908	Awboy River	3.95	0.30	0.0	3.7	27.7	19.3	37.0	8.0	4.3
SMW_bal	51.988328	–8.042892	Ballyeightragh	2.45	0.11	2.3	5.0	40.0	12.7	25.7	5.3	9.0
SMW_lis	52.198791	–8.122335	River Araglin	2.88	0.57	26.3	5.7	18.3	20.7	13.0	9.0	7.0
SMW_t_gle	53.648847	–9.733475	Glenummera River	3.61	0.28	16.7	7.7	2.3	2.7	8.0	47.0	15.7

"Other" flow mesohabitats include boils, waterfalls, backwaters and slackwater.

Substrate								Shading			
Bedrock	Boulder	Cobble	Gravel-pebble	Sand	Silt	Clay	Organic/peat	Broken	Dense	Slope class	TWINSPAN group
5.3	77.7	15.0	1.3	0.7	0.0	0.0	0.0	15	10	Low	1
0.0	1.3	4.0	23.3	71.3	0.0	0.0	0.0	10	0	Very low	4
5.3	69.0	18.0	0.7	0.0	0.0	0.7	6.3	50	25	Low	6
43.0	21.7	2.7	0.3	0.0	0.0	0.0	32.3	20	20	Low	6
13.7	27.7	8.3	1.3	1.0	0.0	0.0	1.0	30	20	Very steep	6
6.0	50.0	38.3	0.0	0.0	0.0	0.0	0.0	10	0	Moderate	5
9.7	45.3	39.7	4.0	1.3	0.0	0.0	0.0	60	20	Steep	2
23.0	50.3	23.0	3.0	0.7	0.0	0.0	0.0	15	2	Steep	5
8.7	14.3	16.0	0.7	0.3	0.0	0.0	0.0	5	0	Very steep	1
26.0	22.0	20.0	2.0	0.0	0.0	0.0	0.0	10	0	Very steep	5
3.0	48.3	45.7	1.7	0.0	0.0	0.0	1.3	25	6	Low	2
3.3	16.3	15.0	0.0	0.0	0.0	0.0	0.0	20	3	Extremely steep	5
10.3	14.3	16.0	3.0	0.0	0.0	0.0	0.0	10	0	Very steep	5
19.3	60.7	16.7	1.0	0.0	0.0	0.0	0.0	5	0	Very steep	1
29.3	37.7	9.3	3.3	0.0	0.0	0.0	0.0	7	0	Extremely steep	5
6.0	37.0	49.7	6.7	0.0	0.0	0.0	0.7	25	10	Very low	1
2.3	34.7	52.7	6.0	4.3	0.0	0.0	0.0	10	5	Low	1
3.3	66.3	26.7	3.7	0.0	0.0	0.0	0.0	8	5	Moderate	3
4.3	47.0	32.3	8.0	2.7	0.0	5.7	0.0	15	0	Very low	1
10.3	49.3	31.0	6.0	3.3	0.0	0.0	0.0	50	20	Moderate	3
14.0	52.3	28.0	2.0	2.0	1.7	0.0	0.0	60	30	Moderate	1
9.7	33.3	3.0	46.0	6.3	0.0	1.7	0.0	30	25	Very low	1
7.3	20.7	10.3	2.3	4.3	0.3	0.0	0.0	40	40	Low	1
3.0	49.7	44.3	2.3	0.0	0.7	0.0	0.0	60	30	Moderate	1
1.0	47.0	46.7	5.3	0.0	0.0	0.0	0.0	9	0	Very low	1
16.3	33.7	10.0	0.0	0.0	0.0	0.0	0.0	10	5	Very steep	2
56.0	20.3	12.3	0.0	0.0	0.0	0.0	0.0	40	40	Extremely steep	2
4.7	23.3	45.3	2.7	3.0	0.0	0.0	0.0	50	40	Steep	2
0.0	35.7	56.7	4.0	3.7	0.0	0.0	0.0	25	5	Very low	1
38.7	37.0	13.3	0.0	0.0	0.0	0.0	0.0	5	0	Steep	5
52.3	31.7	16.0	0.0	0.0	0.0	0.0	0.0	5	0	Extremely steep	3
5.3	41.0	14.7	0.7	0.0	0.0	0.0	0.0	7	0	Steep	1
21.0	57.3	21.7	0.0	0.0	0.0	0.0	0.0	5	0	Very steep	3
0.0	2.3	51.0	33.3	13.3	0.0	0.0	0.0	60	30	Low	6
39.7	41.0	17.7	0.7	0.0	1.0	0.0	0.0	5	0	Extremely steep	2
15.3	37.7	15.7	1.0	0.3	0.0	0.0	0.0	15	10	Very steep	3
11.0	67.7	19.0	0.0	0.0	0.0	0.0	0.0	12	0	Very steep	1
19.3	55.7	25.0	0.0	0.0	0.0	0.0	0.0	7	3	Very steep	3
6.0	67.0	21.3	3.0	2.7	0.0	0.0	0.0	50	10	Low	1
2.0	19.3	58.3	14.3	6.0	0.0	0.0	0.0	40	37	Low	1
9.3	40.3	29.3	6.7	1.7	0.0	0.0	0.0	70	10	Very steep	1
10.0	27.7	11.7	0.7	0.0	0.0	0.0	0.0	3	0	Extremely steep	3

An Ghníomhaireacht Um Chaomhnú Comhshaoil

Tá an GCC freagrach as an gcomhshaol a chosaint agus a fheabhsú, mar shócmhainn luachmhar do mhuintir na hÉireann. Táimid tiomanta do dhaoine agus don chomhshaol a chosaint ar thionchar díobhálach na radaíochta agus an truaillithe.

Is féidir obair na Gníomhaireachta a roinnt ina trí phríomhréimse:

Rialáil: Rialáil agus córais chomhlíonta comhshaoil éifeachtacha a chur i bhfeidhm, chun dea-thorthaí comhshaoil a bhaint amach agus díriú orthu siúd nach mbíonn ag cloí leo.

Eolas: Sonraí, eolas agus measúnú ardchaighdeáin, spriocdhírithe agus tráthúil a chur ar fáil i leith an chomhshaoil chun bonn eolais a chur faoin gcinnteoireacht.

Abhcóideacht: Ag obair le daoine eile ar son timpeallachta glaine, táirgiúla agus dea-chosanta agus ar son cleachtas inbhuanaithe i dtaobh an chomhshaoil.

I measc ár gcuid freagrachtaí tá:

Ceadúnú

> Gníomhaíochtaí tionscail, dramhaíola agus stórála peitril ar scála mór;
> Sceitheadh fuíolluisce uirbigh;
> Úsáid shrianta agus scaoileadh rialaithe Orgánach Géinmhodhnaithe;
> Foinsí radaíochta ianúcháin;
> Astaíochtaí gás ceaptha teasa ó thionscal agus ón eitlíocht trí Scéim an AE um Thrádáil Astaíochtaí.

Forfheidhmiú Náisiúnta i leith Cúrsaí Comhshaoil

> Iniúchadh agus cigireacht ar shaoráidí a bhfuil ceadúnas acu ón GCC;
> Cur i bhfeidhm an dea-chleachtais a stiúradh i ngníomhaíochtaí agus i saoráidí rialáilte;
> Maoirseacht a dhéanamh ar fhreagrachtaí an údaráis áitiúil as cosaint an chomhshaoil;
> Caighdeán an uisce óil phoiblí a rialáil agus údaruithe um sceitheadh fuíolluisce uirbigh a fhorfheidhmiú
> Caighdeán an uisce óil phoiblí agus phríobháidigh a mheasúnú agus tuairisciú air;
> Comhordú a dhéanamh ar líonra d'eagraíochtaí seirbhíse poiblí chun tacú le gníomhú i gcoinne coireachta comhshaoil;
> An dlí a chur orthu siúd a bhriseann dlí an chomhshaoil agus a dhéanann dochar don chomhshaol.

Bainistíocht Dramhaíola agus Ceimiceáin sa Chomhshaol

> Rialacháin dramhaíola a chur i bhfeidhm agus a fhorfheidhmiú lena n-áirítear saincheisteanna forfheidhmithe náisiúnta;
> Staitisticí dramhaíola náisiúnta a ullmhú agus a fhoilsiú chomh maith leis an bPlean Náisiúnta um Bainistíocht Dramhaíola Guaisí;
> An Clár Náisiúnta um Chosc Dramhaíola a fhorbairt agus a chur i bhfeidhm;
> Reachtaíocht ar rialú ceimiceán sa timpeallacht a chur i bhfeidhm agus tuairisciú ar an reachtaíocht sin.

Bainistíocht Uisce

> Plé le struchtúir náisiúnta agus réigiúnacha rialachais agus oibriúcháin chun an Chreat-treoir Uisce a chur i bhfeidhm;
> Monatóireacht, measúnú agus tuairisciú a dhéanamh ar chaighdeán aibhneacha, lochanna, uiscí idirchreasa agus cósta, uiscí snámha agus screamhuisce chomh maith le tomhas ar leibhéil uisce agus sreabhadh abhann.

Eolaíocht Aeráide & Athrú Aeráide

> Fardail agus réamh-mheastacháin a fhoilsiú um astaíochtaí gás ceaptha teasa na hÉireann;
> Rúnaíocht a chur ar fáil don Chomhairle Chomhairleach ar Athrú Aeráide agus tacaíocht a thabhairt don Idirphlé Náisiúnta ar Ghníomhú ar son na hAeráide;
> Tacú le gníomhaíochtaí forbartha Náisiúnta, AE agus NA um Eolaíocht agus Beartas Aeráide.

Monatóireacht & Measúnú ar an gComhshaol

> Córais náisiúnta um monatóireacht an chomhshaoil a cheapadh agus a chur i bhfeidhm: teicneolaíocht, bainistíocht sonraí, anailís agus réamhaisnéisiú;
> Tuairiscí ar Staid Thimpeallacht na hÉireann agus ar Tháscairí a chur ar fáil;
> Monatóireacht a dhéanamh ar chaighdeán an aeir agus Treoir an AE i leith Aeir Ghlain don Eoraip a chur i bhfeidhm chomh maith leis an gCoinbhinsiún ar Aerthruailliú Fadraoin Trasteorann, agus an Treoir i leith na Teorann Náisiúnta Astaíochtaí;
> Maoirseacht a dhéanamh ar chur i bhfeidhm na Treorach i leith Torainn Timpeallachta;
> Measúnú a dhéanamh ar thionchar pleananna agus clár beartaithe ar chomhshaol na hÉireann.

Taighde agus Forbairt Comhshaoil

> Comhordú a dhéanamh ar ghníomhaíochtaí taighde comhshaoil agus iad a mhaoiniú chun brú a aithint, bonn eolais a chur faoin mbeartas agus réitigh a chur ar fáil;
> Comhoibriú le gníomhaíocht náisiúnta agus AE um thaighde comhshaoil.

Cosaint Raideolaíoch

> Monatóireacht a dhéanamh ar leibhéil radaíochta agus nochtadh an phobail do radaíocht ianúcháin agus do réimsí leictreamaighnéadacha a mheas;
> 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 um chosaint ar an radaíocht a sholáthar, nó maoirsiú a dhéanamh ar sholáthar na seirbhísí sin.

Treoir, Ardú Feasachta agus Faisnéis Inrochtana

> Tuairisciú, comhairle agus treoir neamhspleách, fianaise-bhunaithe a chur ar fáil don Rialtas, don tionscal agus don phobal ar ábhair maidir le cosaint comhshaoil agus raideolaíoch;
> An nasc idir sláinte agus folláine, an geilleagar agus timpeallacht ghlan a chur chun cinn;
> Feasacht comhshaoil a chur chun cinn lena n-áirítear tacú le hiompraíocht um éifeachtúlacht acmhainní agus aistriú aeráide;
> Tástáil radóin a chur chun cinn i dtithe agus in ionaid oibre agus feabhsúchán a mholadh áit is gá.

Comhpháirtíocht agus Líonrú

> Oibriú le gníomhaireachtaí idirnáisiúnta agus náisiúnta, údaráis réigiúnacha agus áitiúla, eagraíochtaí neamhrialtais, comhlachtaí ionadaíocha agus ranna rialtais chun cosaint chomhshaoil agus raideolaíoch a chur ar fáil, chomh maith le taighde, comhordú agus cinnteoireacht bunaithe ar an eolaíocht.

Bainistíocht agus struchtúr na Gníomhaireachta um Chaomhnú Comhshaoil

Tá an GCC á bainistiú ag Bord lánaimseartha, ar a bhfuil Ard-Stiúrthóir agus cúigear Stiúrthóir. Déantar an obair ar fud cúig cinn d'Oifigí:

1. An Oifig um Inbhunaitheacht i leith Cúrsaí Comhshaoil
2. An Oifig Forfheidhmithe i leith Cúrsaí Comhshaoil
3. An Oifig um Fhianaise agus Measúnú
4. An Oifig um Chosaint ar Radaíocht agus Monatóireacht Comhshaoil
5. An Oifig Cumarsáide agus Seirbhísí Corparáideacha

Tugann coistí comhairleacha cabhair don Ghníomhaireacht 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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