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Biological Tools to Measure the Impact of Flow on Ecology in Irish Rivers

Authors: Martin Gammell, Heather T. Lally, Conor Graham, Lynda Weekes, Andrés Peredo Arce, Chris Westwood, Mike Dunbar and Chris Extence

Lead organisation: Atlantic Technological University

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Rialtas na hÉireann
Government of Ireland

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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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What did this research aim to address?

Flow is an important determinant of the biological community in a river, as different species are adapted to different flow velocities. Because human-driven factors, such as climate change, water abstraction and the construction of instream barriers, can alter river flows, thereby having negative impacts on river biota, it is important to be able to monitor and mitigate such impacts. There is an increasing number of biomonitoring tools available for measuring the effects of changes in flow on river biota. The main aim of this project was to investigate whether data gathered during biological monitoring programmes in Irish rivers could be used to measure impacts of changes in flow on ecology, and to determine whether hydroecological monitoring tools could provide useful additional information for assessing the ecological status of Irish rivers. In this project, the responses of macroinvertebrates, fish and macrophytes to changing river flows was investigated using biological survey data matched with river flow data from nearby hydrometric stations, and biotic indices for monitoring the impacts of changes in flow on Irish river biota were developed.

What did this research find?

Three macroinvertebrate hydroecological indices that were designed to measure the effects of changes in flow, drought and sedimentation in British rivers were tested, and two were adapted for Irish rivers. Scores calculated with the indices were correlated with river flows, and these indices could therefore provide useful information about flow, and some other hydromorphological conditions, in Irish rivers. No suitable fish-based flow index could be adapted to an Irish context using the available data, and no strong relationships were found between fish data and flow, although this may have been due to limitations with available data. A new macrophyte hydroecological index was developed, and was found to be suitable for characterising a combination of prevailing hydromorphological conditions, including flow, and could potentially be used for monitoring the effects of flow and assessing the lotic environment in Irish rivers. An appropriate method for generating a flow-based multi-metric index was identified and used in this project, and, with further work on larger and overlapping datasets for macroinvertebrates, macrophytes and fish, could lead to the development of a multi-metric index for Irish rivers.

How can the research findings be used?

Hydroecological monitoring tools should be incorporated into Irish river monitoring programmes to address national plans and legislation, such as the Water Action Plan 2024 and the Water Environment (Abstractions and Associated Impoundments) Act 2022. Macroinvertebrate hydroecological indices are likely to be most useful, while macrophyte indices also have potential. Further work should be carried out to test the overall performance of the macrophyte index, and to test the performance of the macroinvertebrate indices in different river types and different water quality. An online dashboard that can be used to calculate the macroinvertebrate indices was also developed as part of this project, and is available at https://mgammell.shinyapps.io/biotic_index_dashboard/. To improve data quality for future investigations of flow–ecology relationships, hydrological reference sites, with hydrometric gauges continuously measuring flow, should be maintained at appropriate locations that represent the full range of Irish river typologies without significant hydromorphological, water quality or abstraction pressures. Biological monitoring sites should be established as close as possible to all hydrological reference sites, and surveys for macroinvertebrates, macrophytes and fish should be carried out at all sites annually.

Project code: 2019-W-MS-45

EPA RESEARCH PROGRAMME 2021–2030

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EPA Research Report

Prepared for the Environmental Protection Agency

by

Atlantic Technological University

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This report is based on research carried out from 2020 to 2024 on data gathered from 2005 to 2022. 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

Although biological monitoring has not played a major role to date in the assessment of flow in Irish rivers, there is an increasing number of biomonitoring tools available for measuring the effects of changes in flow on river biota. The main aim of this project was to investigate whether data gathered during biological monitoring programmes in Irish rivers could be used to measure impacts of changes in flow on ecology, and to determine whether hydroecological monitoring tools could provide useful additional information for assessing the ecological status of Irish rivers. The biological data used in this project were from macrophyte surveys carried out in 2021 at 47 sites; macroinvertebrate surveys carried out in 2021 and 2022 at the same 47 sites; historical EPA macroinvertebrate surveys carried out between 2007 and 2018; historical macrophyte surveys from the River Macrophyte Database carried out between 1998 and 2015; and historical Inland Fisheries Ireland electrofishing surveys carried out between 2008 and 2020. The biological data were matched with hydrological data recorded at EPA and Office of Public Works hydrometric stations between 2005 and 2022.

Flow statistics calculated for Irish rivers were generally positive predictors of index scores calculated with three British hydroecological macroinvertebrate indices designed to measure the effects of changes in flow, drought and sedimentation, following some adjustments to take Irish conditions into account. These macroinvertebrate hydroecological indices could therefore provide useful information about current and antecedent flow conditions and some other hydromorphological conditions in Irish rivers. In most statistical models, there was a significant relationship between river type and index score, with higher index scores generally being associated with river sites from more western and elevated locations, providing further evidence that these indices reflect conditions in Irish rivers. In all statistical models, the effects of water quality on index scores were significant; therefore, water quality is a potential confounding factor to consider when investigating hydroecological relationships. Scores calculated using family-level versions of the indices were strongly correlated with scores calculated with versions that used higher resolution taxonomic data; the identification of macroinvertebrates to family level should therefore be sufficient for the application of these indices in Ireland.

No suitable fish-based flow index currently exists that could be tested in and/or adapted to an Irish context using the available data. Multivariate modelling of historical fish community data from Inland Fisheries Ireland electrofishing surveys found that only 1% of the variation in fish community data could be explained by flow events. Univariate modelling on the density of individual year classes of sensitive species (Atlantic Salmon, *Salmo salar*, and Brown Trout, *S. trutta*) found no relationship with calculated flow metrics. Rather, habitat structure (depth, canopy cover, distance to the sea and river substrate type) in individual survey sites explained up to 20% of the variation in sampled fish communities in these models. However, these results may have been affected by limitations with the available data.

The British macrophyte-based hydroecological index that was assessed in this study does not appear to be suitable for monitoring in Irish rivers in its present form for several reasons, including the exclusion of bryophytes from the metric, the use of indicator species that are not optimised for Irish flora and the differing climatic conditions between Britain and the island of Ireland. A new macrophyte hydroecological index was developed, the Macrophyte Morphological Traits Index. This was found to be suitable for characterising a combination of prevailing hydromorphological conditions, including flow, and could potentially be used for monitoring the effects of flow and assessing the lotic environment in Irish rivers.

An appropriate method for generating a flow-based multi-metric index for Ireland was identified and used in this study, and with further work this method could lead to the development of a suitable multi-metric index for Irish rivers. Factors that hindered the development of a multi-metric index in this study were the lack of overlapping monitoring sites for fish and macroinvertebrates close to hydrometric stations with good-quality continuous flow records, the low taxonomic resolution of the macroinvertebrate data and the lack of data on absolute abundances in the macroinvertebrate dataset.

1 Introduction

1.1 Background

The EU Water Framework Directive (WFD) (Directive 2000/60/EC) sets standards for water quality in Europe. Since its implementation, the WFD has led to an increased focus on the ecological status of waterbodies (e.g. Kelly and Harrison, 2016). The ecological status of a waterbody can be assessed using different elements that are linked to water quality, including biological elements, physico-chemical elements, specific pollutants and hydromorphology (Bowman, 2009). Hydromorphological quality elements include water flow, groundwater connectivity, river continuity, the underlying substrate and the structure of the channel and the riparian zone (Dunbar *et al*., 2010a; Chadd *et al*., 2017; Fleming *et al*., 2021). These hydromorphological quality elements should be directly related to biological quality elements, as they provide the physical conditions within which aquatic species live (Extence *et al*., 1999; IFI, 2013; van Oorschot *et al*., 2016). Many biological water monitoring programmes collect data on the presence and abundance of selected groups of organisms at survey sites and then use those data to assign a water quality-based ecological status to waterbodies (e.g. Bradley *et al*., 2013). However, it is now recognised that the physical condition (i.e. hydromorphology) of a waterbody is as important as water quality when considering ecological status (DHPLG, 2018). Water flow is an important determinant of the biological community at a site, while other morphological features, such as substrate composition and channel structure, also have an important role to play and may mediate the influence of flow on the biological community in various ways (Dunbar *et al*., 2010a; Klaar *et al*., 2014). For example, with declining flows, macroinvertebrate taxa associated with slower velocity habitats and silty conditions would be expected to increase in abundance, whereas taxa associated with faster velocities and coarser substrates would be expected to decrease in abundance (Extence *et al*., 1999).

A wide range of methods have been developed to assess the water quality of rivers using river biota, with a large number of methods focusing on how the presence (or absence) of particular taxa can provide information about the impacts of broad categories of pollutants, including organic and nutrient pollution (Bradley *et al*., 2013; WFD-UKTAG, 2014). In contrast, biotic monitoring has not played a major role (to date) in the assessment of river flow in Ireland, which is usually monitored by other means, such as measured flow dynamics (Bowman, 2009). However, there is a recognition of the need to improve assessments of flow and to incorporate ecology into those assessments (Webster *et al*., 2017; DHPLG, 2018). Some preliminary work on environmental flows in Irish rivers was carried out by Webster *et al*. (2017) and included a characterisation of river flow regimes in different landscapes and an assessment of abstraction pressures. This work identified a major knowledge gap with regard to using flow–ecology relationships for assessing risks associated with water abstraction (Webster *et al*., 2017).

Although numerous methods for assessing river flow have been developed, and are being used globally, the degree to which river biota are included in these methods varies (Tharme, 2003). However, in some areas there is a long history of considering river biota when managing environmental flows (e.g. in Australia (Arthington and Zalucki, 1998) and California (Stein *et al*., 2021)), and a substantial amount of work that is of potential relevance to Irish river monitoring programmes has been carried out in the United Kingdom (UK) to investigate whether river biota can provide useful information for monitoring the impacts of changes in flow, using macroinvertebrates in particular (e.g. Extence *et al*., 1999; Turley *et al*., 2016; Chadd *et al*., 2017) and macrophytes to a lesser extent (e.g. Westwood *et al*., 2021).

The main macroinvertebrate-based indices that have been developed in the UK use the common approach of classifying taxa based on their sensitivities to environmental pressures; in these cases, particular aspects of hydromorphological pressure, such as flow (e.g. Extence *et al*., 1999), drought (e.g. Chadd *et al*., 2017) and fine sediment deposition (e.g. Turley *et al*., 2016), are considered, and then an index score is calculated for a site and weighted according to the sensitivities of the taxa that are present. These macroinvertebrate indices have been found to respond to changes in flow, drought and fine sediments in British rivers (Extence *et al*., 1999; Turley *et al*., 2016; Chadd *et al*., 2017). Although a wide variety of macrophyte-based assessment methods are utilised under the WFD (Holmes *et al*., 1999; Szoszkiewicz, 2004; Haury *et al*., 2006; WFD-UKTAG, 2014), there are few known or established plant-based tools for assessing river flow. However, one such tool is the Plant Flow Index (PFI). This was developed in the UK (Westwood *et al*., 2021) to assess plant assemblage responses to changing discharge (flow volume per unit time) in temporary stream channels in southern England, but it has potential for application in Irish rivers. There is also evidence to suggest that information on macrophyte morphology and traits, along with information on species abundance and assemblages, can be used successfully in identifying both anthropogenic disturbance and hydrological changes (Gurnell *et al*., 2010; Moncão *et al*., 2012; Cavalli *et al*., 2014; van Oorschot *et al*., 2016; Gebler and Szoszkiewicz, 2022), and this also has potential for application in Irish rivers. In addition to macroinvertebrates and macrophytes, another major biotic component of rivers is the fish community. Although biotic indices in relation to flow are much more common for non-fish taxa, there have been some efforts to develop fish-based metrics (e.g. Belmar *et al*., 2018). However, a more established approach used in studies involving fish is to study optimal flow conditions for particular species using aquatic habitat/hydraulic modelling (e.g. Dunbar *et al*., 2012), and this is likely to be the most applicable approach to use in an Irish context.

The main aim of this project was to investigate whether hydroecological biomonitoring tools for flow could be usefully applied in WFD river monitoring programmes in Ireland, thereby providing additional information when assessing the ecological status of Irish rivers.

1.2 Objectives

The broad objectives of this project were to:

- test selected macroinvertebrate hydroecological indices and identify and/or develop suitable indices for use in Irish river monitoring programmes;
- investigate fish-based flow metrics and use statistical modelling to investigate the responses of Irish fish communities to varying flow conditions in rivers;
- investigate the relationship between macrophytes and flow, and develop a macrophyte-based flow index for use in Irish river monitoring programmes;
- investigate whether multi-metric hydroecological indices can be usefully applied in Irish river monitoring programmes;
- provide recommendations on the use of hydroecological biomonitoring tools in Irish river monitoring programmes.

1.3 Data

1.3.1 Biological data

The biological data used in this project were obtained from a variety of sources. First, surveys were carried out at 47 river sites throughout Ireland in 2021 (spring, summer and autumn macroinvertebrate surveys, and summer macrophyte surveys) and 2022 (spring and autumn macroinvertebrate surveys). A number of criteria were used to select those 47 sites, the main ones being that the sites were classified as having “good” or “high” water quality (according to WFD status classifications) by the EPA; had a hydrometric station on the same river, within 1 km of the monitoring site, managed by either the EPA or the Office of Public Works (OPW), which was (in March 2021) measuring discharge (flow volume per unit time, measured in $m^3 s^{-1}$, simply referred to hereafter as flow) and had flow data available for download; had no obvious tributaries entering the main channel between the monitoring site and the hydrometric station; and had no apparent significant abstraction or discharge pressures.

Data from historical EPA macroinvertebrate surveys (2007–2018) that were publicly available in an open-access database (Feeley *et al*., 2020) were used for this project, as were fish data from historical electrofishing surveys (2008–2020) that were provided to the project by Inland Fisheries Ireland (IFI) and open-access historical macrophyte survey data (1998–2015) held within the National Biodiversity Data Centre’s (NBDC’s) River Macrophyte Database (RMD), which is part of the NBDC’s National Vegetation Database.

1.3.2 Hydrological data

1.3.2.1 Hydrological data from historical surveys

The EPA's register of hydrometric stations was filtered to identify the stations listed as measuring both water level and flow, and as being managed by either the EPA or the OPW. All available mean daily flow records for these 619 hydrometric stations were downloaded from the EPA and OPW websites (at https://epawebapp.epa.ie/hydronet/ and https://waterlevel.ie/hydro-data/), and downloaded data were subsequently filtered to include only data from 2005 to 2022 (inclusive) for the calculation of flow statistics relevant to the years covered by this project. Many of the 619 stations had substantial amounts of missing data and/or had substantial amounts of poor-quality mean daily flow data (according to the quality codes assigned by the EPA and the OPW to the estimates of mean daily flow contained in their datasets) for the 2005–2022 period. To mitigate the impact of missing and poor-quality data and to ensure the calculation of accurate flow statistics, only those hydrometric stations that had good-quality (at a minimum) mean daily flow data for at least 70% of days between 2005 and 2022 (inclusive) were retained for further analysis. Following application of these criteria, data from 132 hydrometric stations were retained for further analysis of the relationship between flow statistics and historical biological data for macroinvertebrates, fish and macrophytes. In total, 102 of these hydrometric stations were used for analyses of macroinvertebrate data, 49 were used for analyses of fish and 39 were used for analyses of macrophytes.

1.3.2.2 Hydrological data for 2021 and 2022 surveys

Mean daily flow data were available for 45 of the 47 hydrometric stations that had previously been matched with the 2021 and 2022 macroinvertebrate and macrophyte survey sites (see section 1.3.1); no mean daily flow data were available for download (in March 2024) for two hydrometric stations, so those stations (and their matched macroinvertebrate and macrophyte survey sites) were removed from the database prior to further analysis. The remaining 45 hydrometric stations (and their matched macroinvertebrate and macrophyte survey sites) were used for further analysis of the relationship between flow statistics and 2021 and 2022 survey data. It should be noted that 58% (26) of the 45 hydrometric stations did not meet the 70% good-quality mean daily flow data criterion used for analyses with historical data (see section 1.3.2.1).

1.3.2.3 Calculating flow statistics

Flow statistics were calculated using functions from the hetoolkit R package (Dunbar *et al*., 2023). A brief outline of the approach used is provided here; full details are available on request from M. Gammell (Department of Natural Resources and the Environment, Atlantic Technological University, Galway) and more detailed information about the hetoolkit package can be found in Dunbar *et al*. (2023). Prior to the calculation of flow statistics, the impute_flow function from the hetoolkit was used to estimate mean daily flow values for all dates for which mean daily flow data were missing, for each hydrometric station. The equipercentile method of the impute_flow function was used to impute most of the missing flow data; this method uses measured flows at a donor hydrometric station to estimate missing flow data at a target hydrometric station. The linear method was then used to impute mean daily flow values for dates that still had missing data, using linear extrapolation. Flow statistics were calculated (using measured and imputed mean daily flow data) with the calc_flowstats function from the hetoolkit for different time periods, as required. Three of the flow statistics produced by the calc_flowstats function were used: Q10z (a standardised high-flow statistic based on the flow that was equalled or exceeded at a station for 10% of the time), Q95z (a standardised low-flow statistic based on the flow that was equalled or exceeded at a station for 95% of the time) and min_7day_z (another standardised low-flow statistic based on the minimum 7-day mean flow at a station) (Dunbar *et al*., 2023). These flow statistics are regularly used in ecological studies of river flow (e.g. Dunbar *et al*., 2010a,b; Worrall *et al*., 2014) and have been shown to correlate with at least some hydroecological flow indices (e.g. Dunbar *et al*., 2010a; Worrall *et al*., 2014).

2 Macroinvertebrates

2.1 Introduction

Three British hydroecological macroinvertebrate indices have potential for monitoring the effects of changes in flow in Irish rivers: the Lotic-invertebrate Index for Flow Evaluation (LIFE) (Extence *et al*., 1999), the Drought Effect of Habitat Loss on Invertebrates (DEHLI) index (Chadd *et al*., 2017) and the Empirically-weighted Proportion of Sediment-sensitive Invertebrates (E-PSI) index (Turley *et al*., 2016).

Because different macroinvertebrate taxa have different preferences regarding the microhabitats created by river flow and morphology, the taxa present at a site can reveal information about prevailing and antecedent flow conditions, and, as taxa are lost from or added to a site's community, this can reveal information about changes in flow. As flow velocities decline, there should be an increase in the abundance of taxa associated with slower velocity habitats and silty conditions, and a decrease in the abundance of taxa associated with faster velocities and coarser substrates (Extence *et al*., 1999). This link between the macroinvertebrates at a site and that site's flow history formed the basis for the development of the LIFE index (Extence *et al*., 1999). LIFE scores have strong correlations with historical flow, and the LIFE index is used by the Environment Agency in England to quantify the impacts of water abstraction (Klaar *et al*., 2014). Higher LIFE scores are indicative of higher flows (Extence *et al*., 1999). The LIFE method can be used to calculate a family-level or a species-level index, depending on the taxonomic resolution of the data. Under extreme low-flow/drought conditions, the relationship between LIFE score and flow weakens, so the DEHLI index was developed to measure drought effects (Chadd *et al*., 2017). The DEHLI index classifies macroinvertebrates according to their association with stages of channel drying, and it can reveal impacts of drought that may be missed by the LIFE index (Chadd *et al*., 2017). Lower DEHLI scores (towards 1) are indicative of significant drought effects, while higher scores (towards 10) are indicative of minimal or no drought effects (Chadd *et al*., 2017). The DEHLI method is primarily a family-level index, but it does separate some families into different groups based on the varying drought sensitivities of taxa within those groups. Another impact of low flows is increased fine sediment deposition, which can negatively affect a site's physical characteristics (Extence *et al*., 2013). The E-PSI index weights macroinvertebrate taxa based on their sensitivity to deposited fine sediments, and provides an estimate of ecological degradation at a site based on the degree of sedimentation (Turley *et al*., 2016). E-PSI scores range between 0 and 100, with lower scores being indicative of heavier sedimentation (Turley *et al*., 2016). The E-PSI method can be used to calculate a family-level index or a mixed-level index (mainly a species-level index), depending on the taxonomic resolution of the data.

The main objective of this work was to investigate if the LIFE, DEHLI and E-PSI indices were suitable for measuring flow-related changes to macroinvertebrate communities in Irish rivers and, if so, whether they could be usefully applied in Irish river monitoring programmes. One set of analyses used data from 773 historical macroinvertebrate surveys, carried out at 202 sites between 2007 and 2018, matched with flow data from 102 hydrometric stations within 5 km of a monitored site. Another set of analyses used data from 221 macroinvertebrate surveys, carried out at 45 sites in 2021 and 2022, matched with flow data from 45 hydrometric stations within 1 km of a monitored site.

2.2 Results

2.2.1 Adjusted biotic indices

Adjustments were made so that the LIFE, DEHLI and E-PSI indices could be used with macroinvertebrate data collected by the EPA (Feeley *et al*., 2020); the indices were adjusted to work with family-level data and EPA macroinvertebrate abundance categories. The LIFE and E-PSI indices use simple logarithmic abundance categories; the DEHLI index does not use abundance data. As the EPA does not record data on macroinvertebrate abundances in logarithmic categories, the first adjustment needed was to convert EPA abundance categories into logarithmic abundance categories (Table 2.1); data on actual abundances

Table 2.1. Conversion of the six abundance categories used by the EPA into the four logarithmic abundance categories used to calculate LIFE and E-PSI scores

EPA abundance category	Logarithmic abundance category
Single, few	A (1–9 individuals)
Common	B (10–99 individuals)
Numerous	C (100–999 individuals)
Dominant, excessive	D (1000+ individuals)

were available for the 2021 and 2022 surveys and were converted directly to logarithmic abundance categories. No other adjustments were made for the E-PSI index; both the original family-level and mixed-level versions of this index (Turley *et al*., 2016) were used, and scores were calculated using the formula and methods in Turley *et al*. (2016).

For the LIFE index, macroinvertebrate species are assigned to one of six flow groups based on current velocity preferences (Extence *et al*., 1999). Using these flow groups, in combination with abundance categories, a species-level index can be calculated. Macroinvertebrate families were also assigned to flow groups, so that a family-level index could be calculated. In assigning a flow group to families that contained species with varying current velocity preferences, Extence *et al*. (1999) chose a single flow group to best represent those families. Due to differences between British and Irish fauna, flow groups assigned to families containing species with varying current velocity preferences in Britain may not be the most appropriate flow groups for those same families on the island of Ireland. Therefore, the family-level LIFE index was adjusted for Ireland by investigating the Irish status of all families and species in Extence *et al*. (1999) using published work (Byrne *et al*., 2009; Foster *et al*., 2009; Nelson *et al*., 2011; Kelly-Quinn and Regan, 2012; Foster *et al*., 2014; Elliott and Dobson, 2015; O'Connor, 2015; Feeley *et al*., 2016; Riley, 2020) and records of each species held by the NBDC at https://maps.biodiversityireland.ie/. The family flow groups assigned by Extence *et al*. (1999) were then adjusted for Ireland, if considered necessary (see Table 2.1; full details of how adjustments were made are available on request from M. Gammell (Department of Natural Resources and the Environment, Atlantic Technological University, Galway)), and these adjusted flow groups were used to calculate new family-level LIFE scores specifically adapted for Ireland, using the same formula (equation 2.1) and methods in Extence *et al*. (1999).

$$\text{LIFE} = \frac{\sum fs}{n} \tag{2.1}$$

In equation 2.1, *fs* is the flow score assigned to each macroinvertebrate family used in the calculation of LIFE scores, and *n* is the number of families used in that calculation (Extence *et al*., 1999). A flow score (*fs*) is assigned to each family in a sample based on its abundance category (Table 2.1) and flow group (Table 2.2), as shown in Table 2.3. For example, a macroinvertebrate family from flow group 1 with abundance category A has an *fs* of 9, while a macroinvertebrate family from flow group 4 with abundance category C has an *fs* of 4.

The original species-level (Extence *et al*., 1999) version of the LIFE index was also calculated using the methods in Extence *et al*. (1999). The species-level LIFE index did not need to be adjusted for Ireland because it uses species-level flow groups for the macroinvertebrate taxa in a sample.

For the DEHLI index, macroinvertebrate taxa were assigned to 1 of 10 different drought intolerance score (DIS) groups, based on their drought intolerance. By assigning the taxa found in a sample to DIS groups, a DEHLI score can be calculated. Most of the assignations of DIS groups in Chadd *et al*. (2017) were at the family level, although taxa in a small number of families were assigned to different DIS groups, if those taxa had differences in drought intolerance. The original DEHLI index was converted to a family-level index for use in Ireland by investigating the Irish status of all taxa in families that had more than one DIS in Chadd *et al*. (2017), using published work (Kelly-Quinn and Regan, 2012; O'Connor, 2015; Feeley *et al*., 2016) and records for each taxon held by the NBDC. Based on this, the most appropriate single DIS for those families that had more than one DIS in Chadd *et al*. (2017) was assigned to the family for use in Ireland (see Table 2.4; full details of how the most appropriate DIS was chosen are available on request from M. Gammell (Department of Natural Resources and the Environment, Atlantic Technological University, Galway)). The new family-level DEHLI index, specifically adapted to Ireland, was calculated using

Table 2.2. Flow groups assigned to macroinvertebrate families for the calculation of LIFE scores for Ireland

Family	Flow group	Family	Flow group	Family	Flow group
Ameletidae[a]	1	Physidae[b]	3	Hydrophilidae	4
Heptageniidae	1	Corophiidae	3	Scirtidae	4
Taeniopterygidae[b]	1	Gammaridae[b]	3	Dryopidae[a]	4
Leuctridae[b]	1	Calopterygidae	3	Sialidae	4
Capniidae	1	Psychomyiidae[b]	3	Sisyridae	4
Perlodidae	1	Ecnomidae	3	Hydroptilidae	4
Perlidae	1	Planariidae	4	Phryganeidae	4
Chloroperlidae	1	Dugesiidae	4	Limnephilidae	4
Rhyacophilidae	1	Dendrocoelidae	4	Leptoceridae	4
Philopotamidae	1	Valvatidae	4	Tipulidae	4
Odontoceridae	1	Bithyniidae	4	Dictynidae	5
Goeridae	1	Lymnaeidae	4	Mysidae	5
Neritidae	2	Planorbidae[c]	4	Aeshnidae[b]	5
Ancylidae[c]	2	Acroloxidae	4	Corduliidae[b]	5
Margaritiferidae	2	Unionidae	4	Mesovelidae	5
Piscicolidae	2	Sphaeriidae	4	Nepidae	5
Astacidae	2	Dreissenidae	4	Hygrobiidae	5
Baetidae	2	Glossiphoniidae	4	Hydrochidae[a]	5
Ephemerellidae	2	Hirudinidae	4	Molannidae[b]	5
Ephemeridae	2	Haemopidae[a]	4	Chaoboridae	5
Caenidae[b]	2	Erpobdellidae	4	Culicidae	5
Nemouridae[b]	2	Asellidae	4	Syrphidae	5
Cordulegastridae	2	Crangonyctidae	4	Georissidae[a]	6
Aphelocheiridae	2	Siphlonuridae	4		
Hydraenidae[b]	2	Leptophlebiidae[b]	4		
Elmidae	2	Coenagrionidae	4		
Osmylidae	2	Lestidae	4		
Glossosomatidae	2	Libellulidae	4		
Polycentropodidae[b]	2	Hebridae	4		
Hydropsychidae	2	Hydrometridae	4		
Beraeidae	2	Veliidae	4		
Lepidostomatidae	2	Gerridae	4		
Sericostomatidae	2	Notonectidae	4		
Apatanidae[a]	2	Pleidae	4		
Limoniidae[a]	2	Corixidae	4		
Pediciidae[a]	2	Haliplidae	4		
Ptychopteridae	2	Noteridae	4		
Simuliidae	2	Dytiscidae	4		
Viviparidae	3	Gyrinidae	4		
Hydrobiidae[b]	3	Helophoridae[a]	4		

[a]These families were not included in Extence *et al.* (1999), but were used for calculating LIFE scores for Ireland.

[b]Families that have had their flow groups adjusted for Ireland; these families are in different flow groups in Extence *et al.* (1999).

[c]The family Ancylidae (containing the species *Ancylus fluviatilis*) was considered a separate scoring family in Extence *et al.* (1999). The family Ancylidae is no longer in use and *A. fluviatilis* is now in the family Planorbidae. However, to maintain consistency with Extence *et al.* (1999), and because *A. fluviatilis* has a different flow preference from the other Planorbidae species, is easily identified and is commonly recorded in Ireland, this species is considered separately here; the family Ancylidae has been retained for calculating LIFE scores.

Table 2.3. Flow scores (*fs*) for different combinations of flow groups and abundance categories used to calculate LIFE scores

Flow group	Abundance category	*fs*	Flow group	Abundance category	*fs*	Flow group	Abundance category	*fs*
1	A	9	3	A	7	5	A	5
1	B	10	3	B	7	5	B	4
1	C	11	3	C	7	5	C	3
1	D	12	3	D	7	5	D	2
2	A	8	4	A	6	6	A	4
2	B	9	4	B	5	6	B	3
2	C	10	4	C	4	6	C	2
2	D	11	4	D	3	6	D	1

Source: Compiled by authors based on information provided in Extence *et al.* (1999).

Table 2.4. DISs assigned to macroinvertebrate families for the calculation of the DEHLI index using Irish data

Family	DIS	Family	DIS	Family	DIS
Heptageniidae	10	Glossosomatidae	7	Libellulidae	4
Ameletidae	10	Sericostomatidae	7	Ecnomidae	4
Perlidae	10	Pediciidae	7	Tipulidae	4
Chloroperlidae	10	Stratiomyidae	7	Psychodidae	4
Philopotamidae[a]	10	Gammaridae	6	Tabanidae	4
Rhyacophilidae	10	Polycentropodidae	6	Helophoridae	3
Taeniopterygidae[a]	9	Simuliidae	6	Planariidae	3
Odontoceridae	9	Empididae	6	Dugesiidae	3
Thaumaleidae	8	Unionidae	6	Physidae	3
Dixidae	8	Sisyridae	6	Hydrophilidae	3
Goeridae	8	Hydropsychidae	5	Scirtidae[a]	3
Hydrobiidae	7	Leptoceridae	5	Lymnaeidae	2
Bithyniidae	7	Muscidae	5	Sphaeriidae	2
Valvatidae	7	Coenagrionidae	5	Asellidae	2
Lestidae	7	Siphlonuridae	5	Corixidae	2
Calopterygidae	7	Ephemeridae	5	Gerridae	2
Aeshnidae	7	Caenidae	5	Hydrometridae	2
Nepidae	7	Leptophlebiidae[a]	5	Veliidae	2
Hydraenidae	7	Sialidae	5	Mesoveliidae	2
Phryganeidae	7	Elmidae	5	Gyrinidae[b]	2
Beraeidae	7	Hydrochidae	5	Ephydridae	2
Pyralidae	7	Molannidae	5	Notonectidae	1
Baetidae[a]	7	Limoniidae	5	Dytiscidae	1
Perlodidae	7	Ptychopteridae	5	Culicidae	1
Nemouridae[a]	7	Dolichopodidae	5	Syrphidae	1
Leuctridae	7	Rhagionidae	5	Ceratopogonidae	1
Aphelocheiridae	7	Athericidae	5	Planorbidae[c]	1
Lepidostomatidae	7	Corduliidae	4	Chironomidae[d]	–

[a]Different taxa from this family were assigned different DISs in Chadd *et al.* (2017).

[b]Adult records only (i.e. larval Gyrinidae records are not used in the calculation of the DEHLI index).

[c]Excluding *A. fluviatilis*.

[d]Different taxa from this family were assigned different DISs in Chadd *et al.* (2017). It was not possible to determine the most appropriate DISs for Irish Chironomidae records, so this family was not used to calculate the DEHLI index for Ireland.

the same formula (equation 2.2) and methods as those described in Chadd *et al*. (2017). The original version of the DEHLI index was also calculated, following the methods in Chadd *et al*. (2017).

$$\text{DEHLI} = \frac{\sum DIS}{n} \tag{2.2}$$

In equation 2.2, *DIS* is the DIS assigned to each macroinvertebrate family used in the calculation of the DEHLI index, and *n* is the number of taxa used in that calculation (Chadd *et al*., 2017).

2.2.2 Analysis of historical macroinvertebrate data

Linear mixed models were used to investigate relationships between biotic index scores and flow statistics. All modelling was carried out using cubed index scores to correct for skew. In the models, the response variable was the cubed index score (LIFE, DEHLI or E-PSI score, depending on the model); the fixed effects were Q10z, Q95z or min_7day_z (depending on the model), river type (based on Northern Ireland Environment Agency categories (see the Water Framework Directive (Priority Substances and Classification) (Amendment) Regulations (Northern Ireland) 2015) and Q-value category (an assessment of the water quality at a site, ranging from Q1 for sites with the lowest water quality to Q5 for sites with the highest water quality); and the random effects were monitoring site, hydrometric station and year of survey. Separate models were constructed for four different fixed antecedent periods, namely one each for flow statistics calculated for the following: the 6-month winter–spring period immediately before the 6-month summer–autumn period during which the macroinvertebrate sample had been collected; the 6-month summer–autumn period a year prior; the 6-month winter–spring period a year prior; and the 6-month summer–autumn period 2 years prior. The best-fitting model was selected using Akaike information criterion (AIC) values. A moving time-window analysis using the climwin R package (Bailey and van de Pol, 2016) was used to identify the antecedent period, within the 2-year period prior to a sample being collected, for which Q10z and Q95z statistics were the strongest predictors of biotic index scores; models were also constructed using these (relative) time periods. Table 2.5 shows selected outputs from those models for which flow statistics were the strongest predictors of biotic index values; the output is for the fixed effect of the flow statistic on the index score. The antecedent period for which flow statistics were the strongest predictors of index scores varied between models, but the overall pattern was for flow to be a weak but positive, and often significant, predictor of index score; an exception to this expected pattern was one model with a significant negative relationship between Q95z and DEHLI score. Figure 2.1 represents the relationship between Q10z for the 1-month period 12 months before a sample was taken and LIFE scores; it shows the expected pattern, namely that the LIFE score increases as the value

Table 2.5. Fixed effects of flow statistics on biotic index scores for the best-fitting linear mixed models for different antecedent time periods

Index	Flow statistic	Antecedent period	Statistic	*P*-value[a]
LIFE	Q10z	Fixed: 6-month summer–autumn period, 2 years prior	1.283	0.201
LIFE	Q10z	Relative: 1-month period, 12 months prior	3.301	0.001
LIFE	Q95z	Fixed: 6-month summer–autumn period, 1 year prior	1.707	0.089
LIFE	Q95z	Relative: 1-month period, 12 months prior	2.633	0.009
DEHLI	Q95z	Fixed: 6-month winter–spring period, 1 year prior	–2.098	0.038
DEHLI	Q95z	Relative: 2-month period, 23–24 months prior	–0.070	0.944
DEHLI	min_7day_z	Fixed: 6-month summer–autumn period, 1 year prior	2.297	0.024
E-PSI	Q95z	Fixed: 6-month, winter–spring period, same year	1.672	0.098
E-PSI	Q95z	Relative: 1-month period, 12 months prior	2.046	0.042
E-PSI	min_7day_z	Fixed: 6-month, summer–autumn period, 2 years prior	0.990	0.324

[a]**Significance was tested using Satterthwaite's method for *t*-tests.**

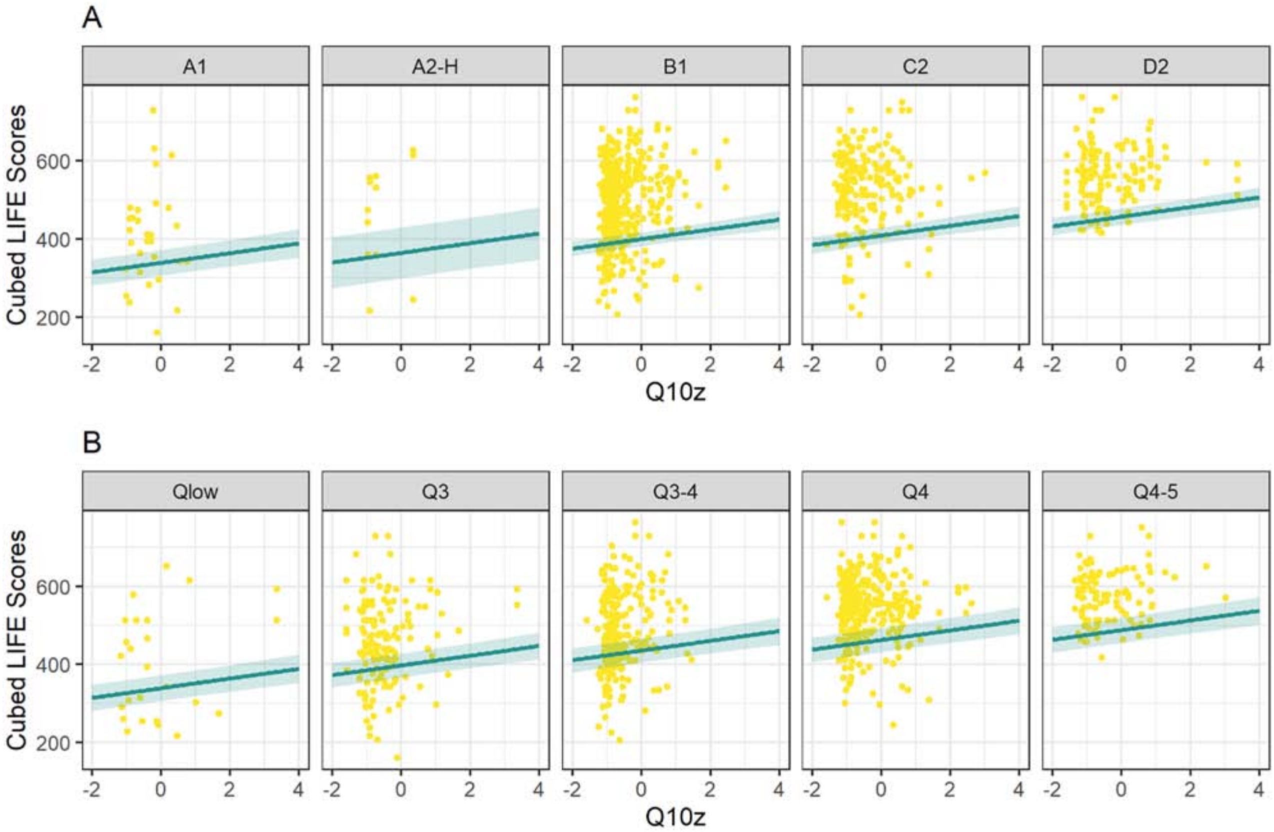

Figure 2.1. Predicted relationships between Q10z for the 1-month period 12 months prior to the date of the macroinvertebrate sample collection and cubed LIFE scores by (A) river type and (B) Q-value; note that Qlow includes Q-values from any category lower than Q3.

of the flow statistic increases. Figure 2.1 also shows the consistent (and generally significant) relationships observed for the other two fixed effects (river type and Q-value) in all models. Figure 2.1A shows a difference in LIFE scores based on river type, with higher LIFE scores being predicted in sites from more western and elevated locations (C2, D2) than in sites from more eastern and central lowland areas (A1, A2-H, B1). Figure 2.1B shows the difference in LIFE scores based on Q-value category, with higher LIFE scores being predicted to be associated with sites with better water quality.

2.2.3 Analysis of 2021 and 2022 macroinvertebrate data

Because the 2021 and 2022 macroinvertebrate data had been identified to species level, species-level LIFE (LIFE-Species) scores, original DEHLI index (DEHLI-Original) scores and mixed-level E-PSI index (E-PSI-Mixed) scores were calculated, as were all family-level indices. Separate linear mixed models were used to examine the relationship between cubed family-level scores for each index and cubed LIFE-Species scores, cubed DEHLI-Original scores and cubed E-PSI-Mixed scores; monitoring site and year of survey were random effects in all models. Family-level LIFE (LIFE-Family) score was a strong and significant positive predictor of LIFE-Species score ($t = 26.514$, $P < 0.001$), with an increase of approximately 0.864 ± 0.033 (95% confidence interval (CI): 0.800, 0.928) in cubed LIFE-Species score for every unit increase in cubed LIFE-Family score; family-level DEHLI index (DEHLI-Family) score was a strong and significant positive predictor of DEHLI-Original score ($t = 133.415$, $P < 0.001$), with an increase of approximately 1.019 ± 0.008 (95% CI: 1.004, 1.034) in cubed DEHLI-Original score for every unit increase in cubed DEHLI-Family score; family-level E-PSI index (E-PSI-Family) score was a strong and significant positive predictor of E-PSI-Mixed score ($t = 26.086$, $P < 0.001$), with an increase of approximately 0.894 ± 0.034 (95% CI: 0.827, 0.961) in cubed E-PSI-Mixed score for every unit increase in cubed E-PSI-Family score.

Separate analyses were carried out for autumn, spring and summer macroinvertebrate surveys. Linear mixed (autumn, spring) and linear (summer) models were used to investigate relationships between LIFE-Family scores and flow statistics. In the models, the response variable was cubed index score; the fixed effects were Q10z or Q95z (depending on the model), river type (based on Northern Ireland Environment Agency categories) and Whalley, Hawkes, Paisley and Trigg (WHPT) average score per taxon (ASPT) (WHPT-ASPT) (a water quality variable) (WFD-UKTAG, 2021); and the random effects in mixed models were monitoring site and year of survey. Separate models were constructed for eight different fixed antecedent periods: one model for flow statistics calculated for the 3-month period immediately before the 3-month survey season (autumn, spring or summer), a second model for the 3-month period immediately before that and so on until the final (eighth) model for the 3-month period 2 years before the survey season. The best-fitting model was selected using AIC values. To compare results from family-level and species-level indices, models were also constructed for the antecedent period in the best-fitting LIFE-Family model, but with LIFE-Species score as the response variable. Table 2.6 shows selected outputs from those models for which flow statistics were the strongest predictors of LIFE-Family scores, as well as output from the associated LIFE-Species models for the same antecedent period; the output is for the fixed effect of the flow statistic on the index score. The antecedent period for which flow statistics were the strongest predictors of index scores varied between models, but the overall pattern for autumn and summer surveys was for flow to be a weak but positive, and sometimes significant, predictor of LIFE scores; an exception to this expected pattern was seen for spring surveys, where flow was found to be a weak negative predictor of LIFE score, although none of the predictions for spring models was significant.

Figure 2.2 shows the relationship between Q10z calculated for the 3-month winter period prior to the autumn survey season and LIFE scores; it shows the expected pattern of increasing LIFE scores with the increasing value of the flow statistic (the relationship is significant for LIFE-Family and non-significant for LIFE-Species scores in this case; see Table 2.6). Figure 2.2 also illustrates a strong and significant positive relationship between WHPT-ASPT and LIFE-Family and LIFE-Species scores, indicating that higher LIFE scores are associated with sites with better water quality. This significant positive relationship between WHPT-ASPT and LIFE scores was seen for all models (Table 2.6). The relationship between river type and LIFE scores was not as clear in 2021/2022 models; some models found a significant difference in LIFE scores based on river type, but other models did not.

Table 2.6. Fixed effects of flow statistics on index scores for the best-fitting LIFE-Family models (and associated LIFE-Species models) for different antecedent periods

Season	Index	Flow statistic	Antecedent period	Statistic	*P*-value[a]
Autumn[b]	LIFE-Family	Q10z	3-month winter period prior to the autumn survey period	2.204	0.031
Autumn[b]	LIFE-Species	Q10z		1.650	0.108
Autumn[b]	LIFE-Family	Q95z	3-month winter period prior to the autumn survey period	1.961	0.055
Autumn[b]	LIFE-Species	Q95z		1.367	0.209
Spring[b]	LIFE-Family	Q10z	3-month spring period 2 years prior to the spring survey period	–1.644	0.104
Spring[b]	LIFE-Species	Q10z		–1.134	0.260
Spring[b]	LIFE-Family	Q95z	3-month spring period 2 years prior to the spring survey period	–1.217	0.227
Spring[b]	LIFE-Species	Q95z		–0.355	0.724
Summer[c]	LIFE-Family	Q10z	3-month spring period prior to the summer survey period	1.919	0.062
Summer[c]	LIFE-Species	Q10z		2.130	0.039
Summer[c]	LIFE-Family	Q95z	3-month winter period prior to the summer survey period	1.591	0.119
Summer[c]	LIFE-Species	Q95z		2.138	0.039

[a]Significance tested with Satterthwaite's method for *t*-tests.
[b]Linear mixed models were used for autumn and spring.
[c]Linear models were used for summer.

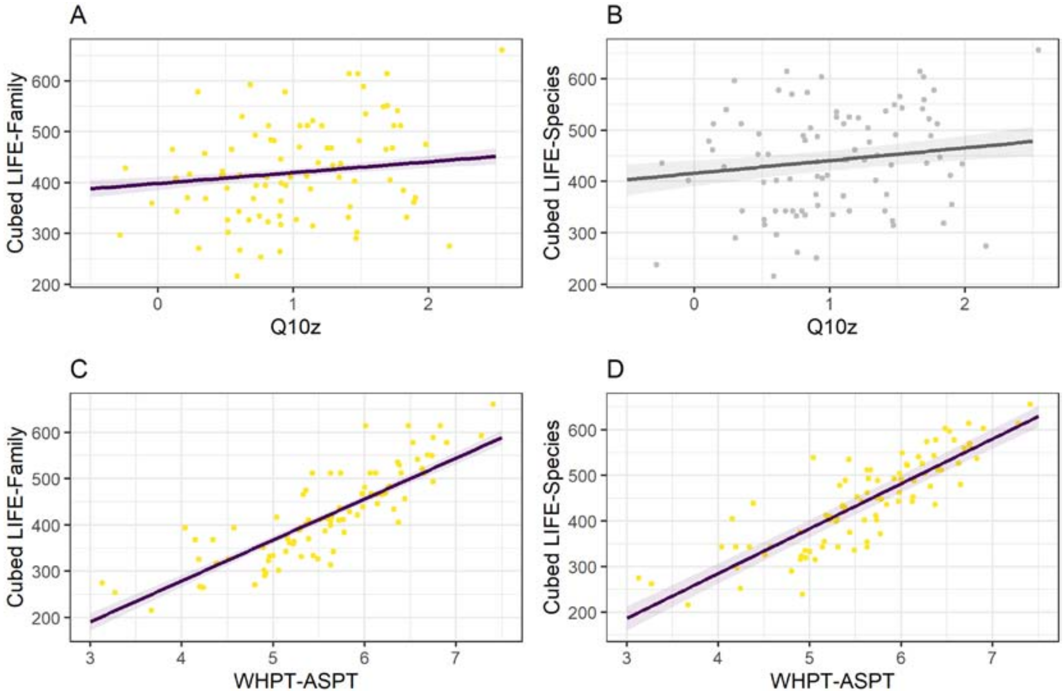

Figure 2.2. Predicted relationships between (A) Q10z and LIFE-Family score (significant); (B) Q10z and LIFE-Species score (non-significant); (C) WHPT-ASPT and LIFE-Family score (significant); and (D) WHPT-ASPT and LIFE-Species score (significant). Q10z was calculated for the 3-month winter period prior to the autumn survey.

2.2.4 LIFE scores and flow at two selected sites

To demonstrate the expected relationship between LIFE scores and flow at individual sites, graphs were generated of LIFE scores calculated on five separate occasions during an approximately 18-month period throughout 2021 and 2022 at two of the macroinvertebrate monitoring sites that showed some of the greatest variation in LIFE scores during that period (Figure 2.3). In addition to LIFE scores, flow estimates from the nearby hydrometric stations that had previously been matched with those macroinvertebrate monitoring sites were also plotted on the same graphs for the same general time period. In Figure 2.3, LIFE scores at both sites can be seen to approximately track changes in flow, with a certain time lag; this time lag is expected because the macroinvertebrate community will have a lagged response to changes in flow. For the first site, there is an approximate 3-month lag in the response of the macroinvertebrate community (Figure 2.3A and B); for the second site there is an approximate 5-month lag (Figure 2.3C and D).

2.2.5 DEHLI and LIFE scores at two selected sites

To demonstrate how DEHLI and LIFE scores are expected to show different responses to changes in flow, and to drought conditions in particular, graphs were generated to show LIFE and DEHLI scores calculated using data gathered between 2007 and 2018 by the EPA (historical data) and in 2021 and 2022 (during this project) at two selected EPA macroinvertebrate monitoring sites that showed a substantial decline in DEHLI scores from 2021 to 2022 (following relatively sustained periods of drought; see, for example, Antwi *et al*., 2022) (Figure 2.4). The graphs show that DEHLI scores approximately tracked LIFE scores at both sites, but that the range in DEHLI scores was greater than the range in LIFE scores, and, under the presumed influence of drought conditions (i.e. the 2021–2022 period on the graphs), DEHLI scores at both sites showed a more obvious decline than LIFE scores.

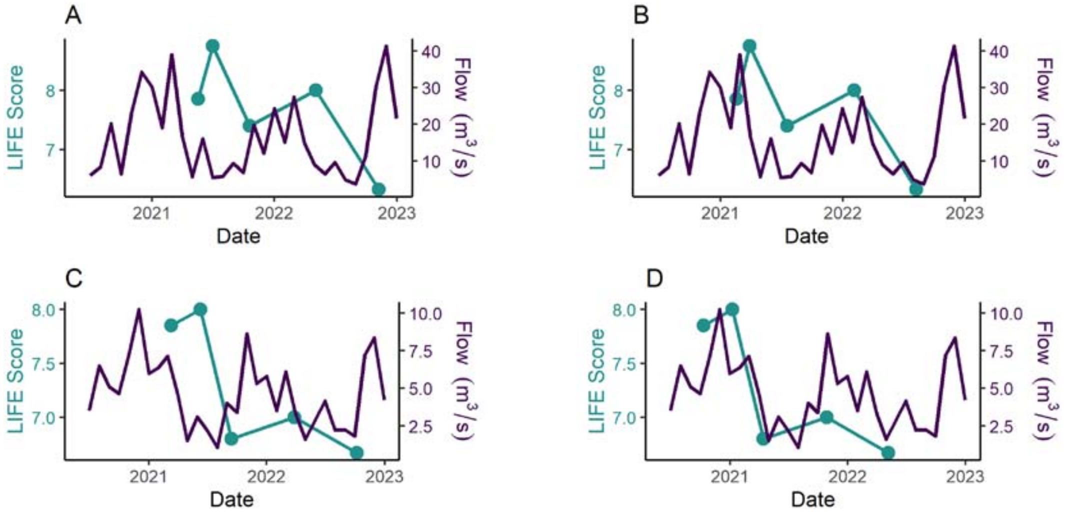

Figure 2.3. Overlap of LIFE scores (family-level index adjusted for Ireland) calculated for two macroinvertebrate monitoring sites and flow estimates (calculated as monthly averages of the mean daily flow values) from their matching hydrometric stations. Panel A shows LIFE scores calculated on different dates for a single macroinvertebrate monitoring site (EPA site RS22F020300) and flow estimates from its matching hydrometric station; Panel B shows the same data but with the LIFE scores offset by 3 months (i.e. to overlap with dates exactly 3 months before the collection dates of the macroinvertebrate samples). Panel C shows LIFE scores calculated on different dates for a second macroinvertebrate monitoring site (EPA site RS32B010200) and flow estimates from its matching hydrometric station; Panel D shows the same data but with the LIFE scores offset by 5 months (i.e. to overlap with dates exactly 5 months before the collection dates of the macroinvertebrate samples).

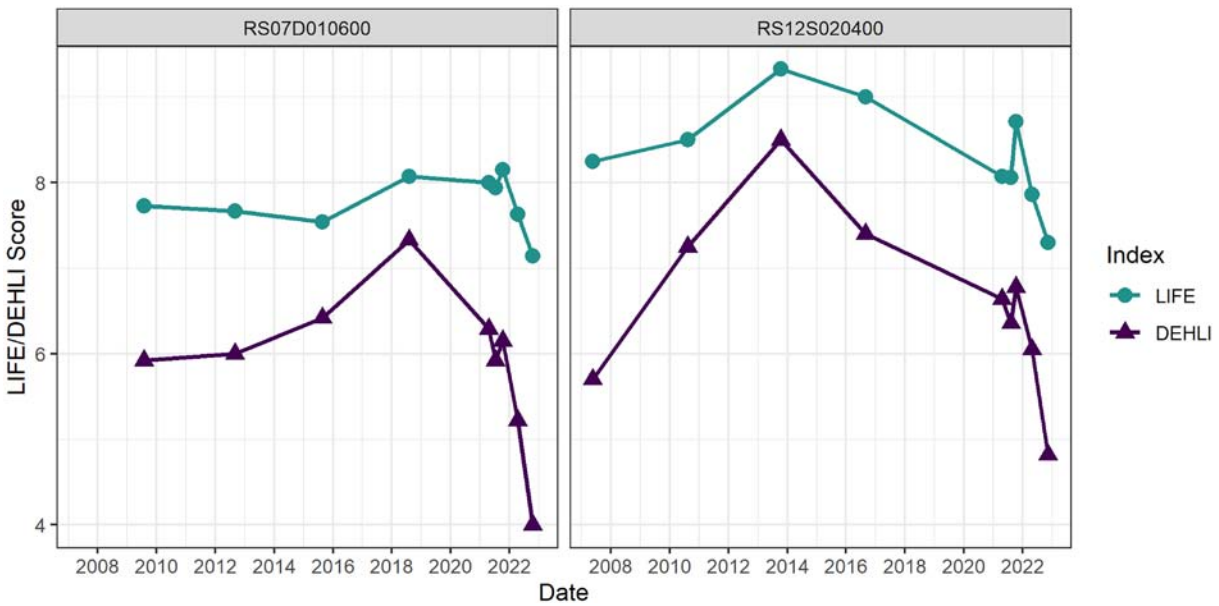

Figure 2.4. DEHLI scores (adjusted for Ireland) and LIFE scores (family-level index adjusted for Ireland) calculated for two EPA macroinvertebrate monitoring sites (RS07D010600 and RS12S020400) using data gathered between 2007 and 2022.

2.2.6 Geographical distribution of LIFE and E-PSI scores

Graphs were generated to compare the geographical distribution of median LIFE and E-PSI scores for all EPA river monitoring sites for which macroinvertebrate data were available during the 2007–2018 period with the distribution of upland areas in Ireland (based on Carlier *et al*., 2021) (Figure 2.5). Lower LIFE scores equate to lower flows (Extence *et al*., 1999), and Figure 2.5 shows that the lower LIFE scores are concentrated in more lowland areas, where lower river flows (in general) would be expected, whereas higher LIFE scores are concentrated in more upland areas, where higher river flows would be expected. Lower E-PSI scores equate to higher levels of sedimentation (Chadd *et al*., 2017), and Figure 2.5 shows that lower E-PSI scores are also concentrated in more lowland areas, where greater levels of sedimentation (in general) would be expected, whereas higher E-PSI scores are concentrated in more upland areas, where lower levels of sedimentation would be expected.

2.3 Conclusions

For hydroecological indices to be useful indicators of flow in Irish rivers, index scores should show significant positive relationships with flow statistics. Therefore, based on the results of this work, the three British hydroecological macroinvertebrate indices adapted here can be considered potentially useful tools for Irish river monitoring programmes. For the historical macroinvertebrate data, flow statistics were, with a few exceptions, consistently positive predictors of biotic index scores. Although relationships were relatively weak, they were significant in some cases, and the strengths of the relationships were similar to those found in other published work (e.g. Dunbar *et al*., 2010b). The relationships were often stronger when flow statistics had been calculated for antecedent periods identified with a moving time-window analysis (Bailey and van de Pol, 2016), suggesting that moving time-window analyses should be used in future work. For the 2021 and 2022 data, flow statistics were weak but consistently positive predictors of biotic index scores for autumn and summer surveys, and some relationships were significant. For spring surveys, the direction of the relationship was unexpectedly negative, but not significant. Previous work has found stronger relationships using autumn data than using spring data (Dunbar *et al*., 2010a). In all models with historical data, the relationship was significant between river type and index score, with higher index scores being associated with sites from more western and elevated locations than with sites from more eastern and central lowland areas. This pattern was not as obvious in the results from the 2021 and 2022 data, but the link between river type and index score in historical data provides further evidence to

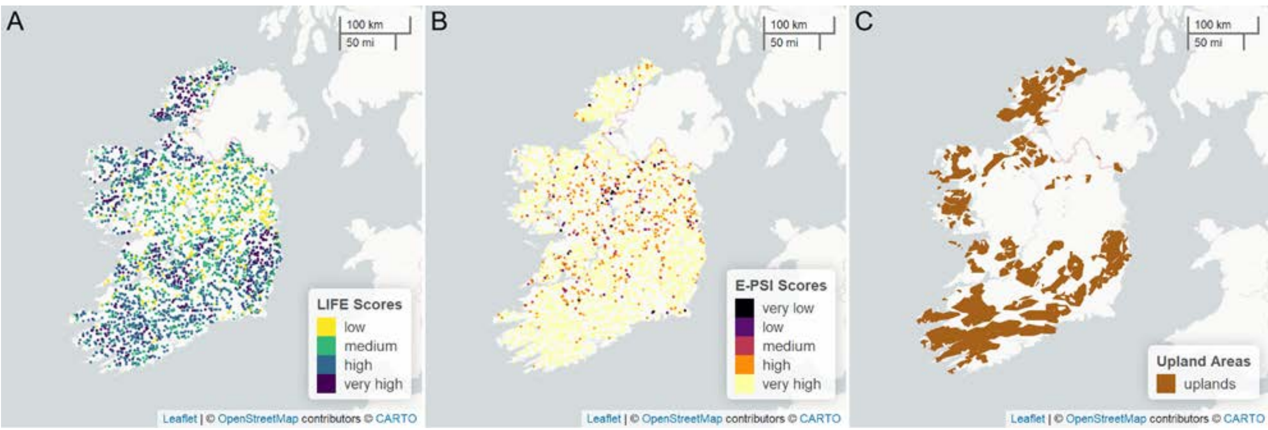

Figure 2.5. Distribution of median LIFE (A) and E-PSI (B) scores for all river monitoring sites for which data were available during the 2007–2018 period compared with the distribution of upland areas (C) (based on Carlier *et al*., 2021). LIFE scores were categorised as follows: ≤7.5 (low), >7.5 and ≤8.0 (medium), >8.0 and ≤8.5 (high), >8.5 (very high). Lower LIFE scores equate to lower flows. E-PSI scores were categorised as follows: ≤20 (very low), >20 and ≤40 (low), >40 and ≤60 (medium), >60 and ≤80 (high), >80 (very high). Lower E-PSI scores equate to heavier sedimentation.

suggest that these hydroecological indices can be usefully applied in Ireland. One possible confounding factor that needs to be considered is water quality, as all models found significant positive effects of water quality on index score. There are also other potential confounding factors that could affect the strength of the relationship between biotic index scores and flow that were not considered here, but that should be taken into account in future work, such as river size (e.g. the effects of drought are likely to be greater in smaller rivers) and fine sediment source (e.g. from peaty or mineral soils). Finally, because the scores calculated using family versions of the indices were strongly correlated with scores calculated with versions that used higher resolution (e.g. species-level) taxonomic data, the identification of macroinvertebrates to family level should be sufficient for the application of these indices in Irish river monitoring programmes.

3 Fish

3.1 Introduction

Similar to both macroinvertebrate (Extence *et al*., 1999; Turley *et al*., 2016; Chadd *et al*., 2017) and macrophyte (Januaer *et al*., 2010; Westwood *et al*., 2021; Vukov *et al*., 2022) communities, fish assemblages respond to changes in flow conditions (Pearson *et al*., 2011; Dunbar *et al*., 2012; Belmar *et al*., 2018; Stein *et al*., 2021). However, the responses of fish to altered flow regimes are much more complex than those of macroinvertebrates and macrophytes. Changes in flow conditions affect fish in multiple ways, for instance through changes to physical habitat and access to components of that habitat, changes in the supply of food resources and effects on life-cycle components. These effects, in turn, can cause alterations in behaviour, energy expenditure, and population and community dynamics (Rytwinski *et al*., 2017; Hansen *et al*., 2024). These responses can also interact with other impacts, such as nutrient status, the introduction of non-native fish species and climate change, and our biological understanding of the interacting effects of altered flow conditions with these other impacts on fish communities is poor (Pont *et al*., 2007).

While a number of studies have attempted to link fish-based biological indicators or indices of riverine ecosystem health with calculated statistics from flow-gauge data (Pearson *et al*., 2011; Taylor *et al*., 2013; Belmar *et al*., 2018), unlike the situation for macroinvertebrates and macrophytes, there are no suitable hydroecological indices for fish that could potentially be adapted for monitoring in Irish rivers. Another approach to the study of relationships between fish abundance at a site and flow is to examine optimal flow and habitat conditions for particular species using statistical modelling approaches (Dunbar *et al*., 2012; Fornaroli *et al*., 2016; Stein *et al*., 2021). For example, Fornaroli *et al*. (2016) examined density–environment relationships for three life stages of Brown Trout (*Salmo trutta*) and found that water velocity, substrate characteristics and refugia availability affected habitat suitability; they then used habitat-based models to determine a suitable flow range to maintain all life stages of this species. Such species-specific hydraulic habitat modelling would be very informative in elucidating the effects of past and future hydromorphology for sensitive species such as salmonids but would be less useful in the development of a fish community-based index for monitoring flow events in Irish rivers. These general methods, including fish-based flow indices (perhaps incorporating aspects of taxonomic and trait-based approaches) and statistical modelling to determine optimal flows, have potential for use in Ireland and could result in improvements to current fish monitoring programmes.

The aims of this work were (a) to investigate, using historical electrofishing data, if there is any relationship between fish communities and calculated flow statistics that would permit the development of a hydroecological fish community index or indices that could have potential for monitoring the impact of changes in flow on fish populations in Irish rivers; and (b) to examine density–environment relationships for individual life stages of sensitive salmonids in Irish rivers, to investigate if a flow-based metric could be developed from such relationships.

3.2 Results

3.2.1 Relationship between fish communities and flow

In order to examine if the fish communities across the study sites varied according to flow histories, a redundancy analysis (RDA) was conducted using the R package vegan. The calculated flow statistics, as provided by the hetoolkit (see section 1.3.2.3), and the low_n statistic (number of days when flow is below the Q95 in the last 6 months) were included as potential explanatory variables along with the following site-specific habitat variables: percentage riffle, percentage pools, percentage bedrock, percentage boulder, percentage cobble, percentage gravel, percentage sand, percentage mud, habitat diversity (Simpson's Diversity Index for habitat types), substrate diversity (Simpson's Diversity Index for substrate types), mean depth (m), distance to the source (km), distance to the sea (km), altitude (m), amount of canopy cover (none = 0; rare = 1; light = 2; occasional = 3; medium = 4; frequent = 5; abundant = 6; heavy = 7), slope and river order. Prior to analysis, the explanatory variables were standardised and fish community composition data were Hellinger transformed using the decostand function (of the vegan package). Model selection was carried out with forward and backward selection using the ordiR2step function in order to determine the most important predictors to include in the final RDA. The anova.cca function was used to test the significance of the terms included in the final model and the envfit function was used to determine which species were significantly correlated ($P = < 0.05$) with the habitat variables included in the RDA.

A number of explanatory variables were significant and included in the final RDA models (Figure 3.1 and Table 3.1), accounting for 21% of the variation in the fish communities. The site-specific habitat variables accounted for the majority of the variation in the fish communities, with these variables accounting for 4.2% (mean depth), 2.0% (percentage canopy cover), 2.0% (percentage riffle), 1.4% (percentage mud), 0.1% (slope), 1.0% (habitat diversity), 1.0% (percentage bedrock) and 0.8% (distance to the sea). The only significant flow metric was low_n, which explained just 1.22% of the variation in the fish communities.

A second RDA was conducted on the fish community data to further explore any possible relationship between low- and high-flow events and the fish

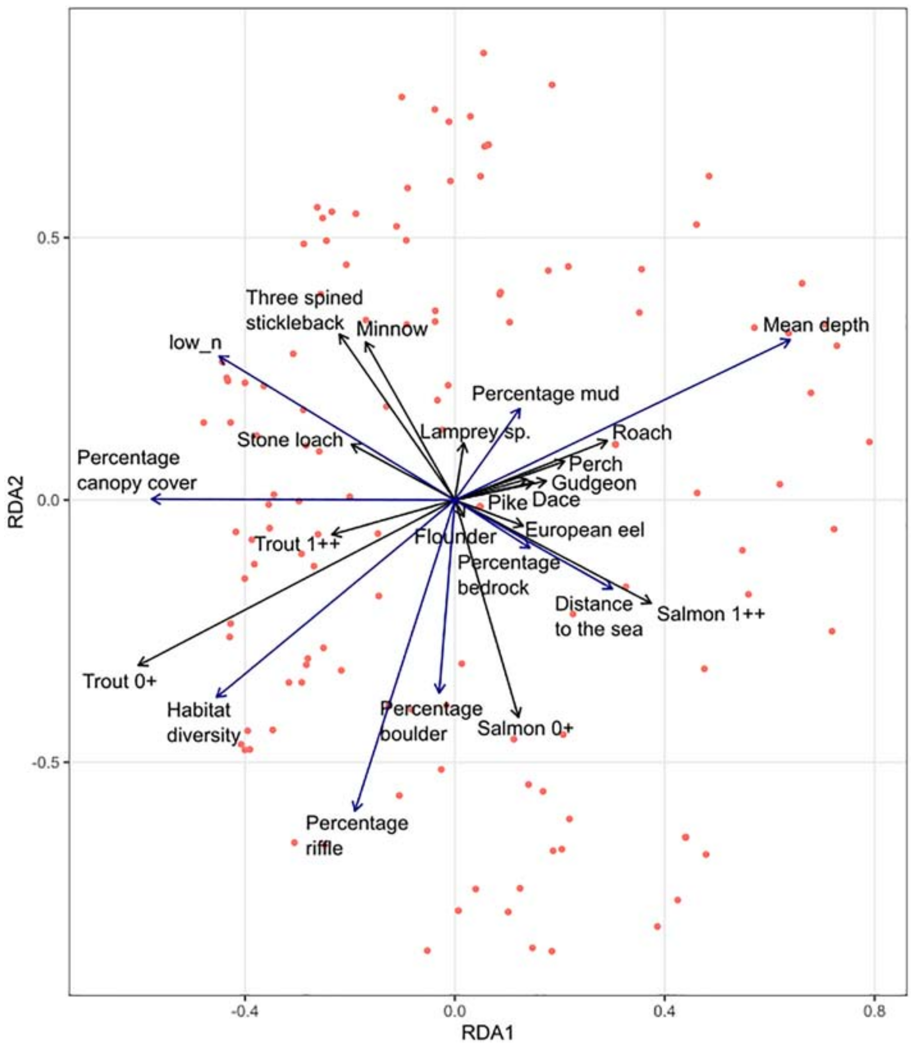

Figure 3.1. RDA biplot showing significant relationships between species and explanatory variables in the final model and indicating the significant explanatory variables that cumulatively explain 21.2% of the variation in the fish communities across surveys. Red dots correspond to individual surveys in the fish community dataset.

Table 3.1. Results of RDA used to estimate the relationship between potential site-specific explanatory variables and fish communities

Explanatory variable	Adjusted R^2	*F* statistic	*P*-value
Mean depth	0.042	12.16	0.001
Percentage canopy cover	0.020	5.88	0.001
Percentage riffle	0.020	5.74	0.001
Percentage mud	0.014	3.95	0.001
Low_n	0.013	3.74	0.002
Slope	0.012	3.53	0.001
Habitat diversity	0.010	2.85	0.007
Percentage bedrock	0.010	2.76	0.013
Distance to the sea	0.008	2.32	0.027

R^2, coefficient of determination.

community. The same site-specific habitat variables used in the previous RDAs were included as explanatory variables. However, instead of using the flow statistics as provided by the hetoolkit (see section 1.3.2.3), a different approach was taken. As floods and droughts have different effects on the biotic community, both effects were considered separately. First, the hetoolkit was used, following the procedure described, to calculate the median flow (Q50) using a window width and a window step of 30 days. In other words, the monthly median flow for every month in the data series was used. Every Q50 value was compared with the overall mean Q50 value from the same station and month, and the result was transformed into a percentage. The months where the Q50 was in the higher 5% were considered to be under flood conditions and the months in the lower 5% to be under drought conditions. For each sample, the number of months under flood and drought conditions during the last 12 months was counted. The results of this second RDA showed that a total of approximately 14% of the variation in fish communities could be explained by site-specific habitat variables, but none of the variation was explained by the selected flow metrics (Table 3.2 and Figure 3.2).

Table 3.2. Results of RDA used to estimate relationships between potential site-specific explanatory variables, as well as numbers of flood and drought months in the preceding 12-month period, and fish communities

Explanatory variable	Adjusted R^2	*F* statistic	*P*-value
River order	0.049	9.82	0.001
Percentage riffle	0.024	4.83	0.002
Distance to the sea	0.024	4.70	0.001
Percentage mud	0.016	3.13	0.004
Percentage boulder	0.015	2.97	0.015
Altitude	0.013	2.66	0.013

R^2, coefficient of determination.

3.2.2 Relationship between salmonids and flow

RDAs examine the effect, if any, that putative explanatory variables have on communities, and, therefore, the effects on individual species or year classes of species can go undetected due to other components of the community having greater influence on the analyses. Therefore, generalised linear models were used to investigate if the densities of salmonids were related to flow regimes. Individual generalised linear models were conducted on 0+ salmon, 1++ salmon (1+ and 2+, combined), all salmon, 0+ trout, 1++ trout (1+ trout and older), all trout, and total salmonid density (number of individuals/m^2), as well as the diversity of the salmonid community, calculated using Simpson's Diversity Index. Each generalised linear model included the calculated flow variables included in the second RDAs, plus the site-specific habitat variables listed in Table 3.2 and also the density of non-native fish species. All species except Atlantic Salmon, Brown Trout, Three-spined Stickleback (*Gasterosteus aculeatus*), Nine-spined Stickleback (*Pungitius pungitius*), European Eel (*Anguilla Anguilla*), Flounder (*Platichthys flesus*) and lamprey species were classified as non-native fish species (Wheeler and Maitland, 1973; Wheller, 1977; Kelly *et al*., 2007; King *et al*., 2011). Data were transformed (Table 3.3) and generalised linear modelling conducted using the glm2 package and quasi-Gaussian distributions. Neither of the flow metrics had a significant impact on the density of salmonids in any of the eight generalised linear models. Both the percentage of riffle habitat (0+ salmon, all salmon, 0+ trout, all salmonids and salmonid diversity) and the distance from the sea (0+ salmon, all salmon, 0+ trout) had significant positive effects on fish density. The percentage of mud as the river substrate had a significant negative effect on the density of all salmon, with altitude having significant positive and negative effects in four of the generalised linear models (Table 3.3).

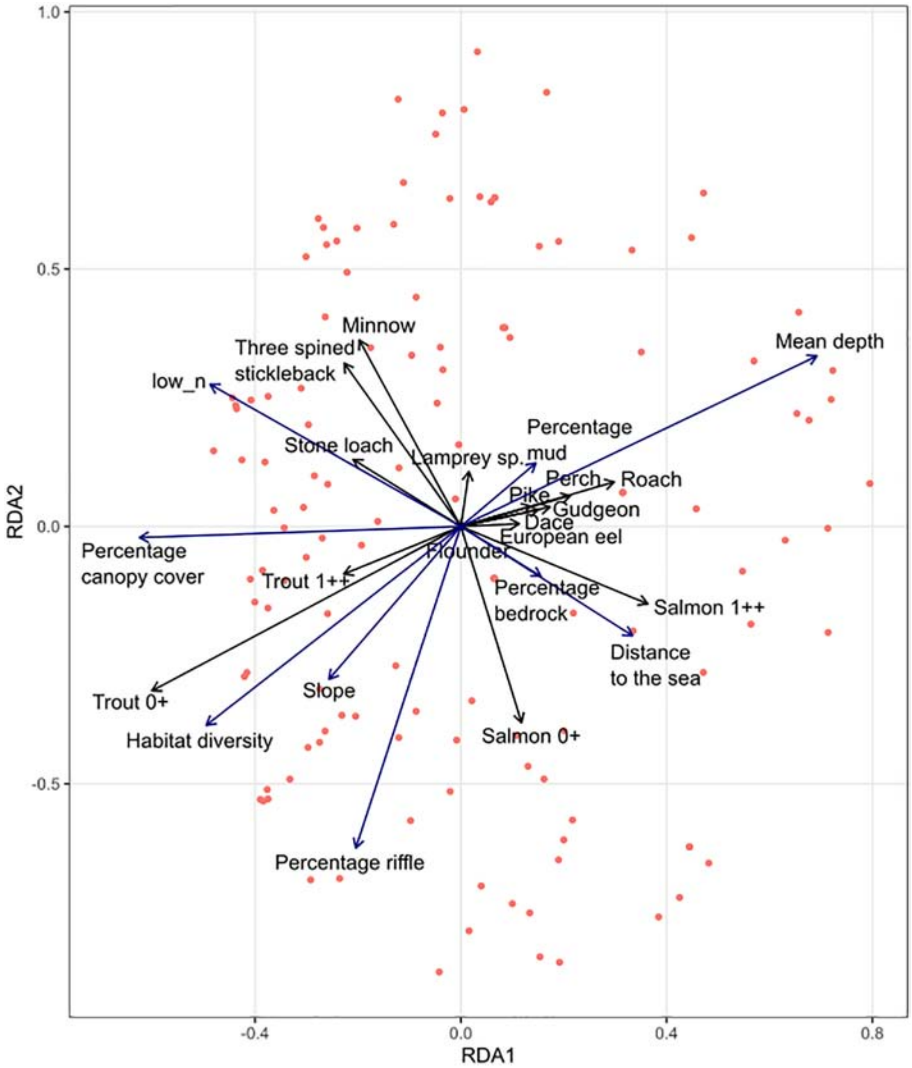

Figure 3.2. An RDA biplot showing significant relationships between species and explanatory variables in the final model and indicating the significant site-specific, but not flow, explanatory variables that cumulatively explain approximately 14% of the variation in the fish communities across surveys. Red dots correspond to individual surveys in the fish community dataset.

3.3 Conclusions

It is clear from the results of the RDAs that habitat structure, such as canopy cover and riverbed composition, explains far more of the variability in fish communities than any of the flow metrics. This finding is similar to those of Kelly *et al*. (2007), who demonstrated that the fish assemblages in Irish rivers are strongly linked to habitat composition. Of all potential explanatory variables, mean depth of river site explained the most variation (6.8%) of fish community structure in the first RDAs. However, depth can obviously be related to flow conditions and therefore it is somewhat unclear if the impact of depth on community structure is due to higher or lower than typical river flow or due to deeper water habitats.

In addition, mean depth was highly correlated with river size and, as the RDAs showed that increasing mean depth was positively associated with roach and perch densities and negatively associated with juvenile Brown Trout densities (Figure 3.1), one might assume that the impact of depth on the fish communities was due to the typical habitat depth, given the preference of roach and perch for deeper river habitats and the preference of young Brown Trout for shallower habitats. For the first RDAs, only 1.03% of the variability in fish communities could be explained by a calculated flow variable, low_n, which represents the number of days during the previous 6 months when flow was below the Q95 value (i.e. 95% of the days had a higher flow than this value). Indeed, for the

Table 3.3. Results of individual generalised linear models of salmonid year classes, species, all salmonids and salmonid diversity (Simpson's Diversity Index) modelled with site-specific habitat variables, density of non-native species and the two calculated flow metrics: the number of previous months that had low-flow or flood conditions

Variable	Estimate	*t*-value	*P*-value	Estimate	*t*-value	*P*-value
	0+ Atlantic Salmon (4th root)			**0+ Brown Trout (cube root)**		
Intercept	0.125	1.83	0.07	0.115	1.60	0.11
Density of non-native fish	0.017	0.18	0.86	–0.070	–0.73	0.47
Percentage riffle	0.003	2.60	0.01*	0.003	2.55	0.01*
Distance from the sea	0.001	2.40	0.02*	–0.001	–2.31	0.02*
Percentage mud	–0.002	–1.73	0.09	–0.000	–0.22	0.82
Percentage boulder	–0.002	–0.74	0.46	–0.002	–0.53	0.60
Altitude	–0.001	–1.12	0.26	0.002	2.56	0.01*
No. of drought months	–0.011	–0.46	0.64	0.009	0.38	0.71
No. of flood months	0.015	0.56	0.57	0.036	1.27	0.21
	1++ Atlantic Salmon (4th root)			**1++ Brown Trout (square root)**		
Intercept	0.116	2.33	0.02*	0.091	2.42	0.02*
Density of non-native fish	–0.108	–1.61	0.11	–0.009	–0.18	0.86
Percentage riffle	0.001	1.49	0.14	0.000	0.26	0.79
Distance from the sea	0.001	2.46	0.02*	–0.000	–1.22	0.23
Percentage mud	–0.002	–1.97	0.05	0.001	1.60	0.11
Percentage boulder	–0.001	–0.19	0.85	0.001	0.80	0.43
Altitude	–0.001	–1.25	0.21	0.001	2.43	0.02*
No. of drought months	0.173	–0.40	0.69	–0.000	–0.04	0.97
No. of flood months	0.023	1.19	0.24	0.015	1.01	0.32
	All Atlantic Salmon (4th root)			**All Brown Trout (4th root)**		
Intercept	0.19	2.69	0.01**	0.292	4.08	< 0.001***
Density of non-native fish	–0.005	–0.06	0.95	–0.038	–0.40	0.69
Percentage riffle	0.003	2.35	0.02*	0.002	1.54	0.13
Distance from the sea	0.001	2.89	0.01**	–0.001	–1.41	0.16
Percentage mud	–0.003	–2.24	0.03*	0.001	1.00	0.32
Percentage boulder	–0.002	–0.51	0.61	0.001	0.17	0.87
Altitude	–0.001	–1.45	0.15	0.002	2.45	0.02*
No. of drought months	–0.019	–0.79	0.43	0.006	0.24	0.81
No. of flood months	0.019	0.68	0.50	0.036	1.26	0.21
	All salmonids (cube root)			**Salmonid diversity (square root)**		
Intercept	0.244	3.46	< 0.001***	0.244	3.46	< 0.001***
Density of non-native fish	–0.027	–0.29	0.77	–0.027	–0.29	0.77
Percentage riffle	0.003	2.17	0.03*	0.003	2.17	0.03*
Distance from the sea	–0.001	–0.77	0.44	–0.000	–0.77	0.44
Percentage mud	0.001	0.32	0.75	0.000	0.32	0.75
Percentage boulder	0.001	0.19	0.85	0.001	0.19	0.85
Altitude	0.002	2.42	0.02*	0.002	2.47	0.16
No. of drought months	0.002	0.06	0.95	0.002	0.06	0.95
No. of flood months	0.029	1.04	0.30	0.029	1.04	0.30

***Significant at < 0.05; **significant at < 0.01; ***significant at < 0.001.**

second RDAs, none of the calculated flow statistics was significant. Similar outcomes were seen for the analyses of the individual life stages of Atlantic Salmon and Brown Trout, and of total salmon, total trout and all salmonids, with none of the selected flow metrics being significant in the respective generalised linear models. Therefore, the results of this project show that available fish data in combination with calculated flow statistics cannot be used to generate a fish community metric to indicate past flow events or to assess the impact of changing flow on fish communities. However, it should be noted that there were limitations that may have affected the results. For example, a limited number of fish surveys had been carried out close to hydrometric stations with continuous mean daily flow data, which meant that sample sizes in this study were relatively small. Of a total of 1797 fish surveys available for analyses, less than 8% could be included in the analyses, mainly because the majority of surveys were conducted at sites that were not located within 5 km of a hydrometric station with the requisite quality of flow data. In addition, functional flow statistics that describe the magnitude, timing, duration, frequency and/or rate of change may be more relevant for fish, and future work should take this into account. Finally, as the fish data that were used in this project were not collected for the purposes of assessing the effects of flow on fish, it is recommended that dedicated surveys for this specific purpose be carried out in the future.

4 Macrophytes

4.1 Introduction

The role of river macrophytes in assessing water quality has been extensively investigated since the WFD came into force in 2000. There has been much debate on their reliability in indicating river water quality (Demars *et al.*, 2012; Baattrup-Pedersen *et al.*, 2017) and thus more focus in recent times on their ability to indicate hydromorphological aspects of the lotic environment (Gebler and Szoszkiewicz, 2022; Vukov *et al.*, 2022). It is well accepted that macrophytes do reflect their prevailing physical surroundings effectively (Haury, 1996; O'Briain *et al.*, 2018; Weekes *et al.*, 2021) and can indicate flow regimes (Januaer *et al.*, 2010). Despite this, there has been limited development of usable macrophyte metrics for this purpose; however, one such metric, the PFI, has been developed by Westwood *et al.* (2021) in the UK to assess the changing hydroecological conditions of temporary rivers and streams. The main principle behind this metric is that plants vary in their tolerance to drying out and, therefore, plant assemblages can change over time in response to changing habitat conditions and water availability. Recent investigations have also shown that macrophyte morphological traits are useful in indicating river hydromorphology (Gurnell *et al.*, 2010; Baattrup-Pedersen *et al.*, 2017; Vukov *et al.*, 2022); plants are categorised according to their growth habit morphology, which were shown to reflect prevailing hydromorphological river conditions, yet there is still a lack of usable plant-trait metrics for river hydromorphological monitoring and assessment.

The objectives of this work were to first calculate and evaluate the PFI in the Irish context, then to develop and evaluate a new proposed macrophyte flow metric named the Macrophyte Morphological Traits Index (MMTI) for Irish rivers, based on easily recognisable macrophyte morphological features.

4.2 Results

Two datasets were used for analysis:
dataset 1 was a collection of 131 river macrophyte samples (89 extracted from the RMD and 42 recorded for this project in 2021) that could be directly related to flow data extracted from nearby hydrometric stations (see section 1.3.2). The advantage of these data was their connection with reliable flow data; however, the size of this dataset is relatively small and it is limited in its range of river habitats, which limits the analytical outcomes. These data were used for initial evaluation of the PFI due to reliable flow data being available. Dataset 2 was a collection of 658 macrophyte samples extracted from the RMD that had a consistent collection of physical data recorded in addition to flow data. The advantage of these data was that there was a wider range of river habitats, from upland streams to more mature rivers; however, flow was measured using various methods across different timelines, which is likely to blur the influence of flow as a key driver of macrophyte distribution. To help mitigate this issue, the flow data were relativised across the dataset. These data were used to further evaluate the PFI and to evaluate the effectiveness of the proposed MMTI at detecting flow.

4.2.1 Calculation of the PFI

The PFI (equation 4.1) is based on 34 indicator species with allocated PFI scores based on their tolerance to changes in water availability. These scores are further weighted by their abundance using the 9-point cover weighting code (CWC) (Holmes *et al.*, 1999).

$$\text{PFI} = \Sigma(\text{PFI Taxon code} \times \text{CWC})/\Sigma\text{CWC} \times 10 \qquad (4.1)$$

Not all scoring species are native to Ireland or recorded consistently to species level; equivalent replacement species were used where possible and choices were based on the Irish river vegetation phytosociological classification system (Weekes *et al.*, 2018). Table 4.1 provides a summary of species changes and issues. Substrate percentage cover values were also scored according to Westwood *et al.* (2021) (more details on methods are available on request from L. Weekes (Department of Biological and Pharmaceutical Sciences, Munster Technological University, Tralee)).

Table 4.1. PFI-scoring species that either are not native to Ireland or have not been recorded consistently to species level in the Irish dataset and their impact on calculating PFI scores for Irish rivers

PFI indicator species	Comment
Myosoton aquaticum	Not native; replaced with *Cardamine flexuosa*
Ranunculus pseudofluitans	Not native and no equivalent Irish replacement
Ranunculus penicillatus *Ranunculus peltatus*	Both species are more difficult to distinguish from each other in Ireland than elsewhere (both morphologically and genetically); grouped together as *Ranunculus* subgenus *Batrachium*, which results in loss of resolution, as both species are in different PFI flow groups
Carex acutiformis *Carex riparia*	*Carex* species not consistently recorded to species level, meaning loss of resolution in the PFI
Glyceria notata *Glyceria maxima*	*Glyceria* species not consistently recorded to species level, meaning loss of resolution in the PFI

4.2.2 Evaluation of the PFI in an Irish context

The PFI scores were examined in relation to Q10z and Q95z lag 1 flows, that is, those recorded 12 months prior to the macrophyte samples being recorded as per Westwood *et al*. (2021); 6-month antecedent flows (Q10z lag0 and Q95z lag0) were also investigated. Parametric modelling techniques were not possible with these data (even after data transformation); therefore, a non-parametric approach was used. Non-metric multidimensional scaling (NMDS) was used to construct an ordination biplot (Figure 4.1) to examine the relationship between the PFI and the distribution of indicator species and to ascertain if the PFI was correlated with any other available environmental factors that may have influenced species distribution. The significance of these factors was tested using a permutation test.

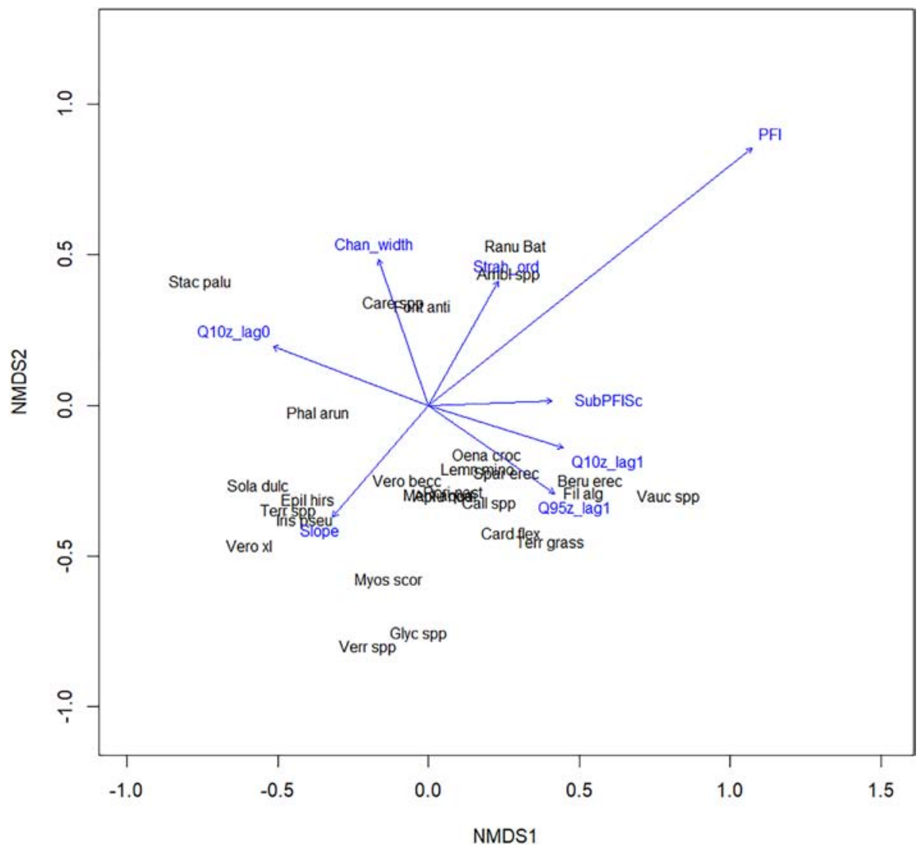

Figure 4.1. NMDS biplot (final stress = 23.2) showing the distribution of the PFI-scoring species in relation to flow, the PFI and other significant environmental drivers.

NMDS indicates that the PFI does not correlate well with flow; this was also confirmed by multiple pairwise tests that were conducted to further investigate the relationships between the PFI, flow and additional available environmental data using Spearman's rank correlation (Figure 4.2 and Table 4.2).

Spearman's ρ values indicate varying degrees of correlation, with $\rho \geq 0.7$ indicating strong correlation, $0.4 \leq \rho \leq 0.7$ indicating moderate correlation, $0.3 \leq \rho \leq 0.4$ indicating weak correlation and $\rho \leq 0.3$ indicating very weak correlation. Values indicating very weak correlations are not reported here, even if significant.

These values show that the PFI had no significant correlation with flow or any of the available environmental factors (altitude (m), Strahler order, channel depth (cm) and width (m), slope (degrees), and substrate type (PFI substrate metric)). However, significant moderate correlations were found between flow and channel width and weak correlations were found between flow and substrate type (Table 4.2). Bryophytes are an integral and important component of Irish macrophyte flora (Weekes *et al*., 2018), especially considering that streams and small rivers, where bryophytes are often dominant, make up more than 77% of the Irish river network (EPA, 2005). The PFI relates only to vascular plants, and some of these species are associated more with canals than streams or rivers in Ireland; thus, the usefulness of the PFI in its present form is limited in the Irish context. The PFI is also likely to be more suited to the drier climate of Britain than to the Atlantic climatic conditions of the island of Ireland (more details on methods are available on request from L. Weekes (Department of Biological and Pharmaceutical Sciences, Munster Technological University, Tralee), particularly as the PFI was originally designed to monitor temporary

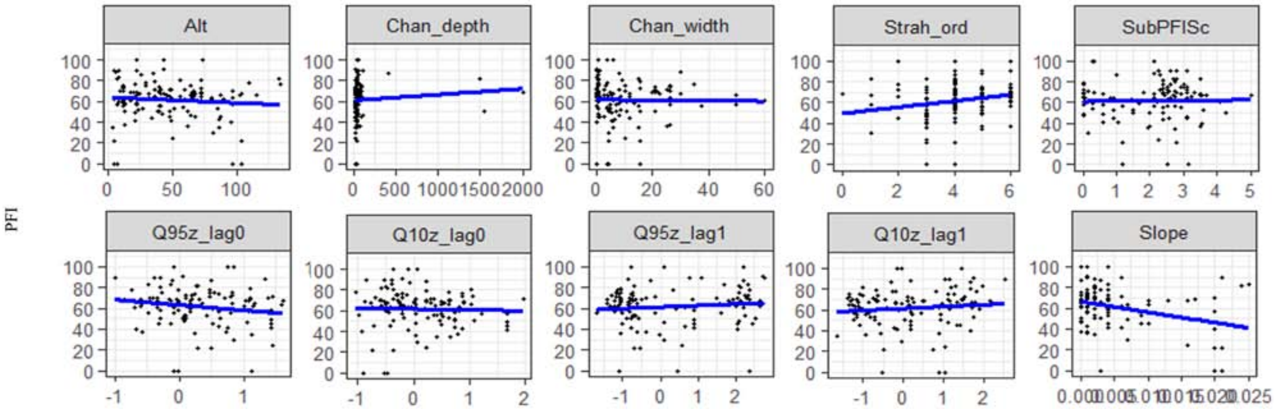

Figure 4.2. Scatterplots of multiple pairwise tests (Spearman's rank correlation) showing the relationship between the PFI and flows Q10z for the antecedent 6 months (lag0) and 12 months (lag1), Q95z lag0 and Q95z lag1 and additional environmental variables. Alt, altitude; Chan_depth, channel depth; Chan_width, channel width; Strah_ord, Strahler order; SubPFISc, substrate.

Table 4.2. Significant correlations with Spearman's *ρ* values and adjusted *P*-values

Correlation strength	Significant correlations	Spearman's *ρ* value	Adjusted *P*-value
Moderate	Altitude vs Strahler order	–0.51	<0.0005
	Flow Q10z lag 0 vs channel width	0.57	<0.0005
	Flow Q95z lag0 vs channel width	0.44	<0.0005
	Flow Q10z lag1 vs channel width	–0.61	<0.0005
	Flow Q95z lag1 vs channel width	–0.64	<0.0005
Weak	Flow Q10z lag 0 vs substrate	–0.37	<0.0005
	Flow Q95z lag0 vs substrate	–0.33	0.01
	Flow Q10z lag1 vs substrate	0.38	<0.0005
	Flow Q95z lag1 vs substrate	0.36	<0.0005
	Slope vs Strahler order	–0.38	<0.0005

watercourses (Westwood *et al*., 2021), although planned development of the PFI may make it more applicable in an Irish context in the future.

4.2.3 Development and calculations of the MMTI

The PFI as it stands is not suitable for use in Irish river systems; however, aspects of its approach can be applied to the proposed MMTI for Ireland. The principle of this new proposed index is based on the river continuum concept for river macrophytes, where morphological changes are observed spatially from the source to the mouth of a river (Muotka and Virtanen, 1995; Matson, 2006; Scarlett and O'Hare, 2006; Gecheva, 2013; Baláži and Hrivnák, 2015; Weekes *et al*., 2018). Macrophyte species have been categorised into 10 morphological groups that reflect prevailing flow conditions. These categories are based on the growth habit of bryophytes and the leaf structure/rooting capacity of vascular plants (Table 4.3). The categories are arranged ordinally, from dominant macrophyte morphology found in fast-flowing/spatey upland streams (Macrophyte Morphological Trait (MMT) value = 1) to dominant macrophyte morphology found in lowland slow-flowing/pooling waters (MMT value = 9). A terrestrial vascular plant group (MMT value = 10) has been added to indicate rivers that have the potential to seasonally dry out, as terrestrial species are important for indicating prolonged periods of drought in river channels (Westwood *et al*., 2021). The choice of species examples and morphological traits in relation to dominant flow regimes is based on a wealth of other research resources in addition to those cited for the river continuum concept above (examples include Heuff, 1987; Preston and Croft, 1997; Haslam, 2006; Hrivnak *et al*., 2010; Janauer *et al*., 2010).

Macrophytes included in these morphological trait groups are those that prevail in the wetted channel and wetted margins (splash zone for bryophytes).

Table 4.3. The 10 morphological trait groups of the proposed MMTI with corresponding MMT values, typical flow conditions, niches and examples of species within each group

MMT value	Morphological trait group	Flow conditions	Niche	Species example
1	Compact cushion-like bryophytes	Torrential/fast – spatey streams	Aquatic/splash zone	*Blindia acuta* *Marsupella emarginata*
2	Mat/weft bryophytes	Fast – streams and small rivers	Aquatic/splash zone	*Hygrohypnum ochraceum* *Chiloscyphus polyanthus*
3	Trailing bryophytes	Moderately fast – streams and small rivers	Aquatic	*Fontinalis antipyretica* *Platyhypnidium riparioides*
4	Linear-leaved vascular plants	Moderately fast – small and larger rivers	Rooted aquatic	*Ranunculus* subgenus *Batrachium* *Sparganium emersum*
5	Ellipsoid-leaved vascular plants	Moderate – small and larger rivers	Rooted aquatic	*Elodea canadensis* *Callitriche* spp.
6	Filamentous macroalgae	Moderate – small and larger rivers	Rooted aquatic	*Cladophora* spp. *Melosira* spp.
7	Round-leaved vascular plants	Slow/still – larger and mature rivers	Rooted aquatic	*Nuphar lutea*
8	Pleustic vascular plants	Slow/still – larger and mature rivers	Not rooted/suspended aquatic	*Lemna minor* *Lemna trisulca*
9	Emergent vascular plants	Slow/still – larger and mature rivers	Rooted with emergent shoots from water, broad leaved and grass/reed like	*Apium nodiflorum* *Rorippa nasturtium-aquaticum* *Phalarus arundinacea*
10	Terrestrial vascular plants	Rivers at any stage that are prone to seasonal drought	Terrestrial species	Grasses (excluding *Phalaris*, *Phragmites* and *Glyceria* ssp.), broad-leaved species

Species excluded from these groups are the thallose liverworts, terrestrial bryophytes and bank vascular plants. Macrophyte percentage cover abundance values in each sample were given a 9-point CWC (Holmes *et al*., 1999). The MMTI was calculated for samples in dataset 2 using a similar formula to the PFI (equation 4.2).

$$\text{MMTI} = \Sigma(\text{MMT value} \times \text{CWC})/\Sigma\text{CWC} \times 10 \qquad (4.2)$$

4.2.4 Evaluation of the MMTI

Other known macrophyte metrics were calculated for comparison with the MMTI; these were the Mean Trophic Rank (MTR) (Holmes *et al*., 1999), the LEAFPACS2 River Macrophyte Nutrient Index (RMNI) (WFD-UKTAG, 2014), the LEAFPACS River Macrophyte Hydraulic Index (RMHI) (WFD-UKTAG, 2008), the PFI (Westwood *et al*., 2021) and abundance-weighted Ellenberg scores (House and Punchard, 2007). It was necessary to follow a non-parametric statistical approach for these data for the same reasons as those explained in section 4.2.2. More details on the methods used are available on request from L. Weekes (Department of Biological and Pharmaceutical Sciences, Munster Technological University, Tralee). Multiple pairwise tests using Spearman's rank correlations were used to compare relationships between the MMTI and the other calculated metrics and additional available environmental factors (channel depth (cm) and width (m), elevation (metres above sea level), substrate (PFI substrate metrics), slope (degrees) and Strahler order) (Figure 4.3 and Table 4.4).

These pairwise tests show that the MMTI was moderately correlated with the other indices except the PFI and weighted Ellenberg values, further confirming that the PFI is not suitable in its present form for Irish rivers. The MMTI was found to be significant but very weakly correlated with flow; however, it was moderately correlated with substrate and weakly correlated with slope (Table 4.4), both of which have been found to be closely related to flow (Grinberga, 2010; Gecheva *et al*., 2013). NMDS was carried out and an NMDS ordination biplot was constructed (excluding the RMNI, RMHI and MTR due to being strongly auto-correlated). This biplot (Figure 4.4) further shows that the MMTI lines up well with changing hydromorphological conditions; for example, the MMTI increases as slope, elevation and substrate size decrease, which is expected (the higher the MMTI, the slower the flow). The MMTI vector also follows a similar direction to the flow vector. The species distributions also follow an expected distribution pattern, from small upland fast-flowing streams dominated by bryophytes (bottom left of Figure 4.4, e.g. *Scapania undulata* – Scap undu, *Hyocomium armoricum* – Hyoc armo) to slower flowing, more mature rivers dominated by vascular

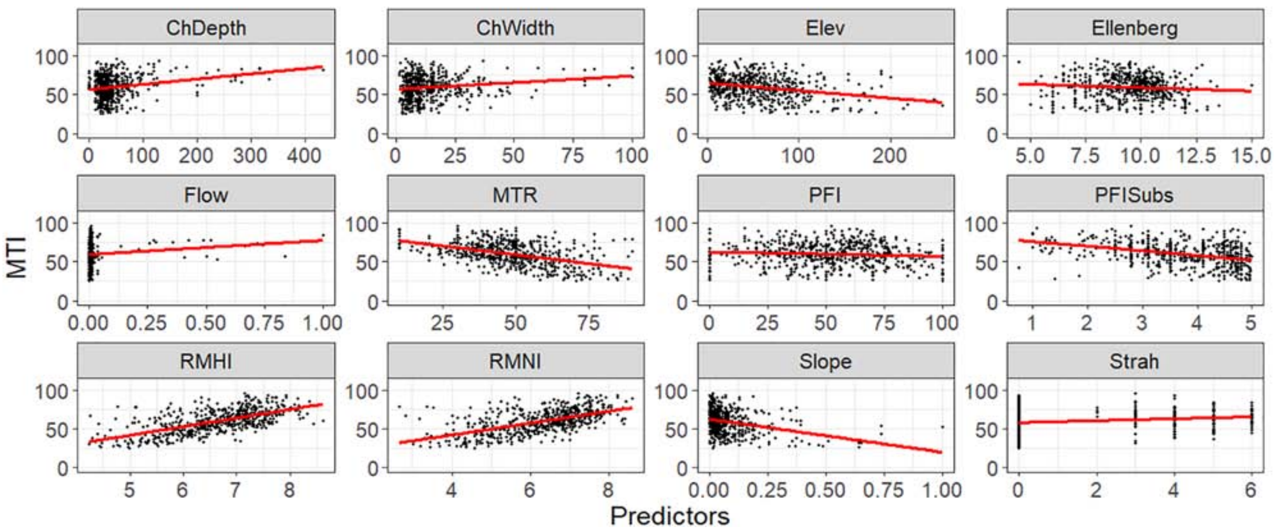

Figure 4.3. Scatterplots of multiple pairwise tests (Spearman's rank) showing the relationship between the MMTI and flow and known macrophyte metrics (weighted Ellenberg values, MTR, RMNI, RMHI, PFI) and additional environmental variables. ChDepth, channel depth; ChWidth, channel width; Elev, elevation; Strah, Strahler order; PFISubs, substrate type.

Table 4.4. Significant Spearman's ρ correlation values and adjusted *P*-values of macrophyte indices and available environmental variables found when evaluating the MMTI

Correlation strength	Significant correlations	Spearman's ρ value	Adjusted *P*-value
Strong	RMNI vs RMHI	0.93	< 0.0005
	RMNI vs MTR	–0.84	< 0.0005
	RMHI vs MTR	–0.75	< 0.0005
Moderate	MMTI vs RMHI	0.63	< 0.0005
	MMTI vs RMNI	0.58	< 0.0005
	MMTI vs MTR	–0.45	< 0.0005
	MMTI vs substrate	–0.41	< 0.0005
	Flow vs channel width	0.50	< 0.0005
	PFI vs Ellenberg value	0.48	< 0.0005
	Slope vs channel depth	–0.41	< 0.0005
	Slope vs substrate	0.48	< 0.0005
	Slope vs MTR	0.45	< 0.0005
	Slope vs RMNI	–0.51	< 0.0005
	Slope vs RMHI	–0.56	< 0.0005
	Substrate vs RMNI	–0.41	< 0.0005
	Substrate vs RMHI	–0.47	< 0.0005
Weak	MMTI vs slope	–0.33	< 0.0005
	Flow vs Strahler order	0.36	< 0.0005
	RMHI vs Ellenberg value	0.36	< 0.0005
	Elevation vs channel width	–0.36	< 0.0005
	Elevation vs RMNI	–0.30	< 0.0005

macrophytes (top right of Figure 4.4, e.g. *Sparganium erectum* – Spar erec, *Berula erecta* – Ber erec). The multiple pairwise tests and the NMDS show that it is difficult to isolate any one physical variable, such as flow, from the others, which has long been a challenge in studying river macrophytes (Suren and Duncan, 1999; Grinberga, 2010; Weekes *et al*., 2021).

4.3 Conclusions

The British PFI is not suitable for Irish river systems in its present form; reasons include the exclusion of bryophytes in the metric, indicator species not being optimised for Irish flora and the differing climatic conditions in Ireland. The results suggest that the proposed MMTI is a suitable index for characterising a combination of prevailing hydromorphological conditions, including flow, and that it could potentially be used for monitoring the lotic environment. However, it should be noted that the signal for flow was blurred in these analyses, mainly due to the RMD being composed of heterogenous data. It is also well accepted that it is difficult to disentangle the effects of various physical environmental factors on macrophyte distribution. Therefore, the index proposed will need to be further tested and developed before being incorporated into any hydromorphological monitoring programme. Temporal changes in prevailing flow conditions could be detected if comprehensive baseline macrophyte surveys were conducted and tested against closely associated and reliable hydrometric data.

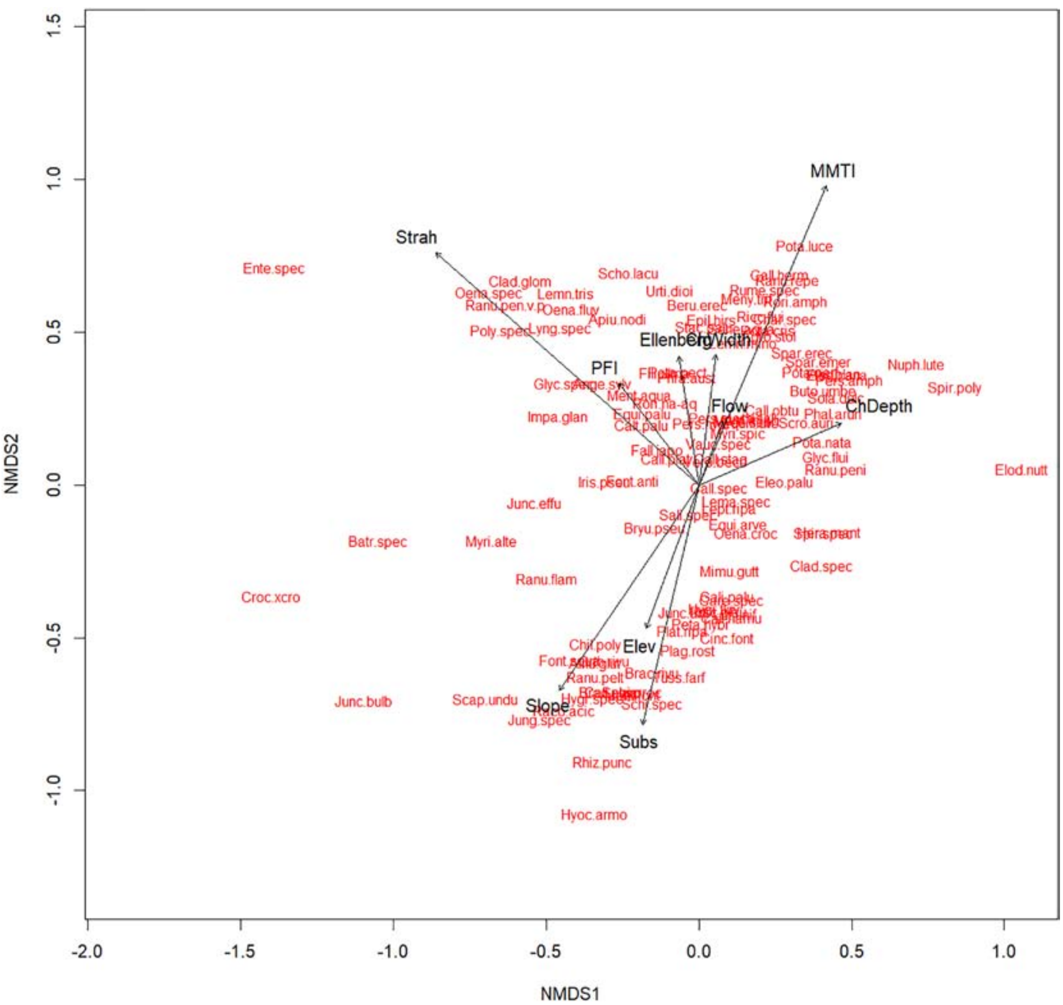

Figure 4.4. NMDS biplot (final stress = 23.6) showing the distribution of species in relation to flow, the MMTI, the PFI and other significant environmental drivers.

5 Multi-metric Indices

5.1 Introduction

The main objective of this work was to investigate whether multi-metric hydroecological indices for flow can be usefully applied in Irish river monitoring programmes, particularly to detect the effects of extreme hydrological events (floods and droughts) in Irish rivers using metrics derived for three major taxon groups (macroinvertebrates, macrophytes and fish), based on some of the work outlined in Chapters 2, 3 and 4. A multi-metric index (MMI) is a mathematical formula that combines several metrics to produce a unique score. The MMI score is expected to more accurately reflect conditions than a single metric (Hering *et al*., 2006). MMIs were initially developed to assess water quality in freshwater ecosystems (Karr, 1981), and since their inception the knowledge and application of these indices has expanded. For example, the number of region-specific MMIs has increased greatly (Vadas *et al*., 2022), as has the number of MMIs used to assess general ecosystem status (Böhmer *et al*., 2004; Lorenz *et al*., 2004), hydromorphological modification (Timm *et al*., 2011; Käiro *et al*., 2012) and extreme flow events, particularly droughts (Straka *et al*., 2019; Theodoropoulos *et al*., 2020). MMIs have several advantages over single metrics. A single metric responds to a certain range of variation in the condition being assessed; including additional metrics can increase this range (Hering *et al*., 2006). In some cases, certain external factors can have a strong effect on a metric (Theodoropoulos *et al*., 2020), dampening its response to the condition of interest; therefore, combining several metrics can ensure a more robust assessment. Furthermore, it is possible to control such external factors by using MMIs based on different metrics for different subsets of samples, such as river type (Böhmer *et al*., 2004), sampling season (Straka *et al*., 2019) or water quality (Theodoropoulos *et al*., 2020). All in all, MMIs tend to be more flexible than single metrics, often producing more reliable estimates of the conditions that are being assessed (Herring *et al*., 2006).

A number of approaches can be followed to create an MMI (Hering *et al*., 2006; Pearson *et al*., 2011), but some important considerations remain true for all approaches. Every individual metric included as part of the MMI should react to the condition that the MMI assesses (Theodoropoulos *et al*., 2020), and a causal link behind the relationship between the metric and the condition being assessed is required. The combination of metrics should provide a more robust and accurate prediction than that produced by a single metric (Pearson *et al*., 2011). The selected metrics should also reflect different aspects of the ecosystem; for example, they should use different components of the biota and/or use different methods for measuring variation in the biota, such as species diversity, functional diversity and abundance (Straka *et al*., 2019). All MMIs should be tested using an independent dataset from the one used to create them (Böhmer *et al*., 2004; Vadas *et al*., 2022).

The main aim of this work was to develop an MMI that (a) is specific to Irish rivers; (b) is based on macroinvertebrate, fish and macrophyte data (in any combination); (c) uses data gathered using current Irish survey methodologies; and (d) can accurately detect the effects of antecedent extreme flood events at river-reach scale.

5.2 Results

5.2.1 Bibliographic review

A literature review was conducted following a rapid evidence assessment, using the methodology described in Collins *et al*. (2015). Before the literature review begins, the search strategy must be clearly defined: first, the specific objectives of the review are defined; second, the search terms and the search engines to be used are chosen; and, third, a clear strategy for discarding the least relevant literature is described. This strategy drastically reduces the time investment needed to find and read relevant literature, while being transparent and reproducible, and hence it was the approach chosen for this literature review. Using this approach, eight MMIs were identified as candidates for use in Irish rivers (Table 5.1), six based on macroinvertebrate metrics (German Fauna Index, Hellenic Flow Index, Biodrought Index, Macroinvertebrates in Estonia: Score of

Table 5.1. Candidate MMIs for measuring flow conditions in Irish rivers

MMI	Biotic indicator	Region	Relationship with flow	Reference
GFIn	Macroinvertebrates	Europe (Germany)	Not correlated	Böhmer *et al.*, 2004; Lorenz *et al.*, 2004
ELF (clean rivers)[a]	Macroinvertebrates	Europe (Greece)	Q10z: r(781) = –0.174 $P<0.001$	Theodoropoulos *et al.*, 2020
ELF (polluted rivers)[a]			Q10z: r(781) = –0.125 $P<0.001$	
Biodrought (spring sampling)	Macroinvertebrates	Europe (Czechia)	Not adaptable	Straka *et al.*, 2019
Biodrought (autumn sampling)			Not correlated	
MESH	Macroinvertebrates	Europe (Estonia)	Not correlated	Timm *et al.*, 2011; Käiro *et al.*, 2012
MSCI[a]	Macroinvertebrates	North America	Q10z: r(781) = –0.165 $P<0.001$	McCord *et al.*, 2009
NRSA Benthic	Macroinvertebrates	North America	Not adaptable	Waite *et al.*, 2021
NRSA Fish	Fish	North America	Not adaptable	Waite *et al.*, 2021
GRFIn	Fish	North America	Not adaptable	Pearson *et al.*, 2011; Taylor *et al.*, 2013

The table includes the biotic indicator, the region where the MMIs were developed and their relationship with the flow statistics (Q95z, Q10z and min_7day_z) calculated for Irish rivers.

[a]MMIs significantly correlated with Q10z.

ELF, Hellenic Flow Index; GRFIn, Great-River Fish Index; MESH, Macroinvertebrates in Estonia: Score of Hydromorphology; MSCI, Missouri Stream Condition Index.

Hydromorphology, Missouri Stream Condition Index, National Rivers and Streams Assessment (NRSA) Benthic Multimetric Index) and two on fish-based metrics (NRSA Fish Multimetric Index, Great-River Fish Index). Four MMIs could not be adapted using the available data, either because they were partially based on categories (taxonomical and functional) not found in Ireland (both fish-based MMIs) or because they required metrics not recorded by the actual survey methods applied in Irish rivers (the spring sampling variant of the Biodrought Index and the NRSA Benthic Multimetric Index). The rest of the MMIs were adapted to Irish rivers and their scores calculated using the historical data for the same surveys and sites as described in section 2.1. The scores of these MMIs were tested for correlation (using Pearson correlation test, $\alpha = 0.08$, as in Böhmer *et al.*, 2004) with the hydrological parameters described in section 1.3.2.3: Q10z, Q95z and min_7day_z. Three of the MMIs were significantly correlated with the Q10z statistic, but the strength of the correlation was less than 0.20 in every case, indicating that these MMIs will not be precise enough to predict flood conditions consistently (Table 5.1).

5.2.2 Hydrological conditions

In creating a new MMI for Irish rivers, the first step was to determine which conditions the MMI would be used to assess. Instead of using the flow statistics previously calculated by the hetoolkit (section 1.3.2.3), a different approach was taken. As floods and droughts have different effects on the biotic community, both effects were considered separately. First, the hetoolkit was used to calculate the monthly median flow (Q50) for every month in the data series. Every Q50 value was compared with the overall mean Q50 value from the same station and month, and the result was transformed into a percentage. The months where the Q50 was in the higher 5% were considered to be under moderate flood conditions, the months in the higher 1% to be under extreme flood conditions, the months in the lower 5% to be under moderate drought conditions and the months in the lower 1% to be under extreme drought conditions. This information was recorded as the variables flood intensity (*FI*) and drought intensity (*DI*): the *FI* score of a given month was 2 if the month was under extreme flood conditions, 1 if it was under moderate flood conditions and 0 in any other case; the same scoring system was used for the *DI* variable.

As antecedent flow conditions over extended periods can have an impact on river biota, two periods of time were considered: 12 months ($t=12$) and 24 months ($t=24$). The following variables were calculated for each month and hydrological station using equations 5.1 and 5.2: flood conditions during the last 12 months (F_{12}); flood conditions during the last 24 months (F_{24}); drought conditions during the last 12 months (D_{12}); and drought conditions during the last 24 months (D_{24}). The parameter m is the number of months ago, with FI_m being the value of FI m number of months ago (e.g. in a given sample, if 4 months ago the flood conditions were extreme, then the FI_4 for that sample would be equal to 2 (equation 5.1)) and DI_m being the value of DI m number of months ago (equation 5.2).

$$F_t = \frac{10}{t}\sum_{m=1}^{t}(1+t-m)\times FI_m \tag{5.1}$$

$$D_t = \frac{10}{t}\sum_{m=1}^{t}(1+t-m)\times DI_m \tag{5.2}$$

The quantiles of the flow variables (F_{12}, F_{24}, D_{12} and D_{24}) were calculated across the complete dataset; the quantile 0.66 was the upper threshold for reference flow conditions and the quantile 0.95 was the lower threshold for extreme flow conditions.

5.2.3 Multi-metric index development

To develop a new MMI, a similar approach to the one described by Böhmer *et al*. (2004) was taken: first, a number of metrics based on the biotic indicators were selected; second, the metrics correlated with the flow variables were classified as candidate metrics; third, the candidate metrics were combined in different MMIs; fourth, the MMIs correlated with the flow variables were classified as candidate MMIs; and, finally, the candidate MMIs were tested using an independent dataset. Macroinvertebrate and fish metrics were selected to represent four descriptive categories of the biotic communities: sensitivity to flow, diversity/richness, composition/abundance and functional guild. The metrics included were selected based on the results of work described in Chapters 2 and 3, and metrics that were identified during the bibliographic review. Some metrics could not be included because of limitations of the data (e.g. metrics based on certain functional macroinvertebrate guilds), and, given the low number of samples including fish data ($n=166$) and the reduced number of fish metrics that were possible to calculate for these data (two composition/abundance metrics and four diversity/richness metrics), fish metrics were discarded from further analyses (Table 5.2). As only a small amount of macrophyte data were available, and no suitable macrophyte

Table 5.2. Macroinvertebrate metrics by category

Macroinvertebrate metrics	Correlated flow variables
Sensitivity to flow	
LIFE[a]	F_{12}, F_{24}, D_{12}
DELHI index[a]	F_{12}, F_{24}
E-PSI index	No correlation
Diversity/richness	
Number of EPT[a]	D_{24}
Number of Plecoptera[a]	F_{12}, F_{24}, D_{24}
Number of Trichoptera[a]	D_{24}
Simpson's Diversity Index	No correlation
Shannon Diversity Index	No correlation
Composition/abundance	
Percentage EPT	No correlation
Percentage Crustacea	No correlation
Percentage Plecoptera[a]	F_{12}, F_{24}
Percentage Trichoptera	No correlation
Percentage GOLD[a]	F_{12}, F_{24}
Functional	
Percentage taxa associated with rapid and moderate to fast flows[a] (see Extence *et al*., 1999)	F_{12}

[a]Candidate metrics and their correlated flow variables (at $\alpha=0.08$).

EPT, Ephemeroptera, Plecoptera and Trichoptera; GOLD, Gastropoda, Oligochaeta and Diptera.

MMIs were found during the bibliographic review, macrophyte data were not used for MMI development.

The biological samples were matched with the hydrological samples by location and date. Pearson correlation tests between each metric and the four flow variables (F_{12}, F_{24}, D_{12} and D_{24}) were carried out, and the metrics that were significantly correlated (at $\alpha = 0.08$, as in Böhmer *et al*., 2004) were selected as candidate metrics to include in the MMIs for each one of the flow variables.

The dataset was randomly divided into two subsets: the first subset ($n = 494$) was used to develop the candidate MMIs and the second subset ($n = 452$) was used to test them. In the first subset, the candidate metrics were transformed and normalised according to Blocksom (2003): a value of 1 always represented reference flow conditions and a value of 0 extreme flow conditions (equation 5.3). Any value above 1 was set as 1, and any value below 0 was set as 0.

$$\text{scored metric} = \frac{\text{value for test sample} - \text{threshold value for extreme conditions}}{\text{threshold value for reference conditions} - \text{threshold value for extreme conditions}} \tag{5.3}$$

To create the MMIs, one candidate metric from each category was selected; the MMI was then calculated as the mean of the selected candidate metrics. The different combinations of candidate metrics resulted in 11 MMIs (2 for F_{12} and 9 for F_{24}). They were tested for correlation with their respective flow variable using a Pearson test (at $\alpha = 0.08$, as in Böhmer *et al*., 2004). One of the 11 tested MMIs was selected as a candidate MMI, as it was significantly correlated with the flow variable F_{12} ($r(297) = -0.107$, $P = 0.064$). This MMI included as metrics the LIFE score of the sample, the number of Plecoptera in the sample and the percentage of Plecoptera in the sample. Following the same procedure, the candidate MMI was calculated for the second data subset and tested for correlation with the F_{12} variable using the Pearson test ($r(286) = -0.104$, $P = 0.077$) (Figure 5.1). The negative correlation is expected, as lower values in the MMI indicate conditions closer to the reference, which are defined as low values in F_{12}.

To account for the differences between biotic communities produced by different environmental conditions across rivers (Böhmer *et al*., 2004), the complete dataset was divided by river type and also by slope, using Irish river typologies (Hannigan and Kelly-Quinn, 2016). In the first case, MMIs were developed for the three typologies with more than 100 samples (type 12, $n = 106$; type 31, $n = 186$; type 32, $n = 180$)

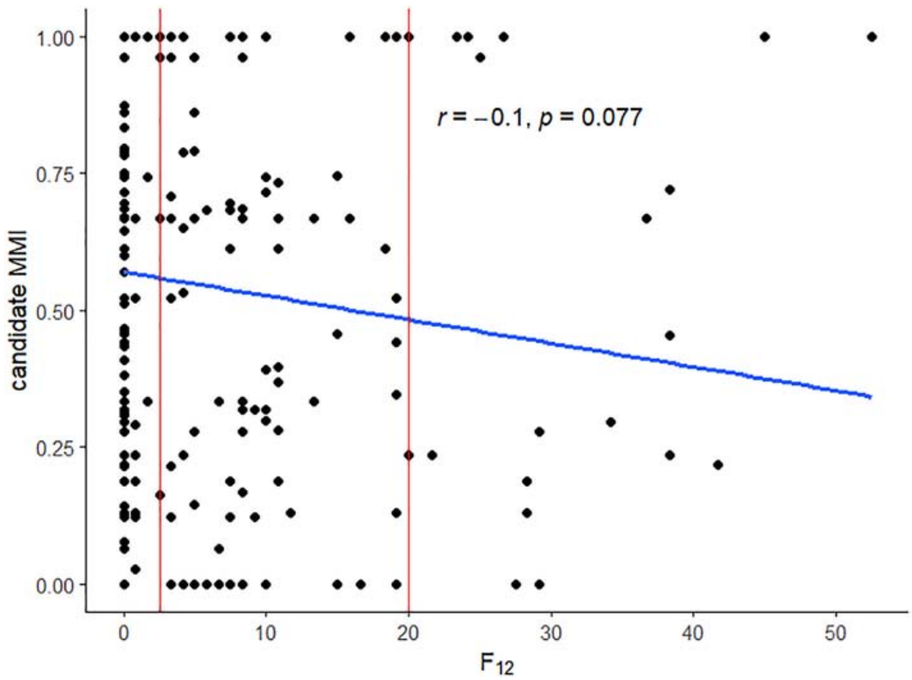

Figure 5.1. Scatterplot showing the relationship between the candidate MMI and the F_{12} variable. The plot shows the *r* value and *P*-value. The regression line is in blue and the red lines show the quantiles 0.66 and 0.95 of F_{12}.

using the same procedure as described above. In the second case, MMIs were developed for each of the four slope classes (low, $n = 203$; medium, $n = 321$; high, $n = 153$; and very high slope, $n = 106$). In all cases, no MMI was significantly correlated with the flow variables of the second data subset.

5.3 Conclusions

Neither the candidate MMI that was developed nor the potentially suitable MMIs that were identified in the bibliographic review appear to be suitable for reliably assessing the occurrence of extreme flow events in Irish rivers, given the low strength of the correlations between metric values and flow variables. Nevertheless, a number of relevant metrics and a valid approach for generating a flow MMI for Ireland that can be further developed in the future were identified. Moreover, the development process and the results obtained shed light on the next steps to follow.

The bibliographic review revealed several restrictions on the development of a new flow MMI for Irish rivers. First, no MMI using macrophytes that reacted consistently to flow conditions was found, indicating a clear knowledge gap. Second, all MMIs based on fish were from the USA, and given the differences between fish communities in North America and Ireland those metrics could not be adapted. Moreover, Irish fish communities are relatively simple (Roset *et al.*, 2007), so it was not possible to use functional metrics or flow sensitivity metrics for fish, and no strong relationships between flow and Irish fish communities were found when fish were considered alone (see sections 3.2 and 3.3), although this may have been due to limitations of the available dataset. And, third, although some of the macroinvertebrate MMIs that were identified are promising, it is recommended that specific flow sensitivity metrics be developed for the macroinvertebrate fauna of Ireland, as macroinvertebrates are one of the principal components used in flow MMIs from other regions.

Another necessary step towards the assessment of flow conditions in Irish rivers is to consider the flow variables in more detail. In this work, specific periods of time (1 and 2 years) were used, as were specific intensity thresholds, to define extreme flow events. In future work, it is recommended that other time periods and thresholds be considered, to determine whether these would result in stronger correlations between the flow variables and the biotic metrics.

Several limitations of the biological data hindered the capacity to calculate reliable metrics. First, the lack of overlapping monitoring sites for fish and macroinvertebrate surveys made it impossible to combine both biological communities when developing MMIs. Second, it was not possible to calculate several macroinvertebrate metrics given the taxonomic resolution of the data and the lack of absolute abundance data. Moreover, some of the metrics that were included in this work were calculated using estimates of the absolute and relative abundances, which is likely to have reduced their accuracy and performance. Therefore, it is desirable to improve the sampling methodologies and designs used in Irish river monitoring programmes to better represent the biotic and environmental characteristics of Irish rivers. All in all, although the goal of creating a new MMI for Irish rivers was not achieved, there are clear indications that such an index could be developed successfully in the future.

6 Recommendations

The overall aim of this project was to investigate whether data gathered during biological monitoring programmes in Irish rivers could be used to measure the impacts of changes in flow on ecology, and to determine whether hydroecological monitoring tools could provide useful additional information for assessing the ecological status of Irish rivers. The results of the project provide evidence to suggest that macroinvertebrate- and macrophyte-based hydroecological indices can potentially be used to estimate effects of changing flow conditions in Irish rivers and that they could therefore contribute to the assessment of ecological status. No fish-based hydroecological indices that were suitable for use in Ireland were identified, and relationships between fish communities and flow were weak, suggesting that fish data may be less useful for assessing the impact of changes in flow in Irish rivers. However, it should be noted that the weak relationships found between fish communities and flow may have been due to limitations with the available fish dataset, and therefore more targeted surveys in the future could yield different results. Individual hydroecological indices were also combined in various ways to generate several MMIs, but no strong relationships were found between the MMI values calculated and flow.

The results from this project show that the adapted family-level LIFE index, the adapted DEHLI index and the E-PSI index can be used to measure impacts on ecology and track changes in flow, and potentially sediment, conditions in Irish rivers; it is therefore recommended that these macroinvertebrate-based hydroecological monitoring tools be incorporated into Irish river monitoring programmes to monitor any changes in macroinvertebrate communities that may reflect deteriorations in ecological status due to changes in flow and sediment. The incorporation of these hydroecological indices into Irish river monitoring programmes should be relatively easy to achieve, as the adjustments that were made to the indices in this project mean that they can be used with macroinvertebrate data that are already collected as part of EPA river monitoring programmes. Furthermore, an online dashboard for calculating the indices was developed and is available for use at https://mgammell.shinyapps.io/biotic_index_dashboard/. Although there are versions of the LIFE and E-PSI indices that can be calculated using species-level data, the species-level and family-level versions of the indices were strongly correlated with each other; therefore, the current taxonomic resolution used in EPA macroinvertebrate monitoring is sufficient for calculating these indices. However, species-level identifications should result in more accurate calculations of the biotic indices, and should be considered if feasible. Even if a change to species-level recording is not possible, a change in the way that EPA macroinvertebrate abundance data are recorded is recommended. Currently, data on macroinvertebrate taxon abundances are recorded in one of six categories (single, few, common, numerous, dominant, excessive), but the LIFE and E-PSI indices use simplified logarithmic abundance categories in their calculations (A = 1–9 individuals; B = 10–99 individuals; C = 100–999 individuals; D = 1000+ individuals). It is likely that the index values calculated for Irish rivers would be more accurate if these simplified logarithmic abundance categories were also recorded during macroinvertebrate surveys (in addition to the abundance categories that are currently recorded), so that they could be used in index calculations; these logarithmic abundance categories would also be beneficial for the calculation of MMIs.

There are additional advantages to using these macroinvertebrate hydroecological indices in Irish river monitoring. For example, the *Water Action Plan 2024: A River Basin Management Plan for Ireland* (DHLGH, 2024) highlights hydromorphological pressures as the second-most significant category of pressure on Irish rivers and proposes the restoration of more natural hydromorphological conditions, primarily by removing or modifying barriers to flow. It is therefore recommended that these hydroecological indices be used to monitor the responses of macroinvertebrate communities to such restoration work, by conducting macroinvertebrate surveys and calculating hydroecological indices before and after restoration. Furthermore, the *Review of Ireland's Heavily Modified Water Body Designations for the Third Cycle River Basin Management Plan* (EPA, 2022) identified 433 heavily modified rivers, which are unlikely to achieve good ecological status, as restoration measures would affect the specified use of the modified river; they will, however, be expected to achieve good ecological potential through the development of appropriate programmes of measures as required. The authors of the review noted that hydromorphological impacts on the biological community in heavily modified rivers may be masked because biological data gathered as part of river monitoring programmes in Ireland are not sensitive to hydromorphological changes. However, this project has shown that macroinvertebrate hydroecological indices do respond to changes in flow in Irish rivers, and they should therefore be used as part of the assessment of the effectiveness of programmes of measures for Ireland's heavily modified rivers. One mitigation measure frequently mentioned in the review is sediment control; the E-PSI index in particular could be used to assess effects of sediment control on the macroinvertebrate community. Finally, these macroinvertebrate indices should also be incorporated into methods for monitoring the effects of abstractions on river macroinvertebrate communities, as required under the Water Environment (Abstractions and Associated Impoundments) Act 2022, and should also be used in the development of guidelines and legislation in relation to minimum environmental flows required to support ecology in Irish rivers. If an abstraction alters a river's flow regime to a degree that significantly impacts on the macroinvertebrate community, it should be possible to detect this change using the LIFE index. If an abstraction increases incidences of channel drying, it should be possible to detect this using the DEHLI index, and, if it increases the amounts of fine sediment in a river, it should be possible to detect this using the E-PSI index. To detect such impacts, index scores would also need to be calculated before any abstraction occurs and/or expected index scores would need to be calculated for relevant river stretches. As noted previously, Webster *et al*. (2017) identified a major knowledge gap with regard to flow–ecology relationships for assessing risks associated with water abstraction; the results of this project have contributed to filling that gap. The three macroinvertebrate indices should also respond to river regulation effects when flow is controlled by non-varying releases from reservoirs or sluices, for example. Furthermore, the use of relevant predictive tools, such as the River Invertebrate Prediction and Classification System in the UK, may help to disentangle hydromorphology and water quality impacts, to identify the predominant pressure affecting a site (Extence *et al*., 1999).

In this project, there was little to no relationship between the available historical fish community data and the calculated metrics of flow, and therefore it was not possible to generate a fish index for flow using these data. However, it must be noted that the fish data provided by IFI were not collected with the aim of elucidating relationships affected by variations in flow, but rather as part of IFI's monitoring to assess the health and classify the ecological status of Irish river habitats. Over the period (2008–2020) for which historical fish data were available, IFI conducted a considerable 1797 electrofishing surveys from a total of 1135 sites. However, many of these survey sites were not within 5 km of a hydrometric station with good-quality flow data, resulting in a reduction in suitable data of approximately 90%, to 169 surveys from 103 sites. A further reduction in surveys prior to modelling was required, as study stretch habitat data were missing from some sites, resulting in a final total of fish community data from 92 surveys at 60 sites being included in the RDAs. An increase in the number of surveys that could be included in such analyses would obviously result in greater statistical power to detect significant relationships between flow and fish community metrics and would facilitate the inclusion of first-order interactions between explanatory variables in models.

Electrofishing surveys for IFI monitoring of rivers of up to 10 m in width typically involve single-pass timed electrofishing surveys typically lasting 10 minutes, rather than resource-intensive density estimates from area-delineated multi-pass depletion electrofishing. Results are then converted to minimum density estimates for each species, which are calculated using species-specific size-based conversion factors that facilitate extrapolation to density estimates. These extrapolations permit a considerable number of electrofishing surveys to be completed every year over large spatial scales (Matson *et al*., 2018). This balance between the more accurate density estimates from area-delineated multi-pass depletion electrofishing and the number of surveys that can be conducted was obviously an important consideration when designing this monitoring programme to ensure achievement of its objectives (Matson *et al*., 2018). While fish densities calculated from multi-pass depletion electrofishing would be considered more accurate than single-pass timed electrofishing surveys, this is not thought to be a limiting factor in identifying flow-based effects on the fish assemblages. The RDAs did identify those explanatory variables that explained variability in the fish communities (water depth, substrate type, etc.) but were unable to detect any significant impact of the calculated flow statistics. Therefore, one must assume that the fish data are of sufficient quality to address the research questions.

The very limited level of variation in the fish communities explained by the calculated flow statistics may, the authors believe, be lower than the level of variation explained in macroinvertebrates and macrophytes because of the behavioural response capabilities of fish. Fish can and do respond to changes in flow by altering their position within a river by taking refuge in pools during drought events (O'Grady *et al*., 2008). During high-flow events, fish are capable of responding to increased water velocity by moving from locations selected primarily to optimise feeding opportunities to microhabitats within rivers where water velocity is reduced due to eddying effects of boulders and other stream-bed components. The calculated flow statistics used in the analyses in this study were based on water velocity, but the impact of water velocity on fish communities is highly dependent on the physical structure of rivers. Such hydromorphological impacts as the interaction between water flow and riverbed structure is a very important consideration when studying the effects of changing flow regimes on fish communities and should be incorporated into future efforts to develop fish community-based metrics of flow impacts in rivers.

The resources required to design a bespoke project aiming to collect fish abundance data and associated site-specific habitat data that can be linked to good-quality flow regimes and, therefore, maximise the likelihood of generating a functional fish-based flow index, would be considerable. However, perhaps some investment in IFI's WFD fish monitoring programme, or other related research programmes, could facilitate this. In the event of such a scenario, the following recommendations may need to be considered:

- Increasing the number of surveys included in the analyses by investing in more flow gauges and/or additional sites to ensure that more survey locations, and hence surveys, could be matched to good-quality flow data.
- Ensuring that future studies benefit from the inclusion of other potential explanatory variables that are known to influence river fish assemblages in Ireland. In particular, hydrochemistry variables that were not available for this study that would be beneficial include water chemistry data, such as nutrient concentrations, and continuously logged data on temperature and dissolved oxygen concentrations. Increasing the number of surveys included in the analyses would facilitate the examination of first-order interactions of explanatory variables such as dissolved oxygen concentrations with low-flow events and/or nutrient status.
- Collecting site-specific habitat data on continuous scales rather than DAFOR (dominant, abundant, frequent, occasional, rare) scales at all study sites, which would enable the detangling of site-specific habitat effects from flow effects. Additionally, characterising the microhabitats of study sites would be beneficial for elucidating the potential behavioural responses of fish in seeking refuge from extreme high-flow events.
- Defining and including unimpacted reference sites, although this will certainly be challenging given the ubiquitous effects of climate change on changing flow events (Pont *et al*., 2007).
- Repeated sampling of the same sites on multiple dates, which could potentially further elucidate the effects of flow events on fish. In particular, sampling soon after extreme flow events could prove to be informative, particularly if potential explanatory variables of fish communities are modelled against the response of fish communities before and after such events. Such analyses would identify short-term effects of changes in flow and therefore may need to be combined with repeated sampling at set time points after such flow events to identify medium- and/or long-term effects on fish assemblages.

The macrophyte PFI in its present form is not suitable for use in Irish rivers; however, its approach to calculating associated metrics was useful in the development of the proposed MMTI. The evaluation of the MMTI suggests that it is a useful tool for characterising a combination of hydroecological conditions, including flow, and it is recommended that it be incorporated into Irish WFD river monitoring programmes. It is well known that it is difficult to disentangle the effects of various physical environmental factors on macrophyte distribution and, as a result, it is recommended that physical factors be considered in combination with each other as an indication of overall hydroecological health. It should be noted that flow had a weak signal in these analyses, as flow was not necessarily the main focus of most of the historical river macrophyte surveys within the RMD. This signal for flow could be augmented in future river macrophyte surveys by encompassing a wide range of river habitats with associated reliable flow data from hydrometric stations, to fully examine the value of the MMTI and indeed macrophytes in specifically monitoring for flow. It may also be more efficient in the future to examine mean annual discharges in rivers in relation to macrophytes rather than flow. This is being investigated at present in the UK by C. Westwood and colleagues (Environmental Research Associates, Exeter, UK). Careful consideration should also be given to matching macrophyte sampling dates with the antecedent hydrological record, as the duration of the macrophyte community response to pressures such as droughts and abstractions will depend on their severity.

The MMTI could complement and strengthen existing methods of monitoring signals of hydroecological change if used in combination with the macroinvertebrate hydroecological indices, as using the MMTI has similar advantages to using macroinvertebrate indices as stated under the macroinvertebrate-related recommendations above. A further advantage is that the MMTI requires a general recognition of macrophyte growth forms and leaf structure, thus making it more accessible for use by surveyors other than botanists/bryologists, and has potential for being incorporated into a citizen science programme. However, there is also a danger of losing valuable detail if the MMTI is solely used for monitoring purposes, as surveys where macrophytes are identified to species level provide greater detail in terms of water quality and water chemistry, and species with abundance values are essential for river classification purposes. Therefore, it is recommended that comprehensive macrophyte surveys are carried out in conjunction with the MMTI.

The MMIs that were identified in the literature review and the candidate MMI that was developed in this project for assessing the impacts of extreme flow events in Irish rivers are not currently suitable for deployment in Irish monitoring programmes, as relationships between all MMI values and flow statistics were weak. An appropriate method for generating a flow-based MMI for Ireland was identified and used in this project and, with further work, this method could lead to the development of a suitable MMI for Irish rivers. The main factors that hindered the development of an MMI were the lack of a large sample of overlapping monitoring sites for fish and macroinvertebrates close to hydrometric stations with good-quality continuous flow records, as well as the low taxonomic resolution of the macroinvertebrate data and the lack of absolute abundance data in the macroinvertebrate dataset.

The strength of the evidence for incorporating fish data and MMIs into hydroecological monitoring in Irish rivers is weak. However, it should be noted that the relatively weak relationships between fish community statistics and flow could be explained by the fact that the available data were more limited for fish than for other taxa. As noted previously, there were also limitations in the macroinvertebrate and macrophyte data that could have affected the strength of the relationships between macroinvertebrate- and macrophyte-based hydroecological indices and flow in this study. For example, biological surveys (for macroinvertebrates, macrophytes and fish) are not carried out annually at most monitoring sites, but annual surveys (and multiple surveys at the same site in a single year) are more appropriate for investigating relationships between flow and ecology. For the MMIs that were used in this study, the lack of data on actual macroinvertebrate abundances caused difficulties when calculating the indices (as discussed for the macroinvertebrate hydroecological indices above); another difficulty when attempting to calculate MMIs was the lack of overlapping survey data for macroinvertebrates, macrophytes and fish from the same monitoring sites, which meant that MMIs that included data for all three of these biological components could not be developed. Another limitation on the amount of data that could be used to investigate relationships between flow and ecology was the relatively small number of hydrometric stations on Irish rivers that had good-quality continuous flow records and were close to biological monitoring sites. Finally, water quality was found to be a probable confounding factor that could potentially mask relationships between biology and flow. Therefore, to facilitate more accurate and detailed investigations of relationships between biology and flow in Irish rivers in the future, it is recommended that hydrological reference sites, with well-maintained hydrometric gauges continuously measuring flow with a high degree of accuracy, should be maintained at appropriate locations throughout Ireland. These reference sites should include many of the hydrometric stations with good-quality continuous flow records that were identified in this project, as well as additional stations established in areas that currently have sparse coverage, such as upland areas and some of the drier areas in the east of the country. Moreover, the reference sites should be located on rivers without significant hydromorphological, water quality or abstraction pressures and should represent the full range of river typologies in Ireland. Biological monitoring sites should be established at appropriate locations as close as possible to all hydrological reference sites, and surveys for macroinvertebrates, macrophytes and fish should be carried out at all sites at least annually. Ideally, surveys for macroinvertebrates, macrophytes and fish at the same site should be carried out at the same time, or at least within a few days of each other, and on more than one occasion (e.g. in different seasons) in a single year; opportunistic sampling should also be carried out at these sites soon after extreme flow events to assess their impacts, as these could be missed during routine planned sampling. The establishment of hydrological reference sites and associated biological monitoring sites is particularly important in the context of predicted future changes to flows in Irish rivers due to climate change (Murphy and Meresa, 2024) and would allow such changes to be monitored using the hydroecological tools developed in this project.

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Abbreviations

AIC	Akaike information criterion
ASPT	Average score per taxon
CWC	Cover weighting code
DEHLI	Drought Effect of Habitat Loss on Invertebrates
DEHLI-Family	Family-level Drought Effect of Habitat Loss on Invertebrates index
DEHLI-Original	Original Drought Effect of Habitat Loss on Invertebrates index
DI	Drought intensity
DIS	Drought intolerance score
E-PSI	Empirically-weighted Proportion of Sediment-sensitive Invertebrates
E-PSI-Family	Family-level Empirically-weighted Proportion of Sediment-sensitive Invertebrates index
E-PSI-Mixed	Mixed-level Empirically-weighted Proportion of Sediment-sensitive Invertebrates index
FI	Flood intensity
IFI	Inland Fisheries Ireland
LIFE	Lotic-invertebrate Index for Flow Evaluation
LIFE-Family	Family-level Lotic-invertebrate Index for Flow Evaluation
LIFE-Species	Species-level Lotic-invertebrate Index for Flow Evaluation
MMI	Multi-metric index
MMT	Macrophyte Morphological Trait
MMTI	Macrophyte Morphological Traits Index
MTR	Mean Trophic Rank
NBDC	National Biodiversity Data Centre
NMDS	Non-metric multidimensional scaling
NRSA	National Rivers and Streams Assessment
OPW	Office of Public Works
PFI	Plant Flow Index
RDA	Redundancy analysis
RMD	River Macrophyte Database
RMHI	River Macrophyte Hydraulic Index
RMNI	River Macrophyte Nutrient Index
WFD	Water Framework Directive
WHPT	Whalley, Hawkes, Paisley and Trigg

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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