Report No. 166

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

Characterisation of Reference Conditions for Rare River Type

Literature Review

Authors: Edel Hannigan and Mary Kelly-Quinn

Comhshaol, Pobal agus Rialtas Áitiúil
Environment, Community and Local Government

www.epa.ie

ENVIRONMENTAL PROTECTION AGENCY

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

The work of the EPA can be divided into three main areas:

Regulation: *We implement effective regulation and environmental compliance systems to deliver good environmental outcomes and target those who don't comply.*

Knowledge: *We provide high quality, targeted and timely environmental data, information and assessment to inform decision making at all levels.*

Advocacy: *We work with others to advocate for a clean, productive and well protected environment and for sustainable environmental behaviour.*

Our Responsibilities

Licensing

We regulate the following activities so that they do not endanger human health or harm the environment:

- waste facilities *(e.g. landfills, incinerators, waste transfer stations)*;
- large scale industrial activities *(e.g. pharmaceutical, cement manufacturing, power plants)*;
- intensive agriculture *(e.g. pigs, poultry)*;
- the contained use and controlled release of Genetically Modified Organisms *(GMOs)*;
- sources of ionising radiation *(e.g. x-ray and radiotherapy equipment, industrial sources);*
- large petrol storage facilities;
- waste water discharges;
- dumping at sea activities.

National Environmental Enforcement

- Conducting an annual programme of audits and inspections of EPA licensed facilities.
- Overseeing local authorities' environmental protection responsibilities.
- Supervising the supply of drinking water by public water suppliers.
- Working with local authorities and other agencies to tackle environmental crime by co-ordinating a national enforcement network, targeting offenders and overseeing remediation.
- Enforcing Regulations such as Waste Electrical and Electronic Equipment (WEEE), Restriction of Hazardous Substances (RoHS) and substances that deplete the ozone layer.
- Prosecuting those who flout environmental law and damage the environment.

Water Management

- Monitoring and reporting on the quality of rivers, lakes, transitional and coastal waters of Ireland and groundwaters; measuring water levels and river flows.
- National coordination and oversight of the Water Framework Directive.
- Monitoring and reporting on Bathing Water Quality.

Monitoring, Analysing and Reporting on the Environment

- Monitoring air quality and implementing the EU Clean Air for Europe (CAFÉ) Directive.
- Independent reporting to inform decision making by national and local government *(e.g. periodic reporting on the State of Ireland's Environment and Indicator Reports).*

Regulating Ireland's Greenhouse Gas Emissions

- Preparing Ireland's greenhouse gas inventories and projections.
- Implementing the Emissions Trading Directive, for over 100 of the largest producers of carbon dioxide in Ireland.

Environmental Research and Development

- Funding environmental research to identify pressures, inform policy and provide solutions in the areas of climate, water and sustainability.

Strategic Environmental Assessment

- Assessing the impact of proposed plans and programmes on the Irish environment *(e.g. major development plans)*.

Radiological Protection

- Monitoring radiation levels, assessing exposure of people in Ireland to ionising radiation.
- Assisting in developing national plans for emergencies arising from nuclear accidents.
- Monitoring developments abroad relating to nuclear installations and radiological safety.
- Providing, or overseeing the provision of, specialist radiation protection services.

Guidance, Accessible Information and Education

- Providing advice and guidance to industry and the public on environmental and radiological protection topics.
- Providing timely and easily accessible environmental information to encourage public participation in environmental decision-making (*e.g. My Local Environment, Radon Maps).*
- Advising Government on matters relating to radiological safety and emergency response.
- Developing a National Hazardous Waste Management Plan to prevent and manage hazardous waste.

Awareness Raising and Behavioural Change

- Generating greater environmental awareness and influencing positive behavioural change by supporting businesses, communities and householders to become more resource efficient.
- Promoting radon testing in homes and workplaces and encouraging remediation where necessary.

Management and structure of the EPA

The EPA is managed by a full time Board, consisting of a Director General and five Directors. The work is carried out across five Offices:

- Office of Environmental Sustainability
- Office of Environmental Enforcement
- Office of Evidence and Assessment
- Office of Radiological Protection
- Office of Communications and Corporate Services

The EPA is assisted by an Advisory Committee of twelve members who meet regularly to discuss issues of concern and provide advice to the Board.

EPA Research Programme 2014–2020

Characterisation of Reference Conditions for Rare River Types

A Literature Review

(2013-W-FS-15)

Prepared for the Environmental Protection Agency

by

University College Dublin

Authors:

Edel Hannigan and Mary Kelly-Quinn

ENVIRONMENTAL PROTECTION AGENCY
An Ghníomhaireacht um Chaomhnú Comhshaoil
PO Box 3000, Johnstown Castle, Co. Wexford, Ireland

Telephone: + 353 53 916 0600 Fax: +353 53 916 0699
Email: info@epa.ie Website: www.epa.ie

ACKNOWLEDGEMENTS

This report is published as part of the EPA Research Programme 2014–2020. The programme is financed by the Irish Government. It is administered on behalf of the Department of the Environment, Community and Local Government by the Environmental Protection Agency which has the statutory function of co-ordinating and promoting environmental research.

The authors acknowledge the Environmental Protection Agency for funding this project under the EPA Research Programme.

The project partners wish to thank the following for their steerage and advice: Dr Catherine Bradley, Dr Jim Bowman and Dr Martin McGarrigle. The project partners also wish to acknowledge the valuable contributions of Dr Matthew Craig, Mr Bryan Kennedy, Dr Wayne Trodd, Mr Anthony Mannix, Dr Donal Daly, Ms Katie Tedd and Dr Imelda O'Neill.

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.

EPA RESEARCH PROGRAMME 2014–2020

Published by the Environmental Protection Agency, Ireland

PRINTED ON RECYCLED PAPER

ISBN: 978-1-04095-625-2 January 2016

Price: Free Online version

Project Partners

Edel Hannigan
School of Biology and Environmental Science
Science Centre West
University College Dublin
Belfield
Dublin 4
Ireland
Tel: +353 1 716 2339
Email: edel.hannigan@ucd.ie

Mary Kelly-Quinn
School of Biology and Environmental Science
Science Centre West
University College Dublin
Belfield
Dublin 4
Ireland
Tel: +353 1 716 2337
Email: mary.kelly-quinn@ucd.ie

Contents

List of Figures and Tables

Figure

Tables

Executive Summary

Under the EU Water Framework Directive, all surface waterbodies must be differentiated according to type in order to establish type-specific reference conditions. This has resulted in Member States developing national typologies that define the main surface water habitats in their country. In addition, common intercalibration types have been established for the Geographical Intercalibration Groups (GIGs) to allow the intercalibration process to proceed. As a result, intercalibration is based on reference conditions for the GIG types or benchmarking within the GIG types rather than on the national types. A consequence of this is that some national types may not fit completely into the common types, leading to their omission from the intercalibration process. It is therefore understandable that there has been limited attention paid to how rare river types are characterised or treated. Rare river types in this context are defined as systems that present the biota with a distinct combination of naturally challenging or distinct environmental conditions. They may or may not be limited in their distribution. The difficulty in separating natural variability from anthropogenic variability makes developing reference conditions for these rivers particularly challenging.

Four river categories were identified in Ireland by Kelly-Quinn et al. (2009) as potentially rare, that is presenting naturally challenging environmental conditions for the biota. These are (1) groundwater-dominated rivers; (2) highly calcareous rivers with calcium precipitation; (3) low alkalinity, naturally acidic rivers; and (4) rivers strongly influenced by lakes. Groundwater-dominated rivers tend to be characterised by low sediment concentrations, a relatively stable thermal and flow regime, and high water clarity. There is a lack of consensus over whether these systems host a distinct set of ecological characteristics. Where calcium carbonate precipitation and deposition occurs in highly calcareous rivers, the effects on the biological communities may be similar to those of sedimentation or siltation. The deposits can result in a loss of interstitial spaces and a reduction in habitat heterogeneity, both of which play a major role in structuring macroinvertebrate communities. Naturally acidic rivers have low acid-neutralising capacity, reducing the ability of the river to neutralise acid input. In systems where acidity occurs naturally through the input of organic ions the pH will be reduced or kept naturally low; however, the effects on the biota appear less severe than those associated with anthropogenic acidity. Lake outlets are the transitional zone between lakes and rivers (usually 1–2 km from the lake opening) and are influenced by lake processes. The literature shows that the transport of phyto- and zooplankton from the lake into the outlets results in macroinvertebrate communities that are dominated by filter-feeders.

Few Member States have looked at how to approach these rare river types, resulting in a paucity of guidelines on how to intercalibrate potentially new national river types. The first step is to determine whether or not these rare rivers are in fact new national types, or if they are a subset of the existing types. Either way, it is essential that appropriate reference conditions are characterised for these groups and, if necessary, metrics are adapted to ensure that their ecological status is accurately assessed.

1. Introduction

1.1 Typology

1.1.1 Defining national river types

All surface waterbodies must be differentiated according to type. This must be done in accordance with the technical specification set out in the Water Framework Directive (WFD) (Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000, Annex II). The WFD sets out two options that may be adopted by European Union (EU) Member States to develop and define river types: System A and System B (EC, 2000). Both systems are top-down approaches consisting of an abiotic framework by which waterbodies can be differentiated. System A is an a priori typology consisting of obligatory categories with fixed boundaries for each factor. The factors include altitude (three categories), catchment size (four categories ranging from small to very large) and geology (three categories, namely siliceous, organic and calcareous) (EC, 2000). System B offers a more flexible approach. Despite having the same obligatory factors as System A, with System B EU Member States can choose from additional factors and can also determine their own descriptor sizes. If a System B approach is used, then at least the same level of discrimination must be achieved as would be the case with System A.

In order for a typology to be useful, the types defined must be ecologically meaningful and representative of the different waterbodies present in a particular location. The typology must also serve to reduce the natural variation in biological parameters within a type (Dodkins *et al.*, 2005). There are several difficulties associated with typologies; for example, the categories defined may be overly broad, resulting in a large variation between sites within a particular type (Dodkins *et al.*, 2005; Kelly-Quinn *et al.*, 2009).

1.1.2 Approaches to developing a typology

Across Europe, EU Member States have developed national typologies using one of the approaches outlined above. Owing to its greater flexibility, most Member States (e.g. Austria, Belgium, Finland, France, Germany, Ireland and Italy) have used the System B approach to define river types, with other Member States, such as Poland and the United Kingdom, preferring to adopt the System A typology. Although all Member States have developed a typology for surface waters, some have yet to provide information on the testing of those typologies using biological data (e.g. Belgium, France, Italy and Latvia). Without this kind of validation, the river types defined may not be ecologically meaningful (EC, 2012a). Many Member States have also reported difficulties in establishing reference conditions for all river types and all biological elements.

Great Britain adopted the System A approach to defining river types (UKTAG, 2003; Anon., 2010). Therefore, information on altitude, catchment size and geology was recorded (UKTAG, 2003). Three categories were defined in each factor, giving 27 potential stream types. In reality, 18 ecologically meaningful stream types were confirmed, as some of the theoretical 27 types did not exist or were found to have very few representatives (UKTAG, 2003). Of the 18 river types defined for Great Britain, three were not found in Scotland and two were not present in England and Wales. The dominant river types also varied between Scotland and England and Wales, with low-altitude, small-catchment and calcareous rivers accounting for one-third of the river network in England and Wales, and medium-altitude, small-size and siliceous rivers more common in Scotland (UKTAG, 2003). With the development of river basin management plans, new river types were added to the initial list of 18 through the introduction of a new catchment size class and geology type. The typology framework was also amended to account for small catchments ($<10\,km^2$) in England and Wales (Anon., 2010). This increased the number of river types to 21, all of which were tested against biological data (EC, 2012b). River types in Northern Ireland were also developed using System A, which resulted in 12 ecologically meaningful stream types (EHS, 2005). Given that three out of the four River Basin Districts (RBDs) found in Northern Ireland are international RBDs, which share catchments with the Republic of Ireland, it was important that types

were comparable between the two countries (EHS, 2005). The North–South Shared Aquatic Resource (NS SHARE) team/project applied the Irish typology to rivers in Northern Ireland, and this approach has been found to work (Murphy and Glasgow, 2009).

Davy-Bowker *et al.* (2006) compared the WFD System A typology with three multivariate predictive River Indicator Prediction and Classification System (RIVPACS)-type tools, one of which is used in the United Kingdom. RIVPACS predictive models differ from the WFD approach in that reference sites are first established based on the macroinvertebrate fauna present. The environmental variables that best predict the biological classifications are then determined (Clarke *et al.*, 2003; Davy-Bowker *et al.*, 2006). The RIVPACS-type models were found to be more effective at predicting macroinvertebrate reference conditions than the System A approach, which is constrained by its use of a limited number of categorical variables. The use of continuous, more ecologically significant, variables, such as substrate composition, at local geographical scales allowed the RIVPACS-type models to better predict the invertebrate communities (Davy-Bowker *et al.*, 2006; Sandin and Verdonschot, 2006). System B allows for local factors such as substratum composition, distance from source and mean water depth to be included in the typology; however, this typology may result in the definition of a large numbers of river types.

Applying the WFD System B approach in each ecoregion and using altitude, geology, stream slope and size as descriptors, Germany defined 24 river types (Pottgiesser and Sommerhäuser, 2004). The number of river types varies between 6 and 24 across the 10 RBDs (EC, 2012c). Types were first divided by ecoregion and then sub-ecoregions were established to enable a more refined typology to be developed; this left four groups (Alps and Alpine Foothills, Central Highlands, Central Plains and ecoregion-independent types), each of which contains varying numbers of river types. The ecoregion-independent stream types included lake outflows, small organic substrate-dominated rivers and small streams in riverine floodplains which were spread across several ecoregions (Pottgiesser and Sommerhäuser, 2004). They were characterised using geomorphological parameters such as geology and soils. A bottom-up approach, using both biotic and abiotic factors, was adopted to determine regional stream typologies (Sommerhäuser, 2002; Lorenz *et al.*, 2004). Rawer-Jost *et al.* (2004) assessed the macroinvertebrate fauna for 15 of the 24 stream types and confirmed at least 6 of the 15 as being independent types. Some overlap was found between other rivers, and typological factors such as stream size and bottom substrate were highlighted as important in determining lowland types. Four types could not be included in the study owing to a lack of reference samples, while a further two types were omitted because they represented large rivers, which are subject to different assessment methods (Rawer-Jost *et al.*, 2004). Lake outflows were also omitted from this study. A study conducted by Brunke (2004) in northern Germany succeeded in validating six of the seven potential river types identified for the region as distinct types, including lake outlets and marsh streams with a predominantly clay substrate, although it should be noted that the selected sites were the best available examples of reference conditions rather than “true” reference sites.

1.1.3 River typology in Ireland

In Ireland, the System B approach was chosen, and a number of typologies based on various combinations of physical descriptors were tested in order to determine the most ecologically meaningful ones. The work was carried out as part of the Characterisation of Reference Conditions and Testing of Typology of Rivers (RIVTYPE) project using 50 sites that were determined to be of high ecological status (Kelly-Quinn *et al.*, 2005). This dataset generated a number of types based on a range of methods such as expert opinion and permutational analysis of environmental boundaries (Kelly-Quinn *et al.*, 2005). A System A typology was also generated. The study concluded that a permutation-based 12-category typology, based on geology and slope (Table 1.1), best discriminated the biological elements across all groups and was therefore accepted by the Environmental Protection Agency (EPA) (Kelly-Quinn *et al.*, 2005). It must be pointed out, however, that the utility of the national types is at this stage debatable, as the EU intercalibration process was based on reference conditions for common intercalibration types within Geographical Intercalibration Groups (GIGs) (section 3.3).

Table 1.1. Irish national river types based on geology and river slope (EPA, 2005)

Code	Catchment geology (% of bedrock in upstream catchment by type)	Description	Hardness
1	100% siliceous	Low alkalinity	< 35 mg $CaCO_3$/L
2	1–25% calcareous (mixed geology)	Medium alkalinity	35–100 mg $CaCO_3$/L
3	> 25% calcareous	High alkalinity	> 100 mg $CaCO_3$/L
Code	*Slope (m/m)*		
1	≤ 0.005	Low slope	
2	0.005–0.02	Medium slope	
3	0.02–0.04	High slope	
4	> 0.04	Very high slope	

Examples of type codes:

The two codes from above are combined in order, with geology providing the first digit and slope the second digit. For example, a code of 31 indicates a calcareous low-slope site, and a code of 23 indicates a mixed geology and high slope of between 2% and 4% gradient.

$CaCO_3$, calcium carbonate.

2. Establishing Reference Conditions

As described above, typology provides a framework of abiotic factors by which rivers with similar hydromorphological conditions can be grouped. Once a typology has been developed, using the approaches outlined above, and the river types defined, reference conditions for each type must be established in order to determine waterbody status pursuant to the WFD. Given that pristine conditions do not exist, the Common Implementation Strategy working group 2.3 – REFCOND recommended reference conditions to be a state in which "the values of the physico-chemical, hydromorphological and biological quality elements correspond to totally or nearly totally undisturbed conditions" (Wallin *et al*., 2003, cited in EC, 2003). Preferably, potential reference sites are initially identified using hydromorphological, physico-chemical and pressure criteria, with the use of biological quality elements (BQEs) excluded where possible to avoid circularity and minimise the risk of rare waterbody types not being detected (EC, 2003). However, in many cases all available data, including biological data and expert judgement, may be needed to identify potential reference sites. For sites to be considered reference condition, the level of impact must be minimal and, therefore, acceptable non-impact thresholds must be determined for critical pressures (Pardo *et al.*, 2012).

Guidelines on establishing reference conditions were set out in the REFCOND guidance document (EC, 2003), with a number of approaches available to EU Member States. Where potential reference condition sites are available, a spatial network of sites should be established using existing biological, hydromorphological and chemical data from monitoring sites. In circumstances in which potential reference sites are not available, then predictive modelling, historical data from formerly undisturbed sites or palaeoreconstruction can be used. Expert judgement can also be used, particularly in combination with other approaches. Each approach has advantages and disadvantages (EC, 2003). One of the main difficulties in defining reference conditions is determining what level of anthropogenic impact is acceptable at these sites, given that completely natural conditions no longer exist.

The ECOSTAT (2011) document on intercalibration also sets out guidelines on deriving reference conditions and defining alternative benchmarks for reference conditions to harmonise national class boundaries across EU Member States for the purposes of intercalibration. True reference condition sites must fulfil all the criteria for reference conditions for all BQEs. There must also be sufficient numbers of these sites to enable reliable reference conditions for common intercalibration types to be established. Where true reference sites do not exist, which tends to be the case in many countries in each GIG, then "partial" reference sites can be used. These are sites that experience greater anthropogenic impact but not for all BQEs. This means that for some BQEs the sites represent reference conditions; for example, morphological disturbance and change will not affect the metrics used to assess how phytobenthic communities respond to nutrients but they will affect macroinvertebrate communities (Kelly *et al.*, 2012). If sufficient numbers of either true or partial reference sites from which reference conditions can be established do not exist, then alternative reference or benchmark sites can be used for intercalibration. Sites used to establish alternative reference or benchmark conditions must represent a similar level of anthropogenic impact, all relevant pressures must be accounted for, and, if multiple pressures exist, they must be combined in a meaningful way. These sites must be identified from the common intercalibration dataset for the GIG. How far the benchmark status deviates from "true" reference conditions must be determined by common opinion and the "benchmark communities" that have been defined (ECOSTAT, 2011). Virtual reference sites (i.e. those that do not exist in reality) may be used for this purpose.

As accurate ecological status assessment depends heavily on the existence of reliable type-specific reference conditions, it is vital that natural spatial variation in the biological elements is not too large within any one type. Where this issue does arise, it is important to try and establish more reliable reference conditions by refining the typology using additional factors that result in reduced variation in the biological element (ECOSTAT, 2011). An example of this procedure is seen in the United Kingdom, where the extra factor of

humic level was incorporated into the typology during the development of the Acid Water Indicator Community (AWIC) index to account for natural acidity in rivers in which acidity was driven by organic acids rather than anthropogenic impact (McFarland, 2010). Before establishing a new river type using additional factors, whether or not there are sufficient relevant reference sites to establish reliable reference conditions must be determined (ECOSTAT, 2011).

The concept of reference conditions as a means by which to assess and evaluate ecosystem health has come under scrutiny (Bouleau and Pont, 2015). In dynamic systems, where there are gradients of change attributable to natural variability, it is very difficult to separate natural effects from those of anthropogenic impact. As a result, type-specific reference conditions may be too restrictive and may not represent the natural gradients present in these ecosystems. An extreme example of the natural variability and gradient of change observed in a river system is the cementing of some river reaches by calcium carbonate ($CaCO_3$) deposition while further upstream and downstream is unaffected.

3. Status Assessment

The WFD requires all EU Member States to achieve at least “good” status for all surface waters by 2015, except where derogations have been sought, by preventing the deterioration of systems already achieving “high” or “good” status and improving or rehabilitating those that fail to meet the required level. In the WFD, “‘Good surface water status’ means the status achieved by a surface waterbody when both its ecological status and its chemical status are at least ‘good’” (EC, 2000). The overall surface water status is determined on a “one out all out” principle whereby the status is determined by the element with the lowest value (Figure 3.1).

3.1 Ecological Status and Bioassessment Metrics

Ecological status encompasses biological elements, hydromorphological elements, relevant pollutants and physico-chemical elements (EC, 2000). BQEs include phytobenthos, aquatic flora, invertebrates and fish. Hydromorphological elements refer to the physical morphology of the waterbody (i.e. the hydrology, morphology and continuity of the system). The general conditions supporting the biota, such as nutrients, oxygen and temperature, are also assessed as part of the ecological status. In order to assign ecological status, the output of the quality metric used for each BQE must be expressed as an ecological quality ratio (EQR). The EQR represents the relationship between the observed and expected biological communities in a reference sample for the waterbody. It is expressed as a value between 0 and 1, with values close to 1 indicating high ecological status and those close to 0 signifying low ecological status. The ecological status of the waterbody is determined based on the assessment of the BQEs, and the general chemical, physico-chemical and hydromorphological conditions of the waterbody. Chemical status is assessed by monitoring and measuring priority and other dangerous substances and comparing them with approved standards that are indicative of good status (see Figure 3.1).

In order to assess the BQEs, indices or metrics that measure deviation from reference conditions must be developed, refined, validated and intercalibrated using the common EU typology set out for each GIG to ensure accurate assessment. The WFD requires certain characteristics of the different BQEs to be assessed (e.g. community composition, abundance, etc.); however, it leaves the decision on what metric should be used to each Member State. Hundreds of biological metrics, some of which are newly developed and others of which are adapted from those used prior to the implementation of the WFD and cover a range of BQEs and pressures, have been used across EU Member States to assess the status of surface waters. The metric used to assess a particular BQE is optimised for that element and is usually linked with a particular pressure (e.g. nutrient enrichment, habitat degradation or artificial acidity inputs). An overview of the biological metrics used in the WFD by Birk *et al.* (2012) found that sensitivity and ecological trait metrics were most often used to assess the ecological status of rivers. In terms of detecting the effects of different pressures, the number of metrics designed to detect hydrological or morphological degradation was found to increase from autotrophic to heterotrophic elements (i.e. from phytoplankton to fish fauna), whereas the opposite trend was found for metrics detecting nutrient enrichment or organic pollution (Birk *et al.*, 2012). Phytobenthos, phytoplankton (in lakes) and benthic invertebrate assessment methods were the best validated of the biological elements across the water categories for the pressure–impact relationship (Birk *et al.*, 2012); however, this relationship remains untested for a large number of metrics.

3.2 Metrics Used in Ireland

Many of the metrics used to assess ecological status in Irish waters have been developed over a number of decades. The standards for the supporting physico-chemical and hydromorphological quality elements have been set out and adopted in Irish law under the European Communities environmental objectives (surface water) regulations for 2009 (S.I. No. 272).

The EPA river-quality rating scheme (Q-value) was developed in the 1970s to measure the impact of organic pollution based on macroinvertebrate taxon sensitivity. It was subsequently found to measure accurately the

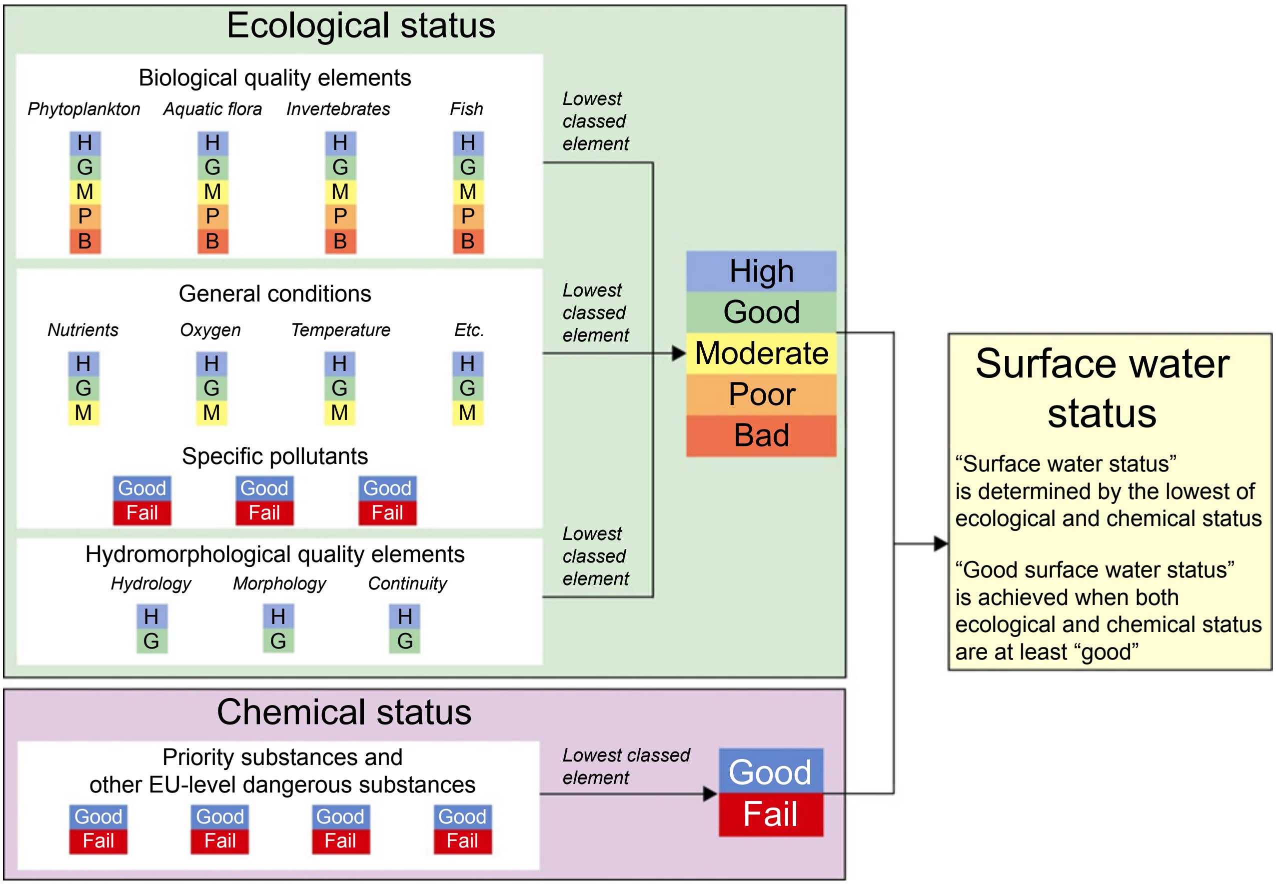

Figure 3.1. Overview of the various steps involved in establishing ecological and chemical status for the determination of surface water status (diagram from EPA, 2008).

impact of nutrient enrichment. This metric was developed independently of river type, and the relationship between the Q-value rating and land-use has been well documented (Donohue *et al.*, 2006). Although this metric works effectively on the more common river types, there are certain situations in which the Q-value may score a system too harshly (i.e. sensitive species are missing as a result of the natural characteristics of the river rather than as a consequence of pollution). In addition, this metric does not detect the effects of anthropogenic acidification, which is identified as a potential pressure that requires monitoring under the WFD. In the United Kingdom, the AWIC index, using species-level data, was developed to detect acidification. It is an abundance-derived metric based on the sensitivity of species to acid conditions (McFarland, 2010) and has been successfully intercalibrated. Thresholds for Acid Water Indicator Community species (AWICsp) reference values in the United Kingdom have recently been published (WFD-UKTAG, 2014). This metric has been adopted in Ireland to assess acidity.

Although the Trophic Diatom Index (TDI) has been intercalibrated, it has not been used in the latest status assessment and is undergoing further refinement. There are disadvantages and limitations with this method of assessment. The TDI is based on the preference of diatom taxa for varying nutrient concentrations, which makes it suitable for detecting organic pollution, but it may not successfully detect other types of pollution (Kelly *et al.*, 2009). By assessing only diatom communities, other parts of the phytobenthos may be ignored, and so metrics based on other phytobenthic groups in conjunction with the TDI would lead to a more accurate assessment of ecological status for this quality element. The Diatom Acidification Metric (DAM) has been developed to detect the effects of acidification (Juggins and Kelly, 2012).

The method recently adopted to assess the status of macrophytes in Irish rivers is the Mean Trophic Rank (MTR). It was designed to detect organic pollution and excessive nutrient enrichment, with a low MTR indicating plant communities associated with low nutrient

status (Holmes *et al.*, 1999). This metric is useful but, again, it may not be effective in detecting other pressures. Another disadvantage of this metric is that it does not allow direct comparison across river types (Holmes *et al.*, 1999). The MTR has been successfully intercalibrated for only one river type in the Central Baltic GIG (medium, lowland, mixed geology rivers (R-C4)), and intercalibration is not yet completed in the Northern GIG (EC, 2013). This metric has not been applied to any sites in the latest status assessments, and further testing and development of the metric was conducted in 2015.

3.3 Intercalibration

To ensure comparability of ecological status across the EU, it is important that the interpretation of status classifications is the same across all Member States. For the purposes of intercalibration, Member States with similar biogeographical water types were placed into GIGs. The process of intercalibrating all the national metrics used to assess each BQE was undertaken to enable the comparison of the boundaries between "high and good" status and "good and moderate" status across Member States (EC, 2003; ECOSTAT, 2011). Intercalibration is carried out separately for each anthropogenic pressure and each BQE. To date, two phases of intercalibration have been completed. Phase 1 was completed in 2008 with limited success. Issues identified during Phase 1 were addressed during the second phase of intercalibration, which was completed in 2011 with the intercalibration of 230 methods from 28 countries (Poikane *et al.*, 2014), including the Irish metrics mentioned in section 3.2. Ireland is involved in two GIGs – the Northern GIG and the Central Baltic GIG – for the intercalibration of rivers. There are still some gaps in a number of GIGs for particular BQEs (e.g. assessment methods for macrophytes in the Northern GIG have yet to be intercalibrated).

Three intercalibration options are available to the GIGs depending on how comparable the national assessment methods of the Member States are (ECOSTAT, 2011). The first option is applicable where data acquisition and numerical evaluation are the same across all Member States within the GIG (i.e. the same sampling and processing methods are used). Where data sampling and evaluation procedures differ between countries, then Option 2, the use of a common intercalibration metric, must be applied. Option 3 is used when data are acquired using similar methods but numerical evaluation varies. Option 1 represents the most straightforward approach. However, most EU Member States use pre-existing assessment methods, which can complicate the harmonisation of reference conditions and the setting of classification boundaries. Option 2 requires the use of a common dataset consisting of reference sites across Member States in each GIG from which the common metric can be calculated and a common intercalibration benchmark established (ECOSTAT, 2011). By establishing a common benchmark for reference conditions across the GIG for each intercalibration type, a clearer assessment of status can be made, and the criteria for identifying reference conditions are also more transparent (ECOSTAT, 2011). Where a common metric is used, there should be a high level of correlation between the common metric and the national assessment metrics. This approach was used by the Northern and Central Baltic GIGs.

Given the timing of goals and aims under the WFD schedule, the national typology reports (Article 5 characterisation reports) were not completed by the time the process of intercalibrating the classification of status across Member States was scheduled to begin. As a result, a common typology [Intercalibration Common (IC) types] for each GIG based on System A characteristics, namely catchment size, altitude and geomorphology, alkalinity and organic matter, was used to define river types (Table 3.1). Irish rivers were represented by two intercalibration common types in the Northern GIG (R-N1, small, lowland, siliceous rivers with moderate alkalinity; and R-N3, small/medium, lowland, organic rivers) and four intercalibration common types in the Central Baltic GIG (R-C2, small, lowland rivers with siliceous rock geology; R-C4, medium, lowland rivers with mixed geology; R-C5, large, lowland rivers with mixed geology; and R-C6, small, lowland rivers with calcareous geology) (Table 3.1) (McGarrigle and Lucey, 2009). The use of common types for the purposes of intercalibration was necessary but by no means ideal, as they were designed to cover all BQEs and thus are not optimised for any one BQE (Kelly *et al.*, 2012), which may potentially lead to variability in reference conditions for the different elements. Another consequence of using a common typology for each GIG is that some national river types may not fit completely into the common types, which may lead to their omission from the intercalibration process (Poikane *et al.*, 2014). A misalignment between the national and

Table 3.1. Common intercalibration types for the Northern and Central Baltic GIG

GIG	Type	River characterisation	Catchment area (of stretch)	Altitude and geomorphology	Alkalinity (meq/L)	Organic material (mg Pt/L)
Northern GIG	R-N1[a]	Small lowland siliceous, moderate alkalinity	10–100 km^2	< 200 m or below the highest coastline	0.2–1	< 30 (< 150 in Ireland)
	R-N3[a]	Small lowland organic	10–1000 km^2	< 200 m or below the highest coastline	< 0.2	> 30
	R-N4	Medium lowland siliceous, moderate alkalinity	100–1000 km^2	< 200 m or below the highest coastline	0.2–1	< 30
	R-N5	Small mid-altitude siliceous	10–100 km^2	Between lowland and highland	< 0.2	< 30
Central Baltic GIG	R-C1	Small lowland siliceous sand	10–100 km^2	Lowland, dominated by sandy substrate (small particle size), 3–8 m width (bankfull size)	> 0.4	N/A
	R-C2[a]	Small lowland siliceous rock	10–100 km^2	Lowland, rock material, 3–8 m width (bankfull size)	< 0.4	N/A
	R-C3	Small mid-altitude siliceous	10–100 km^2	Mid-altitude, rock (granite) – gravel substrate, 2–10 m width (bankfull size)	< 0.4	N/A
	R-C4[a]	Medium lowland mixed	100–1000 km^2	Lowland, sandy to gravel substrate, 8–25 m width (bankfull size)	> 0.4	N/A
	R-C5[a]	Large lowland mixed	1000–10,000 km^2	Lowland, barbel zone, variation in velocity, maximum altitude in catchment 800 m, > 25 m width (bankfull size)	> 0.4	N/A
	R-C6[a]	Small lowland calcareous	10–300 km^2	Lowland, gravel substrate (limestone), width 3–10 m (bankfull size)	> 2	N/A

[a]Denotes the IC types representing Irish river types.

N/A, not applicable to Central Baltic types.

intercalibration common types may also hinder the process of translating the intercalibration results into the national typology (Poikane *et al.*, 2014).

Recently, descriptions of the national river and lake types defined by all EU Member States and the range of typological factors used in their development were compiled to enable waterbody status and pressures to be assessed at a European level (ECOSTAT, 2014, 2015). From these data, all national types were grouped based on similarity into one of 20 broad European types, including glacial rivers, which were introduced as a new broad type at the request of the Alpine countries. The Scandinavian countries requested that lowland organic rivers be redefined and split into two sub-types based on their geology, with mid-altitude organic rivers introduced as two new broad river types (broad Types 12 and 13). These broad types were then linked to the IC common types mentioned previously as well as to types defined under the Habitats Directive (Council Directive 92/43/EEC of 21 May 1992) (ECOSTAT, 2014). Although this most recent work links broad European types, national types and intercalibration common types, it does not in any way alter the national river types set out by Member States. The 12 Irish national river types (discussed in section 1.1.3) are grouped into two broad European types: Type 3, which covers lowland, siliceous rivers in very small to small catchments, and Type 5, which refers to lowland rivers in very small to small catchments with a calcareous or mixed geology (ECOSTAT, 2014, 2015).

Intercalibration has been completed in all GIGs for most of the BQEs; however, some gaps remain; for example, metrics for macrophytes have yet to be intercalibrated for the Northern GIG (Poikane *et al.*, 2014).

4. Rare River Types

4.1 Rare River Types

Rare river types (e.g. lake outlets or highly calcareous rivers) are stretches of river that present naturally challenging combinations of environmental conditions for the aquatic biota. Difficulty in differentiating natural variability from anthropogenic variability, a challenge that is particularly prevalent when it comes to developing reference conditions for transitional waters (Elliott and Quintino, 2007; Basset *et al.*, 2013), may explain the lack of clarity in how to characterise or treat rare rivers. The need to categorise the dominant waterbody types, along with the fact that some EU Member States have yet to validate their typologies using biological data, resulting in the possibility that some of the types do not exist, may also contribute to the paucity of guidelines on intercalibrating new types. Some EU Member States have identified and considered rare or unusual lake and river types that may not be represented in the national typology. In Sweden, for example, it was suggested that a lake type could be considered rare if it was represented by fewer than 10 waterbodies (Fölster *et al.*, 2004). These rare types may be excluded from the typology and the intercalibration process and instead dealt with through special management plans (Fölster *et al.*, 2004).

A number of the river types found in Great Britain have been identified as rare, for example rivers with a medium-size catchment area (100–1000 km^2), a predominantly organic surface geology and a low (< 200 m) catchment altitude. Other rare types include those rivers characterised by a large catchment area, a low catchment altitude and a predominantly siliceous geology (UKTAG, 2004). Each of these types represents < 1% of the categorised river network and is generally restricted in terms of geographical distribution, occurring only in particular areas (UKTAG, 2004). The initial selection of river types may change and develop as more information is collected. In Great Britain, factors have already been added to the typology to account for small rivers (< 10 km^2) that were not included in the initial System A process (Anon., 2010). The need to account for natural variation attributable to humic acids when assessing the effects of acidification has also been recognised in the development of an acid water indicator metric in Great Britain (McFarland, 2010; WFD-UKTAG, 2014), as previously mentioned. Although some countries have included rare streams in the development and validation of their typology (e.g. Germany has identified lake outlets and predominantly clay marsh streams as distinct types), rare river types have, generally, not been dealt with.

In an Irish context, it is suggested that rare river types, which may not be represented in the national typology, be treated separately through special management plans (EPA, 2005). Rare in this situation refers to systems with a naturally variable or unusual combination of environmental conditions rather than a limited distribution. Four river types were identified as being potentially rare by Kelly-Quinn *et al*. (2009): (1) groundwater-dominated rivers; (2) highly calcareous rivers with calcium precipitation; (3) low-alkalinity, naturally acidic rivers; and (4) rivers strongly influenced by lakes. Examples of these systems were omitted from the initial development of the current 12-type national typology. Each of these rare types potentially presents different abiotic challenges for the biota and, therefore, the current reference conditions may not accurately represent their communities, leading to an inaccurate assessment of their ecological status. A variable body of knowledge exists on these types of rivers and knowledge is particularly scarce for some types, especially in Ireland.

Each of the four river groups identified as potentially representing separate river types or sub-types presents different environmental conditions for the aquatic biota. Whether or not the variation in the physical and chemical conditions observed in these rivers results in them hosting distinct biological communities has yet to be established, with little research undertaken for some rivers and a lack of consensus among researchers for others.

4.1.1 Groundwater-dominated rivers

The importance of groundwater and its role in maintaining stream flow and mesohabitat heterogeneity (Boulton and Hancock, 2006) cannot be underestimated. As a

result, the scientific community has begun to take a more holistic approach to water resources by investigating groundwater and surface water interactions in terms of hydrology, hydrogeology and chemistry.

All streams and rivers receive a groundwater input; for example, approximately 30% of the annual flow of the majority of Irish rivers is derived from groundwater (Daly, 2009). The local and regional geology and the permeability of the underlying aquifer determine the input of groundwater, referred to as baseflow, to the total stream flow (Boulton and Hancock, 2006). The level of dependence on baseflow or groundwater-derived sources varies across scales (i.e. reach scale vs. catchment scale) (Sear *et al*., 1999; Boulton and Hancock, 2006) and season; for example, during periods of low surface flow the percentage flow derived from groundwater can increase from 30% to 90% (Daly, 2009). Defining groundwater-dominated rivers is dependent mainly on scale (reach, catchment) and the geology of the underlying aquifer (Sear *et al*., 1999, Boulton and Hancock, 2006). The permeability and geology of the aquifer and the surface soils will influence the flow regime and the rate of discharge into surface water systems. These also affect the chemical make-up of the water and the constituents transferred to the stream through the groundwater. The majority of the bedrock aquifers in Ireland are unconfined and have fissure permeability. Four aquifer types are defined in Ireland under the WFD, based on their flow regime: (1) sand/gravel aquifers; (2) karstified limestone; (3) productive fissured bedrock; and (4) poorly productive fissured bedrock (Daly, 2009). With poorly productive bedrock underlying 71% of the country, this waterbody type contributes a substantial amount of baseflow to surface waters (Daly, 2009).

In general, groundwater and surface water differ in their hydrochemical characteristics, with groundwater tending to have a chemical regime reflective of the aquifer geology and a more stable thermal and flow regime (Crisp and Westlake, 1982; Sear *et al*., 1999). Dissolved oxygen (DO) levels can vary in groundwater depending on a number of factors, including whether the aquifer is confined or unconfined. This makes establishing the range of natural background DO levels difficult. Younger (2009) recorded the typical range of DO in natural groundwater as ranging from 0 to 3 mg/L (0–25% saturation) with extreme values of 12 mg/L (100% saturation) in shallow permeable aquifers. Where aquifers are unconfined, as is the case for many aquifers in Ireland, they can recharge quickly and thus typically have much higher (usually close to 100% saturation) DO content (Matthew Craig, EPA, personal communication, 13 October 2015). Recent work aimed at establishing natural background levels of Irish groundwater quality found that, to some extent, DO concentrations were controlled by groundwater vulnerability (Tedd *et al*., in preparation). DO concentrations were highest at monitoring points that were in highly vulnerable settings[1] (Tedd *et al*., in preparation). Where groundwater sites were categorised as being extremely vulnerable, the lower and upper limits recorded for natural background levels of DO were 4.9 mg/L O_2 and 12.0 mg/L O_2, respectively (Tedd *et al*., in preparation). Alternatively, where groundwater vulnerability was low, the upper and lower limits reported for the natural background levels of DO were 0.5 mg/L O_2 and 6.0 mg/L O_2, respectively (Tedd *et al*., in preparation). The lithology of the groundwater sites had no significant impact on DO levels (Tedd *et al*., in preparation). The presence of elements such as iron can lead to reducing conditions and can potentially reduce DO concentrations. Consequently, groundwater-dominated rivers are generally characterised by low sediment concentrations, high water clarity and a relatively stable thermal and flow regime (Whiting and Stamm, 1995; Sear *et al*., 1999), although this may not hold true for all rivers. Some groundwater-fed rivers also tend to have naturally low oxygen levels. Temperature, DO and flow regime are extremely important for biological communities. Oxygen-sensitive species will not be found in stretches of rivers with low oxygen saturation, and stenothermic species will favour habitats with a stable temperature regime. Conductivity, hardness and alkalinity are influenced by the underlying geology; therefore, in limestone areas or karst springs, high levels of hardness will be recorded and may result in the deposition of $CaCO_3$ on benthic substrates as well as on the biota. An increased influx of groundwater, resulting in more stable physico-chemical characteristics, may affect the biological communities and may culminate in the presence of obligate groundwater invertebrates (stygofauna) in the stream community (Boulton and Hancock, 2006).

Much of the research on groundwater-dominated rivers has focused primarily on rivers draining karst aquifers

1 The highest vulnerability category (Extreme X) represents sites at which the bedrock is at or near the surface and within 30 m of a location of point recharge (DELG/EPA/GSI, 1999, in Tedd *et al*., in preparation).

(Cannan and Armitage, 1999; Petts *et al*., 1999; Smith and Wood, 2002). Water temperature, leaf litter inputs and flow permanence are important factors in structuring macroinvertebrate communities in all river systems including karst springs (Petts *et al*., 1999; Smith and Wood, 2002; Smith *et al*., 2003). Inputs of groundwater have been linked with increased invertebrate richness and abundance (Mattson *et al*., 1995; Smith and Wood, 2002). Flow permanence has been highlighted as a factor that strongly affects community structure, more so than temperature or DO levels in some cases, with periods of drought or reduced flow resulting in a decrease in species abundance (Smith and Wood, 2002; Wood and Armitage, 2004). Macrophyte assemblages also benefit from flow stability, which enables increased growth and results in the development of dense macrophyte beds (Sear *et al*., 1999). Wood *et al*. (1999) found changes in macroinvertebrate taxon richness at groundwater-dominated sites to be a result of both hydrological conditions (i.e. severe drought) and management activities.

Whether or not all groundwater-dominated rivers host a distinct set of ecological characteristics remains unclear. Although conditions presented by groundwater dominance, particularly increased flow stability, may favour macrophyte growth, resulting in dense mats of vegetation (Sear *et al*., 1999), the effect on invertebrate communities is not as clear. The RIVPAC assessment tool was used in the United Kingdom to investigate this and the results suggested that communities varied significantly in response to local geology, regardless of the fact that they were all groundwater dominated (Sear *et al*., 1999). This emphasises the importance of local-scale factors such as changes in geology (Williams *et al*., 1997; Cannan and Armitage, 1999) and soils. Originally, however, RIVPAC contained a number of polluted sites and so may not be sufficiently sensitive to detect a groundwater difference.

4.1.2 Highly calcareous rivers

The occurrence of springs in areas with karst limestone bedrock can give rise to highly calcareous rivers, which may experience precipitation and deposition of $CaCO_3$ on the surrounding substrate, flora and fauna. This $CaCO_3$ precipitate is referred to as marl, tufa or travertine deposits, and although all of these differ slightly in their formation, most studies do not differentiate between them. Marl is freshwater sediment that contains a variable amount of $CaCO_3$ (35–65%). It tends to be soft and friable. Tufa, however, is a porous, soft limestone formed by chemical precipitation that creates surficial deposits around springs. $CaCO_3$ deposition in Irish rivers tends to be in the form of marl. The degree of deposition can range from small amounts on macrophytes and substrate to a complete cementing of the substrate, leading to a reduction in both substrate and microhabitat heterogeneity (Pentecost, 2005). This phenomenon may be caused by a number of physical, chemical and biological processes, including evaporation. However, the principal driver of this phenomenon appears to be the reaction of carbon dioxide (CO_2) and calcium bicarbonate solutions, which results in the precipitation of $CaCO_3$ (Pentecost, 2005), although not all studies agree with this assessment (Pitois *et al.*, 2003).

Biological action may also contribute to this process of precipitation and deposition (Pedley, 1992) through the removal of CO_2 from the system during photosynthesis (Butcher, 1946). Stream velocity can lead to a heterogeneous distribution of carbonate deposits, as observed by Casas and Gessner (1999), who noted a greater extent of travertine deposition on leaves on the outer edges of leaf litterbags than on those in the inner part of the bags. Other studies have indicated the importance of photosynthesising green algae and diatoms in both initiating the precipitation reaction by raising the pH (green algae) (Hartley *et al.*, 1997) and minimising the precipitation of $CaCO_3$ (Lu *et al.*, 2000). Lu *et al.* (2000), using scanning electron microscopy, found that calcite surfaces with attached diatoms had an etched appearance, which suggests that the diatoms may in some way affect calcite dissolution.

Similar to the effects of siltation and sedimentation, cementing of the substrate by $CaCO_3$ reduces habitat heterogeneity and interstitial spaces in the streambed (Casas and Gessner, 1999; Pentecost, 2005; Rundio, 2009), thereby impacting perhaps the two most important factors affecting invertebrate structure. The loss of interstitial spaces and microhabitats potentially alters the invertebrate communities present, which will affect salmonids in terms of food availability and habitat for spawning (Kelly-Quinn *et al.*, 2003; Pitois *et al.,* 2003; Rundio, 2009). Invertebrate abundance also tends to be lower in stream reaches where deposition occurs (Pitois *et al.*, 2003; Álvarez and Pardo, 2007; Rundio, 2009). Kelly-Quinn *et al.* (2003) reported higher invertebrate diversity in the highly alkaline Caher River at an uncompacted river site with a lower gradient than

further downstream where the flow was relatively faster and the substrate was compacted by $CaCO_3$ deposits. Compaction of the streambed reduces or eliminates burrowing species from the fauna (Kelly-Quinn *et al.*, 2003; Rundio, 2009), as was the case in the Caher River, where epilithic chironomid larvae dominated and Oligochaeta, a common burrowing taxon, was found in very low numbers. The lack of substrate heterogeneity in rivers experiencing compaction may also cause a reduction in the diversity of species of Trichoptera with a habitat preference for stony or weedy substrata found to be absent from compacted sites of the Caher River (Kelly-Quinn *et al.*, 2003).

Ecosystem functioning may also be negatively impacted by the deposition of $CaCO_3$ through a reduction in the decomposition rate of leaves (Casas and Gessner, 1999) as a result of the activity of decomposers being impeded. This is similar to the delay or reduction in leaf litter breakdown observed in streams where siltation occurs (Reice, 1974). A reduction in the time taken to break down organic matter will directly affect nutrient cycling within the system. Individual species may benefit from varying degrees of deposition; for example, the grazing limnephild *Melampophylax mucoreus* displays faster larval growth and higher adult biomass when reared on stones with a carbonate tufa crust (Kock *et al.*, 2006). For other species, the effect varies, as deposition on the organism itself may reduce mobility and impair sensory organs (Dürrenfeld, 1978, cited in Ruff and Maier, 2000); however, these effects are poorly studied.

4.1.3 Naturally acidic rivers

River acidity occurs when the acid-neutralising capacity of the river is low, thereby reducing the ability of the system to neutralise acid inputs. These inputs can be natural, whereby the input of organic acids, such as humic acids, reduces the pH or keeps it naturally low (Driscoll *et al.*, 1989; Collier *et al.*, 1990). This is in contrast to the anthropogenic acidity that arises from an influx of inorganic ions, mainly sulphates and nitrates, through atmospheric deposition. In recent years, the amount of anthropogenic acidification has decreased significantly, with concentrations of marine and non-marine sulphates much reduced in Irish lakes and rivers (Burton and Aherne, 2012; Feeley *et al.*, 2013). The recovery observed in terms of hydrochemistry is also occurring in both Irish and international biological communities but at a much slower rate (Kowalik *et al.*, 2007; Feeley and Kelly-Quinn, 2014).

The effects of natural acidity appear less severe than those associated with anthropogenic acidity (Dangles *et al.*, 2004; Petrin *et al.*, 2008) as it occurs at a much slower rate over a long period of time (i.e. hundreds of years), which enables organisms to acclimatise and adapt to the altered levels of acidity. The low pH observed in naturally acidic rivers does not appear to be accompanied by an increase in the inorganic labile monomeric aluminium, a form toxic to fish and insects, associated with anthropogenic acidity. This has been attributed to the fact that the organic acids that drive natural acidity, such as humic acids, can bind up to 90% of the dissolved aluminium in the stream (Collier *et al.*, 1990), thereby rendering it non-toxic through complexation processes (Driscoll *et al.*, 1980; Dangles *et al.*, 2004).

However, the role of anthropogenic acidity has complicated our understanding of the communities to be expected in our naturally acid streams. Some studies have shown that the invertebrate communities of naturally acidified rivers tend to have a species richness more similar to that of circumneutral rivers (Winterbourn and Collier, 1987; Collier *et al.*, 1990; Dangles *et al.*, 2004) than to that of anthropogenically acidified rivers (Petrin *et al.*, 2008). However, community composition is relatively distinct in naturally acidic streams (e.g. 27% of the taxa found in Swedish streams studied by Dangles *et al.* (2004) were exclusive to naturally acidic streams). Similarly, in Great Britain, Kowalik *et al.* (2007) found that, regardless of the source of acidity, the macroinvertebrate communities of acid streams varied from those of circumneutral streams. However, they also found that individual species were affected by the nature of the acidity. For example, only three out of nine species classified as acid sensitive were present (*Baetis rhodani*, *Isoperla grammatica* and *Hydropsyche* spp.) in sites where the acid episode was driven by non-marine sulphate (i.e. artificial acidity), whereas eight out of the nine were present in sites where acidity was driven by base cation dilution and organic acids (i.e. natural acidity) (Kowalik *et al.* 2007). The only acid-sensitive species absent from the sites where acidity was natural was *Gammarus pulex* (Kowalik *et al.*, 2007). This result indicates two main findings: (1) that naturally acidic systems have a higher diversity of species than anthropogenically acidified streams, although this is likely to be limited on a site-by-site basis; and

(2) that local factors other than pH (e.g. low calcium levels) are important drivers of community composition and structure in naturally acidic systems. Furthermore, although it is clear that the biological communities of acid rivers differ from those of circumneutral streams in terms of community structure, more research is needed to determine if invertebrate taxa have in fact adapted or acclimatised to localised natural acidity conditions. If this is the case, then the biological communities of naturally acidic rivers are likely to differ from anthropogenically acidified systems. There is a difficulty in validating this in that, despite the large reduction in anthropogenic acidification, slow biological recovery rates and the historical legacy of anthropogenic acidity itself make it difficult to determine the baseline characteristics of a naturally acidic system.

In many acid-sensitive Irish streams, acidity is episodic in nature (Kelly-Quinn, 1997; Kelly-Quinn *et al.*, 1996; Feeley *et al.*, 2013), which further complicates our ability to understand and elucidate the subtle characteristics of naturally acidic systems. In Ireland at present, the primary driver of episodic acidity in peat-dominated catchments, and to a lesser extent, in non-peaty systems, is the addition of organic acids and base cation dilution (Burton and Aherne, 2012; Feeley *et al.*, 2013). Land-use (e.g. conifer forestry) may magnify the dissolved organic carbon (DOC) load and the pH response, resulting in periods of high acidity and more prolonged reduction in pH. Studies have shown that episodic acidification (i.e. severe pulses of acidity during periods of high precipitation, resulting in significant drops in pH) is typical of low-order, peat-covered, acid-sensitive catchments (Kelly-Quinn *et al.*, 1996). During storm flow events, pH values as low as 3.93 in the east and 4.0 in the south and west of the country have been recorded by Feeley *et al.* (2013); this is similar to the results of previous studies in both the west and east of the country (Bowman, 1986, 1991; Kelly-Quinn *et al.*, 1996; Allott *et al.*, 1997; Kelly-Quinn, 1997). This can have consequences for the biota, as pH alone can be extremely toxic to individual species such as *B. rhodani* (Kowalik and Ormerod, 2006). Nevertheless, Feeley and Kelly-Quinn (2014) recorded acid-sensitive invertebrate assemblages at all episodically acidic sites examined. Acid-sensitive streams in the east, on igneous geology (alkalinity of 1.09–3.59 mg $CaCO_3$/L), had lower numbers of acid-sensitive taxa than streams on more calcium-rich sedimentary and metamorphic geologies in the south (alkalinity of 10–40.10 mg $CaCO_3$/L) or west (alkalinity of 7.80–17.77 mg $CaCO_3$/L) of the country, indicating that underlying geology (and calcium concentrations) plays a significant part in structuring biological communities.

The AWICsp index is based on the occurrence of acid-sensitive or -tolerant taxa and the observed mean pH. The index takes into account taxon abundance and ranges from 1, indicating a community composed entirely of highly acid-tolerant species, to a score of between 10 and 14, representing a community with numerous highly acid-sensitive taxa. For AWICsp scores in the United Kingdom, a threshold of 7.38 has been adopted for reference communities in acid-sensitive humic waters (> 10 mg/L DOC), whereas the reference value for acid-sensitive clear waters in England and Wales is 7.68, and for Scottish clear waters is 8.61 (WFD-UKTAG, 2014). Non-forested sites in the south and west of Ireland recorded AWICsp scores above this threshold (Feeley and Kelly-Quinn, 2014), with forested sites having a lower score. Generally, non-forested acid-sensitive sites recorded higher AWICsp values than sites with high percentage forest cover within a catchment (Feeley and Kelly-Quinn, 2014). These results indicate that in the switch from anthropogenic to natural (although magnified by forest cover) acidity in recent years, the biological community has begun to return to a less acid-tolerant community. However, determining the end point of recovery and what constitutes a natural community in Irish organically acid streams is difficult to determine at this point in time.

Few studies have looked at the diatom communities of acid-sensitive rivers (Kelly, 2008; O’Driscoll *et al.*, 2012, 2014). A reasonable body of work exists on the phytobenthic algal communities of acidic habitats (both natural and anthropogenic) and on how acidification affects community structure (Müller, 1980; Mulholland *et al.*, 1986; Collier and Winterbourn, 1990; Kinross *et al.*, 1993; Ledger and Hildrew, 1998; DeNicola, 2000; Passy, 2006; Kelly, 2008; MacDougall *et al.*, 2008; O’Driscoll *et al.*, 2012, 2014). Acidic mountain streams in Poland hosted communities consisting mainly of acidophilic species, with diversity and richness positively correlated with pH (Kwandrans, 1993). DeNicola (2000) conducted a review of the published literature on the diatom communities of highly acidic environments, both naturally acidic sites and sites anthropogenically acidified mainly by acid mine drainage. The study concluded that pH has a profound effect on diatom richness, with the number of taxa decreasing significantly

in environments with a pH of 3.5 compared with those in the range of pH 4.5–5, suggesting a pH threshold for many species (DeNicola, 2000). Other factors, such as low silica concentrations, may limit diatom communities. An increase in algal biomass (green algae, filamentous algae and diatoms) with decreasing pH has been observed in some studies (Müller, 1980) but not others (Kinross *et al.*, 1993). The phytobenthic communities of naturally acidic streams varied compared with circumneutral streams (Collier and Winterbourn, 1990). While the epilithon in both the humic, brown-water streams and circumneutral streams consisted mainly of diatoms, the species differed between stream types. In addition, the epilithon of the acid streams contained filamentous algae, in contrast to the cyanobacteria present in the circumneutral streams (Collier and Winterbourn, 1990).

4.1.4 Rivers strongly influenced by lakes (lake outlets)

Lake outlets can be defined as the transitional zone between a lake and a stream (Malmqvist and Eriksson, 1995; Brunke, 2004). Lake proximity influences a number of physical and chemical processes in the outflowing river, the effects of which dissipate with distance from the lake. Outlet streams may have a more stable flow regime than areas further downstream, as the lake reduces the fluctuations in discharge resulting from precipitation and leading to mild changes in the water level (Giller and Malmqvist, 1998). This, however, tends not to be the case for Irish lake outlets, where lake level, and therefore outlet flow, is influenced by precipitation. The flow at many lake outlets may be very low during periods of low precipitation owing to the presence of a sill at the outlet. Outlets can be either lentic or lotic habitats depending on slope and flow regime (Brunke, 2004). They tend to have a higher, more stable temperature regime and a higher concentration of suspended particulate organic matter than further downstream (Harding, 1992; Giller and Malmqvist, 1998; Hieber *et al.*, 2002), as well as lower quantities of DOC (Larson *et al.*, 2007). Transportation of plankton can result in higher secondary productivity in outflows, which in turn influences the composition of the biological communities (Giller and Malmqvist, 1998; Donath and Robinson, 2001; Brunke 2004; Goodman *et al.*, 2010).

A reasonable body of research has examined the aquatic biota of lake outlet streams (Valett and Stanford, 1987; Robinson and Minshall, 1990; Richardson and Mackay, 1991; Malmqvist and Eriksson, 1995; Hoffsten, 1999; Hieber *et al.*, 2002). Interestingly, factors such as substrate and current flow were found to have more of an effect on invertebrate community structure and richness (Malmqvist and Eriksson, 1995; Brunke, 2004) than the trophic status or physical structure of the lake itself. The macroinvertebrate communities of lake outlets tend to be dominated by filter-feeders such as Simuliidae and Hydropsychidae (Valett and Stanford, 1987; Richardson and Mackay, 1991; Harding, 1992; Hoffsten, 1999), with filter-feeder densities declining sharply downstream in a similar trend to that observed for phytoplankton (Sheldon and Oswood, 1977; Morin *et al.*, 1988; Robinson and Minshall, 1990). How far downstream these effects reach varies depending on the size and flow rate of both the inflowing and outflowing rivers. Generally, a sharp reduction in filter-feeder densities and plankton biomass has been reported within 1–2 km of the outlet (Valett and Stanford, 1987; Robinson and Minshall, 1990; Hoffsten, 1999). Potential factors driving the change in invertebrate distribution patterns and structure include quantity and quality of organic matter/seston, flow, substrate, colonisation cycles and competition (Oswood, 1979; Richardson and Clifford, 1983; Richardson and Mackay, 1991; Malmqvist and Eriksson, 1995; Hoffsten, 1999). A pattern of peak filter-feeder densities occurring in the lake outlet and decreasing further downstream has been demonstrated in lakes of all trophic status; however, not all lake types exhibit this pattern. Unlike in lowland lakes, there was no significant increase in filter-feeder densities in the outlets of oligotrophic Alpine lakes (Maiolini *et al.*, 2006), possibly because concentrations of organic matter and seston may be limited and the lake may act as a sink for organic matter rather than a source (Robinson and Minshall, 1990; Hieber *et al.*, 2002; Maiolini *et al.*, 2006). Brunke (2004) demonstrated that lake outlets host a distinct invertebrate community from other river types in northern Germany and thus represent a separate type.

Relatively few studies have investigated the phytobenthic communities of lake outlets (Hieber *et al.*, 2001; Robinson and Kawecka, 2005). Despite recording a higher number of genera, the structures of Alpine lake outlet communities were observed to be similar to those in other Alpine streams and comprised mainly diatoms and cyanobacteria (Hieber *et al.*, 2001). Some taxa were present only in the outlets, including the diatom genera *Amphora*, *Denticula* and *Nitzschia*, and the cyanobacteria *Oscillatoria* and *Phormidium* (Hieber *et al.*,

2001). Similarly, a number of taxa found in other Alpine streams were absent from the outlets. Robinson and Kawecka (2005) reported similar assemblages in both inlet and outlet streams, although the distance between them was short. The lake did, however, exert a strong influence on the composition of the diatom assemblages by acting as a continuous source of colonists and thereby resulting in a higher species abundance in the outlets than in the inlets.

4.2 How to Deal with Rare River Types

Natural variability in ecosystems can present unusual physico-chemical and hydromorphological conditions that may ultimately determine the structure of the biological communities. In order to assess accurately the status of these unusual systems, it is essential that appropriate reference conditions are established and metrics developed to allow the effects of the natural variability to be disentangled from those of potential anthropogenic impacts from multiple pressure sources. The first step in tackling this issue is to characterise the biological communities of these systems and to determine if they do in fact represent a distinct national river type or are best described as a sub-type. The effectiveness of the assessment methods used to determine the ecological status of the system must be tested and may warrant modification to account for the natural variation observed. Few countries have addressed how we might deal with categorising and assessing rare river types.

Clearly, natural variability in biological communities needs to be characterised, and the key drivers of natural variation that are likely to affect the assessment methods need to be identified and incorporated into the typology (Basset *et al*., 2013), which may necessitate the development of new assessment metrics. This approach was taken in the United Kingdom when developing a bioassessment method to detect and measure the impact of acidification. The metric developed is the WFD-AWICsp index, which was adapted from the AWIC index (Davy-Bowker *et al*., 2006). While developing typology and class boundaries for this metric, McFarland (2010) found that humic waters in Scotland with DOC concentrations of > 10 mg/L hosted a distinct macroinvertebrate community compared with both Scottish sites with lower concentrations of DOC and other acidic sites in the United Kingdom. Humic acids tend to reduce the amount of inorganic monomeric aluminium associated with acidification and so may reduce the impact on the fauna, something which must be considered when developing a metric. As a result, DOC was included as a factor in the reference typology (WFD-UKTAG, 2014). A new method was also developed for detecting acidification impacts using diatoms. The DAM highlighted the importance of addressing the issue of natural acidity when setting appropriate reference or "expected" values (Juggins and Kelly, 2012), something which other metrics have not taken into account. Juggins and Kelly (2012) deem a waterbody "unimpacted" when the strong acid anions are in balance with base cations.

Alternatively, a new assessment metric may be developed to assess accurately a particular river type. In northern Germany, lake outlets were validated as a distinct river type, which led to the development of the Lake outlet Typology Index (LTI). This index follows a similar approach to that used in the Potamon Type Index for large rivers developed by Schöll and Haybach (2001), which works on the basis that all taxa on the species list showing a preference for that particular habitat are indicators of high quality (Brunke, 2004; Birk and Schmedtje, 2005). The LTI consists of five levels based on the preference a species has for this habitat; the preferences were checked using published autecological information. Some taxa also show a strong preference for either lentic or lotic outlets, with lentic outlet communities being more similar to dammed-up sites than lotic outlets. Lake outlet preference (LP) values were assigned to all 132 taxa, although not all were included in the total LP list based on results from ordinations, frequency tables and abundance distributions (Brunke, 2004). LP 1 species are species that are typical of lake outlets, for example *Hydropsyche angustipennis* and *Neureclipsis bimaculata*. Species considered characteristic of lake outlets include *Bithynia leachii* and *Athripsodes cinererus* (LP 2). LP 3 species in the LTI have a high affinity for lake outlets (e.g. *Hydropsyche pellucidula*, *Cloeon dipterum* and *Polycentropus flavomaculatus*), whereas LP 4 species are habitat generalists that can live in most habitats (e.g. *Ephemera danica*). LP 5 species are taxa that may occasionally be found in this habitat type but do not show a preference for it, for example *B. rhodani* and *Chaetopteryx villosa* (Brunke, 2004). The LTI is calculated based on the sum of the LP values for each taxon squared and divided by the total number of LP taxa. A quantitative version of the LTI was developed in which a weighting was applied to downplay the frequently lower occurrence of taxa most

typical of lake outlets compared with the more generalist taxa of LP 5. Brunke (2004) recommended the use of the LTI based on quantitative data (LTI_{quan}) to assess lake outlets in the lowlands of northern Germany, as it was able to classify the clearly good, very good and strongly degraded or impaired sites. The LTI_{quan} was also found to be more accurate and robust than the LTI based on qualitative data and was affected less by seasonal variations (Brunke, 2004). This approach may prove useful in dealing with lake outlets; however, it would have to be developed and adapted for the macroinvertebrate species occurring in Irish rivers and lakes.

5. Conclusion

There is little guidance at present specifically relating to intercalibrating new national river types. Currently, new river types must meet two criteria before being intercalibrated. First, the proposed new river type must correspond to one of the common intercalibration types of the appropriate GIG and, second, it must be found in at least two countries within that GIG. In the case of a rare or distinct river type, intercalibration may not be possible, and in such cases an explanation detailing the barriers to intercalibration should be given (ECOSTAT, 2011). The most recent technical report on intercalibration outlines procedures for intercalibrating new classification methods of ecological status (Willby *et al.*, 2014). A new or adapted classification method, combined with the addition of appropriate environmental factors to the typology to narrow variation within types, may be the best approach to ensure that unusual river types are accurately assessed under the WFD.

References

Allott, N., Brennan M., Cooke, D. *et al*., 1997. *Stream Chemistry, Hydrology and Biota, Galway-Mayo Region*. COFORD, Dublin, Ireland.

Álvarez, M. and Pardo, I., 2007. Factors controlling epilithon biomass in a temporary, karstic stream: the interaction between substratum and grazing. *Journal of the North American Benthological Society* 26:207–220.

Anon., 2010. *Delineation and Typology of Surface Water Bodies for Cycle 1 of the Water Framework Directive: River Basin Management*. Environment Agency, Rotherham, UK.

Basset, A., Barbone, E., Borja, A. *et al*., 2013. Natural variability and reference conditions: setting type-specific classification boundaries for lagoon macroinvertebrates in the Mediterranean and Black Seas. *Hydrobiologia* 704:325–345.

Birk, S. and Schmedtje, U., 2005. Towards harmonisation of water quality classification in the Danube River Basin: Overview of biological assessment methods for running waters. *Archiv für Hydrobiologie Supplement* 158:171–196.

Birk, S., Bonne, W., Borja, A. *et al*., 2012. Three hundred ways to assess Europe's surface waters: An almost complete overview of biological methods to implement the Water Framework Directive. *Ecological Indicators* 18:31–41.

Bouleau, G. and Pont, D., 2015. Did you say reference conditions? Ecological and socio-economic perspectives on the European Water Framework Directive. *Environmental Science & Policy* 47:32–41.

Boulton, A.J. and Hancock, P.J., 2006. Rivers as groundwater-dependent ecosystems: a review of degrees of dependency, riverine processes and management implications. *Australian Journal of Botany* 54:133–144.

Bowman, J.J., 1986. *Precipitation Characteristics and the Chemistry and Biology of Poorly Buffered Irish Lakes: a Western European Baseline for 'Acid Rain' Impact Assessment*. An Foras Forbartha, Dublin, Ireland.

Bowman, J, 1991. *Acid Sensitive Surface Waters in Ireland: The Impact of a New Major New Sulphur Emission on Sensitive Surface Waters in an Unacidified Region*. Report of the Environmental Research Unit, Dublin, Ireland.

Brunke, M., 2004. Stream typology and lake outlets – a perspective towards validation and assessment from northern Germany (Schleswig-Holstein). *Limnologica – Ecology and Management of Inland Waters* 34:460–478.

Burton, A. and Aherne, J., 2012. Changes in the Chemistry of Small Irish lakes. *AMBIO* 41:170–179.

Butcher, R.W., 1946. Studies in the ecology of rivers: VI. the algal growth in certain highly calcareous streams. *Journal of Ecology* 33:268–283.

Cannan, C.E. and Armitage, P.D., 1999. The influence of catchment geology on the longitudinal distribution of macroinvertebrate assemblages in a groundwater dominated river. *Hydrological Processes* 13:355–369.

Casas, J.J. and Gessner, M.O., 1999. Leaf litter breakdown in a Mediterranean stream characterised by travertine precipitation. *Freshwater Biology* 41:781–793.

Clarke, R.T., Wright, J.F. and Furse, M.T., 2003. RIVPACS models for predicting the expected macroinvertebrate fauna and assessing the ecological quality of rivers. *Ecological Modelling* 160:219–233.

Collier, K. and Winterbourn, M., 1990. Structure of epilithon in some acidic and circumneutral streams in South Westland, New Zealand. *New Zealand Natural Sciences* 17:1–11.

Collier, K.J., Ball, O.J., Graesser, A.K. *et al*., 1990. Do organic and anthropogenic acidity have similar effects on aquatic fauna? *Oikos* 59:33–38.

Crisp, D.T. and Westlake, D.F., 1982. The temperatures of three different small streams in north west England. *Hydrobiologia* 35:305–323.

Daly, D, 2009. Groundwater – the "hidden resource". *Biology and Environment: Proceedings of the Royal Irish Academy* 109B:221–236.

Dangles, O., Malmqvist B. and Laudon, H., 2004. Naturally acid freshwater ecosystems are diverse and functional: evidence from boreal streams. *Oikos* 104:149–155.

Davy-Bowker, J., Clarke, R., Johnson, R. *et al*., 2006. A comparison of the European Water Framework Directive physical typology and RIVPACS-type models as alternative methods of establishing reference conditions for benthic macroinvertebrates. *Hydrobiologia* 566:91–105.

DeNicola, D., 2000. A review of diatoms found in highly acidic environments. *Hydrobiologia* 433:111–122.

Dodkins, I., Rippey, B., Harrington, T.J. *et al.*, 2005. Developing an optimal river typology for biological elements within the Water Framework Directive. *Water Research* 39:3479–3486.

Donath, U. and Robinson, C.T., 2001. Ecological characteristics of lake outlets in Alpine environments of the Swiss Alps. *Archiv für Hydrobiologie* 150:207–225.

Donohue, I., McGarrigle, M.L. and Mills, P., 2006. Linking catchment characteristics and water chemistry with the ecological status of Irish rivers. *Water Research* 40:91–98.

Driscoll, C.T., Baker, J.P., Bisogni, J.J. *et al.*, 1980. Effect of aluminium speciation on fish in dilute acidified waters. *Nature* 284:161–164.

Driscoll, C., Fuller, R. and Schecher, W., 1989. The role of organic acids in the acidification of surface waters in the Eastern U.S. *Water, Air, and Soil Pollution* 43:21–40.

EC, 2000. Directive 2000/60/EC of the European Parliament and of the Council establishing a framework for the Community action in the field of water policy. OJ L 327, 22.12.2000, p. 1–73.

EC, 2003. Guidance Document No 10, *Rivers and Lakes – Typology, Reference Conditions and Classification Systems*. REFCOND. Office for Official Publications of the European Communities, Luxembourg.

EC, 2012a. Report from the Commission to the European Parliament and the Council on the implementation of the Water Framework Directive (2000/60/EC) river basin management plans. European Commission, Luxembourg.

EC, 2012b. Member State: United Kingdom. Report from the Commission to the European Parliament and Council on the Implementation of the Water Framework Directive (2000/60/EC) River Basin Management Plans. European Commission, Brussels, Belgium.

EC, 2012c. Member State: Germany. Report from the Commission to the European Parliament and Council on the Implementation of the Water Framework Directive (2000/60/EC) River Basin Management Plans. European Commission, Brussels, Belgium.

EC, 2013. Commission Decision 2013/480/EU of 20 September 2013 establishing, pursuant to Directive 2000/60/EC of the European Parliament and of the Council, the values of the Member State monitoring system classifications as a result of the intercalibration exercise and repealing Decision 2008/915/EC. OJ L 226, 8.10.2013, p 1–47.

ECOSTAT, 2011. Common implementation strategy for the Water Framework Directive (2000/60/EC). Guidance document No. 14. Guidance document on the intercalibration process 2008–2011. Office for Official Publications of European Communities, European Commission, Luxembourg.

ECOSTAT, 2014. WFD CIS Working Group Ecological Status. Final outcome of the work on broad types, links to national types and their use for European assessments. ECOSTAT meeting 22–23 October 2014. European Commission, Brussels, Belgium.

ECOSTAT, 2015. WFD CIS Working Group Ecological Status. Typology note: Definition of broad European lake and river types and way forward. Meeting March 2015. European Commission, Brussels, Belgium.

EHS, 2005. Water Framework Directive Summary Report of the characterisation and impact analyses required by Article 5 Northern Ireland. Environment and Heritage Service Northern Ireland (Environment Agency NI), Belfast, Northern Ireland.

Elliott, M. and Quintino, V., 2007. The estuarine quality paradox, environmental homeostasis and the difficulty of detecting anthropogenic stress in naturally stressed areas. *Marine Pollution Bulletin* 54:640–645.

EPA, 2005. *The Characterisation and Analysis of Ireland's River Basin Districts*. National Summary Report. EPA, Johnstown Castle, Ireland.

EPA, 2008. The Water Framework Directive – a new management approach. In EPA (ed.), *Ireland's Environment 2008*. EPA, Johnstown Castle, Ireland.

Feeley, H.B. and Kelly-Quinn, M., 2014. Re-examining the effects of episodic acidity on macroinvertebrates in small conifer-forested streams in Ireland and empirical evidence for biological recovery. *Biology & Environment: Proceedings of the Royal Irish Academy* 114B:205–218.

Feeley, H., Bruen, M., Blacklocke, S. *et al.*, 2013. A regional examination of episodic acidification response to reduced acidic deposition and the influence of plantation forests in Irish headwater streams. *Science of The Total Environment* 443:173–183.

Fölster, J., Sandin, L. and Wallin, M., 2004. A suggestion to a typology for Swedish inland surface waters according to the EU Water Framework Directive. Report 2004:13. Department of Environmental Assessment, Swedish University of Agricultural Sciences, Uppsala, Sweden. Available online: http://info1.ma.slu.se/IMA/Publikationer/internserie/2004–13.pdf (accessed 14/10/2015).

Giller, P.S. and Malmqvist, B., 1998. *The Biology of Streams and Rivers*. Oxford University Press, New York, NY, USA.

Goodman, K.J., Baker, M.A. and Wurtsbaugh, W.A., 2010. Mountain lakes increase organic matter decomposition rates in streams. *Journal of the North American Benthological Society* 29:521–529.

Harding, J.S., 1992. Physico-chemical parameters and invertebrate faunas of three lake inflows and outlets in Westland, New Zealand. *New Zealand Journal of Marine and Freshwater Research* 26:95–102.

Hartley, A.M., House, W.A., Callow, M.E. *et al*., 1997. Coprecipitation of phosphate with calcite in the presence of photosynthesizing green algae. *Water Research* 31:2261–2268.

Hieber, M., Robinson, C.T., Rushforth, S.R. *et al*., 2001. Algal communities associated with different Alpine stream types. *Arctic, Antarctic, and Alpine Research* 33:447–456.

Hieber, M., Robinson, C.T., Uehlinger, U. *et al*., 2002. Are alpine lake outlets less harsh than other alpine streams? *Archiv für Hydrobiologie* 154:199–223.

Hoffsten, P.-O., 1999. Distribution of filter-feeding caddisflies (Trichoptera) and plankton drift in a Swedish lake-outlet stream. *Aquatic Ecology* 33:377–386.

Holmes, N., Newman, J. and Chadd, S., 1999. *Mean Trophic Rank: a User's Manual*. Environment Agency, Bristol, UK.

Juggins, S. and Kelly, M.G., 2012. *Development of a Water Framework Directive-Compatible Metric for Assessing Acidification in UK and Irish Rivers*. Report SC070034/TR2. Environment Agency, Bristol, UK.

Kelly, M.G., 2008. Forestry and surface water acidification: effects on diatoms. Report to Western River Basins District, Galway, Ireland.

Kelly, M., Bennett, C., Coste, M. *et al*., 2009. A comparison of national approaches to setting ecological status boundaries in phytobenthos assessment for the European Water Framework Directive: results of an intercalibration exercise. *Hydrobiologia* 621:169–182.

Kelly, M.G., Gómez-Rodríguez, C., Kahlert, M. *et al*., 2012. Establishing expectations for pan-European diatom based ecological status assessments. *Ecological Indicators* 20:177–186.

Kelly-Quinn, M., 1997. *Aquafor Report: A Study of the Effects of Stream Hydrology and Water Quality in Forested Catchments on Fish and Invertebrates: Eolas contract HEIC91304A: Final Report*. COFORD, Dublin, Ireland.

Kelly-Quinn, M., Tierney, D., Coyle, C. *et al*., 1996. Factors affecting the susceptibility of Irish soft-water streams to forest-mediated acidification. *Fisheries Management and Ecology* 3:287–301.

Kelly-Quinn, M., Bradley, C., Murray, D.A., *et al*., 2003. Physico-chemical characteristics and macroinvertebrate communities of the Caher river. *Biology and Environment: Proceedings of the Royal Irish Academy* 103B:187–196.

Kelly-Quinn, M., Bradley, C., Dodkins, I. *et al*., 2005. *Water-Framework Directive – Characterisation of Reference Conditions and Testing of Typology of Rivers (2002-W-LS-7). Final Report*. EPA, Wexford, Ireland.

Kelly-Quinn, M., Tierney, D. and McGarrigle, M., 2009. Progress and challenges in the selection of type-specific reference conditions for Irish rivers and lakes. *Biology and Environment: Proceedings of the Royal Irish Academy* 109B:151–160.

Kinross, J.H., Christofi, N., Read, P.A. *et al*., 1993. Filamentous algal communities related to pH in streams in The Trossachs, Scotland. *Freshwater Biology* 30:301–317.

Kock, C., Meyer, A., Spänhoff, B. *et al*., 2006. Tufa deposition in karst streams can enhance the food supply of the grazing caddisfly *Melampophylax mucoreus* (Limnephilidae). *International Review of Hydrobiology* 91:242–249.

Kowalik, R.A. and Ormerod, S.J., 2006. Intensive sampling and transplantation experiments reveal continued effects of episodic acidification on sensitive stream invertebrates. *Freshwater Biology* 51:180–191.

Kowalik, R.A., Cooper, D.M., Evans, C.D. *et al*., 2007. Acidic episodes retard the biological recovery of upland British streams from chronic acidification. *Global Change Biology* 13:2439–2452.

Kwandrans, J., 1993. Diatom communities of acidic mountain streams in Poland. In Dam, H. (ed.), *Twelfth International Diatom Symposium*. Springer Netherlands, Heidelberg, Germany, pp. 335–342.

Larson, J.H., Frost, P.C., Zhiyu, Z. *et al*., 2007. Effects of upstream lakes on dissolved organic matter in streams. *Limnology and Oceanography* 52:60–69.

Ledger, M.E. and Hildrew, A.G., 1998. Temporal and spatial variation in the epilithic biofilm of an acid stream. *Freshwater Biology* 40:655–670.

Lorenz, A., Feld, C.K. and Hering, D., 2004. Typology of streams in Germany based on benthic invertebrates: Ecoregions, zonation, geology and substrate. *Limnologica – Ecology and Management of Inland Waters* 34:379–389.

Lu, G., Zheng, C., Donahoe, R.J. *et al*., 2000. Controlling processes in a $CaCO_3$ precipitating stream in Huanglong Natural Scenic District, Sichuan, China. *Journal of Hydrology* 230:34–54.

MacDougall, S.E., Carrick, H.J. and DeWalle, D.R., 2008. Benthic algae in episodically acidified Pennsylvania streams. *Northeastern Naturalist* 15:189–208.

McFarland, B.E.N., 2010. *Developing Typology and Class Boundaries for WFD-AWICsp to Assess Acidification in UK Rivers Using Macroinvertebrates*. UKTAG Freshwater Task Team, Environment Agency, Rotherham, UK.

McGarrigle, M. and Lucey, J., 2009. Intercalibration of ecological status of rivers in Ireland for the purpose of the water Framework Directive. *Biology and Environment: Proceedings of the Royal Irish Academy* 109B:237–246.

Maiolini, B., Lencioni, V., Boggero, A. *et al*., 2006. Zoobenthic communities of inlets and outlets of high altitude Alpine lakes. *Hydrobiologia* 562:217–229.

Malmqvist, B. and Eriksson, Å., 1995. Benthic insects in Swedish lake-outlet streams: patterns in species richness and assemblage structure. *Freshwater Biology* 34:285–296.

Mattson, R.A., Epler, J.H. and Hein, M.K., 1995. Description of benthic communities in karst, spring-fed streams of North Central Florida. *Journal of the Kansas Entomological Society* 68:18–41.

Morin, A., Back, C., Chalifour, A. *et al*., 1988. Effect of black fly ingestion and assimilation on seston transport in a Quebec lake outlet. *Canadian Journal of Fisheries and Aquatic Sciences* 45:705–714.

Mulholland, P.J., Elwood, J.W., Palumbo, A.V. *et al*., 1986. Effect of stream acidification on periphyton composition, chlorophyll, and productivity. *Canadian Journal of Fisheries and Aquatic Sciences* 43:1846–1858.

Müller, P., 1980. Effects of artificial acidification on the growth of periphyton. *Canadian Journal of Fisheries and Aquatic Sciences* 37:355–363.

Murphy, K. and Glasgow, G., 2009. North–South coordination in Ireland's international river basin districts. *Biology and Environment: Proceedings of the Royal Irish Academy* 109B:139–150.

O'Driscoll, C., de Eyto, E., Rodgers, M. *et al*., 2012. Diatom assemblages and their associated environmental factors in upland peat forest rivers. *Ecological Indicators* 18:443–451.

O'Driscoll, C., de Eyto, E., Rodgers, M. *et al*., 2014. Spatial and seasonal variation of peatland-fed riverine macroinvertebrate and benthic diatom assemblages and implications for assessment: a case study from Ireland. *Hydrobiologia* 728:67–87.

Oswood, M., 1979. Abundance patterns of filter-feeding Caddisflies (Trichoptera: Hydropsychidae) and seston in a Montana (U.S.A.) lake outlet. *Hydrobiologia* 63:177–183.

Pardo, I., Gómez-Rodríguez, C., Wasson, J.-G. *et al*., 2012. The European reference condition concept: A scientific and technical approach to identify minimally-impacted river ecosystems. *Science of the Total Environment* 420:33–42.

Passy, S. I., 2006. Diatom community dynamics in streams of chronic and episodic acidification: the roles of environment and time. *Journal of Phycology* 42:312–323.

Pedley, M., 1992. Freshwater (phytoherm) reefs: the role of biofilms and their bearing on marine reef cementation. *Sedimentary Geology* 79:255–274.

Pentecost, A., 2005. *Travertine*. Springer, Berlin, Germany.

Petrin, Z., Laudon, H. and Malmqvist, B., 2008. Diverging effects of anthropogenic acidification and natural acidity on community structure in Swedish streams. *Science of the Total Environment* 394:321–330.

Petts, G.E., Bickerton, M.A., Crawford, C. *et al*., 1999. Flow management to sustain groundwater-dominated stream ecosystems. *Hydrological Processes* 13:497–513.

Pitois, F., Jigorel, A. and Bertru, G., 2003. Development of cyanobacterial build-up and evolution of river bed morphology in the chalk stream Eaulne (Upper-Normandy, France). *Biodiversity & Conservation* 12:621–636.

Poikane, S., Zampoukas, N., Borja, A. *et al*., 2014. Intercalibration of aquatic ecological assessment methods in the European Union: Lessons learned and way forward. *Environmental Science & Policy* 44:237–246.

Pottgiesser, T. and Sommerhäuser, M., 2004. *Profiles of German Stream Types*. Available online: http://reform.gisinternet.nl/images/2/2b/Pottgiesser_Sommerhaueser_2004.pdf (accessed 14 October 2015).

Rawer-Jost, C., Zenker, A. and Böhmer, J., 2004. Reference conditions of German stream types analysed and revised with macroinvertebrate fauna. *Limnologica – Ecology and Management of Inland Waters* 34:390–397.

Reice, S.R., 1974. Environmental patchiness and the breakdown of leaf litter in a woodland stream. *Ecology* 55:1271–1282.

Richardson, J.S. and Clifford, H.F., 1983. Life history and microdistribution of *Neureclipsis bimaculata* (Trichoptera: Polycentropodidae) in a lake outflow stream of Alberta, Canada. *Canadian Journal of Zoology* 61:2434–2445.

Richardson, J.S. and Mackay, R.J., 1991. Lake outlets and the distribution of filter feeders: an assessment of hypotheses. *Oikos* 62:370–380.

Robinson, C. and Kawecka, B., 2005. Benthic diatoms of an Alpine stream/lake network in Switzerland. *Aquatic Sciences* 67:492–506.

Robinson, C.T. and Minshall, G.T., 1990. Longitudinal development of macroinvertebrate communities below oligotrophic lake outlets. *The Great Basin Naturalist* 50:303–311.

Ruff, H. and Maier, G. 2000. Calcium carbonate deposits reduce predation pressure on *Gammarus fossarum* from salamander larvae. *Freshwater Biology* 43:99–105.

Rundio, D.E., 2009. Community–habitat relationships in coastal streams in Big Sur, California, USA: travertine influences macroinvertebrate abundance and community structure. *Hydrobiologia* 620:91–108.

Sandin, L. and Verdonschot, P.M., 2006. Stream and river typologies – major results and conclusions from the STAR project. *Hydrobiologia* 566:33–37.

Schöll, F. and Haybach, A., 2001. Bewertung von großen Fließgewässern mittels Potamon-Typie-In-dex (PTI). *Verfahrensbeschreibung und Anwendungsbeispiele.-Bundesanstalt für Gewässerkunde, Mitteilung* 23:1–28.

Sear, D.A., Armitage, P.D. and Dawson, F.H., 1999. Groundwater dominated rivers. *Hydrological Processes* 13:255–276.

Sheldon, A. and Oswood, M., 1977. Blackfly (Diptera: Simuliidae) abundance in a lake outlet: test of a predictive model. *Hydrobiologia* 56:113–120.

Smith, H. and Wood, P.J., 2002. Flow permanence and macroinvertebrate community variability in limestone spring systems. *Hydrobiologia* 487:45–58.

Smith, H., Wood, P.J. and Gunn, J., 2003. The influence of habitat structure and flow permanence on invertebrate communities in karst spring systems. *Hydrobiologia* 510:53–66.

Sommerhäuser, M., 2002. Stream typology, reference conditions and stream assessment in Germany. In Rouppa, M. and Karttunen, K. (eds.), *Typology and Ecological Classification of Lakes and Rivers*. TemaNord 2002:566. Nordic Council of Ministers, Copenhagen, Denmark.

Tedd, K., Coxon, C., Misstear, B. *et al.*, in preparation. *Natural Background Levels of Irish Groundwater Quality. Report to the Environmental Protection Agency.* EPA, Johnstown Castle, Ireland.

UKTAG, 2003. Guidance on typology for rivers for Scotland, England and Wales (Final Working Paper). UKTAG Work Program – Task 2a. Typology for Rivers. UK Technical Advisory Group on the Water Framework Directive, Bristol, UK.

UKTAG, 2004. Type Specific Reference Condition Descriptions for Rivers in Great Britain. TAG Work Programme 8a (02) Reference conditions for Rivers. UK Technical Advisory Group on the Water Framework Directive. UK (v2. PR1 15-03-05). UK Technical Advisory Group on the Water Framework Directive, Bristol, UK.

Valett, H.M. and Stanford, J.A., 1987. Food Quality and Hydropsychid Caddisfly Density in a Lake Outlet Stream in Glacier National Park, Montana, USA. *Canadian Journal of Fisheries and Aquatic Sciences* 44:77–82.

WFD-UKTAG, 2014. UKTAG River Assessment Method. Benthic Invertebrate Fauna. Invertebrates (Anthropogenic Acidification): WFD Acid Water Indictor Community (WFD-AWIC). Water Framework Directive – United Kingdom Advisory Group (WFD-UKTAG), Stirling, Scotland.

Whiting, P.J. and Stamm, J., 1995. The hydrology and form of spring-dominated channels. *Geomorphology* 12:233–240.

Willby, N.J., Birk, S., Poikane, S. *et al.*, 2014. Water Framework Directive intercalibration manual. Procedure to fit new or updated classification methods to the results of a completed intercalibration. European Commission Joint Research Centre Institute for Environment and Sustainability, Publications Office of the European Union, Luxembourg.

Williams, D.D., Williams, N.E. and Cao, Y., 1997. Spatial differences in macroinvertebrate community structure in springs in southeastern Ontario in relation to their chemical and physical environments. *Canadian Journal of Zoology* 75:1404–1414.

Winterbourn, M.J. and Collier, K.J., 1987. Distribution of benthic invertebrates in acid, brown water streams in the South Island of New Zealand. *Hydrobiologia* 153:277–286.

Wood, P.J. and Armitage, P.D., 2004. The response of the macroinvertebrate community to low-flow variability and supra-seasonal drought within a groundwater dominated stream. *Archiv für Hydrobiologie* 161:1–20.

Wood, P.J., Armitage, P.D., Cannan, C.E. *et al.*, 1999. Instream mesohabitat biodiversity in three groundwater streams under base-flow conditions. *Aquatic Conservation: Marine and Freshwater Ecosystems* 9:265–278.

Younger, P.L., 2009. *Groundwater in the Environment: an Introduction*. John Wiley & Sons, Chichester, UK.

Abbreviations

AWIC	Acid Water Indicator Community index
AWICsp	Acid Water Indicator Community species (index)
BQE	biological quality element
$CaCO_3$	calcium carbonate
CO_2	carbon dioxide
DAM	Diatom Acidification Metric
DO	dissolved oxygen
DOC	dissolved organic carbon
EQR	ecological quality ratio
EPA	Environmental Protection Agency
EU	European Union
GIG	Geographical Intercalibration Group
LP	lake outlet preference (value)
LTI	Lake outlet Typology Index
MTR	Mean Trophic Rank
RDB	River Basin District
RIVPACS	River Indicator Prediction and Classification System
TDI	Trophic Diatom Index
WFD	Water Framework Directive

AN GHNÍOMHAIREACHT UM CHAOMHNÚ COMHSHAOIL

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

Is féidir obair na Gníomhaireachta a roinnt ina trí phríomhréimse:

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

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

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

Ár bhFreagrachtaí

Ceadúnú

Déanaimid na gníomhaíochtaí seo a leanas a rialú ionas nach ndéanann siad dochar do shláinte an phobail ná don chomhshaol:

- saoráidí dramhaíola *(m.sh. láithreáin líonta talún, loisceoirí, stáisiúin aistrithe dramhaíola)*;
- gníomhaíochtaí tionsclaíocha ar scála mór *(m.sh. déantúsaíocht cógaisíochta, déantúsaíocht stroighne, stáisiúin chumhachta)*;
- an diantalmhaíocht *(m.sh. muca, éanlaith)*;
- úsáid shrianta agus scaoileadh rialaithe Orgánach Géinmhodhnaithe *(OGM)*;
- foinsí radaíochta ianúcháin *(m.sh. trealamh x-gha agus radaiteiripe, foinsí tionsclaíocha);*
- áiseanna móra stórála peitril;
- scardadh dramhuisce;
- gníomhaíochtaí dumpála ar farraige.

Forfheidhmiú Náisiúnta i leith Cúrsaí Comhshaoil

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

Bainistíocht Uisce

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

Monatóireacht, Anailís agus Tuairisciú ar an gComhshaol

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

Rialú Astaíochtaí na nGás Ceaptha Teasa in Éirinn

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

Taighde agus Forbairt Comhshaoil

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

Measúnacht Straitéiseach Timpeallachta

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

Cosaint Raideolaíoch

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

Treoir, Faisnéis Inrochtana agus Oideachas

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

Múscailt Feasachta agus Athrú Iompraíochta

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

Bainistíocht agus struchtúr na Gníomhaireachta um Chaomhnú Comhshaoil

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

- An Oifig um Inmharthanacht Comhshaoil
- An Oifig Forfheidhmithe i leith cúrsaí Comhshaoil
- An Oifig um Fianaise is Measúnú
- An Oifig um Cosaint Raideolaíoch
- An Oifig Cumarsáide agus Seirbhísí Corparáideacha

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

EPA Research Report 166

Characterisation of Referenece Conditions for Rare River Types Literature Review

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

Authors: Edel Hannigan and Mary Kelly-Quinn

Identifying Pressures

Ireland's freshwaters are under pressure from a range of land-use and anthropogenic stressors. In the 2010 - 2012 monitoring period, 47% of the river water bodies monitored by the Environmental Protection Agency in Ireland failed to reach good ecological status, the minimum level required under the Water Framework Directive. Climate change coupled with future land-use intensification may further stress river systems potentially resulting in a reduction in the ecological status of numerous waterbodies. These challenges emphasise the need for the accurate assessment of the ecological status of aquatic resources. This report is the Literature Review of the RareType project, which will seek to determine if the current 12 national river types adequately represent rare river types; and if the existing metrics accurately assess their status.

Informing Policy

The Water Framework Directive is the key EU legislation requiring Member States to improve and sustainably manage and protect water quality in all surface water bodies. Under the WFD, all EU Member States are obliged to develop a river typology upon which type-specific reference conditions can be defined to enable accurate evaluation of ecological status. Ecological status is determined on the basis of deviation from these type-specific reference conditions. Rare river types were not adequately represented in the development of the existing national typology and so it is not known whether they are sufficiently characterised by the current 12 national river types or represent distinct types. This report is the Literature Review of the RareType project, which will contribute to further refinement of the national typology by characterising the biological communities of rare rivers to determine if they represent distinct river types, and will assess the effectiveness of current metrics in determining their ecological status.

Developing Solutions

The first key step is to determine if the four categories of rare river type (naturally acid rivers, lake outlets, highly calcareous rivers, and groundwater-dominated rivers) examined in the RARETYPE project do in fact represent river types distinct from the 12 national types. If this is the case, then type-specific reference conditions must be established for the new types. This report is the Literature Review of the Rare Type project, which will help ensure that rare river types are categorised and assessed correctly as required by the WFD.

Comhshaol, Pobal agus Rialtas Áitiúil
Environment, Community and Local Government

EPA Research: McCumiskey House, Richiew, Clonskeagh, Dublin 14.

Phone: 01 268 0100
Twitter: @EPAResearchNews
Email: research@epa.ie

www.epa.ie

EPA Research Webpages
www.epa.ie/researchandeducation/research/