FRESHWATER BIOLOGY AND ECOLOGY HANDBOOK

Practitioners' Guide to Improving and Protecting River Health

Focus on Invertebrate Monitoring and Assessment

JOHN MURRAY-BLIGH AND MARTIN GRIFFITHS

FWR FOUNDATION *for* WATER RESEARCH

FRESHWATER BIOLOGICAL ASSOCIATION

PUBLISHED BY

FOUNDATION for WATER RESEARCH

Our Mission

FWR's mission is to advance the education of the public in the science, engineering and management of water resources, water treatment, water supply and use of water, the collection, treatment and re-use of wastewaters, the water environment in general and related subjects.

Our Vision

FWR's vision is a society well informed on water and related environmental matters: one whose citizens are knowledgeable about the issues affecting the sustainable management of water and are thus empowered to contribute to environmental stewardship.

www.fwr.org

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IN PARTNERSHIP WITH

FRESHWATER BIOLOGICAL ASSOCIATION

Our Vision

Our vision is to be the leading independent UK organisation for freshwater information and advice. It's imperative that we support our freshwaters as best we can, so it is our mission to promote the sustainable management of freshwater ecosystems and resources, using the best available science.

The aims and goals of the Freshwater Biological Association are:

- to widen our membership and enthusiasm in the freshwater environment
- to provide evidence and information that helps to protect and conserve unique environments
- to influence and broaden advocacy through outreach and public engagement
- to facilitate the setting of the research agenda, ensuring that science benefits wider environmental aspirations in freshwater

www.fba.org.uk

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Overview

This handbook provides an overview of the biological and ecological methods used to assess the status of the freshwater environment. Good river health is the key outcome and aim of this work. The principles of the delivery of the Water Framework Directive remain constant and feed forward into future river management approaches.

Chapter 1 provides an overview, including the legal framework for freshwater biological monitoring

Chapter 2 is a practitioner's guide to the standard methods for invertebrate sampling and data collection

Chapter 3 provides an understanding of current river invertebrate classification methodologies, focussing on RIVPACS and Surveillance Monitoring.

Chapter 4 looks at other sampling methods for Investigative Monitoring

Chapter 5 looks at indices and data analyses for investigations, including the increasing contribution from citizen science programmes.

Chapter 6 considers the reporting methods used in the UK and the EU, specifically with links to investment programmes, driven by the monitoring and assessment information. It also provides links to publicly available datasets.

The focus is on river invertebrate methodologies and on status classification using UK RIVPACS to provide a working example of what is needed to set up a biological monitoring programme for a national initiative, a river catchment or a specific tributary. Most invertebrate methods utilise these key principles and we expect users to modify and adapt methods to their specific situations as needed. Several key biological and ecological methods are not covered in this handbook, including fish, macrophytes, diatoms, river restoration methodologies, still-water methods, and statistics and computing methods. We invite other specialists to work with FWR and FBA to add additional chapters or sections to expand its coverage.

The core elements described here are the basis for training programmes and university teaching, to provide the expertise to consolidate the improvement of river health into the future. We also hope that this provides a useful insight for civil servants, water managers, specialists, and river conservation groups working to improve and protect our invaluable freshwater environment.

We invite you to contribute.

FWR
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Freshwater Biological Association

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FRESHWATER BIOLOGY AND ECOLOGY HANDBOOK

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Foreword

Dr Eric Valentine

Chair
Foundation for Water Research

On behalf of the Foundation for Water Research (FWR) I am pleased to introduce the *Freshwater Biology and Ecology Handbook*. It is a unique document providing access to the policy, regulation, guidance, practical and operational methods for assessing ecosystem health.

FWR, with its core aim to advance the education of the public in the science, engineering and management of water resources, is ideally placed to sponsor and promote the knowledge exchange material embodied in this handbook. Our independent and charitable status allows us to make this e-handbook available free of charge, and for public good.

The Handbook gives access to the considerable body of work that has been undertaken by the UK and European Union to bring biological monitoring and assessment methods into the mainstream of water quality assessment, classification and reporting. Expensive infrastructure and water management decisions are being driven by our understanding of the biology and ecology of our rivers and lakes. This brings the need to have consistent and high quality information on the status of our water environment.

The extensive research and development undertaken to implement the EU Water Framework Directive has accelerated the development of biological and ecological methods. This builds on the significant biological expertise in the UK and Europe, developed over the past 50 years. Most of this information resides in difficult to find policy documents, reports, technical guidance and academic papers. The book brings this information together in one place to allow structured access to this extensive body of work.

FWR worked with the authors – John Murray Bligh, Martin Griffiths and their colleagues – to produce this book. We are also pleased to jointly badge and promote this document in partnership with the Freshwater Biological Association (FBA). FBA and the Environment Agency have provided expertise and early scoping and design ideas for the book. We regard this as an open document, and we hope that others will continue to contribute additional material and add new chapters to the book, to increase its breadth and to keep information current as policy, science and practical guidance moves forward.

This is the second in a series of handbooks, and is a companion document to the *Regulation for Water Quality* book sponsored by FWR in 2014. This is also available via our website. We hope that the combination of the information made available in these books will accelerate the understanding of the steps necessary to improve and protect our aquatic ecosystems in the UK, in the EU and worldwide. The principles of water regulation, biological assessment and water management are common across the globe. Sharing good practice through these handbooks will allow methods to be developed and adapted for all aquatic systems. In this way we can adapt to climate change, improve biodiversity and protect essential water resources in a sustainable way.

Finally, FWR is undergoing a number of transformational changes to its structure. The trustees are intent on maintaining the key aims and mission of FWR within a new organisational body. This book will be one of the key documents passed on to the new structure which should be operating in its new format from mid 2022.

Dr Eric Valentine – May 2022

Simon Johnson

Executive Director
Freshwater Biological Association

I am pleased to introduce this *Freshwater Biology and Ecology Handbook* which closely aligns with our core aims and vision of advancing freshwater science and conservation action.

Freshwater Biological Association (FBA) is dedicated to understanding and protecting freshwaters across the globe. Over the past 100 years FBA has contributed to the fundamental science behind current monitoring and assessment methods. FBA expertise, with partners, shaped the core monitoring and assessment methods in use by the environment agencies in the UK, the European Union and across the globe. We continue to work at the front line of aquatic biology, often with partners in government, academia, environment agencies and the Rivers Trusts. This book has been produced in partnership with the Foundation for Water Research, and FBA Fellows have contributed their expertise and considerable knowledge to the book.

The EU Water Framework Directive establishes core principles for the long-term management of the water environment, with a focus on Good Ecological Status. The UK, through its 25 Year Environment Plan, and the Environment Act 2021, has committed to take forward and develop these key aims. The research and development undertaken to implement the Directive across Europe and the UK is a rich source of information, bringing aquatic biology into the mainstream alongside the traditional water chemistry driven regulatory frameworks. The book provides a structured way of gaining access to this extraordinary knowledge base.

This book brings together, in one place, the significant biological and ecological expertise and knowledge that has been developed in the UK and across Europe. Much of this information is contained in the 'grey literature' within government reports, technical guidance documents and operational instructions. It is difficult to find.

This handbook is unique in bringing together the underlying principles of biological monitoring and assessment, allowing access to the underlying methodology and documents, and providing practical information for users. It has been designed such that it can be read at different levels and should be useful for a range of users including policymakers, water managers, field biologists, volunteer organisations and interested individuals. It should allow those wishing to protect water resources to set up biological monitoring programmes in water catchments across the globe.

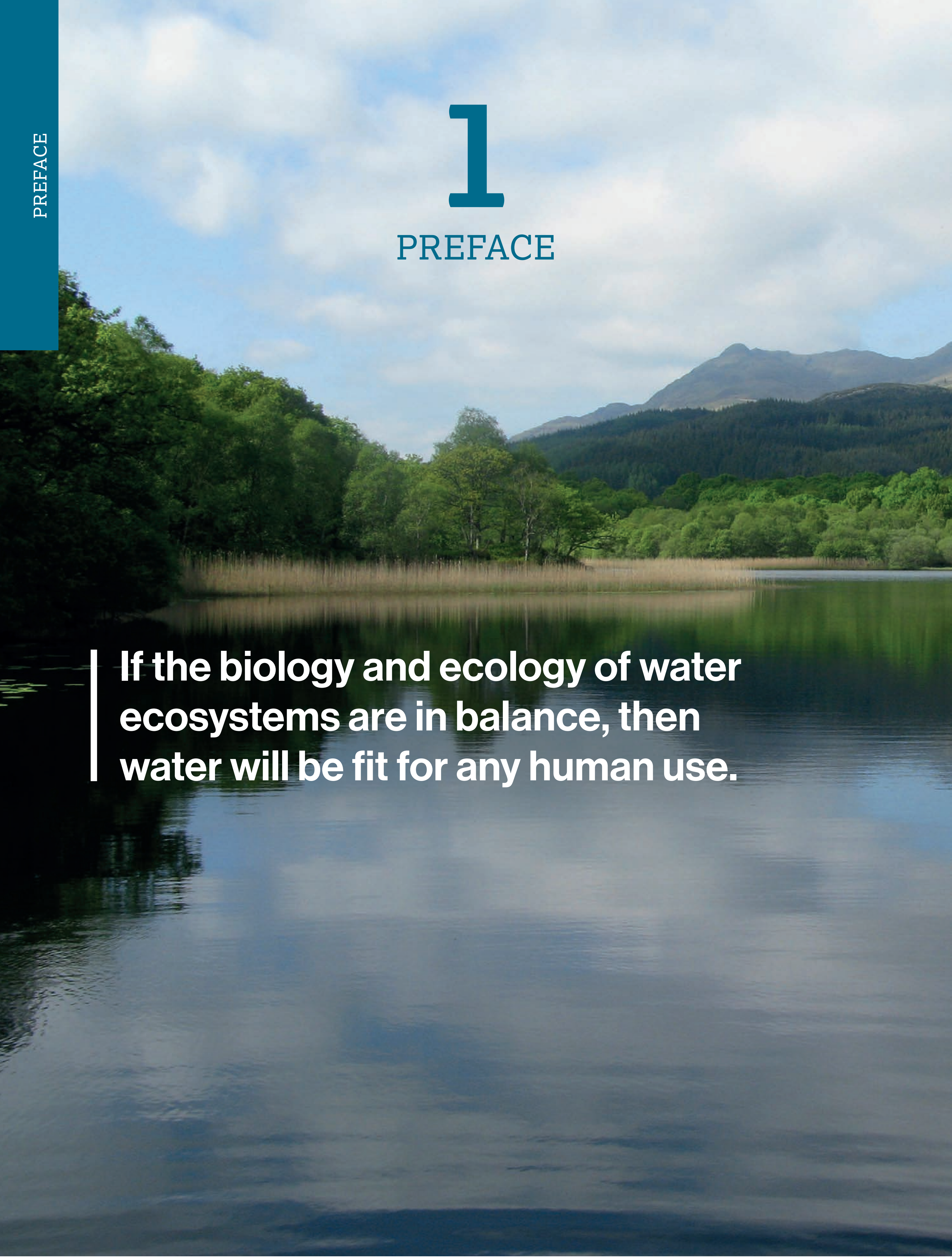
Field monitoring and practical research and development is a fundamental part of our work at FBA. We continue to develop training programmes and use our partnerships and networks to encourage further protection and improvement of the water environment. We hope that the book will provide the focus for university courses, in-house training and knowledge exchange programmes, nationally and internationally.

We regard this as an open and live document. This initial publication focuses on invertebrate biological methods. There are other methods, including those using fish, macrophytes and algae, that in combination give a wider picture of aquatic health. We invite others to consider writing or contributing additional chapters.

By making this freely available via our website, and in partnership with FWR, we hope others can build and improve methods to suit aquatic habitats across the globe.

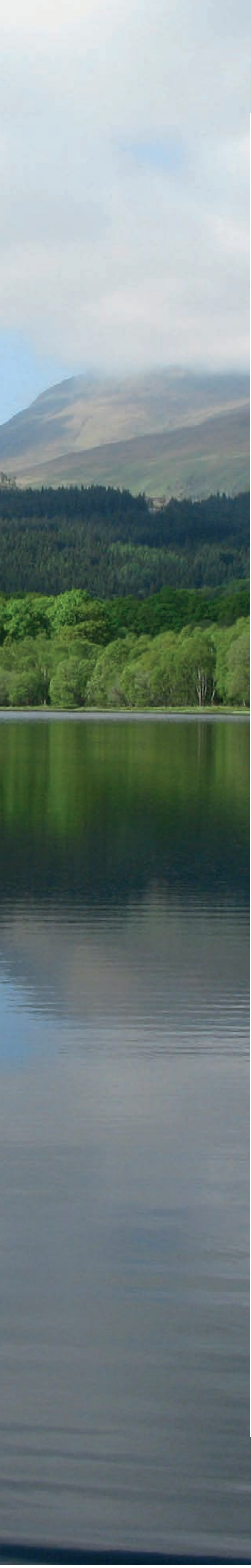
Open access to information will allow the partnerships necessary to improve our rivers and lakes. There is no doubt that high biodiversity in healthy rivers and lakes is a key indicator of the sustainable water resource management to which we must aspire.

Simon Johnson – May 2022



1
PREFACE

If the biology and ecology of water ecosystems are in balance, then water will be fit for any human use.



The health of our water ecosystem is critical to the environment on which we depend and is therefore directly related to human wellbeing. We increasingly realise that the ultimate test of ecosystem health is to monitor the biological and ecological indicators present in the environment. We must protect this precious natural resource and manage the environment sustainably, using long-term river basin planning and management. These principles are enshrined in UK and European Union (EU) legislation under the **Water Framework Directive** (WFD).[1]

What we value as 'the environment' is widely thought of as a physical entity, but it is in fact mainly biologically driven. It is the natural biota and its ecological interactions with the abiotic environment that maintains the health of our environment, essential to our existence. These beneficial processes are known as ecosystem services. We can damage the physical environment, but it is the biotic component of freshwaters that is particularly sensitive to damage from human activity. Not only pollution, but activities such as land use, abstraction, impoundment, physical modification, air pollution, climate change and allowing alien invasive species and diseases to spread: all affect the natural biota and therefore the health of aquatic environments.

We manage the pressures caused by human activities in an integrated way using ecological quality objectives. The best way to define ecological quality and the objectives based on it, is to focus on the actual composition of the biota. Specifically, the degree to which biota are impacted by human activities.

We define biological quality as its proportional deviation from a theoretical near-natural condition. The deviation is a measure of human impact. Biological monitoring provides a direct way to measure environmental quality in relation to biological and ecological quality objectives. In effect, we are measuring the natural capital that provides the ecosystem services. We support our ecological objectives with chemical and physical objectives (including hydrological objectives) to make it easier to regulate the release of industrial, agricultural and domestic waste, and the abstraction of water.

To prioritise our regulatory activities and to focus essential environmental monitoring, we assess the risks and pressures on aquatic environments. These pressures are summarised in Figure 1. Understanding the geographical, geological and anthropogenic pressures within a river basin is often the starting point for river basin planning. In WFD terms, this is known as 'river basin characterisation'. This allows chemical and physical objectives to be set at levels that support the biota necessary to achieve our environmental objectives, and protect human health.

Within river basins we set our ecological objectives at a level that maximises the benefits to our wellbeing, including our health and economy; in other words, at a level that maximises the ecosystem services that the environment provides. We try to avoid over-exploitation and unnecessary damage because this reduces the resilience of the ecosystem and its ability to recover from minor damage, and may cause a long-term or permanent reduction in the services that it provides. Meeting these ecological objectives aims to ensure that the water environment is managed as sustainably as possible.

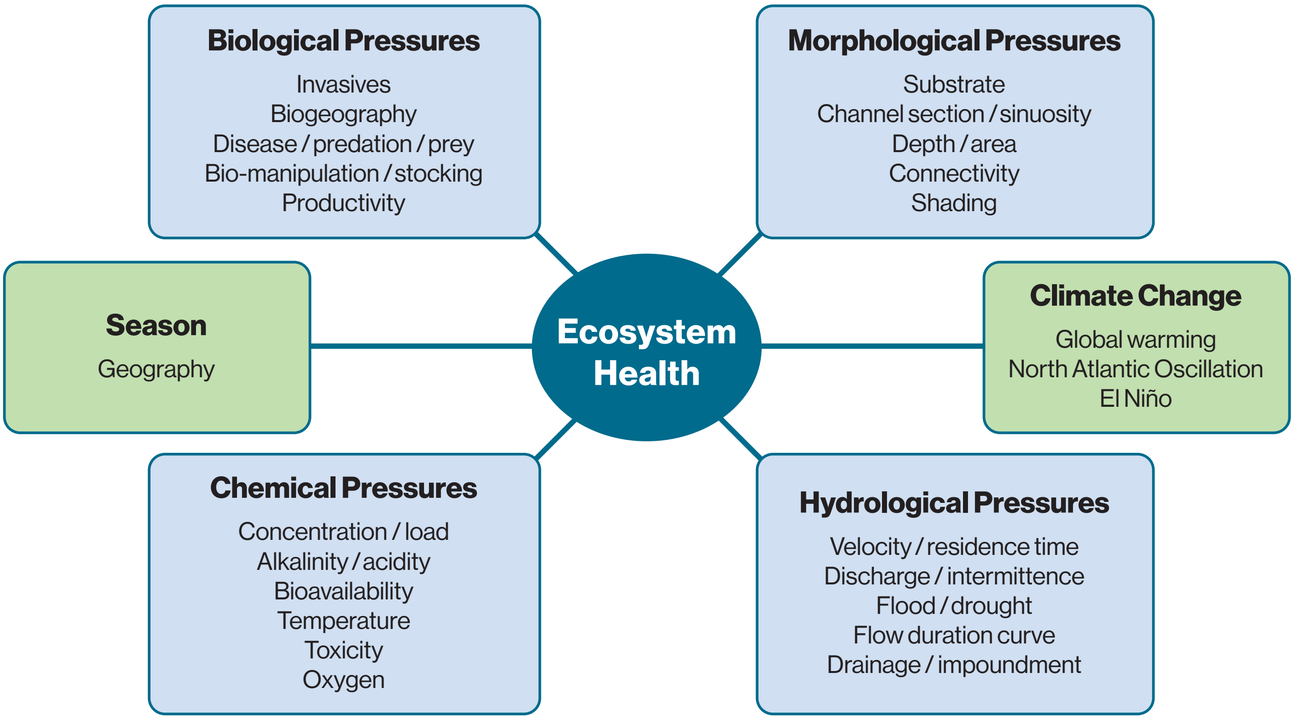


Figure 1

Main environmental pressures and their units of measure. *Source - Adapted from TAG Guidance.*[2]

In the past, when freshwater quality was dominated by pollution from untreated or partially treated sewage and industrial effluent, we were able to improve environmental quality simply by measuring and controlling physico-chemical parameters. Now that these sources of gross pollution have largely been brought under control, we are left with a wide range of 'multiple' pressures that prevent the biota from achieving the quality that maximises its ecosystem services. It is impossible to measure these pressures directly on a common scale, but biological measurements provide a uniform way to measure their combined impacts.

With a changing climate and a growing population, people are putting ever greater pressure on natural water systems. We must adapt to climate change. Water adaptation strategies must be made in a sustainable way, whilst protecting and improving aquatic ecosystems and biodiversity. Understanding the impacts of our activities on aquatic ecosystems and distinguishing these from the effects of natural processes is critical.

Baseline assessments of the state of our environment are important because they enable us to monitor changes in quality, and predictive techniques are crucial for identifying the mitigation strategies that will achieve the desired environmental outcomes. Understanding minimum flow requirements, the effects of climate change, the influence of chemical pollution and physical alterations on the biota are important. They are the focus of ongoing research and development of our biological and ecological monitoring techniques.

The European Union (EU) Water Framework Directive is the core regulatory instrument to maintain and improve the water environment across the EU to maintain and improve the water environment. The WFD came into force in 2000 and set a timetable for implementation. It provides a long-term water planning framework for all river basins across Europe. The first of three six-year river basin planning cycles began in 2010 with the publication of River Basin Plans for all EU rivers, and this was repeated in 2015. The next cycle began in 2021 and is timetabled to complete in 2027.

The breakthrough provided by the WFD was to focus on biological and ecological protection and improvement. For the first time, in the UK and the rest of Europe, environmental objectives were defined principally in biological terms and assessed in biological monitoring programmes, supported by physico-chemical standards and monitoring.

The Water Framework Directive

Prevents further deterioration and protects and enhances the status of aquatic ecosystems and, with regard to their water needs, terrestrial ecosystems and wetlands directly depending on the aquatic ecosystems.

The role of biological and ecological approaches to water management within the WFD will be developed further in **Chapter 1**.

This focus on ecological outcomes has required significant development of monitoring and assessment systems which, for the first time, will be used to drive significant investment and infrastructure development to meet the new objectives. This has promoted significant co-operative research and method development across Europe, aligned with the needs of the WFD.

This handbook aims to capture essential technical information and guidance derived from the EU, UK and international research and make it available in a structured way to practitioners and interested parties.

Substantial progress in biological monitoring and assessment methods have been made in the UK and across Europe, driven by the WFD. Similar approaches are being developed and adopted across the world, in both developed and developing countries. Requests for technical assistance, knowledge exchange and capacity building in biological and ecological methods are frequently made to the UK and the EU. Bringing this information into a single book aims to help this exchange and development process.

The WFD is focussed on achieving ecological outcomes in river basins. The overall aim is to achieve and maintain 'good ecological status' in all EU waters. This is subject to sustainable water use and defined social and economic tests. In some cases, water objectives can be reduced where intervention would be disproportionately expensive, or, if waters are designated as heavily modified or are artificial, where the target is 'good ecological potential'.

Quantifying these terms and the degree to which waters comply with them is a key element of the biological monitoring and assessment process.

With the focus on outcomes, rather than process, the WFD assumes that the most cost-effective combination of actions will be put in place, which aim to achieve agreed status objectives for any given river basin. This allows a wide range of interventions most suited to local circumstance. Figure 2 is useful for considering the range of options available to achieve outcomes.

Physical habitat, flow, water quality, or a combination of all three, could be improved to allow good status to be achieved. Much of the historic clean-up of water bodies has focussed on water quality and pollution control. However, final outcomes may actually be due to improved physical habitat and minimum flow controls, linked to ongoing water quality protection. It is the understanding of how these factors interact that will optimise ecological and biological protection and improvement.

It is important to note that although the overall aim of the WFD is to achieve 'good ecological status' the actual objectives for each water body and river basin are set by the River Basin Planning process, and may be different to this.

The WFD stipulates that implementation progress should be assessed against delivery of the 'programmes of measures' and ultimately the actual objectives set in the plan.

Integrated options for ecological improvement

A wide range of people need to be engaged in influencing water management. Much of the focus needs to be on associated land management, both urban and rural.

Often, it is the impact of human activity on land that affects surface water and groundwater, and which must therefore be managed. No one individual or organisation can do this alone and all need an understanding of the ultimate outcome – a healthy and sustainable water environment.

Building teams across industry and society, with the capability to manage water in a complementary and optimal way, is the key to success. Educating and providing information to these partners is a key aim of this handbook.

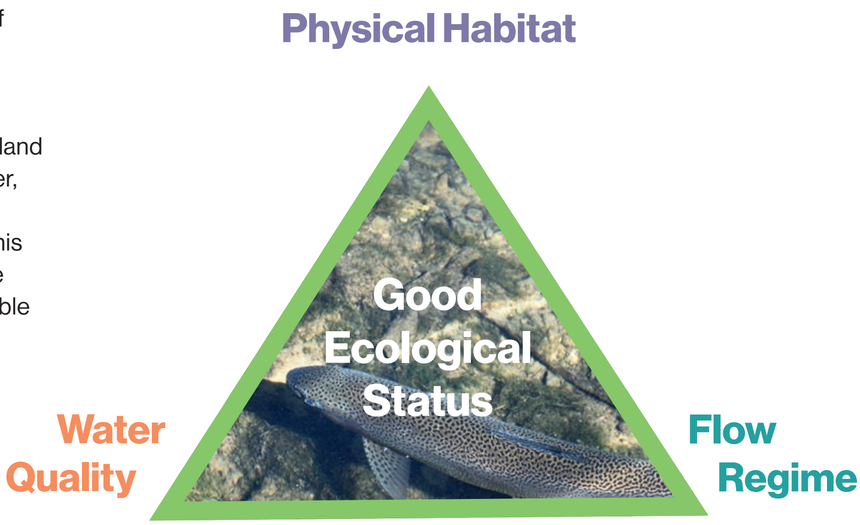


Figure 2
Consider the most effective option or combination of options to achieve good ecological status. *Paul Logan, Environment Agency*

Target Audience

The book is aimed at river management practitioners at all levels and is applicable to the UK, Europe and to river basins across the world.

It is valuable for senior and middle managers and scientists engaged in river basin planning and water resource protection. It can be read at high level by those influencing the broader policy and implementation.

River management officers, water company and environmental protection agency staff and consultant engineers will also find this invaluable.

It provides a depth of technical information and reference to help those directly engaged in developing, commissioning and implementing biological and ecological monitoring programmes and initiatives aimed at improving river and lake health.

Academic staff and researchers working on the natural environment need access to practical biological and ecological assessment methods and the management policies that they support. This information is also critical to the development of new methods of management and river health assessment.

The book can also be used by:

- Lecturers and students at all levels in universities, and is linked to training courses and teaching material.
- NGOs, Rivers Trusts, partner organisations and volunteers.
- Water recreational users, anglers and boat users.

The UK leaving the EU brings new challenges and uncertainty to river management both in the UK and across the EU.

Capturing current knowledge and best practice will help this transition and allow the next generation of biologists and ecologists to build and enhance the knowledge and skills needed to protect our water environment.

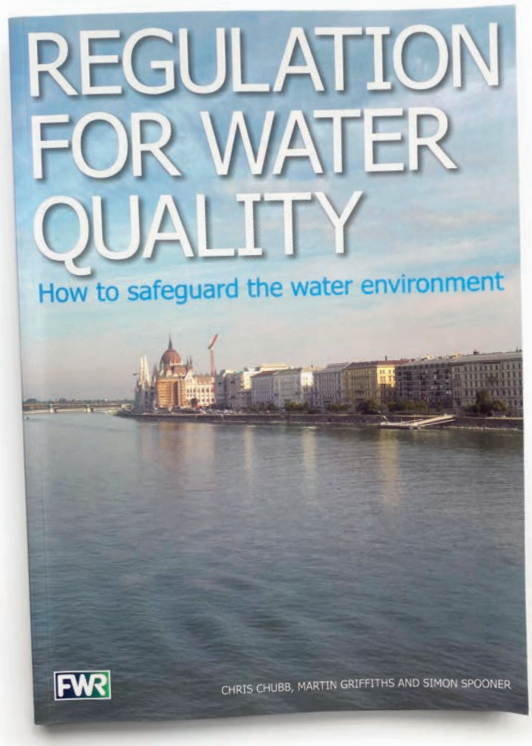
We hope that this handbook will assist in this process.

2
BACKGROUND TO THE HANDBOOK

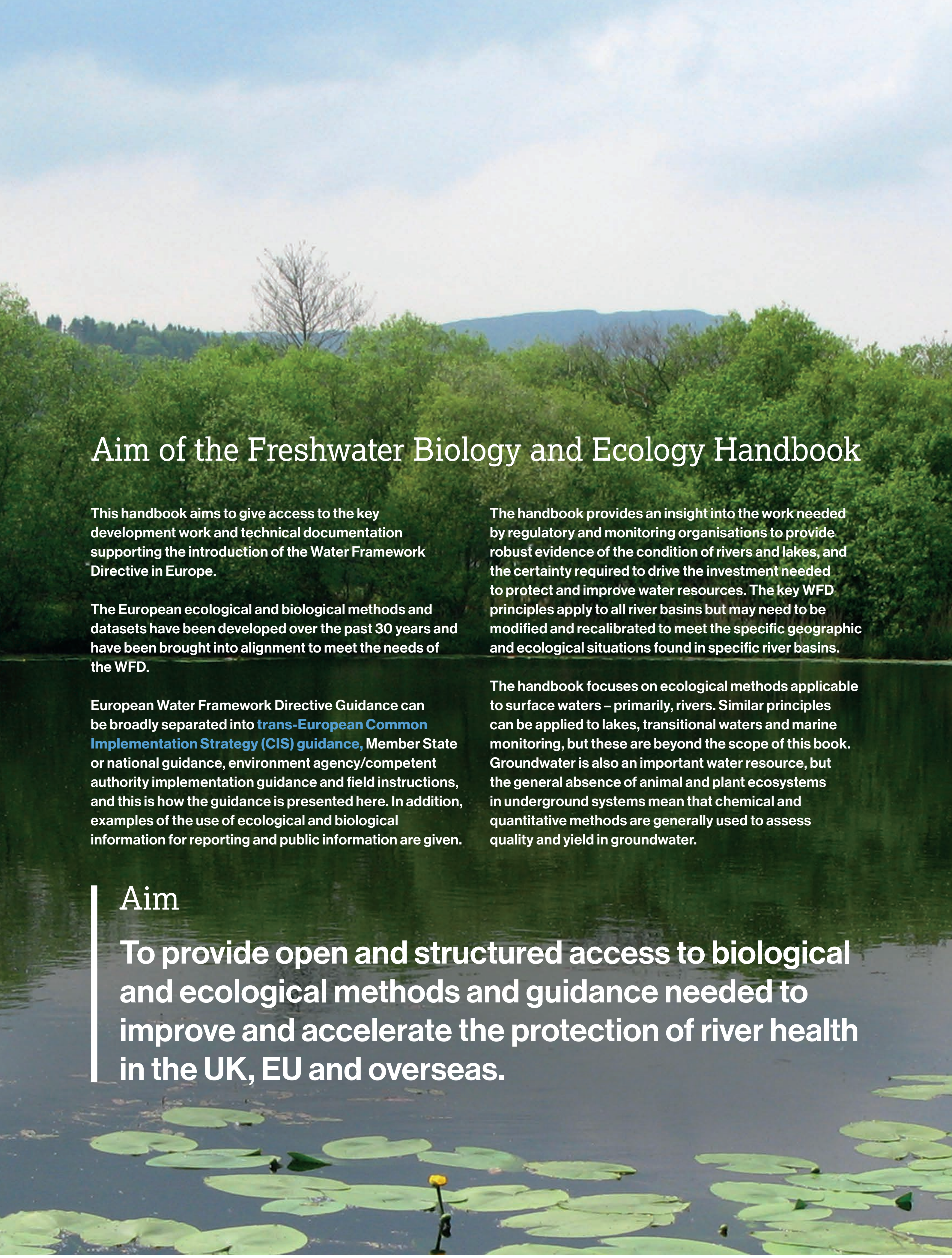
This book has been developed through a partnership led by the Foundation for Water Research (FWR). Other key partners are the Freshwater Biological Association (FBA) and the Environment Agency. Additional expertise, technical and editorial assistance has been made available from other organisations and individuals. These contributions are fully acknowledged at the end of the book.

This handbook is closely linked to a sister book *Regulation for Water Quality: how to Safeguard the Water Environment* by Chris Chubb, Martin Griffiths and Simon Spooner, published by FWR in July 2014.[3] This book is published free of charge and for public good in PDF format.

An initial handbook was written in 2011 for the EU/China River Basin Management Programme as part of the ongoing dialogue and transfer of knowledge between Europe and China.
The original handbook *Ecological and Biological Monitoring, European Water Framework Directive Guidance and Methods* by Martin Griffiths, Reinder Torenbeek and Simon Spooner was published in Chinese in June 2012. PDF copies in Chinese are available via the **China EU Water Platform** website.
That book was never published in English. It was the starting point for this handbook. This new book contains significantly more information, updated technical best practice and provides access to the key components for freshwater management using biological approaches. Our aim is to make this information available to a global audience and provide the tools necessary for practitioners to protect and improve freshwaters.



***Regulation for Water Quality* provided an overview of the essential regulatory framework necessary to maintain and improve the water environment.**



Aim of the Freshwater Biology and Ecology Handbook

This handbook aims to give access to the key development work and technical documentation supporting the introduction of the Water Framework Directive in Europe.

The European ecological and biological methods and datasets have been developed over the past 30 years and have been brought into alignment to meet the needs of the WFD.

European Water Framework Directive Guidance can be broadly separated into trans-European Common Implementation Strategy (CIS) guidance, Member State or national guidance, environment agency/competent authority implementation guidance and field instructions, and this is how the guidance is presented here. In addition, examples of the use of ecological and biological information for reporting and public information are given.

The handbook provides an insight into the work needed by regulatory and monitoring organisations to provide robust evidence of the condition of rivers and lakes, and the certainty required to drive the investment needed to protect and improve water resources. The key WFD principles apply to all river basins but may need to be modified and recalibrated to meet the specific geographic and ecological situations found in specific river basins.

The handbook focuses on ecological methods applicable to surface waters – primarily, rivers. Similar principles can be applied to lakes, transitional waters and marine monitoring, but these are beyond the scope of this book. Groundwater is also an important water resource, but the general absence of animal and plant ecosystems in underground systems mean that chemical and quantitative methods are generally used to assess quality and yield in groundwater.

Aim

To provide open and structured access to biological and ecological methods and guidance needed to improve and accelerate the protection of river health in the UK, EU and overseas.

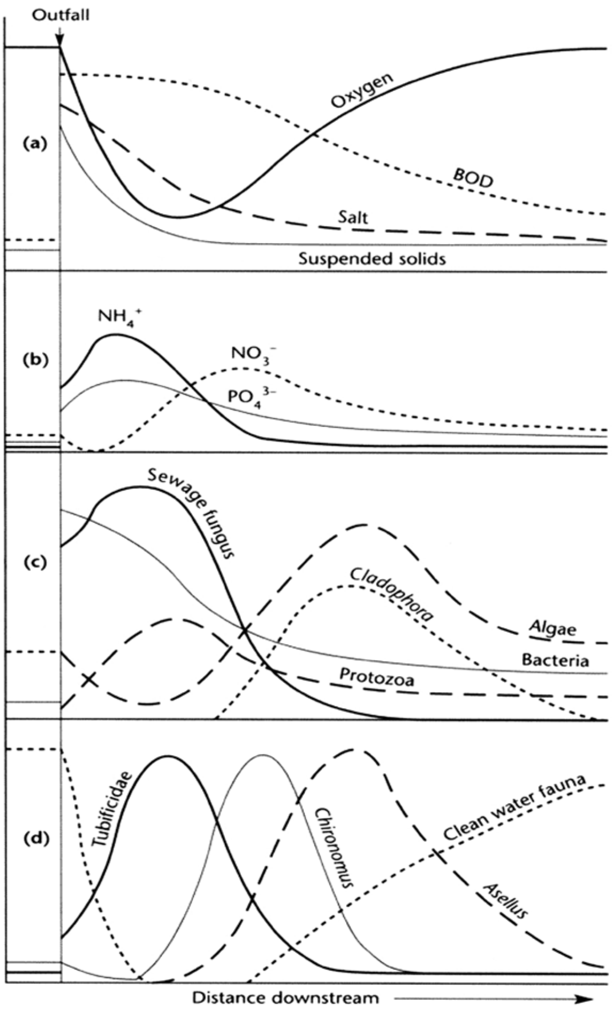
3
DEVELOPMENT OF BIOLOGICAL MONITORING AND ASSESSMENT METHODS

In the past we have managed freshwater using only chemical assessments linked to pollution from industrial and municipal discharges to the water environment.

Being able to assess water ecosystem health has been the result of significant research and development over the past 100 years or so.

This was connected to progressive water environmental legislation, regulation and enforcement.

The fundamental principles and links between sewage pollution, water chemistry and biological impact are shown in Figure 3, from Hynes' book *The Biology of Polluted Waters* (1960).[4] The understanding of these interactions underlies all our current methodologies, and this figure remains highly relevant today.



(a) Represents physical changes
(b) Represents chemical changes
(c) Represents changes in micro-organisms
(d) Represents changes in macro-invertebrates

Figure 3

Showing the effects of an organic effluent on a river and the downstream changes to the chemistry, micro-organisms and the invertebrates as the biota metabolises the waste and restores river quality. *Source: Hynes, Biology of Polluted Waters (1960)* [4]

Increasingly, we realise that chemical indicators alone are insufficient to understand ecosystem health. The damage that chemicals cause depends on the nature of the ecosystem and its sensitivity. This varies according to the nature of the water body and its geography. Untreated waste discharges may be a thing of the past but biological systems do not fully recover. This is because all water bodies are subject to combinations of other pressures that were masked by the impacts of gross pollution. To achieve true assessments of environmental health, biological and ecological methods and targets are needed, complementing chemical and hydrological assessments and targets. These are used to set permit conditions and to help optimise the management of discharges, abstractions and other activities, to meet the ecosystem targets.

River flow and water level has significant impacts on biological communities and is a key ecological component of water management. Developing and enforcing minimum flow conditions to our river systems is one of the most crucial issues. Minimum flows are often neglected in developing countries, and rivers that should naturally have permanent flows may dry up for prolonged periods. Historically, minimum flows have been set to optimise and maintain potable, industrial and agricultural water supply, but increasingly they are linked to biological needs, expressed through ecological quality objectives.

The most straightforward examples are where flow regimes are managed to allow migratory fish to move and ensure connection to upriver spawning grounds. In some cases, increased flow is maintained for a short time to allow a 'freshet' to stimulate the migration of salmon. This 'spate sparing' is common in operating rules for reservoirs. Increasingly, research into the needs of invertebrates for amount and speed of water over the riverbed and seasonal and other natural patterns of flow are enabling improved outcomes to be achieved.

River habitat improvement is a fundamental component of river restoration. Often, the chemical pollution has been improved, but habitat remains unfavourable. Significant progress is being made into options for river restoration and habitat improvement. Many rivers were widened and deepened (resectioned), straightened and river banks stabilised with concrete or other material, to promote drainage. In the UK about 10% of the Environment Agency's engineering budget is now spent on restoring river channels by removing these structures and recreating natural meanders to restore ecological quality. Weirs and other impoundments to fish migration are being removed to not only restore fish populations, but also to meet specific legislation, such as the **Eel Regulation 2007** [(5)]

The **River Restoration Centre** based in the UK at Cranfield University, holds significant resources and expertise on this.

Regulators have been slow to adopt biological assessment methods, possibly because of the cultural links to engineering and chemical disciplines of developing industrial and regulatory organisations. In addition, the complexity of biological systems is difficult to communicate to wider audiences and to link directly to cause and effect. Translating complex indicators of water ecosystem health into simple indexes and regulatory tools has been problematic.

One of the earliest significant attempts to systematically assess river health was the Saprobic Index developed by Kolkwitz and Marsson in the early 1900s.[(6)] This was a system of categorising water quality through levels of organic waste (pollution) in rivers and streams. It was based upon the abundance and distribution of biological species in four saprobic zones. This methodology has been extensively developed in continental Europe and is still used for Water Framework Directive assessment in Austria and Germany.

One of the more important early attempts to enumerate biological quality in the UK was the Trent Biotic Index, developed by Woodiwiss in 1964, to assess the quality of the River Trent and other rivers in the British Midlands.[(7)] This assessed the presence and absence of sensitive invertebrates to polluted environments and allowed the environment to be described as an index ranging from one to ten, one being the most polluted and ten the least polluted. This and similar biotic indices were utilised extensively by biologists and river managers for many years. However, the Trent Biotic Index was insensitive to a number of situations and pollution types and more sensitive systems have since been developed.

In 1970 Chandler took these concepts and added a semi-quantitative component or weighting which transferred presence and absence of indicator organisms into a scoring system – The Chandler Score. [(8)] This provided a numeric basis for the classification of polluted waters and is the precursor of the current biological monitoring and evaluation scoring systems. The Chandler score was sensitive to both organic pollution from sewage, and toxic pollution from industrial discharges and acidic mine drainage.

One of the most established systems of biological assessment is the Biological Monitoring Working Party Score (BMWP), which underlies river invertebrate assessment methods in the UK and many European countries today (BMWP, 1978 [(9)]; Hawkes, 1997 [(10)]). This index was designed as a national system suitable for the biotas of all types of rivers in the UK, being devised for the National River Quality Surveys. Like the Chandler Score on which it was based, it was sensitive to a wide range of pressures.

The Freshwater Biological Association's River Communities Project (1977) developed the underlying BMWP scoring system into a predictive tool known as the **River Invertebrate Prediction and Classification System (RIVPACS)** This is based on a statistical model that estimates the ecological health of river sites. It utilises reference datasets of macroinvertebrates in 'unpolluted' conditions and can predict which macroinvertebrates should be expected in similar sites. **The difference between the expected fauna and the observed fauna indicates the ecological status of the water.** This is an invaluable tool for water managers to target protection and improvement activity. This, and similar approaches, allowed the development and implementation of the WFD and is used for determining the status of all types of surface waters across Europe based on their biota. RIVPACS is still regarded as an example of good practice internationally and more information is given in **Chapter 3**.

Bringing biological and ecological assessment methods such as RIVPACS into mainstream water management across Europe has been a key breakthrough. The importance of dialogue and the international political processes, facilitated by the EU, should not be underestimated, and biologists and research communities neglect this at their peril!

Alastair Ferguson reviewed the evolution of the WFD from the perspective of a biologist and environmental policymaker engaged in the development of the WFD. In a personal communication he noted that:

The early development of the WFD can be traced back to the conclusions of the EU Community Water Policy Ministerial Seminar in Frankfurt in 1988, which highlighted the need for Community legislation covering ecological quality of the aquatic environment. The Council in its resolution of 28 June 1988 asked the European Commission to submit proposals to improve ecological quality in Community surface waters. As a result, Alessandre Barisich of DGXI and Jean J Fried of the Institute de L'eau France organised a European "Study and Reflection Seminar" to discuss "The Ecological Quality of Surface Water" in preparation of a Community Directive, held in May 1989 and published later that year by Barisich and Fried (1989). The Seminar was held at the Villa Olmo in Como hosted by the Centro di Cultura Scientifica Alessandro Volta. Centro Volta is a non-profit organization, created since 1983 and aims to provide scientists all over the world with a distinctive environment for fostering scientific communication, interaction and debate.

*The overall conclusion was that **'a Directive governing the ecological quality of waters should aim to fix high standards for Community waters'.***

The recommended structure of the Directive resulting from the Seminar was quite remarkable in that it provided a framework for assessing ecological quality, setting targets, presenting the results, planning for future improvements and reporting on progress. The way this was set out in the conclusions and recommendations of the Seminar formed the basis for the WFD and the overall structure changed very little during its evolution from an ecologically-based Directive to the far more holistic approach in the WFD.

In this way, the bringing together of technical expertise, within a political and regulatory framework, is essential. It needs to be done at the correct time, in the correct way and when there is an established social and environmental need to make progress in water protection and improvement. It took significant effort over many years to ensure this alignment. These forces came together in 2000 when the WFD was adopted by the European Parliament and the Council of Ministers.

European Commission

4

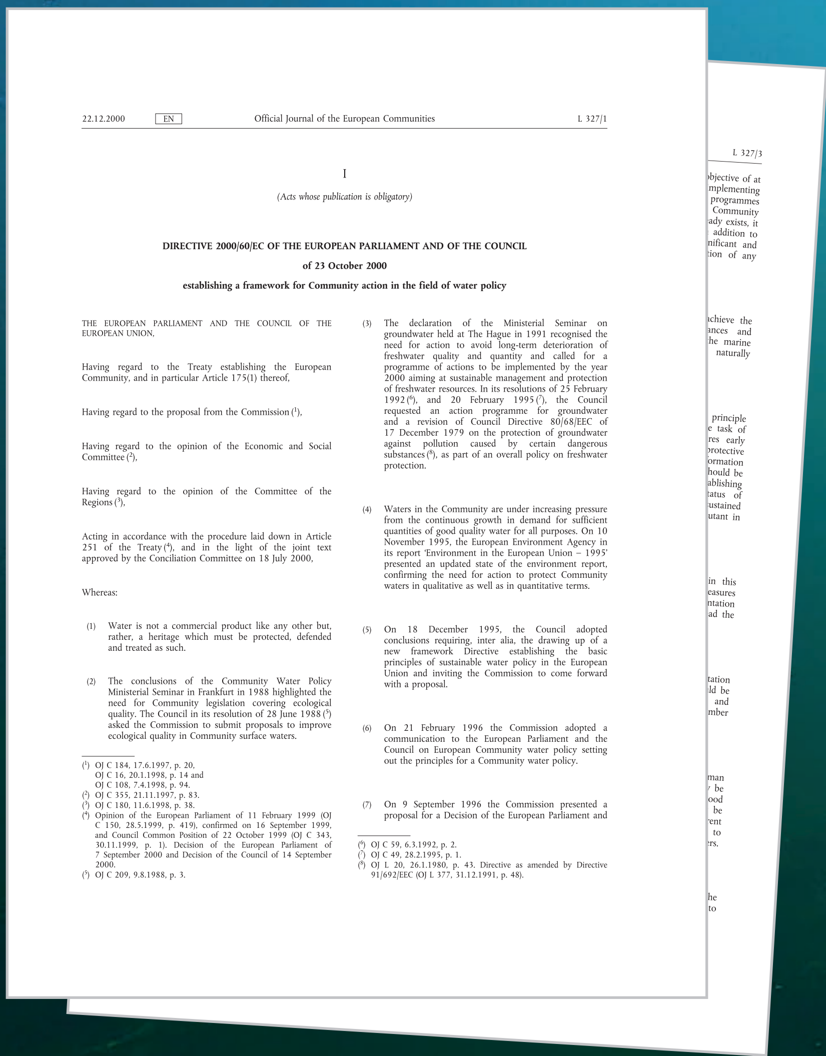
THE WATER FRAMEWORK DIRECTIVE AND ITS DAUGHTER DIRECTIVES

The **Water Framework Directive**[8] was adopted by the EU in 2000

Some elements of the WFD were not ready for incorporation, or were seen as too contentious in 2000 to be included in the full WFD.

These were set aside for further development and negotiation.

They were designated as WFD Daughter Directives. The Groundwater Directive (2006/118/EC) and Priority Substances Directive (2008/105/EC) were the two key elements that were reserved for later agreement.


22.12.2000 EN Official Journal of the European Communities L 327/1

I

(Acts whose publication is obligatory)

DIRECTIVE 2000/60/EC OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL

of 23 October 2000

establishing a framework for Community action in the field of water policy

THE EUROPEAN PARLIAMENT AND THE COUNCIL OF THE EUROPEAN UNION,

Having regard to the Treaty establishing the European Community, and in particular Article 175(1) thereof,

Having regard to the proposal from the Commission (1),

Having regard to the opinion of the Economic and Social Committee (2),

Having regard to the opinion of the Committee of the Regions (3),

Acting in accordance with the procedure laid down in Article 251 of the Treaty (4), and in the light of the joint text approved by the Conciliation Committee on 18 July 2000,

Whereas:

(1) Water is not a commercial product like any other but, rather, a heritage which must be protected, defended and treated as such.

(2) The conclusions of the Community Water Policy Ministerial Seminar in Frankfurt in 1988 highlighted the need for Community legislation covering ecological quality. The Council in its resolution of 28 June 1988 (5) asked the Commission to submit proposals to improve ecological quality in Community surface waters.

(3) The declaration of the Ministerial Seminar on groundwater held at The Hague in 1991 recognised the need for action to avoid long-term deterioration of freshwater quality and quantity and called for a programme of actions to be implemented by the year 2000 aiming at sustainable management and protection of freshwater resources. In its resolutions of 25 February 1992 (6), and 20 February 1995 (7), the Council requested an action programme for groundwater and a revision of Council Directive 80/68/EEC of 17 December 1979 on the protection of groundwater against pollution caused by certain dangerous substances (8), as part of an overall policy on freshwater protection.

(4) Waters in the Community are under increasing pressure from the continuous growth in demand for sufficient quantities of good quality water for all purposes. On 10 November 1995, the European Environment Agency in its report 'Environment in the European Union – 1995' presented an updated state of the environment report, confirming the need for action to protect Community waters in qualitative as well as in quantitative terms.

(5) On 18 December 1995, the Council adopted conclusions requiring, inter alia, the drawing up of a new framework Directive establishing the basic principles of sustainable water policy in the European Union and inviting the Commission to come forward with a proposal.

(6) On 21 February 1996 the Commission adopted a communication to the European Parliament and the Council on European Community water policy setting out the principles for a Community water policy.

(7) On 9 September 1996 the Commission presented a proposal for a Decision of the European Parliament and

(1) OJ C 184, 17.6.1997, p. 20, OJ C 16, 20.1.1998, p. 14 and OJ C 108, 7.4.1998, p. 94.
(2) OJ C 355, 21.11.1997, p. 83.
(3) OJ C 180, 11.6.1998, p. 38.
(4) Opinion of the European Parliament of 11 February 1999 (OJ C 150, 28.5.1999, p. 419), confirmed on 16 September 1999, and Council Common Position of 22 October 1999 (OJ C 343, 30.11.1999, p. 1). Decision of the European Parliament of 7 September 2000 and Decision of the Council of 14 September 2000.
(5) OJ C 209, 9.8.1988, p. 3.
(6) OJ C 59, 6.3.1992, p. 2.
(7) OJ C 49, 28.2.1995, p. 1.
(8) OJ L 20, 26.1.1980, p. 43. Directive as amended by Directive 91/692/EEC (OJ L 377, 31.12.1991, p. 48).

Figure 2
The Water Framework Directive

Table 1
Water Framework Directive and the main daughter directives in force (June 2018)

Directive 2000/60/EC of 23 October 2000 establishing a framework for Community action in the field of water policy
Groundwater Directive 2006/118/EC on the protection of groundwater against pollution and deterioration
Directive 2008/32/EC of 11 March 2008 introducing technical and procedural amendments
Directive 2009/90/EC of 31 July 2009 on technical specifications for chemical analysis and monitoring of water status for WFD
Directive 2013/39/EU of 12 August 2013 amending Directives 2000/60/EC and 2008/105/EC regarding priority substances in the field of water policy
Directive 2014/101/EU of 30 October 2014 in relation to European standards applicable to WFD

Helpfully, the European Commission published a consolidated version of the Water Framework Directive that incorporates the daughter directives and other official decisions https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:02000L0060-20141120

Table 2
Other relevant directives

Water management
Directive 2007/60/EC on the assessment and management of flood risks
Directive 91/271/EEC concerning urban wastewater treatment
Council Directive 98/83/EC of 3 November 1998 on the quality of water intended for human consumption
Directive 2006/7/EC of 15 February 2006 concerning the management of bathing water quality and repealing Directive 76/160/EEC
Directive 91/271/EEC concerning urban waste water treatment
Directive 98/15/EC amending Directive 91/271/EEC to clarify the requirements of the directive in relation to discharges from urban waste water treatment plants to sensitive areas which are subject to eutrophication
Directive 2010/75/EU of 24 November 2010 on industrial emissions (integrated pollution prevention and control)
Directive 91/676/EEC of 12 December 1991 concerning the protection of waters against pollution caused by nitrates from agricultural sources
Nature conservation
Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora
Directive 2009/147/EC of the European Parliament and of the Council of 30 November 2009 on the conservation of wild birds

Chapter 1

WATER FRAMEWORK DIRECTIVE APPROACH

Brexit and the Water Framework Directive

The UK left the EU through the Brexit process on 31st January 2020.

However, the WFD remains in force within the UK and the River Basin Planning and implementation process will continue for the foreseeable future.

Elements of this will be taken forward as an integral part of the **UK 25 Year Environment Plan**

In spite of potential future changes we expect that the core principles of the WFD approach will continue within current and future UK River Basin Management approaches.

More information on the current UK river management approach is given in **Chapter 6**.

1
WATER FRAMEWORK DIRECTIVE CONTEXT

1.1 Background

The major advances in water management and regulation in recent years have centred on the concepts of integrated river basin management and long-term planning of water resources. The EU agreed to adopt the European Water Framework Directive (WFD) in 2000. This provides a long-term planning framework for all the river basins across the EU, based on achieving biological and ecological outcomes. The WFD has been accepted as a model that can be adapted to other river basins across the world. It has stimulated discussion and knowledge exchange programmes in many countries, sponsored by the EU and Member States. Technical advances have taken place with international partners and some elements of the WFD have been adopted, or modified, to suit specific river basin situations.

The WFD introduces new ways of protecting and improving rivers, lakes, groundwater, estuaries and coastal waters. It provides a structure to enable us to develop a sustainable future for our natural waters. The emphasis is on promoting aquatic environments that support balanced plant and animal communities. Healthy ecosystems indicate that water quality is sufficiently high to be available for a variety of human uses.

River Basin Planning through the WFD should consider strategic aims and at the same time accommodate local needs. In this way, river basin planning is seen as a 'top down and bottom up' process. Getting this balance correct is essential.

The adoption of integrated river basin planning provides an opportunity to address strategic challenges such as climate change, sustainable development and other water-based activities. For example, climate change scenarios, including water flows and temperatures, can be modelled and can inform future water planning decisions.

Well planned use of key natural resources makes certain that a balance is struck between socio-economic requirements and environmental needs, and ensures that high quality water is made available for drinking water, industry and agricultural and recreational use. In addition, good planning will allow local water and environmental needs to be met.

The primary aims of the WDF are presented in Box 1.1 and are directly quoted from the Directive.

Box 1.1

Primary aim of the EU WFD is to:
Establish a framework for the protection of inland surface waters, transitional waters, coastal waters and groundwater which:

(a)	prevents further deterioration and protects and enhances the status of aquatic ecosystems and, with regard to their water needs, terrestrial ecosystems and wetlands directly depending on the aquatic ecosystems
(b)	promotes sustainable water use based on a long-term protection of available water resources
(c)	aims at enhanced protection and improvement of the aquatic environment, inter alia, through specific measures for the progressive reduction of discharges, emissions and losses of priority substances and the cessation or phasing-out of discharges, emissions and losses of the priority hazardous substances
(d)	ensures the progressive reduction of pollution of groundwater and prevents its further pollution
(e)	contributes to mitigating the effects of floods and droughts

The WFD introduces ecological objectives to the heart of water environmental protection. Biological indicators are a major subset of those indicating ecological status. These are designed to protect, and where necessary, restore the structure and function of aquatic ecosystems, and thereby safeguard the sustainable use of water resources. The effectiveness of our water management strategies will be judged on ecological outcomes, based on these objectives.

To achieve this, a clear view of the current status of aquatic ecosystems is required, including a view on the pressures and risks impacting each catchment. This requires comprehensive and risk-based ecological monitoring programmes: should, for example, the land use in the catchment comprise largely of arable farming, the monitoring programmes would reflect this; they would be targeted towards nitrates, phosphates and possibly agricultural pesticides. In a catchment containing an industrial process, manufacturing or storing pesticides or specific chemicals, those chemicals would be identified as a risk and monitoring would be put in place to ensure that there was minimum detrimental impact on the water bodies.

Information to determine these pressures and risks is known as **Characterisation** in the WFD. The links between Characterisation and Monitoring are shown in Figure 2.1 below.

Classification schemes are fundamental to the assessment of compliance against objectives and are the primary driver for water management and improvement. Accurate and reproducible assessment against objectives becomes a critical issue as this drives investment and management actions to protect and improve water resources. Importantly, the WFD demands that status classifications are accompanied by measures of 'confidence of class'. In the UK this is commonly defined as 95% confidence that a designated water body is within its class. This requires information about the accuracy and the precision of the methods used for monitoring.

The EU WFD introduces a formal river basin management planning system which is the key mechanism for ensuring the integrated management of water resources. It is also the mechanism for achieving good ecological outcomes and for driving improvements in the most effective way.

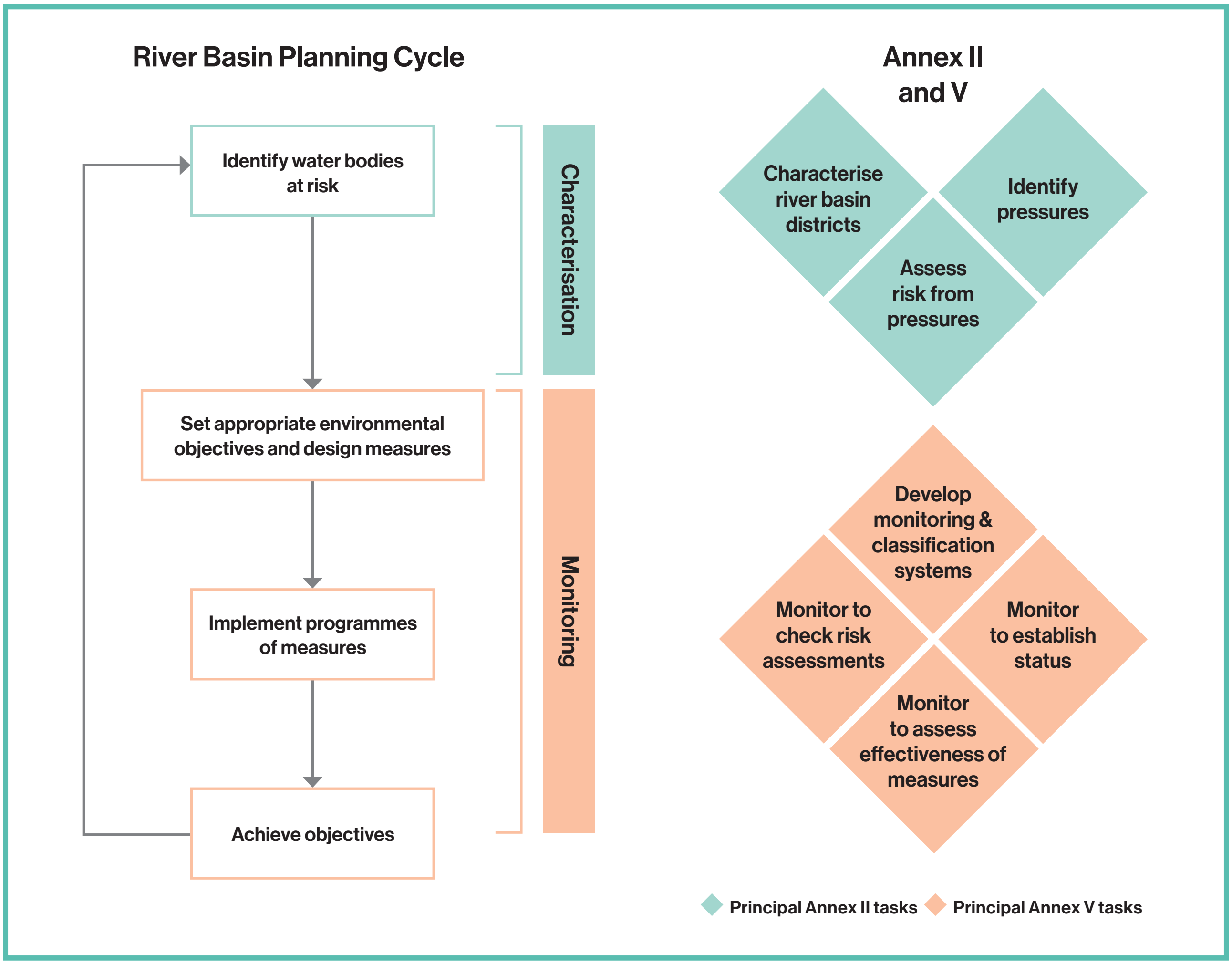
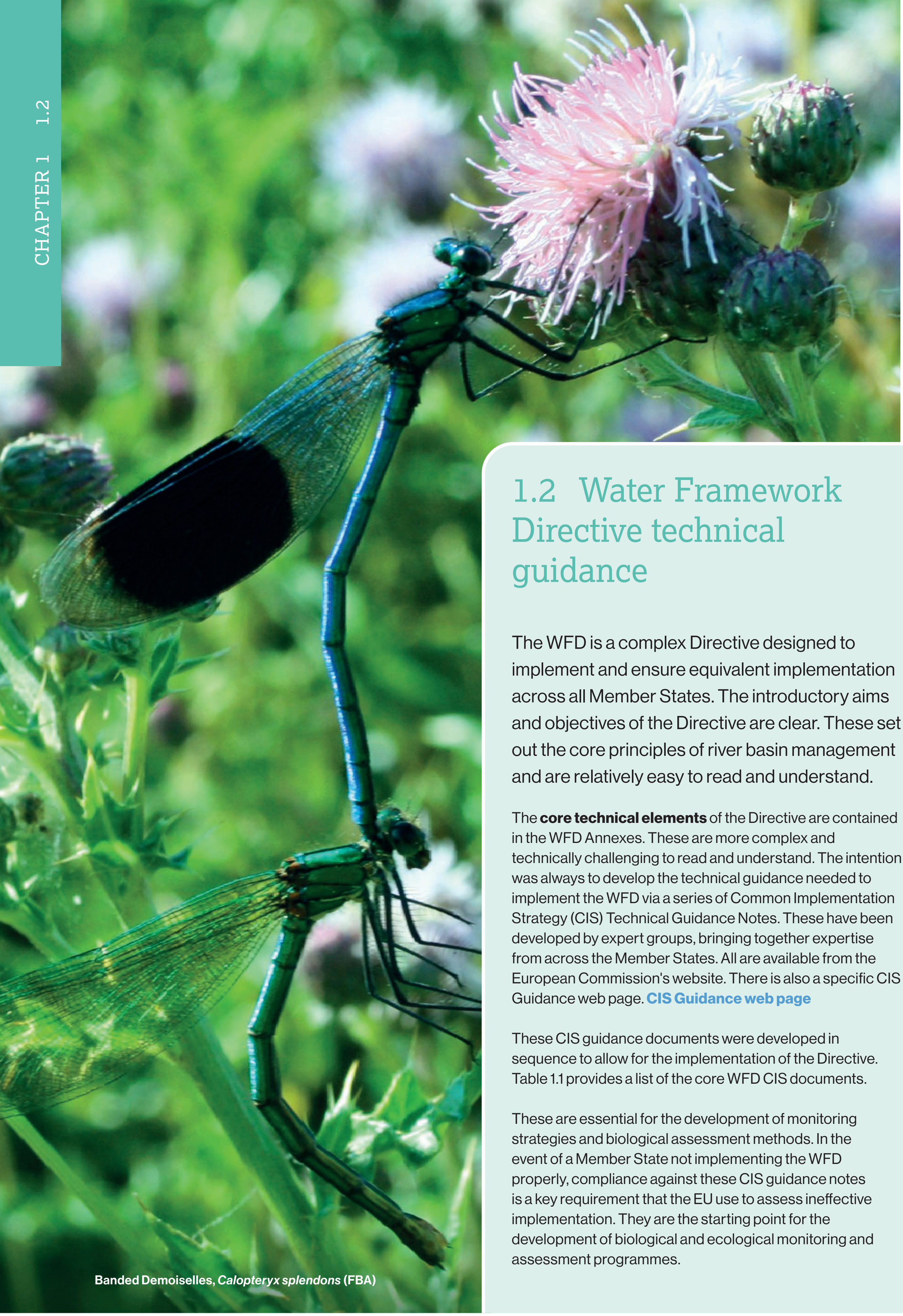


Figure 2.1

Relationship between river basin planning, characterisation and monitoring

Source - Scottish Environmental Protection Agency (SEPA), 2002, Future for Scotland's Waters. [12]



Banded Demoiselles, *Calopteryx splendons* (FBA)

1.2 Water Framework Directive technical guidance

The WFD is a complex Directive designed to implement and ensure equivalent implementation across all Member States. The introductory aims and objectives of the Directive are clear. These set out the core principles of river basin management and are relatively easy to read and understand.

The **core technical elements** of the Directive are contained in the WFD Annexes. These are more complex and technically challenging to read and understand. The intention was always to develop the technical guidance needed to implement the WFD via a series of Common Implementation Strategy (CIS) Technical Guidance Notes. These have been developed by expert groups, bringing together expertise from across the Member States. All are available from the European Commission's website. There is also a specific CIS Guidance web page. **CIS Guidance web page**

These CIS guidance documents were developed in sequence to allow for the implementation of the Directive. Table 1.1 provides a list of the core WFD CIS documents.

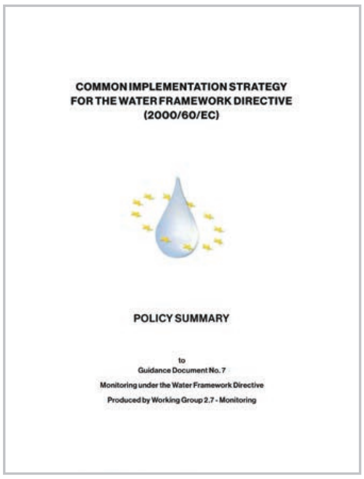
These are essential for the development of monitoring strategies and biological assessment methods. In the event of a Member State not implementing the WFD properly, compliance against these CIS guidance notes is a key requirement that the EU use to assess ineffective implementation. They are the starting point for the development of biological and ecological monitoring and assessment programmes.

Table 1.1

Common Implementation Strategy Guidance Documents

N° 1	Economics and the Environment – The Implementation Challenge of the Water Framework Directive
N° 2	Identification of Water Bodies
N° 3	Analysis of Pressures and Impacts
N° 4	Identification and Designation of Heavily Modified and Artificial Water Bodies
N° 5	Transitional and Coastal Waters – Typology, Reference Conditions and Classification Systems
N° 6	Towards a Guidance on Establishment of the Intercalibration Network and the Process on the Intercalibration Exercise
N° 7	Monitoring under the Water Framework Directive
N° 8	Public Participation in Relation to the Water Framework Directive
N° 9	Implementing the Geographical Information System Elements (GIS) of the Water Framework Directive *[Now replaced by N° 22]*
N° 10	Rivers and Lakes – Typology, Reference Conditions and Classification Systems
N° 11	Planning Processes
N° 12	The Role of Wetlands in the Water Framework Directive
N° 13	Overall Approach to the Classification of Ecological Status and Ecological Potential
N° 14	Guidance on the Intercalibration Process (2008 – 2011)
N° 15	Groundwater Monitoring (WG C)
N° 16	Groundwater in Drinking Water Protected Areas
N° 17	Direct and Indirect Inputs in the light of the 2006/118/EC Directive
N° 18	Groundwater Status and Trend Assessment
N° 19	Surface Water Chemical Monitoring
N° 20	Exemptions to the Environmental Objectives
N° 21	Guidance for Reporting under the WFD
N° 22	Updated WISE GIS guidance (Nov 2008)
	[N° 21 and 22 are replaced by N° 35]
N° 23	Eutrophication Assessment in the Context of European Water Policies
N° 24	River Basin Management in a Changing Climate
N° 25	Chemical Monitoring of Sediment and Biota
N° 26	Risk Assessment and the Use of Conceptual Models for Groundwater
N° 27	Deriving Environmental Quality Standards Version 2018
N° 28	Preparation of Priority Substances Emissions Inventory
N° 29	Reporting under the Floods Directive
N° 30	Procedure to fit new or updated classification methods to the results of a completed Intercalibration Exercise
N° 31	Ecological Flows (final version)
N° 31	Ecological Flows Policy summary (Original English version)
N° 31	Ecological Flows (French version)
N° 32	Biota Monitoring
N° 33	Analytical Methods for Biota Monitoring
N° 34	Water Balances Guidance (final version)
N° 35	WFD Reporting Guidance
N° 35	WFD Reporting Guidance Annex 5
N° 35	WFD Reporting Guidance Annex 6
N° 36	Article 4(7) Exemptions to the Environmental Objectives
N° 37	Steps for defining and assessing ecological potential for improving comparability of Heavily Modified Water Bodies
N° 37	Mitigation Measures Library

List of other CIS thematic documents available on CIRCABC interface **https://circabc.europa.eu/ui**


COMMON IMPLEMENTATION STRATEGY
FOR THE WATER FRAMEWORK DIRECTIVE
(2000/60/EC)

POLICY SUMMARY

to
Guidance Document No. 7
Monitoring under the Water Framework Directive
Produced by Working Group 2.7 - Monitoring

Common Implementation Strategy (CIS) Technical Guidance Notes

1.3 WFD timetable

The EU WFD set a clear timetable for action across Europe (see Table 1.2). Key elements relating to characterisation and ecological and biological monitoring are highlighted in blue and were undertaken between 2004 and 2007, as scheduled by the timetable.

However, it should be noted that this built on a significant body of ecological and biological data and previous development of biological scores and indexes in different Member States. Characterisation was not undertaken from a zero information base, although the amount of data, and its relevance, varied considerably in each river basin and Member State.

It is this element of the WFD that forms the focus of this handbook.

Table 1.2

Water Framework Directive timetable (adapted from Foundation for Water Research). Colour coding indicates the time allowed for each activity.

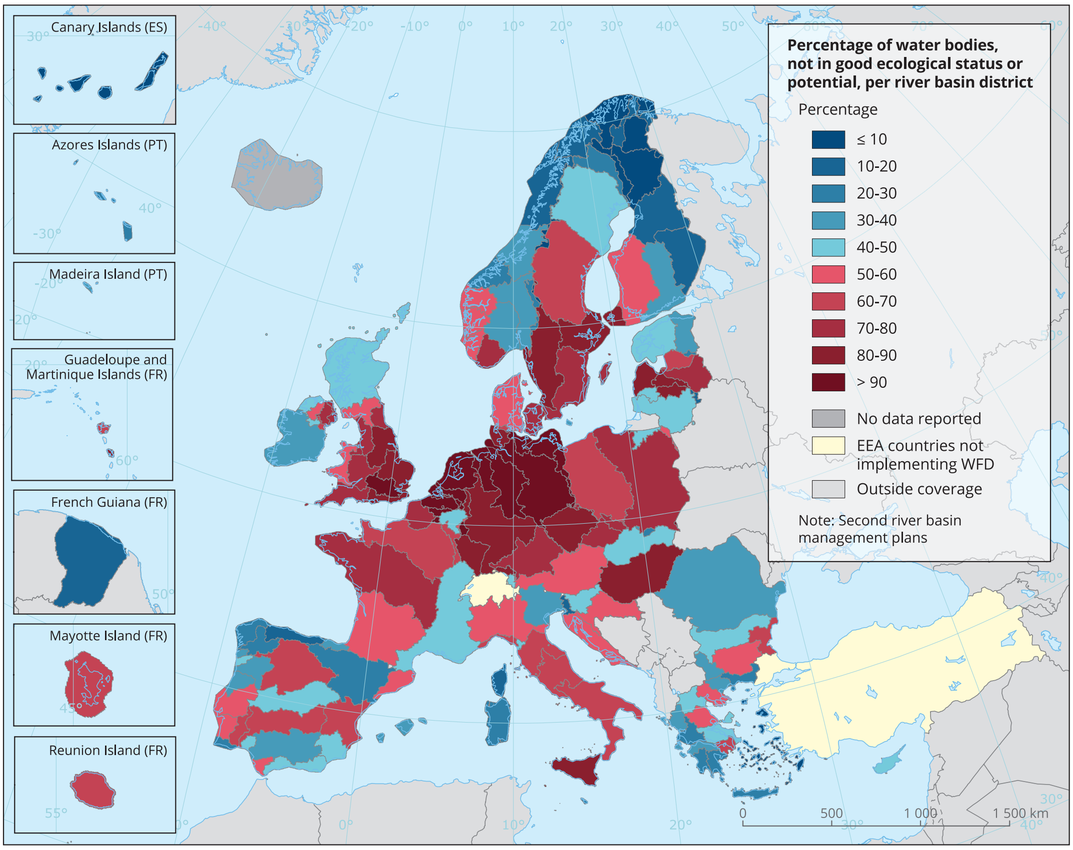
Complete action by year end	Action Required	EU Directive Articles	Overview
2000	Water Framework Directive entered into force	Article 22 Article 25	3 years for Member States to prepare
2003	• Transpose requirements to national legislation	Article 23	
	• **Define River Basin Districts and Authorities**	**Article 3**	
2004	**Characterise river basins: pressures, impact and economic analysis**	**Article 5**	6 years to analyse issues and prepare the River Basin Management Plans
2005	• Identify significant trends in groundwater pollution	Article 17	
2006	• Establish environmental monitoring programmes	Article 8	
	• Publish and consult on a work programme for the production of the first River Basin Management Plans **(RBMPs)**	Article 14	
	• Establish environmental quality standards **(EQSs) for surface waters**	Article 16	
2007	• **Report monitoring programmes to the EC**	**Article 14**	
	• Publish and consult on summary of Significant Water Management Issues **(SWMI) for each River Basin District**		
2008	**Publish and consult on drafts of the RBMPs**	**Article 14**	
2009	• **Publish the first RBMP for each River Basin District**	**Article 13**	
	• Establish programmes of measures **(PoMs), in each River Basin District in order to deliver environmental objectives**	Article 11	
2010	• **Report RBMPs, including PoMs, to the EC**	**Article 9**	3 years to put Programmes of Measures in place
	• Introduce water pricing policies		
2012	• Ensure all POMs are fully operational	Article 11	
	• **Report progress in implementing the first RBMPs**	**Article 15**	
2013	• Review progress of the first RBMP cycle		3 years to achieve specified objectives
2015	• First management cycle ends		
	• Main environmental objectives specified in the first RBMPs met?	**Article 4**	
2015	**Second River Basin Management Plans** (review and update first RBMPs) **and first Flood Risk Management Plans**	Articles 13, 14 and 15	Further 6 years planning, consultation and implementation cycles
2021	**Second management cycle ends**		Further 6 years planning, consultation and implementation cycles
	• **Main environmental objectives specified in the second RBMPs met?**	**Article 4**	
	• Third management plans (review and update second RBMPs)	Articles 13, 14 and 15	
2027	**Third management cycle ends**		Further 6 years planning, consultation and implementation cycles
	• **Final deadline for environmental objectives**	**Article 4**	
	• **Main environmental objectives specified in the third RBMPs met?**	Article 13, 14 and 15	
	• Review and update third RBMPs		

1.4 WFD progress in Europe

Progress in implementing EU Directives, specifically the WFD, can be monitored in two ways.

Firstly, by examining the state of the water environment. The outcomes of regulatory activities are judged by Member State monitoring programmes, and reported to the European Environment Agency (EEA), based in Copenhagen. Periodic *State of Environment* reports are published, the current being *The European environment – state and outlook 2020: knowledge for transition to a sustainable Europe.*[12] **https://www.eea.europa.eu/soer/2020**

Figure 1.3 is an example of outcomes and shows the percentage of water bodies not meeting good ecological status from all river basins across the EU, updated 8 November 2021.



Reference data: ©ESRI | ©EuroGeographics

Figure 1.3

Percentage of classified water bodies in less than good ecological status or potential in rivers and lakes in Water Framework Directive river basin districts

The EEA makes the following observation on this position:

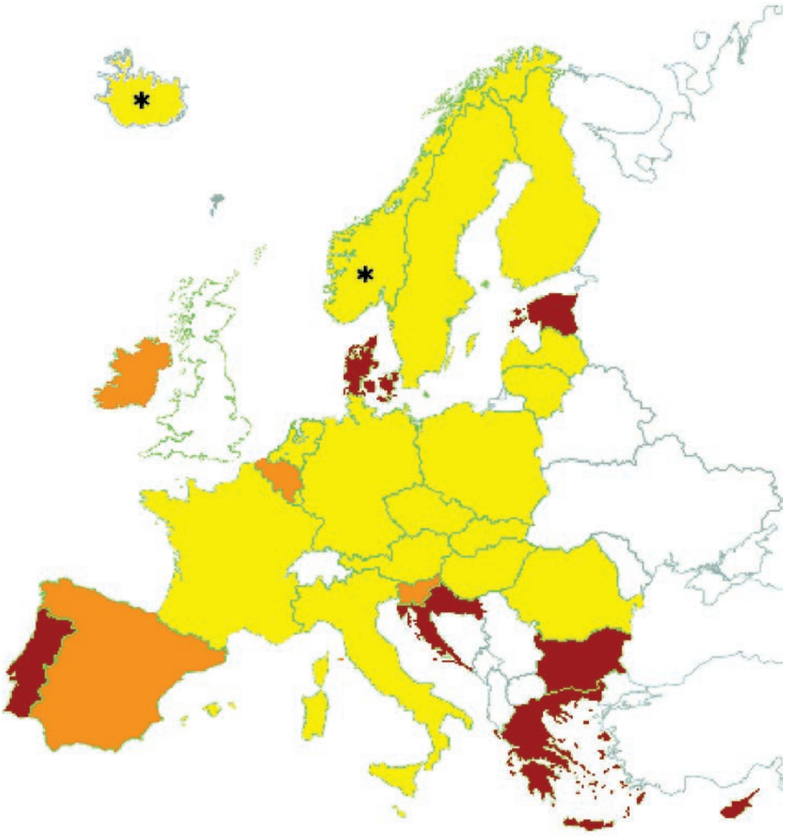
Achieving healthy aquatic ecosystems requires taking a systemic view, as the state of aquatic ecosystems is closely connected to how we manage land and water resources, and to pressures from sectors such as agriculture, energy and transport. There are ample opportunities to improve water management to achieve policy objectives. These include stringent implementation of existing water policy, and integration of water policy objectives into other areas such as the Common Agricultural Policy, EU Cohesion and Structural Funds, and sectoral policies.

This information will inform future water protection strategy and shape future rounds of WFD River Basin Plans.

The second method is to monitor WFD implementation progress and the publishing of WFD River Basin Plans for the second planning round, 2015 to 2021. The EU published Figure 1.4 showing the status of adoption of the second WFD River Basin Plans. (Note, the map reflects that the UK has now left the EU.)

This can be seen on the EU's web pages relating to water at: **http://ec.europa.eu/environment/water/participation/map_mc/map.htm** (see below).

The two approaches, outcomes and implementation process provide an indication of current status and where regulatory and monitoring effort needs to be placed to optimise progress against EU strategic aims for the water environment.



Status of implementation of WFD, January 2022.

GREEN – all third River Basin Management Plans adopted.

YELLOW – public consultation concluded but third River Basin Management Plans not adopted yet.

ORANGE – public consultation ongoing.

RED – public consultation not yet started.

**Norway and Iceland are both implementing the Water Framework Directive under a specific timetable agreed pursuant to the Agreement on the European Economic Area (EEA). The plans for 2022-2027 represent the second cycle under formal WFD obligations for Norway and Iceland.*

Figure 1.4

2
MONITORING REQUIREMENTS OF THE EU WFD

2.1 Role of monitoring programmes

Environmental monitoring programmes are developed to provide targeted information to assess overall status or potential, to identify where environmental quality objectives are not being met, to help identify the causes and the measures needed to be taken to meet those objectives, and to measure progress in meeting the objectives.

The level of monitoring must reflect the pressures and risks in a river basin and be focussed on providing the level of certainty needed to make appropriate decisions.

If decisions are taken on the basis of unreliable monitoring information, then expensive mistakes may be made, and unnecessary regulatory pressure applied to the users and communities within the catchment.

To help with this, status classifications are accompanied by measures of confidence of class. This is combined with other information, including the accuracy of monitoring, to provide an estimate of the weight of evidence.

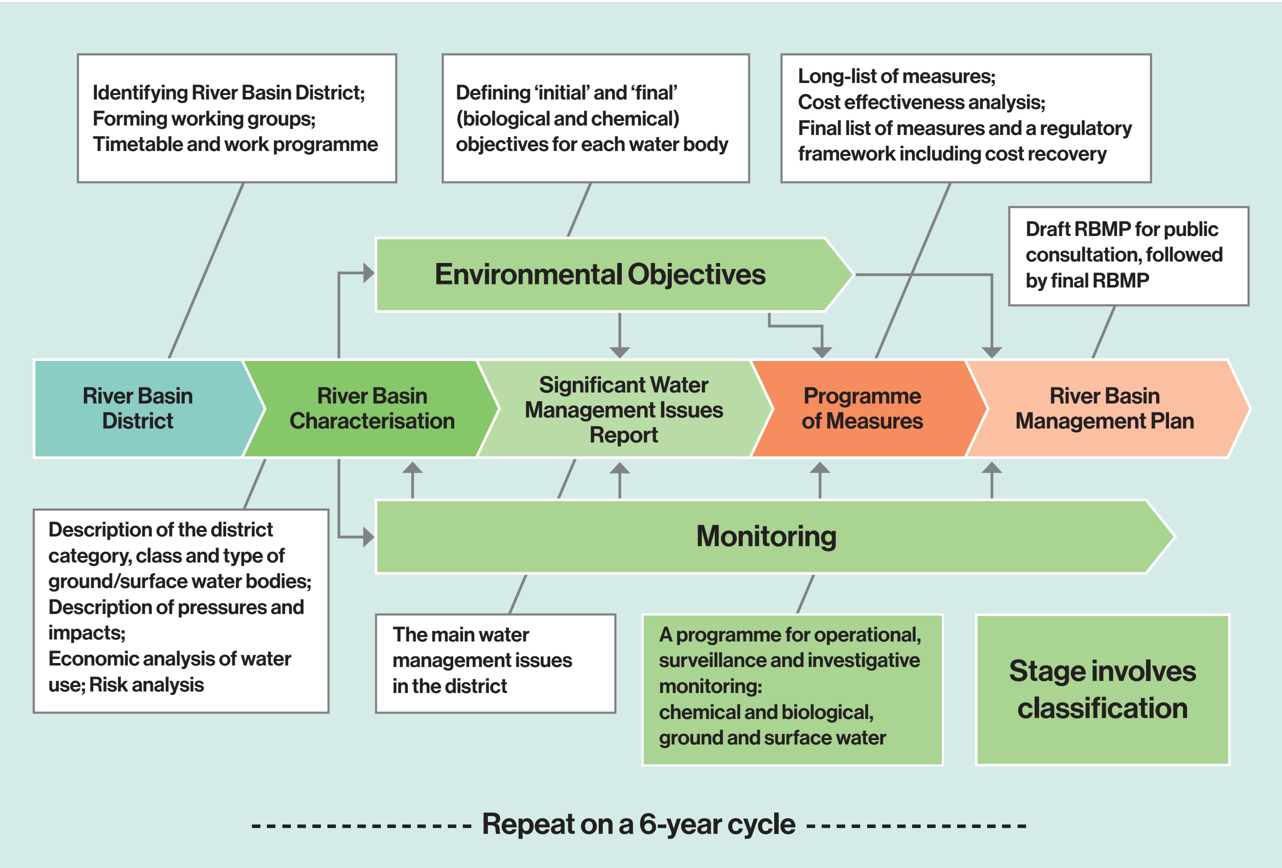
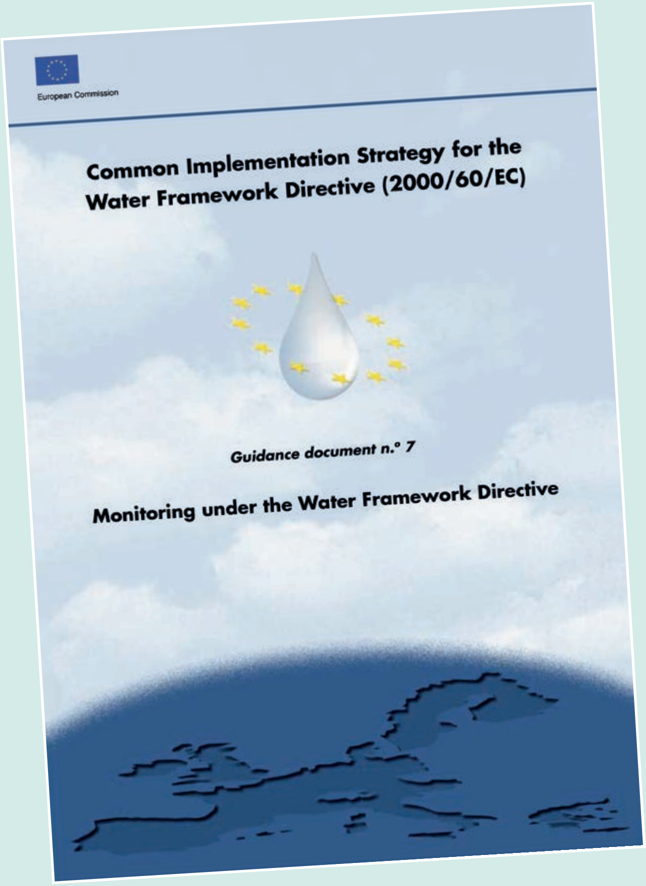


Figure 1.5

The influence of status classification on the stages of river basin management planning for WFD. *Based on a slide by Kees Blok*

The WFD timetable required the establishment of monitoring programmes for surface waters by the end of 2006. **Common Implementation Strategy (CIS) Guidance Document 7** [13] focuses on the monitoring requirements of the WFD and 'aims to guide experts and stakeholders in the design and implementation of the monitoring networks and programmes required to meet the requirements of the WFD for all categories of waters'.



CIS Note 7 guidance[13] sets out that surface water monitoring information is required for:

- classifying status
- supplementing and validating the risk assessment procedure
- designing efficiently and effectively future monitoring programmes
- assessing long-term changes in natural conditions.
- assessing long-term changes resulting from widespread anthropogenic activity
- estimating pollutant loads transferred across international boundaries or discharging into seas
- assessing changes in status of those bodies identified as being at risk in response to the application of measures for improvement or prevention of deterioration
- ascertaining causes of water bodies failing to achieve environmental objectives where the reason for failure has not been identified
- ascertaining the magnitude and impacts of accidental pollution
- intercalibrating biological classifications
- assessing compliance with the standards and objectives of Protected Areas
- quantifying reference conditions for surface water bodies.

2.2 Basic principles of monitoring

Translating the policy and theory into practice is crucial, and field methods and sample analysis provide the basic information on which to base river management decisions.

Collecting field information is expensive, but not as expensive as the infrastructure investment and the management and regulatory actions that may follow. Therefore, to ensure that cost-effective protection and improvement plans are developed, accurate and informative monitoring programmes and procedures must be put in place.

Field methods must be practicable, reproducible and be flexible enough to deal with real situations under a variety of circumstances. For this reason, monitoring organisations, environment agencies and commercial monitoring companies have invested in operational instructions, field guidance and training to ensure that this is done consistently, efficiently and effectively.

Field sampling methods, quality assurance and guidance are well established for chemical monitoring and flow monitoring. However, ecological and biological monitoring and field monitoring are relatively new. Methods have been developed to ensure additional quality and consistency to meet the needs of statutory requirements, such as the WFD.

This book describes the methods implemented in the UK. They are generally examples of good practice. Biological and ecological methods are being refined in every country to improve their robustness, utility and practicality. In the European Union, official methods for WFD status assessment are usually implemented at the beginning of the 6-yearly River Basin Management Plan cycles (see Table 1.2). The methods described in this book should therefore remain current until at least 2027.

2.3 Types of monitoring

The WFD describes the three main types of monitoring for surface waters: surveillance, operational and investigational.

Surveillance monitoring

Surveillance monitoring is undertaken to supplement and validate the initial pressure and impact assessments, to assess long-term changes in natural conditions, and to facilitate efficient and effective design of future monitoring and management activities.

Operational monitoring

Operational monitoring is undertaken to establish the status of those water bodies identified as being at risk of failing their environmental objectives, and to assess any changes in status resulting from the WFD improvement programme, known as the Programme of Measures. In this way, scarce monitoring resource is focused on waters at risk.

Investigative monitoring

Investigative monitoring is carried out when the reason for failure of environmental objectives is not known, or to ascertain the magnitude and impacts of accidental pollution.

Further types of monitoring

The WFD also requires monitoring in relation to Protected Areas, where existing monitoring requirements must be fed into the management, protection and improvement programmes.

Groundwater level monitoring is also required to assess quantitative status. In addition, groundwater qualitative status, especially the upward trend in contaminants, must be monitored.

Whilst these definitions of monitoring are useful, in practice information provided by a wide range of monitoring programmes and historical data sets is used to ensure effective decision making. Predictive modelling techniques are also an important component of decision making. These may be used to estimate current and future water quality, water quantity and ecological status. However, models are only as good as the information fed into them and monitoring programmes need to take this into account in their design.


trakker
trakker

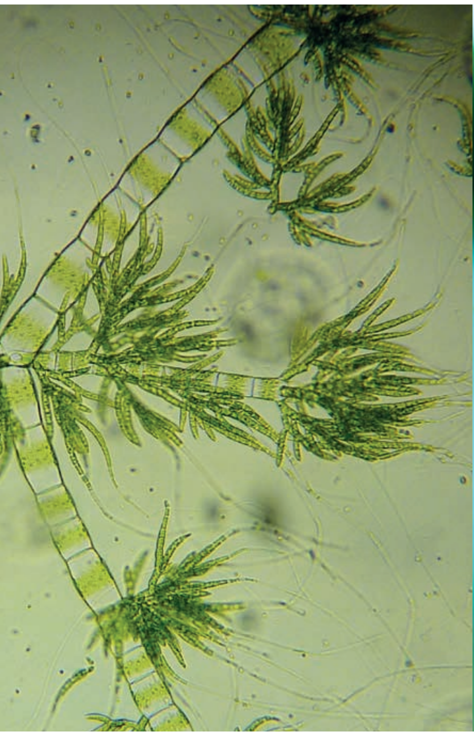
2.4 Monitoring frequency

Monitoring frequency is determined by a number of factors, including variability as a result of the natural variability of the quality element parameter, and natural and anthropogenic pressures.

These form the basis for the statutory requirements for surveillance monitoring specified in Section 1.3 of WFD's Annex V. The frequencies for **surveillance monitoring** are set out in Table 1.3. These intervals can be exceeded if longer intervals can be 'justified on the basis of technical knowledge or expert judgement', but monitoring for all biological and hydromorphological quality elements must be undertaken 'at least once during the surveillance monitoring period', ie in every 6-yearly River Basin Management Plan cycle.

Table 1.3
Surveillance monitoring frequencies, from WFD Annex V

Quality element	Rivers	Lakes	Transitional	Coastal
Biological				
Phytoplankton	6 months	6 months	6 months	6 months
Other aquatic flora	3 years	3 years	3 years	3 years
Macroinvertebrates	3 years	3 years	3 years	3 years
Fish	3 years	3 years	3 years	




The Directive also requires that the monitoring frequencies take account of the variability of parameters resulting from natural and anthropogenic conditions.

Ultimately, the frequencies should allow for an acceptable level of confidence and precision, which must be stated in the River Basin Management Plan.

Low confidence and precision can cause changes of class that are not easily explained, as well as making it more difficult to justify more expensive programmes of measures that may be necessary to restore quality to meet the environmental objectives.

The Directive also points out that the times at which monitoring is undertaken must minimise the impact of seasonal variation so that the results reflect changes only as a result of anthropogenic pressure. Additional monitoring during different seasons may be necessary to achieve this objective. However, samples for the same element must always be collected in the same seasons to avoid natural seasonal effects from influencing the monitoring results.

In the temperate northern hemisphere climates of Europe and the UK, the normal default frequencies are presented in Table 1.4 below.

Table 1.4

UK Normal Monitoring Frequencies for WFD

Parameter	Frequency
Macroinvertebrates	Twice in the appropriate year (Mar – May and Sept – Nov)
Macroinvertebrates (acidification Wales only)	Once in the appropriate year (Mar – May)
Macrophytes	Once in the appropriate year (June – Sept)
Diatoms	Twice in the appropriate year (Mar – May and Sept – Nov)
River Habitat Survey	Once in the appropriate year (April – Sept) for surveillance monitoring (should be performed once every 6 years)

Similar frequencies should be used for operational monitoring because of the similar requirements for status classifications based on this.

Unless it is to be used for status classification, investigative monitoring is not subject to the same restrictions. Most investigative surveys are not repeated. However, seasonal effects must be taken into consideration and may dictate when investigative monitoring is undertaken.

Sometimes, investigative surveys have to be undertaken at times of the year that are not ideal: for example, surveys to investigate the impacts of pollution incidents. Most models for predicting reference conditions assume that samples are collected in particular seasons, so it may not be possible to use samples collected outside these seasons to estimate status.

2.5 The use of ecological monitoring information – links to pressures

In the UK, the Department for Environment, Food and Rural Affairs (Defra) issued guidance on how biological failures identified by monitoring programmes can be attributed to environmental pressures (land use, pollution, hydrology, etc). See Defra (2011) How we determine which pressure is causing a biological failure in the context of the Water Framework Directive. [14]

As outlined in the WFD timetable, there is a 6-yearly cycle in WFD-based water management. **Water bodies are assessed every 6 years.** If a water body fails to meet the environmental goals, the cause must be determined, and measures must be taken to improve the environmental status. The reasons for failure of biological status are not always clear, so the question of how to determine which pressure is causing a biological failure needs to be addressed.

The text which follows is from the above-mentioned Defra document and is written with reference to Figure 1.6 which gives an overview of the current procedure used to determine which pressure is causing a biological failure.

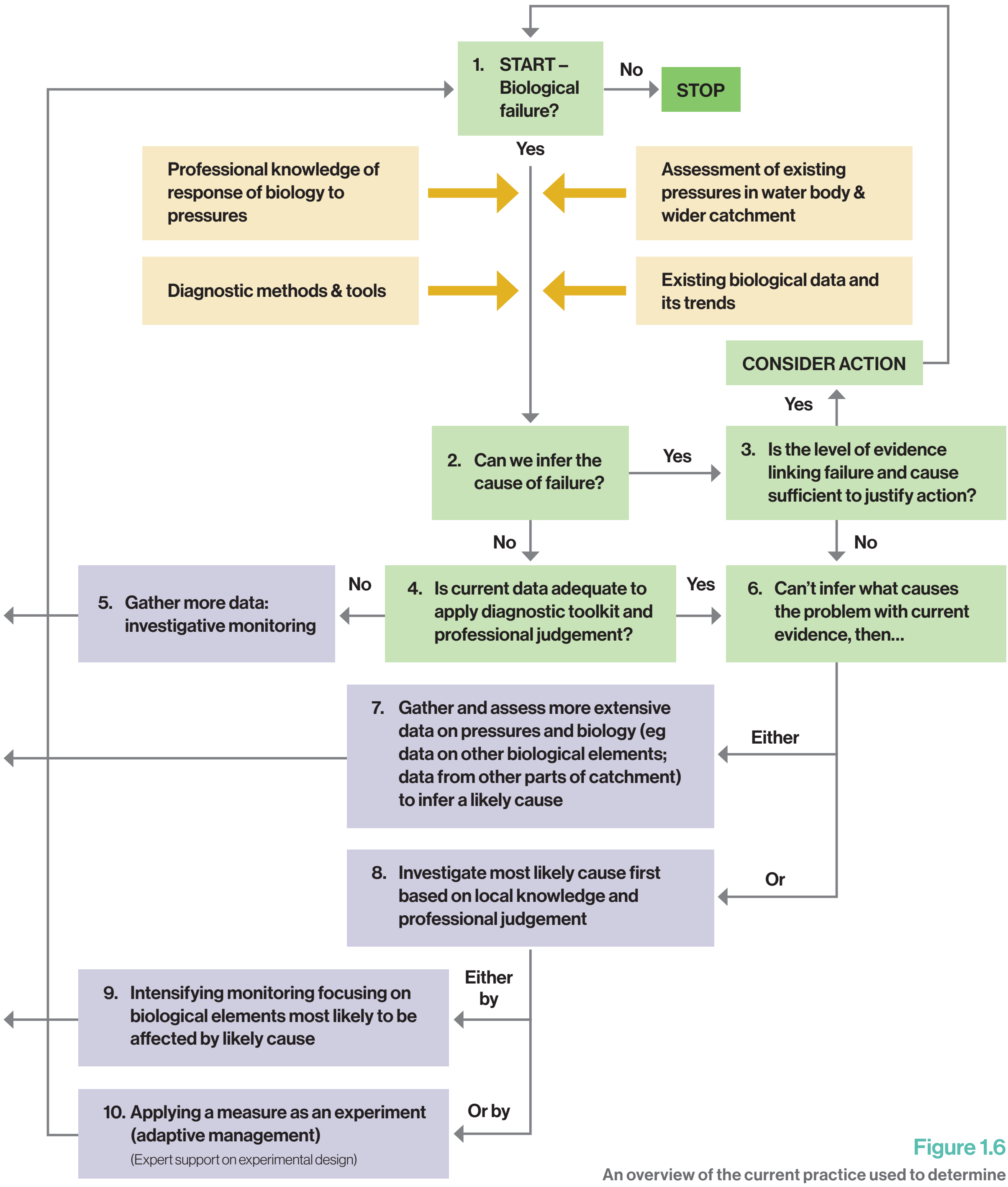


Figure 1.6

An overview of the current practice used to determine which pressure is causing a biological failure. *Defra 2011* [14]

When considering the causes of biological failure of a water body (Figure 1.6, Box 1), we generally take four things into account:

- **Our professional knowledge of the response of biology to pressures**
- **The existing pressures in the water body and wider catchment, taking local knowledge into account**
- **The tools and methods that we use to diagnose the causes of biological pressure**
- **Existing biological data (including external data), its trends and its statistical associations with pressures**

These enable us to identify failures to meet objectives and to infer the causes of failures (Fig 1.6, Box 2). When we do this, we also assess whether the level of evidence linking the pressure with the biological failure is sufficient to support action (Fig 1.6, Box 3) as set out in the guidance on 'Levels of evidence for completing investigations and selecting measures'. Where there is sufficient evidence, the next steps, eg an investigation to determine the source of the pressure and/or implementing measures, can proceed. The pressure might be high phosphate from agriculture and the proposed action may be revised guidance on fertiliser application. However, if the level of evidence does not support action, ie there is insufficient evidence linking the pressure to the biological impact to justify action, then we conclude that we can't infer the cause of the failure with sufficient confidence (Fig 1.6, Box 6).

If we can't infer the likely cause of failure based on the initial assessment (Fig 1.6, Box 2), then we need to judge if the current data is adequate for the application of the diagnostic tools or to apply professional judgement (Fig 1.6, Box 4). Where the data is inadequate, we then gather more or different data (Fig 1.6, Box 5). If the data is sufficient to apply the tools but we can't infer what causes the failure (Fig 1.6, Box 6) our next step depends on the level of certainty associated with the cause of failure.

Where we are uncertain about what causes the failure, we need to explore the situation by gathering and assessing more extensive data (Fig 1.6, Box 7). This might include increasing the number of biological elements sampled at the water body, and/or it may include collecting more data on pressures.

Where we have a good idea of what causes the failure (Fig 1.6, Box 8), we would normally intensify monitoring, focusing on the biological elements most likely to be affected by the pressure in question (Fig 1.6, Box 9). For example, where the suspected pressure is flow, invertebrate analysis might be taken to species (rather than family) level to improve the strength of evidence linking the pressure to the failure. Occasionally, we might undertake an experimental application of a measure to reduce the pressure to demonstrate if this improves the biology (ie 'Adaptive Management') (Fig 1.6, Box 10).

Our knowledge of biological responses to pressures and our diagnostic toolkit will improve further as we repeat this process over time and at multiple water bodies.

3

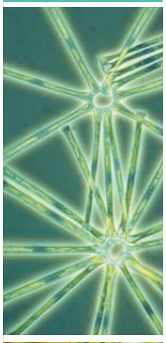
EUROPEAN AND INTERNATIONAL STANDARDS FOR BIOLOGICAL MONITORING

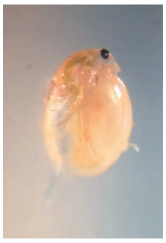
The importance of consistent monitoring methods and quality assurance was recognised by the EU in the implementation of the WFD. An amending **Directive, 2014/101/EU of 30 October 2014** [15] was introduced to consolidate and enforce monitoring standards. Annex 1.3.6 from the amending Directive states:

Annex 1.3.6. Standards for monitoring of quality elements
Methods used for the monitoring of type parameters shall conform to the international standards listed below in so far as they cover monitoring, or to such other national or international standards which will ensure the provision of data of an equivalent scientific quality and comparability.

In this amended Directive, many of the standards relevant to the WFD, particularly for biological sampling and analysis, are not mandatory but are for guidance. This is usually mentioned in its title but is not always clear in the text other than by reading the introductory sections of the standard very carefully. Table 1.5 provides an overview of current standards.

Table 1.5
List of standards relevant to biological and hydromorphological monitoring of surface freshwaters for WFD (correct to July 2018). *This list does not include methods for physico-chemical, hydrological, bacteriological, marine or groundwater assessments.*

	Quality element	Number	Title	Listed in Directive 2014/101/EU
	**Phytoplankton**	**EN 16698:2015**	Water quality – Guidance on quantitative and qualitative sampling of phytoplankton from inland waters	Yes
		EN 16695:2015	Water quality – Guidance on the estimation of phytoplankton biovolume	
		EN 15204:2006	Water quality – Guidance standard on the enumeration of phytoplankton using inverted microscopy (Utermöhl technique)	Yes
		ISO 10260:1992	Water quality – Measurement of biochemical parameters – Spectrometric determination of the chlorophyll-a concentration	Yes
	**Macrophytes and phytobenthos**	**EN 15460:2007**	Water quality – Guidance standard for the surveying of macrophytes in lakes	Yes
		EN 14184:2014	Water quality – Guidance for the surveying of aquatic macrophytes in running waters	Yes
		EN 15708:2009	Water quality – Guidance standard for the surveying, sampling, and laboratory analysis of phytobenthos in shallow running water	Yes
		EN 13946:2014	Water quality – Guidance for the routine sampling and preparation of benthic diatoms from rivers and lakes	Yes
		EN 14407:2014	Water quality – Guidance for the identification and enumeration of benthic diatom samples from rivers and lakes	Yes

Quality element	Number	Title	Listed in Directive 2014/101/EU	
Benthic invertebrates	**EN ISO 10870:2012**	Water quality – Guidelines for the selection of sampling methods and devices for benthic macroinvertebrates in fresh waters	Yes	
	EN 15196:2006	Water quality – Guidance on sampling and processing of the pupal exuviae of Chironomidae (order Diptera) for ecological assessment	Yes	
	EN 16150:2012	Water quality – Guidance on pro rata multi-habitat sampling of benthic macroinvertebrates from wadeable rivers	Yes	
	ISO 8689-1:2000	Water quality – Biological classification of rivers – Part 1: Guidance on the interpretation of biological quality data from surveys of benthic macroinvertebrates		
	ISO 8689-2:2000	Water quality – Biological classification of rivers – Part 2: Guidance on the presentation of biological quality data from surveys of benthic macroinvertebrates		
	EN 16772:2016	Water quality – Guidance on methods for sampling invertebrates in the hyporheic zone of rivers		
	EN 16859:2017	Water quality – Guidance standard on monitoring freshwater pearl mussel *(Margaritifera margaritifera)* populations and their environment		
Fish	**EN 14962:2006**	Water quality – Guidance on the scope and selection of fish sampling methods	Yes	
	EN 14011:2003	Water quality – Sampling of fish with electricity	Yes	
	EN 15910:2014	Water quality – Guidance on the estimation of fish abundance with mobile hydroacoustic methods	Yes	
	EN 14757:2015	Water quality – Sampling of fish with multi-mesh gill nets	Yes	
Hydromorphology	**EN 14614:2004**	Water quality – Guidance standard for assessing the hydromorphological features of rivers	Yes	
	EN 16039:2011	Water quality – Guidance standard on assessing the hydromorphological features of lakes	Yes	
	EN 16870:2017	Water quality – Guidance standard on determining the degree of modification of lake hydromorphology		
Other standards	**EN 16493:2014**	Water quality – Nomenclatural requirements for the recording of biodiversity data, taxonomic checklists, and keys		
	EN ISO 5667-3:2018	Water quality – Sampling – Part 3: Guidance on the preservation and handling of samples	Yes	
	EN 14996:2006	Water quality – Guidance on assuring the quality of biological and ecological assessments in the aquatic environment		
	EN 15110:2006	Water quality – Guidance standard for the sampling of zooplankton from standing waters		
New proposed standards	**prEN 14614 rev**	Water Quality – Guidance standard for assessing the hydromorphological features of rivers		
	prEN 17136	Water quality – Guidance on field and laboratory procedures for quantitative analysis and identification of macroinvertebrates from inland surface waters		
	prEN 17233	Water quality – Guidance for assessing the efficiency and related metrics of fish passage solutions using telemetry		

European standards are administered by the European Committee for Standardisation (CEN, Comité Européen de Normalisation). Each country is represented on CEN by a national standardisation body. Most national standardisation bodies charge for copies of national and European standards. Standards relevant to WFD are considered by CEN Technical Committee CEN/TC 230 Water Analysis

4

RIVER BASINS – WATER BODIES – CATCHMENTS – TYPOLOGY – SAMPLING LOCATIONS

These are the key hydrological and geographical elements of river basin management. Understanding of the issues within a river basin is essential to the integrated management of these complex systems.

4.1 River basins

River basins comprise rivers, lakes, groundwaters, transitional (ie estuarine) and coastal waters draining to a single river mouth. These are known in WFD as **water categories.** A stylised river basin is shown in Figure 1.7.

River basin districts are the main units for managing water resources. The WFD is based on the recognition that decisions taken in one part of a river cannot be taken in isolation, as actions taken at the top of a water catchment influence waters at the bottom. We have often seen over-abstraction or pollution in the headwaters of rivers impacting on downstream users.

The river basin planning and monitoring systems must be designed to ensure the heath of the river basin as a whole. In the case of transboundary rivers that cross national boundaries, two or more Member States have to share river basin management plans.

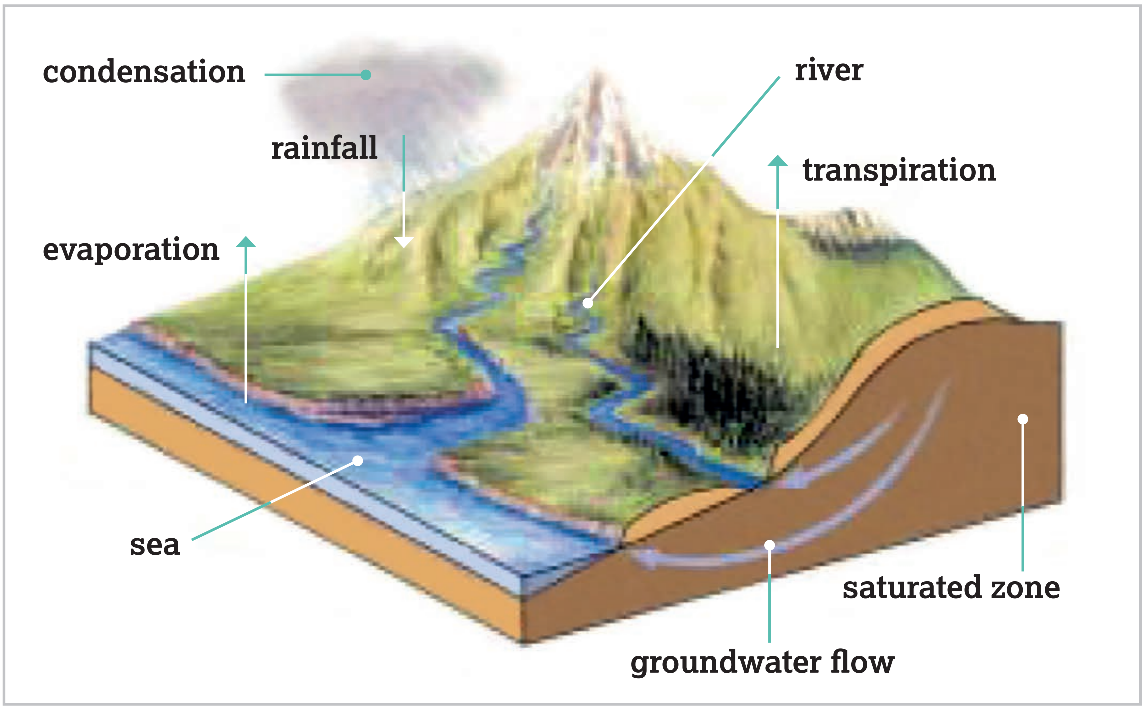


Figure 1.7

River Basin and the water cycle. *Source - Scottish Environmental Protection Agency (SEPA), 2002, Future for Scotland's Waters* [11]

4.2 Water bodies

Water bodies form the base units for monitoring programmes and river basin management.

The key purpose of identifying water bodies is to allow ecological objectives to be defined clearly in relation to the pressures acting upon river basins. Consequently, the water bodies must represent appropriate units for managing particular pressures or sets of pressures. One of the first tasks in developing water management programmes is to identify and define water bodies that are representative of that particular situation. They should be discrete and environmentally significant elements of a river basin. They can be parts or 'reaches' of a river system, a lake, estuary or coastal water, but not combinations of these water body categories. Individual water bodies must be subject to the same risks for which the same measures are needed. For example, a water body within an agricultural catchment would need to be separate from a water body downstream of a town where a sewage treatment works discharge enters the river. This would be a different risk and require different measures for improvement. Natural and heavily modified or artificial areas must be designated as separate water bodies.

Correctly identifying these major natural sub-divisions will make the future management and monitoring more effective and will ensure accurate reporting and assessment. Figure 1.8 shows the key elements of water body identification and introduces the concept of **typology.**

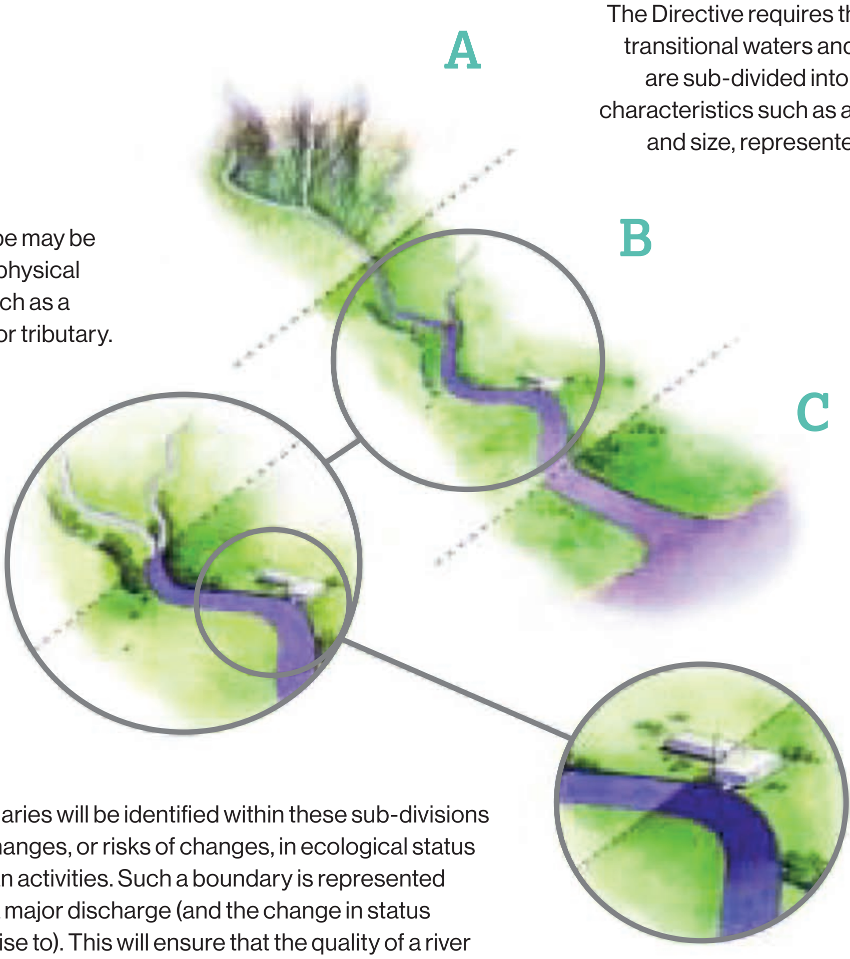


Figure 1.8

Identification of water bodies. *Source - Scottish Environmental Protection Agency (SEPA), 2002, Future for Scotland's Waters* [11]

Catchments are groups of water bodies at a geographical scale that is suitable for effective management.

4.3 Catchments

River basins are often too large to manage effectively, whereas water bodies are often too small. Catchments are groups of water bodies at a geographical scale that is suitable for effective management, in particular so that participation with local partners and stakeholders is possible.

To promote local management initiatives, the UK has introduced the **Catchment Based Approach initiative** Government funding is matched with local industry and voluntary groups to assist in the delivery of the WFD. The catchment based approach is not an official part of WFD, but it was introduced in England in 2013 to help produce river basin management plans with partners in 2015 and undertake the actions highlighted in the WFD River Basin Plans.

Figure 1.9 shows the catchments for the UK catchment based approach initiatives.

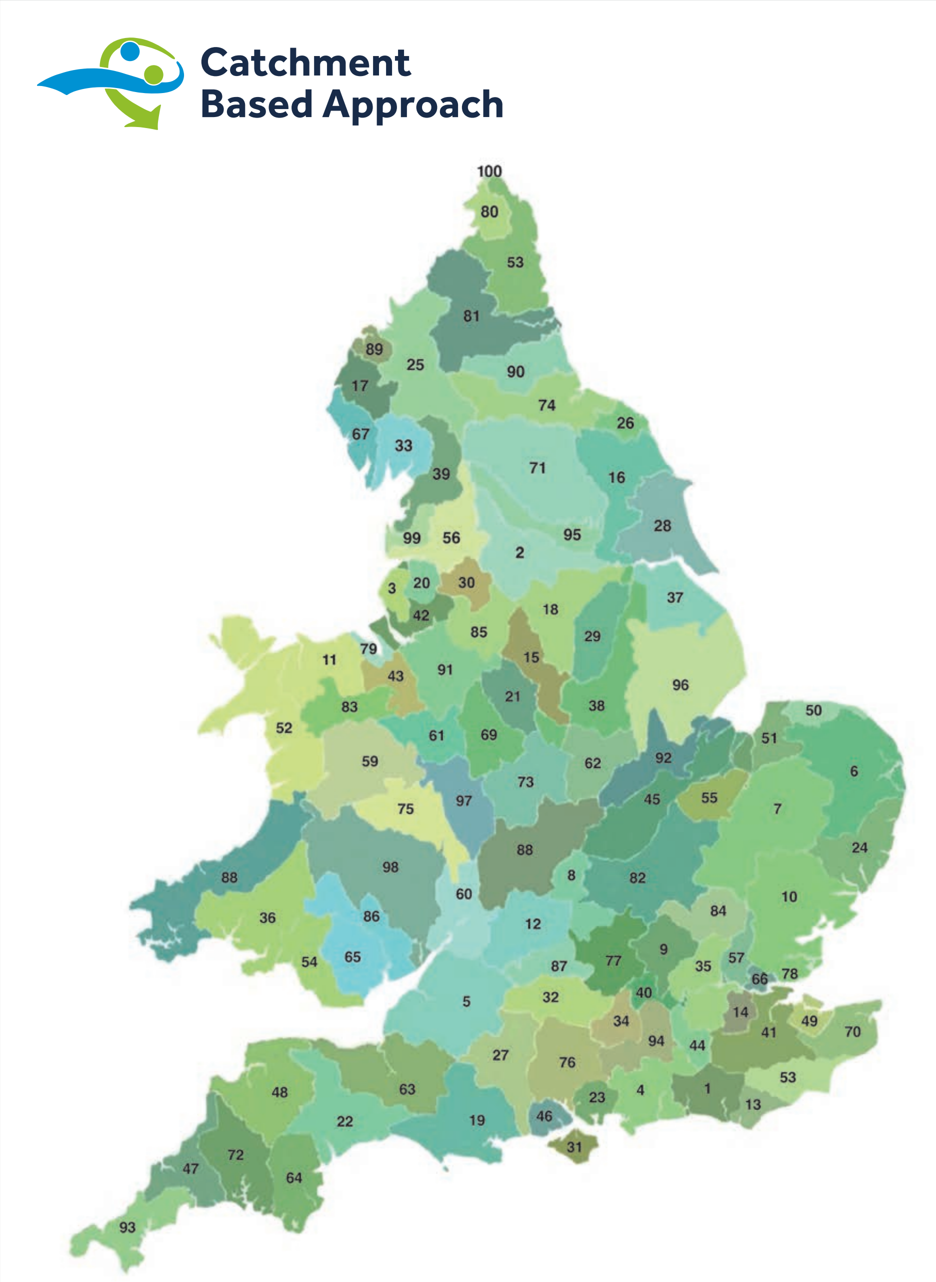


Figure 1.9

UK Catchment Based Approach – map of catchments *(from CaBA website)*

4.4 Typology

Typology, or surface water type, describes the physical and physico-chemical characteristics of a water body that determines the sorts of plants and animals that would be present under natural conditions. Water body typology therefore defines the biotas at reference conditions and therefore the reference values against which biological quality is measured.

Different types reflect the geology, morphology, climate and altitude of water bodies, all of which define the reference flora and fauna. In this way, a typology map of each river basin can be built up, using Geographical Information Systems, if possible, to inform the distribution of the monitoring network. Results from early monitoring can be used to build up an initial assessment. This typology can be refined as information increases.

The aim of river (or stream) typology is explained in the **AQEM project** (AQEM consortium, 2002). [16] River typology is defined as:

A river type is an artificially delineated but potentially ecologically meaningful entity with limited internal biotic and abiotic variation and a biotic and abiotic discontinuity toward other types.

River types might serve as 'units', for which an assessment system can be applied. A river type should always be defined on the basis of natural or near-natural reference sites, since the comparison with undisturbed sites of a certain river type allows defining and classifying different states of degradation. Biological assessment requires sufficiently stable, integrated river typologies, which consider both abiotic and biotic criteria. The most prominent abiotic factors are river morphology, geo-chemistry, altitude, river size and hydrology.

Typology is an essential component of surface water classification systems. For each surface water type an estimation can be made of what the biology would be if there were no (or very minor) alterations to the water body resulting from human interference. This allows the fundamental concept of **'Reference Conditions'** to be established around which the classification systems will be developed.

The WFD provides two methods for Member States to define their national typology:

System A
Eco-regions and some obligatory factors.

System B
No eco-regions, but obligatory factors are altitude, latitude, longitude, geology and size; and there are also optional factors. See Tables 1.6 and 1.7.

Table 1.6

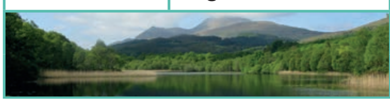
System A and B for typology of **rivers** (from WFD)

System A		System B	
Ecoregion	**Shown on map**	**Obligatory**	Altitude
Altitude	**>800 m**		Latitude
	200–800 m		Longitude
	<200 m		Geology
Size of catchment area	**10–100 km²**		Size
	100–1000 km²	**Optional**	Distance from river source
	1000–10,000 km²		Energy flow
	>10,000 km²		Mean water width
Geology	**Calcareous**		Mean water depth
	Siliceous		Mean water slope
	Organic		Form and shape of main river bed
			River discharge category
			Valley shape
			Transport of solids
			Acid neutralising capacity
			Mean substratum composition
			Chloride
			Air temperature range
			Mean air temperature
			Precipitation

Table 1.7

System A and B for typology of **lakes** (from WFD)

System A		System B	
Ecoregion	**Shown on map**	**Obligatory**	Altitude
Altitude	**>800 m**		Latitude
	200–800 m		Longitude
	<200 m		Depth
Mean depth	**<3 m**		Geology
	3–15 m		Size
	>15 m	**Optional**	Mean water depth
Surface area	**0.5–1 km²**		Lake shape
	1–10 km²		Residence time
	10–100 km²		Mean air temperature
	>100 km²		Air temperature range
Geology	**Calcareous**		Mixing characteristics
	Siliceous		Acid neutralising capacity
	Organic		Background nutrient status
			Mean substratum composition
			Water level fluctuation



In practice, these typologies were not widely used to define reference values. This is because different environmental parameters determine the distribution of different biological quality elements and therefore the reference values of metrics used to measure biological quality.

For example, the reference values for the UK's diatom classification are based only on alkalinity, whereas the reference values for river invertebrates are based on twelve different parameters. In this handbook, the prediction of reference values for each quality element is described in the section relating to that element.

The UK's national typology was used only for reporting monitoring results to the European Commission.

4.5 Sampling site selection within river basins

Different surveys require different survey designs.

Surveillance to monitor changes in status at a national scale requires a different distribution of sites compared to a programme to monitor impacts at high risk sites such as major discharges or abstractions.

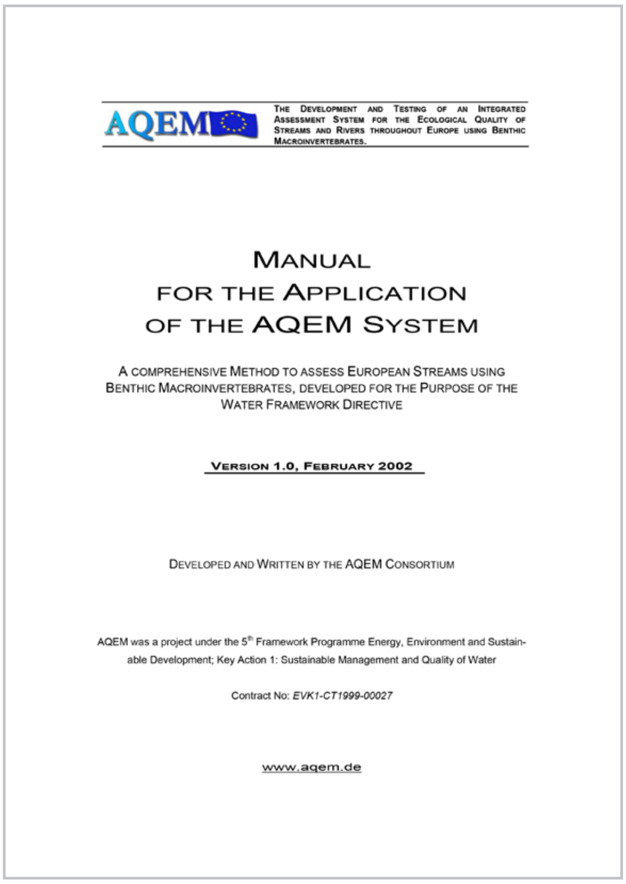
Operational monitoring to assess the performance of management measures implemented at a local scale will require a different network of sites compared to operational monitoring to assess the impact of national measures such as changes to chemical standards.

Investigative monitoring is likely to follow a BACI design (BACI = before, after, control, impact) which, in rivers, usually involves combinations of upstream control and downstream impact sites. Investigative monitoring to locate sources of pollution will be based on upstream-downstream comparisons at successive sites.

4.6 Sampling site selection within water bodies

Sample sites are selected within water bodies and are chosen to represent the quality of that water body as a whole.

The **AQEM project** provides a useful overview of the considerations and rules important in the selection of sampling sites. Much of this guidance is contained within the **AQEM Manual** (16)



AQEM — THE DEVELOPMENT AND TESTING OF AN INTEGRATED ASSESSMENT SYSTEM FOR THE ECOLOGICAL QUALITY OF STREAMS AND RIVERS THROUGHOUT EUROPE USING BENTHIC MACROINVERTEBRATES.

MANUAL
FOR THE APPLICATION
OF THE AQEM SYSTEM

A COMPREHENSIVE METHOD TO ASSESS EUROPEAN STREAMS USING BENTHIC MACROINVERTEBRATES, DEVELOPED FOR THE PURPOSE OF THE WATER FRAMEWORK DIRECTIVE

VERSION 1.0, FEBRUARY 2002

DEVELOPED AND WRITTEN BY THE AQEM CONSORTIUM

AQEM was a project under the 5th Framework Programme Energy, Environment and Sustainable Development; Key Action 1: Sustainable Management and Quality of Water

Contract No: *EVK1-CT1999-00027*

www.aqem.de

Figure 1.10
AQEM Manual

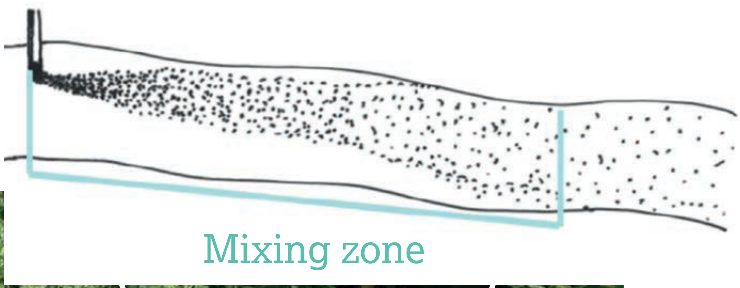
The main goal of a monitoring programme is not to assess local features of a stream but to gain understanding of the ecological quality of a whole water body or a complete catchment. Sampling sites should be representative of the reach being monitored and therefore representative of its biota.

In general, monitoring sites should usually be at the lower end of water bodies so that they can detect impacts from activities further upstream. However, a disadvantage of monitoring at the downstream end of reaches is that smaller, upstream types will be under-represented, especially head waters that are particularly sensitive to pressure and very important as sources of recolonisation.

Biological samples usually require different sites to those used for chemical monitoring. Chemical monitoring sites tend to be on bridges for convenience and ease of access. However, they should be avoided for biological sampling as they will influence the sample because of shading, because of rubble and other debris below the bridge, and because bridges are often located where the river channel is narrowest and therefore not typical of the water body as a whole. The physical characteristics of the sampling site should be as natural as possible so that the samples are representative of the water body, and the biota reflects the pressures in the water body as a whole rather than those in the immediate vicinity of the monitoring site.

Aim to avoid sites that are:

- close to artificial influences, such as dams, bridges, fords, weirs, piers, moorings, reinforced or artificial banks, and livestock watering areas. If this is not possible, the site must represent the water body as a whole. Record any artificial influences on the field data form and take them into account in data analysis
- immediately downstream of confluences or discharges where waters are not fully mixed (see Figure 1.11)
- close to the influence of in-stream lakes and reservoirs
- on stretches subject to dredging or regular weed removal
- in isolated habitats, such as in riffles when they are uncommon in the reach; isolation causes biological communities to be intrinsically less diverse)
- on braided or divided channels – if the site cannot be located elsewhere, such as on a fully braided river, sample within the largest natural channel
- predominantly on bedrock, as it is difficult to sample the invertebrate fauna.



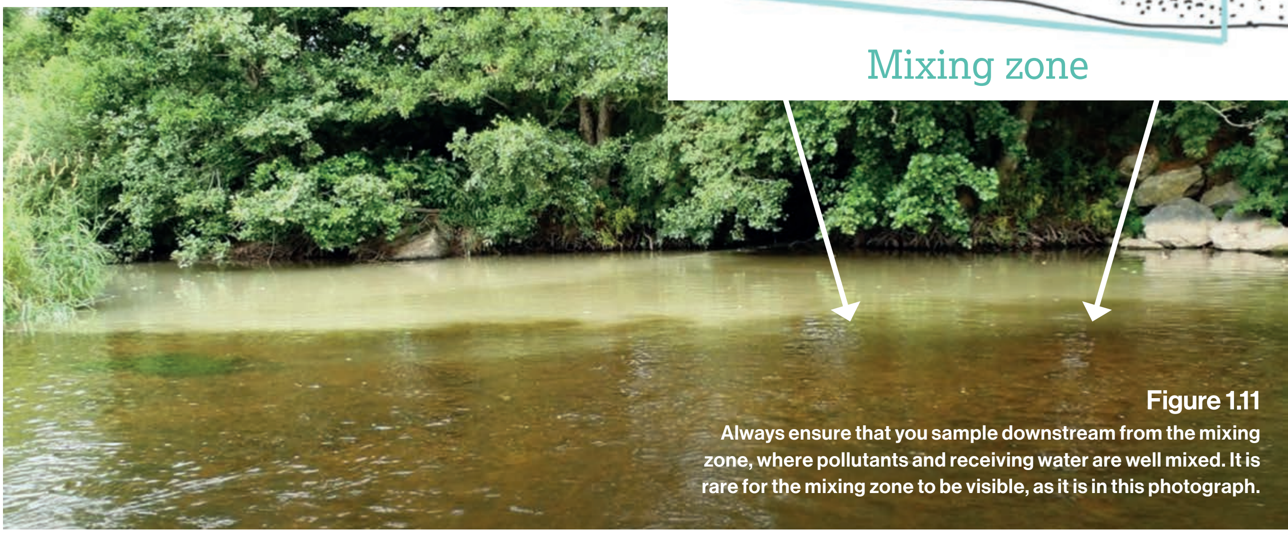
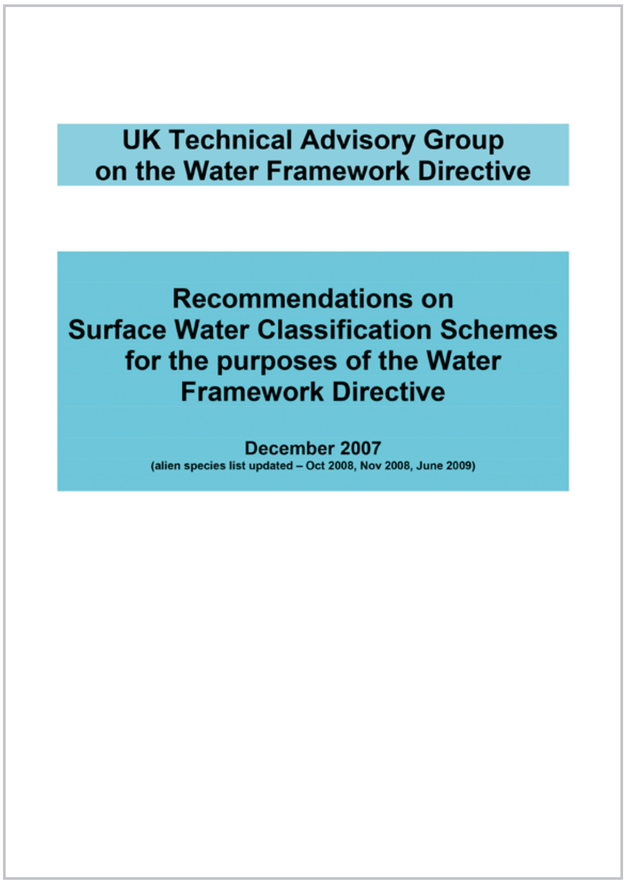


Figure 1.11
Always ensure that you sample downstream from the mixing zone, where pollutants and receiving water are well mixed. It is rare for the mixing zone to be visible, as it is in this photograph.

The 'sampling area' from which the sample is collected should be within a larger 'survey area' with similar characteristics. The sampling area is the spot where the biological sample is taken and should be representative in order for the stream reach to be assessed. The sampling area must reflect the habitat composition of the survey area. The size of the sampling area depends on the stream width and the quality element being sampled. The survey area might cover a section of several hundred metres of stream length up to a complete catchment area of a small stream for which the sampling site should be representative. This ensures that the sampling area is not an isolated habitat, enables a new sampling area to be used if part of the site is damaged, and minimises the effect of inaccurate relocation of the site by a different surveyor.

It is also useful to have biological monitoring sites close to sites for monitoring other elements including chemistry and hydromorphology so that they can be analysed together.

5
WFD SURFACE WATER CLASSIFICATION


UK Technical Advisory Group on the Water Framework Directive

Recommendations on Surface Water Classification Schemes for the purposes of the Water Framework Directive

December 2007
(alien species list updated – Oct 2008, Nov 2008, June 2009)

Figure 1.12
UK Technical Advisory Group Paper on Surface Water Classification [17]

wfd Water Framework Directive UK TAG

The key overview document for the UK on surface water classification is the **UK Technical Advisory Group Paper (UK TAG), 2007 (revised 2009), Recommendations on Surface Water Classification Schemes for the Purposes of the Water Framework Directive** [17] Front cover shown in Figure 1.12.

UK TAG website (http://www.wfduk.org/) includes documents that describe how the UK has implemented the WFD, including detailed descriptions of all the methods used for surface water classification.

The official status class boundaries, including chemical, hydromorphological and biological quality environmental standards implemented in England and Wales, are published together in https://www.legislation.gov.uk/uksi/2015/1623/pdfs/uksiod_20151623_en_003.pdf [18]

legislation.gov.uk

5.1 Introduction

In order to assess and report the quality of surface waters it is essential to develop a method of comparing water bodies in a consistent and transparent way that is easy for anyone to understand. This is known as **classification.** Knowledge of compliance against classifications drives the river basin planning process and is used to target investment to meet the agreed objectives.

The classification process results in each surface water body being assigned a status class. The WFD uses a five-class system for each surface water type. These status classes are termed: high, good, moderate, poor, and bad. Each class represents a different degree of degradation from human interference.

Surface water quality is expressed as **Chemical Status and Ecological Status.**

The Directive uses the term 'quality elements' to refer to the different indicators of ecological quality that comprise its ecological status classification scheme, and different chemicals that comprise its chemical classification scheme. Each quality element contributes to status according to the 'one out, all out' principle, in which overall status is deemed to be that of the individual element indicating the worst class. **'One out, all out' is an important principle of water classification and management. It ensures that all the quality elements needed to provide for a balanced ecosystem are in place within a given class**. However, it causes difficulty because as more elements are considered, the greater the risk of a downgraded misclassification for purely statistical reasons caused by errors being additive. This tends to give a pessimistic view of quality. Also, improvements in individual elements may be masked by failures of other elements. For example, the biological quality may have improved due to pollution control actions, but one or more chemical elements may have failed, leading to an overall failure of good status. This is shown schematically in Figure 1.13.

Chemical status is based on the concentrations of priority substances. There are European standards for all priority substances. Chemical status can be good or bad.

Ecological status is based on:

- biological quality elements
- chemical and physico-chemical quality elements that support the biological elements
- pollutants being discharged in significant quantities, which are referred to as 'specific pollutants'
- hydromorphological quality elements comprising hydrological and morphological elements.

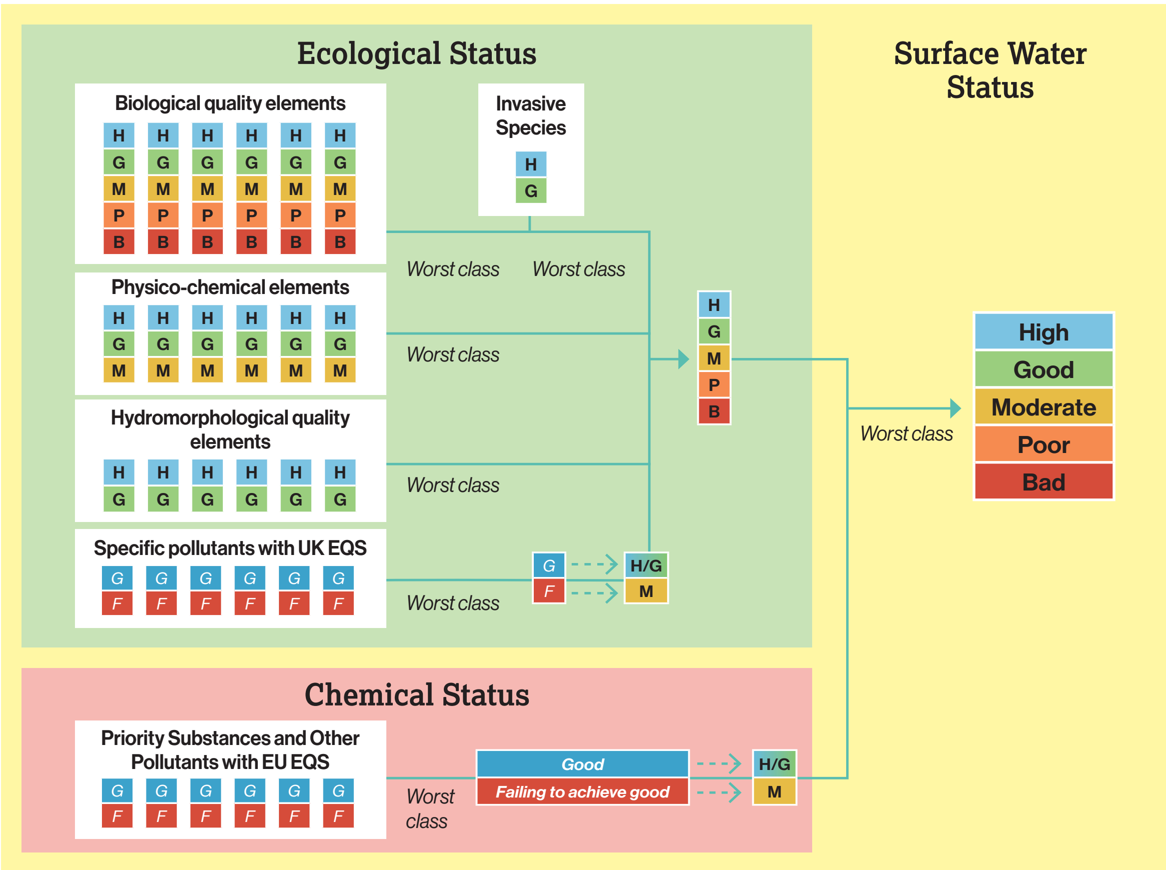


Figure 1.13

Schematic representation of how results for different quality elements are combined to classify ecological status, chemical status and overall surface water status under the WFD, based on 'one out, all out'. Note that only the biological quality elements define poor and bad status. *(After a diagram produced by Peter Pollard)*

There are no European standards for ecological status; each Member State has to develop its own ecological assessment method and ecological standards. A consequence of this is that the only difference between a priority substance and a specific pollutant is that standards for priority substances are set at European level and standards for specific pollutants are set nationally. Many specific pollutants are elevated to priority substances when Europe-wide standards are set.

'Ecological potential' replaces 'ecological status' in artificial and heavily modified waters. Ecological potential takes account of the reduction in biological and morphological status caused by a desirable physical modification that we do not want to remove in order to restore biological status. Physical modification in this context includes structures such as dams or flood defences; it does not include chemical, physico-chemical, or hydrological elements. Ecological potential can be either good or not good.

Chemical and ecological status or potential are combined to produce an overall water body status. The class given to a particular water body will represent an estimate of the degree to which the structure and functioning of the aquatic ecosystem have been altered by the different pressures to which that water body is subject. Figures 1.14 and 1.15 provide a description of the five status classes.

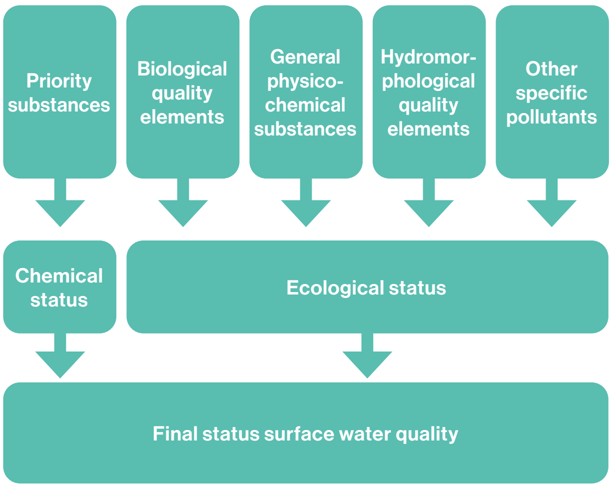
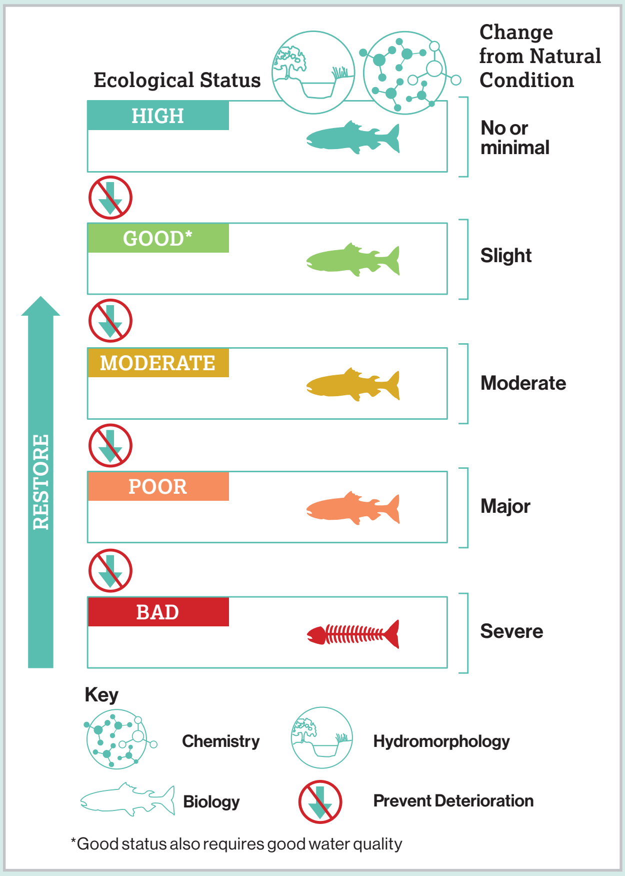


Figure 1.14

Individual WFD status classifications contributing to the overall WFD Surface Water Status. *(Torenbeek, 2007, translated)*

High status represents only very minor changes to the hydromorphology, physico-chemistry and biology of a water body. Good status requires no more than slight changes to the biology of the water body and compliance with quality standards for pollutants. The other status classes are defined according to the level of impact upon their biology. The aim is to restore quality and to prevent deterioration.



High ecological status

Each of the relevant biological, hydromorphological and physico-chemical quality elements match their reference conditions.

Good ecological status

The relevant biological quality elements are only slightly changed from their reference conditions as a result of human activities. Environmental quality standards are achieved for the relevant physico-chemical quality elements.

Moderate ecological status

The relevant biological quality elements are moderately changed from their reference conditions as a result of human activities.

Poor ecological status

The relevant quality biological elements show major changes from their reference conditions as a result of human activities (ie there are substantial changes to the reference biological communities).

Bad ecological status

The relevant biological quality elements are severely changed from their reference conditions as a result of human activities (ie large portions of the reference biological communities are absent).

Figure 1.15

Graphic representation and text summary of the five stage classification system

5.2 Chemical status – priority substances

The WFD specifies a list of Priority Substances and other pollutants that must be taken into account in status classification.

Table 1.8

Priority substances and other pollutants listed in the 2013 Directive amending the original WFD

No	Name of substance
1	Alachlor
2	Anthracene
3	Atrazine
4	Benzene
5	Brominated diphenylethers
6	Cadmium and its compounds (depending on water hardness classes)
6a	Carbon tetrachloride
7	C10-13 Chloroalkanes
8	Chlorfenvinphos
9	Chlorpyrifos (Chlorpyrifos-ethyl)
9a	Cyclodiene pesticides: Aldrin, Dieldrin, Endrin, Isodrin
9b	DDT total
	para-para- DDT
10	1,2-Dichloroethane
11	Dichloromethane
12	Di(2-ethylhexyl)phthalate (DEHP)
13	Diuron
14	Endosulfan
15	Fluoranthene
16	Hexachlorobenzene
17	Hexachlorobutadiene
18	Hexachlorocyclohexane
19	Isoproturon
20	Lead and its compounds
21	Mercury and its compounds
22	Naphthalene
23	Nickel and its compounds
24	Nonylphenols (4-Nonylphenol)
25	Octylphenols ((4-(1,1',3,3'-tetramethylbutyl)-phenol))
26	Pentachlorobenzene
27	Pentachlorophenol
28	Polyaromatic hydrocarbons (PAH) Benzo(a)pyrene Benzo(b)fluoranthene Benzo(k)fluoranthene Benzo(g,h,i)perylene Indeno(1,2,3-cd)pyrene
29	Simazine
29a	Tetrachloroethylene
29b	Trichloroethylene
30	Tributyltin compounds (Tributyltin-cation)
31	Trichlorobenzenes
32	Trichloromethane
33	Trifluralin
34	Dicofol
35	Perfluorooctane sulfonic acid and its derivatives (PFOS)
36	Quinoxyfen
37	Dioxins and dioxin-like compounds
38	Aclonifen
39	Bifenox
40	Cybutryne
41	Cypermethrin
42	Dichlorvos
43	Hexabromocyclododecane (HBCDD)
44	Heptachlor and heptachlor epoxide
45	Terbutryn

The status of a water body is classed as failing to achieve good surface water chemical status if an environmental quality standard for one or more of the priority substances or other dangerous substances listed in Table 1.8 fails.

5.3 Ecological Status

Ecological status is an expression of the quality of the structure and functioning of surface water ecosystems as indicated by the condition of a number of 'quality elements'.

The biological quality elements drive ecological status assessment. The other quality elements are supporting factors. They are environmental parameters that, in the right amounts, are essential to biological communities. Their class boundaries are set at values that ensure good or better biological status.

The WFD provides descriptive 'normative definitions' for high, good and moderate status of all the ecological quality elements. Figure 1.16 provides these important definitions, and specific text from the WFD is included here.

Element	High Status	Good Status	Moderate Status
General	There are no, or only very minor, anthropogenic alterations to the values of the physico-chemical and hydromorphological quality elements for the surface water body type from those normally associated with that type under undisturbed conditions. The values of the biological quality elements for the surface water body reflect those normally associated with that type under undisturbed conditions, and show no, or only very minor, evidence of distortion. These are the type-specific conditions and communities.	The values of the biological quality elements for the surface water body type show low levels of distortion resulting from human activity, but deviate only slightly from those normally associated with the surface water body type under undisturbed conditions.	The values of the biological quality elements for the surface water body type deviate moderately from those normally associated with the surface water body type under undisturbed conditions. The values show moderate signs of distortion resulting from human activity and are significantly more disturbed than under conditions of good status.

Figure 1.16

Normative definitions of ecological status classifications (from the WFD). See also the bullet points below.

Mafly larave like this *Ecdyonurus*, are often associated with high status water bodies.

- Waters achieving a status below moderate shall be classified as **poor** or **bad**.
- Waters showing evidence of **major alterations** to the values of the biological quality elements for the surface water body type and in which the relevant biological communities ***deviate substantially*** from those normally associated with the surface water body type under undisturbed conditions, shall be classified as **poor**.
- Waters showing evidence of **severe alterations** to the values of the biological quality elements for the surface water body type and in which large portions of the relevant biological communities normally associated with the surface water body type under undisturbed conditions ***are absent***, shall be classified as **bad**.

As mentioned previously, there are five classes of ecological status: high, good, moderate, poor and bad. The Directive requires that the overall ecological status of a water body be determined by the results for the biological or physico-chemical quality element that has the worst status class (ie the quality element worst affected by human activity). This is called the 'one out – all out' principle.

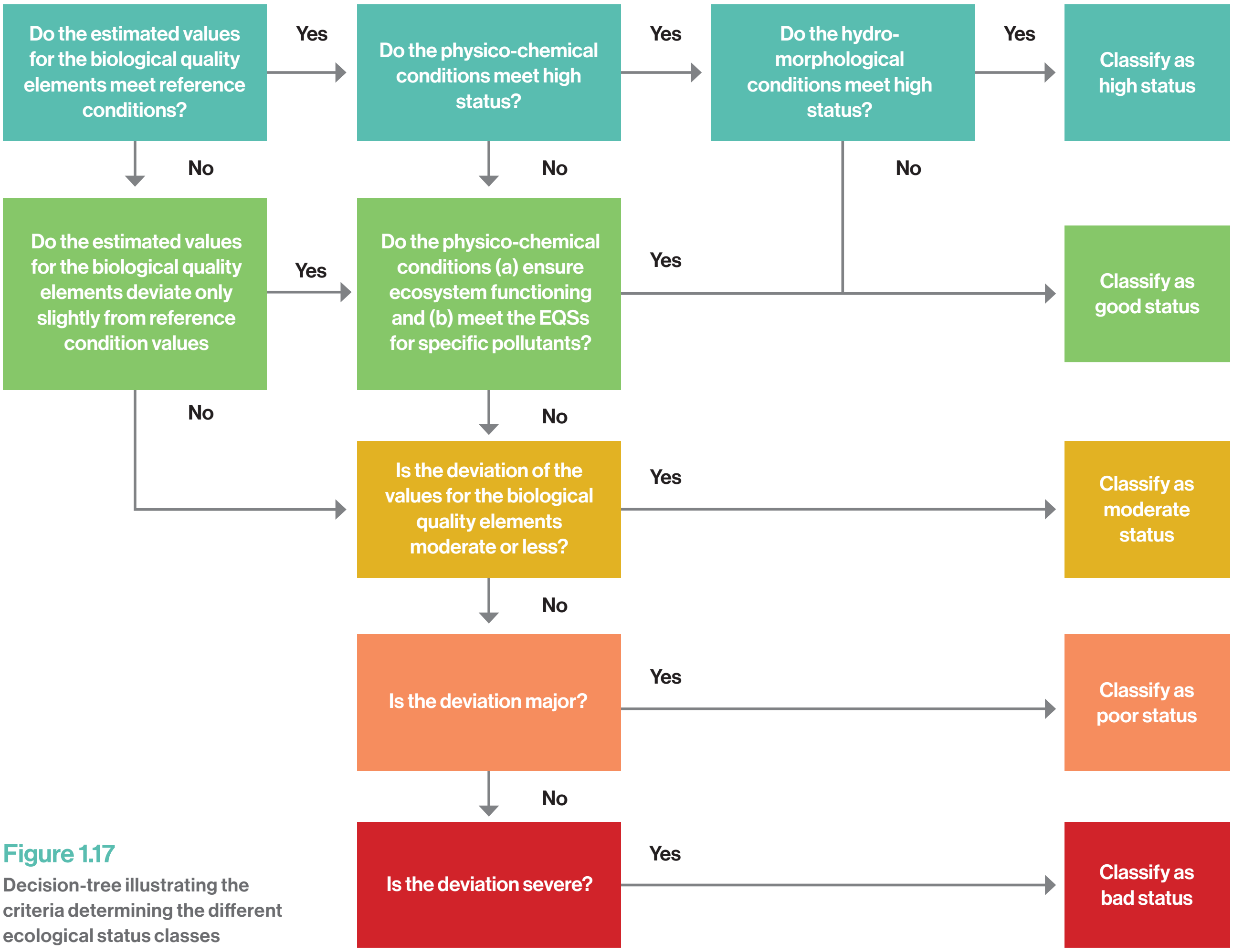


Figure 1.17
Decision-tree illustrating the criteria determining the different ecological status classes

The five-stage classification is used to define environmental status or potential of the water environment and environmental quality objectives. It is also used to assess progress towards achieving environmental objectives and the effectiveness of measures to restore quality.

Biological status or potential is supplemented by assessments of various supporting physico-chemical elements that are necessary to support the biota (such as oxygen, inorganic plant nutrients and pH) and hydromorphological quality elements (such as flow requirements, connection to groundwaters, residence time, and nature of the substrate). **These are combined to define ecological status**. In addition, certain toxic chemicals are included for which standards to protect biological status are set at national level, and these are known as 'specific pollutants'. Chemicals that are to be severely limited or eliminated, with standards set at European level and specified in WFD or subsequent amendments, contribute to a chemical classification. (See previous section on chemicals.) **The ecological and chemical classifications are combined into an overall status classification.**

Development and implementation of the WFD focuses on biological and ecological outcomes in the aquatic environment and has brought a step change in our approach to water management across the EU and the UK. The WFD approach sets out the principles of water management across all EU and UK River Basins for the near 30 year lifespan of the Directive and for the foreseeable future.

Implementing the WFD means that expensive regulatory and infrastructure investments are now focused on meeting biological objectives. This required significant improvements to biological and ecological monitoring and assessment. It has accelerated research and the development of biological methods and guidance across the EU to improve accuracy, quality assurance and equivalence of assessment methods. This handbook provides background to this development and gives open access to the wealth of technical information available to water managers.

A further key principle of biological and ecological assessment is management at river basin or catchment management level (sometimes referred to as watershed management). This also underlies the WFD and has been a core element of water management in the UK for the past 50 years, since the establishment of the UK Regional Water Authorities in 1974. In Western Europe, this approach has facilitated the successful clean-up of many river systems and the rivers Rhine, Danube and Thames are examples of how this restoration has progressed.

Figure 1.18 illustrates the progressive deterioration and restoration of the **River Rhine** during the last century – a situation replicated in most industrial European rivers. This is a positive example of how extensive pollution clean-up can restore a damaged river. The invertebrate monitoring information, here shown simply as species number, shows the sensitive species being heavily damaged during the most polluted period, as indicated by the significant reduction in dissolved oxygen concentration. The species number recovers as the pollution is reduced, until a similar, but slightly different species mix is re-established naturally.

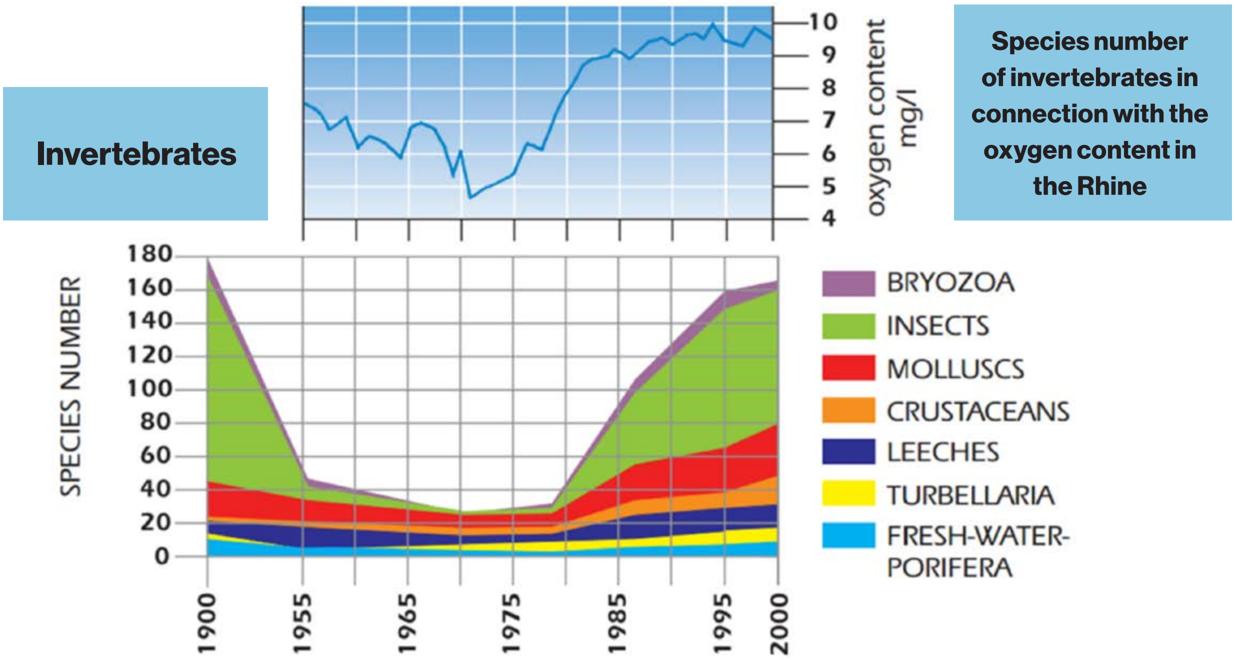


Figure 1.18

River Rhine deterioration and clean-up 1900 to 2000. Shows links between chemical and biological monitoring and assessment
https://www.iksr.org

5.4 General physico-chemical quality elements

The general chemical and physico-chemical quality elements describe water quality. They include chemical substances, such as nutrients, and physical properties, such as the thermal regime. At high ecological status, the condition of each element must be within the range of conditions normally associated with undisturbed conditions. At good ecological status, the Directive requires that the general physico-chemical quality elements comply with standards established by the Member State to protect the functioning of the ecosystem.

Table 1.9

General chemical and physico-chemical quality elements relevant to the different categories of surface water

Water category	Quality elements	Indicators for which standards have been proposed by UKTAG
Rivers	1 **Thermal conditions** 2 **Oxygenation conditions** 3 Salinity 4 **Acidification status** 5 **Nutrient conditions**	1 Temperature 2 Dissolved oxygen concentration 3 - 4 pH 5 Soluble reactive phosphorus concentration
Lakes	1 Transparency 2 Thermal conditions 3 **Oxygenation conditions** 4 **Salinity** 5 **Acidification status** 6 **Nutrient conditions**	1 - 2 - 3 Dissolved oxygen concentration 4 Conductivity 5 Acid neutralising capacity 6 Total phosphorus concentration
Transitional waters (eg estuaries)	1 Transparency 2 Thermal conditions 3 **Oxygenation conditions** 4 **Nutrient conditions**	1 - 2 - 3 Dissolved oxygen concentration 4 Dissolved inorganic nitrogen
Coastal waters	1 Transparency 2 Thermal conditions 3 **Oxygenation conditions** 4 **Nutrient conditions**	1 - 2 - 3 Dissolved oxygen concentration 4 Dissolved inorganic nitrogen

Table 1.10

Specific pollutants *(any pollutant from the list below which is being discharged in significant quantities into the water body)*

The WFD also identifies a list of specific pollutants which are discharged in significant quantities from processes across Europe and which are known to impact on ecological quality:	
(i)	Organohalogen compounds and substances which may form such compounds in the aquatic environment
(ii)	Organophosphorous compounds
(iii)	Organotin compounds
(iv)	Substances and preparations, or the breakdown products of such, which have been proved to possess carcinogenic or mutagenic properties or properties which may affect steroidogenic, thyroid, reproduction or other endocrine-related functions in or via the aquatic environment
(v)	Persistent hydrocarbons and persistent and bioaccumulable organic toxic substances
(vi)	Cyanides
(vii)	Metals and their compounds
(viii)	Arsenic and its compounds
(ix)	Biocides and plant protection products

For good ecological status, the environmental quality standards established for specific pollutants must not be exceeded. With the exception of ammonia in freshwaters, environmental quality standards for the specific pollutants have been set in such a way that, where the standards are met, no adverse effects on aquatic plants and animals should occur.

5.5 Hydromorphological quality elements

For high status to be achieved, the Directive requires that there are no more than very minor human alterations to the hydromorphological quality elements.

At good, moderate, poor and bad status, the required values for the hydromorphological quality elements must support the required biological quality element values for the relevant class. The standards and condition limits recommended by UKTAG are intended to help assess the risk of failing to achieve the necessary values.

Table 1.11
Hydromorphological quality elements

Rivers	Lakes	Transitional waters	Coastal waters
(i) Quantity and dynamics of water flow	(i) Quantity and dynamics of water flow	(i) Depth variation	(i) Depth variation
(ii) Connection to groundwater bodies	(ii) Residence time	(ii) Quantity, structure and substrate of the bed	(ii) Structure and substrate of the coastal bed
(iii) River continuity	(iii) Connection to groundwater bodies	(iii) Structure of the intertidal zone	(iii) Structure of the intertidal zone
(iv) River depth and width variation	(iv) Lake depth variation	(iv) Freshwater flow	(iv) Direction of dominant currents
(v) Structure and substrate of the river bed	(v) Quantity, structure and substrate of the lake bed	(v) Wave exposure	(v) Wave exposure
(v) Structure of the riparian zone	(v) Structure of the lake shore		

5.6 Biological quality elements

The WFD normative definitions give qualitative descriptions of the high, good and moderate status of each biological quality element in each surface water category. High ecological status is achieved when each of the relevant biological, hydromorphological and physico-chemical elements match their reference conditions. At good ecological status, none of the biological quality elements can be more than slightly altered from their reference conditions. At moderate status, one or more of the biological elements may be moderately altered. At poor status, the alterations to one or more of the biological quality elements are major and, at bad status, there are severe alterations such that a large proportion of the reference biological community is absent. Biological status is defined by a wide range of aquatic biota, depending on the water category – river, lake, coastal or transitional (Figure 1.19). Although the monitoring of phytoplankton is a requirement for the classification of rivers, it is not used in the UK because, with a few exceptions, rivers in the UK are rarely long enough to allow growth of phytoplankton. In freshwaters, macrophytes and phytobenthos are combined as one quality element. In the UK, only diatoms are used to define phytobenthos status.

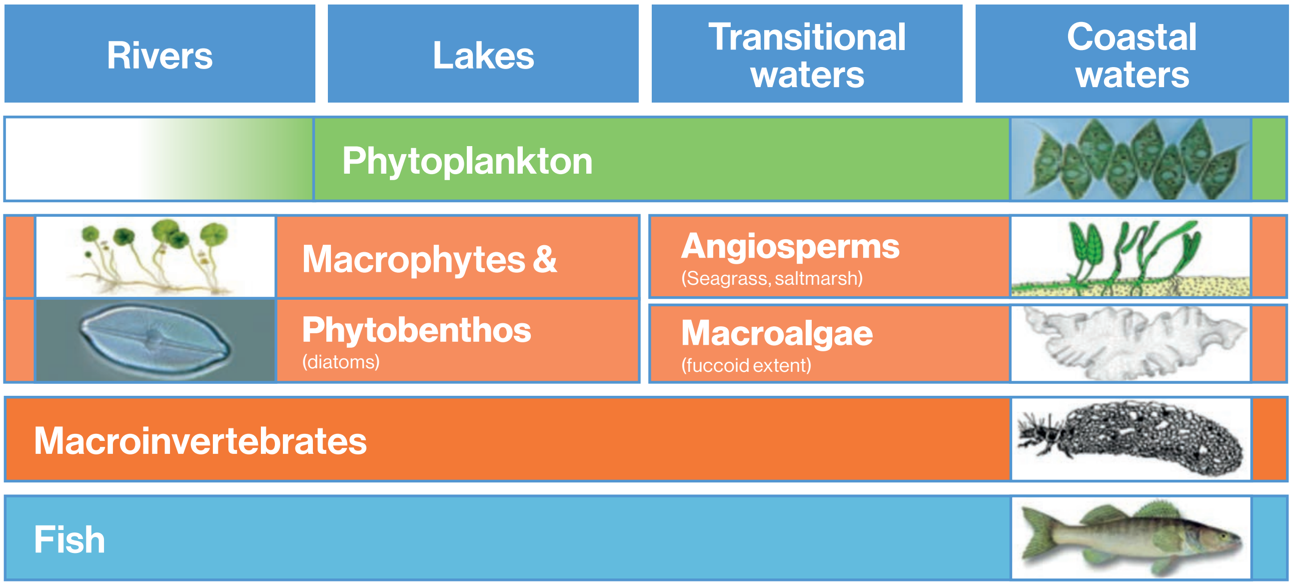
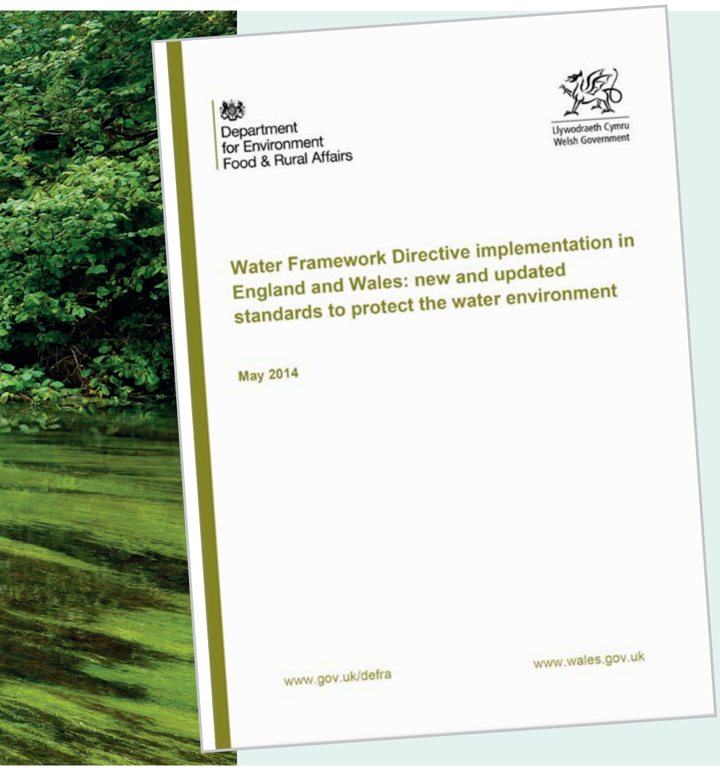


Figure 1.19

Biological quality elements used to classify the status of different surface water categories for the WFD. (River phytoplankton are shaded out because they are not used for status assessment in the UK.)


Department for Environment Food & Rural Affairs

Llywodraeth Cymru Welsh Government

Water Framework Directive implementation in England and Wales: new and updated standards to protect the water environment

May 2014

www.gov.uk/defra

www.wales.gov.uk

For the second WFD cycle the UK improved some of the metrics and methods used to assess water body status in its key document published by Defra in May 2014, entitled ***Water Framework Directive implementation in England and Wales: new and updated standards to protect the water environment*** [19]

This ensured consistency across the UK as the second WFD cycle was developed. It provides additional information on metrics and software tools that might be considered for use outside the UK.

These changes were incorporated in legislation in *The Water Framework Directive (Standards and Classification) Directions (England and Wales) 2015.* [18] **https://www.legislation.gov.uk/uksi/2015/1623/pdfs/uksiod_20151623_en_003.pdf**

Figure 1.20

Front cover Defra 2014, Water Framework Directive implementation in England and Wales: new and updated standards to protect the water environment [19]

Table 1.12

Current official UK biological status classifications for WFD (2nd cycle), from Defra 2014.

Water body Category	Element	Metric	Software Tool
All	Alien invasive	Table in Defra standards	
River	Phytobenthos	Trophic Diatom Index v4 (TDI 4)	River DARLEQ 2
River	Macrophytes	Mean Trophic Rank (MTR)	River LEAFPACS 2
River	Invertebrates	Number of Walley Hawkes Paisley Trigg-scoring taxa (WHPT Ntaxa)	River Invertebrate Classification Tool (RICT)
River	Invertebrates	Average Walley Hawkes Paisley Trigg score per taxon (WHPT ASPT)	RICT
River	Invertebrates	WFD species-level Acid Water Indicator Community (WFD AWICsp)	
River	Fish		FCS2
Lake	Phytoplankton		PLUTO
	Phytobenthos	TDLI 2	Lake DARLEQ2
	Macrophytes		Lake LEAFPACS2
	Invertebrates	Chironomid Pupal Exuviae Technique (CPET)	
		LAMM	

The overall biological status is the status class indicated by whichever biological quality element indicates the worst status, following the 'one out, all out' principle. Certain invasive non-native species also contribute to the classification, but only in so much that they prevent sites where they exist from being classified as high status.

A number of different parameters (eg the balance of different groups of species, the number of different species, the overall abundance of species, etc) may be used to estimate the status of a quality element. These parameters are quantified in various metrics or biotic indices. Different metrics indicate the impact of different pressures (eg the effects of pollution or the effects of morphological alterations) on the element. Results for different metrics may be combined to give a representative picture of the impact of a particular type of pressure (or range of pressures) on the quality element. Multi-metrics may be appropriate where none of the metrics on their own give a sufficiently reliable indication that the quality element has been adversely impacted as a result of human activities. Within a quality element, different metrics are combined in the most appropriate way, most commonly either 'one out, all out', or an average. An overview of the UK classifications and tools is given in Table 1.12.

Operational monitoring only needs to include the quality elements most sensitive to the pressures affecting the status of the water body. Table 1.13 provides an expert judgement view (from UK Technical Advisory Group) as to which elements are most appropriate for specific pressures. The table provides a means of focusing monitoring effort to aid efficient use of resources. Where more than one element is sensitive to a pressure, eg all are sensitive to eutrophication, expert opinion should be employed to choose the most sensitive elements for the category of water concerned.

The use of multiple metrics can improve confidence in the final classification. For example, when assessing phytoplankton, biomass is an important metric because it determines the overall amount of phytoplankton which in turn influences light penetration and oxygen concentration in a water body. Taxonomic composition is also an important metric of phytoplankton because it shows when highly undesirable species (such as cyanobacteria and other opportunistic taxa) are starting to dominate the phytoplankton community and when taxa indicative of particular environmental pressures are prevalent.

The tools developed for classification should continue to be refined to reflect newly emerging environmental pressures and to improve the accuracy of status assessment in order to improve the certainty that programmes of measures are implemented where they are needed. This development work should take account of new data collected through monitoring programmes, together with improvements in scientific understanding on causes and effects. New or modified tools should also be developed where the existing tools are currently unable to properly reflect the impact of particular pressures on the water environment.

Details of specific aquatic invertebrate metrics are dealt with in detail in **Chapter 3**.

Table 1.13

A simplified list of quality elements particularly sensitive to the pressures affecting rivers. Biological elements actually respond to the integrated effect of all pressures. **From UK TAG, 2005, 12. Monitoring Networks** [2]

Source Pressure	Category of Effect	Exposure Pressure	Macrophyte	Phytobenthos	Macroinvertebrates	Fish	Morphology	Hydrology	General Physico-chemical	Specific Pollutants	Priority substances	Priority hazardous substances
Nutrient Enrichment	Primary effect on biology	Change in nutrient concentration in defined water body. Enhanced biomass, changes to other primary producers.	X	X				X	Nutrient suite			
Organic Enrichment	Primary effect on biology	Increased organic enrichment; change in biological community structure			X			X	Organic suite			
Annex 8 and Annex 10 pollutants	Primary effects on sediment and water quality	Increased concentrations of contaminants (water column and sediments)			X			X	General suite	X	X	X
Hydrological	Primary effect on biology	Changed water levels from abstraction; altered flow regime impacting biology	X	X	X	X	X	X	General suite			
Morphological	Primary effect on biology	Riparian and channel modification; altered sediment characteristics (eg size); smothering and damage to river bed	X		X	X	X	X				
Acidification	Primary effect on biology	Change in ANC & pH; change in biological community & toxicity synergies		X	X	X			Acidification suite			

We can only tell if the biology is impacted if we know what the biological composition should be under natural conditions.

$$\text{EQR} = \frac{\text{value of metric observed}}{\text{value of metric at reference condition}}$$

Natural biological communities can vary enormously between locations because of differences in natural conditions such as geology, temperature, altitude and hydrology. These variations can be as great as impacts from anthropogenic pressures. Knowing what the biological composition should be at any given location is therefore imperative if we want to know the degree to which biological communities are impacted at that location. Natural biological communities are grouped into different types with similar composition, and these are associated with different types of natural environmental conditions. These typologies are recognised by WFD. A general typology is described in WFD Annex II, based on altitude, catchment area and geology for rivers, and altitude, depth, surface area and geology for lakes; but individual biological quality elements have their own physico-chemical definitions or 'typologies' because different parameters determine the distribution of different natural biological communities.

The degree to which a biological quality element is impacted (ie the biological quality for a particular quality element) is quantified by the degree to which it deviates from its natural state. This is expressed as an ecological quality ratio (EQR), which is actually a decimal fraction. A number of metrics can be used to quantify biological communities, such as number of taxa, abundance of particular species or groups, biotic indices and diversity indices. Any of these can be expressed as an ecological quality ratio.

The observed value is what is seen from a sample and the reference value is what it should be under near natural conditions. EQRs normally range from zero to one, but they can exceed one. This is a consequence of defining reference value as the average value of a metric at reference sites: by definition, at some sites the value of the metric must be greater than the reference value. If an EQR is substantially less than one, the value of the metric is much lower than we expect under natural unimpacted conditions, and, assuming that the metric is one that decreases with increasing impact, we can assume that the biological community is poorer than it should be. If the EQR is around one, the site is in high status around reference condition. If the value is greater than one, the site is of high status and the biological community is better than average under natural conditions – eg an exceptionally diverse site in a nature reserve. Class boundaries for WFD biological status classifications are always expressed as EQRs. The boundary EQR values represent particular degrees of deviation from the reference values. High status is represented by values relatively close to one (ie little or no deviation) and bad status by values much less than one (ie substantial deviation). The EQR scale allows the biological quality of communities from very different environments to be directly comparable, despite large differences in their natural biological communities. The concept of reference is described more fully in **Section 5.7**.

5.7 Reference conditions

The main goal of stream assessment, according to the Water Framework Directive, is to classify water bodies into status classes (high, good, moderate, poor or bad) and these are defined by their deviation from type-specific reference conditions. **The basic principle of biological quality assessment in the WFD is to compare the actual biological communities in the water body with the biological communities that should be there if it was in a natural state** – specifically, the actual values of biological metrics with their values at reference conditions of the relevant water type.

The reference condition is, by definition, 'high status'. The general definition of high status in the WFD is: *There are no, or only very minor, anthropogenic alterations to the values of the physico-chemical and hydromorphological quality elements for the surface water body type from those normally associated with that type under undisturbed conditions. The values of the biological quality elements for the surface water body reflect those normally associated with that type under undisturbed conditions, and show no, or only very minor, evidence of distortion. These are the type-specific conditions and communities.* (Table 1.2, Annex V, WFD).

Reference conditions are described in more detail in CIS Guidance No 10 *Rivers & Lakes – typology, reference conditions and classification systems.* The definitions of reference were refined further during intercalibration and are described later in this section.

Ideally, reference values are derived from reference sites that have no (or minimal) alterations as a result of human interference or pressure. UK TAG recommends that: *reference conditions should reflect a state in the present or in the past corresponding to very low pressure, without the effects of major industrialisation, urbanization and intensification of agriculture, and with only very minor modification of physico-chemistry, hydromorphology and biology.*

This is not always possible, so reference values may be determined using:

- **networks of reference sites**
- **modelling approaches**
- **or, where the above are not possible (even in combination), expert judgement.**

5.7.1 Network of reference sites

In the **AQEM project,** criteria for the selection of reference condition sites were described as follows:

A reference stream should fulfil all requirements necessary to allow a completely undisturbed fauna to develop and establish itself. Therefore, reference sites should not only be characterised by clean water but also by undisturbed stream morphology and near-natural catchment characteristics. Though it is impossible to find sites in such a pristine condition for many stream types, AQEM has defined the following criteria, which should be met by realistic reference sites:

Basic statements

- The reference condition must be practical, achievable within a river basin and reasonable.
- A reference site, or process for determining it, must hold or consider important aspects of natural conditions.
- The reference conditions must reflect only minimal anthropogenic disturbance.

Land use practices in the catchment area

- In most countries there is anthropogenic influence within the catchment area. Therefore, the degree of urbanisation, agriculture and silviculture (forestry) should be as low as possible for a site to serve as a reference site. No absolute minimum or maximum values have been set for the defining reference conditions (eg % arable land use, % native forest); instead, the least-influenced sites with the most natural vegetation are to be chosen.

River channel and habitats

- The reference site floodplain should not be cultivated. If possible, it should be covered with natural climax vegetation and/or unmanaged forest.
- Coarse woody debris must not be removed (minimum demand: presence of coarse woody debris).
- Stream bottoms and stream margins must not be fixed.
- Preferably, there should be no migration barriers (affecting the sediment transport and/or the biota of the sampling site).
- Only moderate influence due to flood protection measures can be accepted.

Riparian vegetation and floodplain

- Natural riparian vegetation and floodplain conditions must be retained, making lateral connectivity between the stream and its floodplain possible; depending on the stream type, the riparian buffer zone should be greater or equal to 3x channel width.

Hydrologic conditions and regulation

- No alterations of the natural hydrograph and discharge regime should occur.
- There should be no, or only minor upstream impoundments, reservoirs, weirs and reservoirs retaining sediment; no effect on the biota of the sampling site should be recognisable.
- There should be no effective hydrological alterations such as water diversion, abstraction or pulse releases.

Physical and chemical conditions

There should be:

- no point sources of pollution or nutrient input affecting the site
- no point sources of eutrophication affecting the site
- no sign of diffuse inputs or factors which suggest that diffuse inputs are to be expected
- normal background levels of nutrient and chemical base load, which reflect a specific catchment area
- no sign of acidification
- no liming activities
- no impairments due to physical conditions; in particular, thermal conditions must be close to natural
- no local impairments due to chemical conditions; in particular, no known point-sources of significant pollution, all the while considering the near-natural pollution capacity of the water body
- no sign of salinity.

Biological conditions

There must not be any:

- significant impairment of the indigenous biota by introduction of fish, crustaceans, mussels or any other kind of plants and animals
- significant impairment of the indigenous biota by fish farming.

In many cases, particularly in lowland stream types or larger rivers, no reference sites meeting the above criteria are available. For these stream types, the best available sites, which meet most of the criteria, should only be a starting point. The description of these reference communities should be supplemented by an evaluation of historical data and possibly the biotic composition of comparable stream types, eg streams of a similar size but located in a different ecoregion (AQEM consortium, 2002).

So, reference values should be based on information obtained from sites at which the quality element concerned is in reference condition (ie at high status). UK TAG state that this does not mean that at these sites the quality element will be entirely unaffected by human activities. However, it does mean that alterations to it are expected to be minor. There are relatively few sites at which all quality elements are in reference condition and from which data suitable for establishing reference values are available. Consequently, reference values can be derived from sites at which the quality element concerned is estimated to be in its reference condition but other elements at the sites may not be so.

White-clawed crayfish, *Austropotamobius pallipes* (FBA)

5.7.2 Temporally based reference condition

Instead of observed reference conditions, temporally based reference conditions may be used. Temporally based reference conditions can be based on either historical data or paleo-reconstruction, or a combination of both approaches. These approaches are commonly used in areas where human-induced stress is widespread and unperturbed references are few or lacking entirely. For example, paleo-reconstruction of past conditions may be determined either **(i) directly, based on species presence/absence from fossil remains, or (ii) indirectly, using relationships between fossil remains and inference to determine other values such as the reference pH situation**.

One of the strengths of a paleo-reconstruction approach is that it can often be used to validate the efficacy of other approaches if the conditions are stable **(CIS Guidance No. 10, 2003)** [(20)] Another advantage is that recent step changes in ecological status are more easily determined. A further strength of palaeo-reconstruction is that if strong relationships exist between land use and ecosystem composition and function, a predictive approach (hindcasting or extrapolating dose-response relationships) may be used to predict quality elements prior to major alterations in land use (eg pre-intensive agriculture).

Both of these approaches share some of the same weakness. They are usually site and organism-specific, and hence may be of limited value for establishing type-specific values. Regarding palaeo-reconstruction, caution should be exercised in unequivocal reliance on this method for providing the definitive value, because a different choice of calibration dataset used to infer ecological status may result in different values. The widespread use of historical data may be limited by its availability and unknown quality **(CIS Guidance No. 10, 2003)** [(20)]

5.7.3 Modelling approaches

In CIS guidance 10 *(Rivers and lakes – typology, reference conditions and classification systems, 2003)* [(20)] some remarks are made about modelling approaches:

When adequate numbers of representative reference sites are not available in a region/type, predictive modelling, using the data available within a region/type or borrowing data from other similar regions/types, can be used to determine reference values. One of the advantages of using predictive approaches is that the number of sites needed for reliable estimates of mean or median and error are usually lower than those needed if spatial approaches are used. This usually results in fewer sites that need to be sampled, together with lower implementation costs. A second advantage of using predictive approaches is that the models can often be inverted to examine the likely effects of mitigation measures. It must be stressed that predictive models only are valid for the ecoregion and water body type for which they are created.

5.7.4 Expert judgement

Expert judgement usually consists of a narrative statement of expected reference condition. Although an expert's opinion may be expressed semi-quantitatively, qualitative articulation is probably most common. Use of expert judgment may be warranted in areas where reference sites are few or absent.

One of the strengths of this approach is that it may also be used in combination with other methods. For example, expert judgment may be used to extrapolate findings from one quality element to another (eg paleo-reconstruction using fossil diatom remains may be used to infer invertebrate community composition), or to extrapolate dose-response relationships to those expected in unperturbed sites.

Another strength of this approach is that both empirical data and opinion can be amalgamated with present-day concepts of ecosystem structure and function. However, as a number of weaknesses are inherently associated with this approach, caution should be exercised when using it as the sole means of establishing reference conditions. For example, subjectivity, eg the common perception that it was always better in the past, and bias, eg even sites with low diversity can be representative, may limit its usefulness. Other drawbacks include the lack of clarity or low degree of transparency in assumptions used to establish reference and the lack of quantitative measures, eg mean or median values for validation. A further weakness of this, and many other approaches is that the measure obtained is often static, and hence does not include the dynamic, inherent variability often associated with natural ecosystems (CIS Guidance Nº 10).

5.8 Intercalibration

Intercalibration is an important way of ensuring consistency in boundary setting and assessment across all European Member States. It allows the use of different assessment methodologies but ensures consistency and comparability. This is a process undertaken infrequently (at most, once per WFD planning cycle) but is important to ensure comparability.

5.8.1 Aim of intercalibration

Intercalibration is a component of the Water Framework Directive for ensuring that every Member State's ecological quality objectives and assessments of quality against those objectives are consistent across the EU. This ensures that the High-Good and Good-Moderate status boundaries for each biological quality element in each water body category relates to the same quality in each Member State, despite differences in their biotas and assessment methods. It also ensures that their methods comply with the normative definitions in the Directive (Annex V, 1.2).

Different countries have different biological assessment methods because they already had well-established methods before WFD, because of biogeographical differences in their biotas, and because of differences in the environmental pressures as a result of differences in human activities. Intercalibration enabled countries to continue to use and develop their existing methods and avoided the need to develop new methods specifically for the Directive. Another strength of this approach is that both empirical data and opinion can be amalgamated with present-day concepts of ecosystem structure and function.

5.8.2 Technical overview of intercalibration

A short overview is provided with access to some of the key documents.

The intercalibration process described in WFD (Annex V, 1.4.1) and CIS Guidance N° 6 *Towards a Guidance on Establishment of the Intercalibration Network and the Process on the Intercalibration Exercise,*[21] based on a network of sites defining high/good and good/moderate status, was impracticable.

Shortly after intercalibration began, it was realised that alternative methods based on a larger amount of monitoring data were needed. These are described in CIS Guidance N° 14 *Guidance on the Intercalibration Process (2004–2006),*[22] (Figure 1.21). It was not possible to intercalibrate every quality element in every water body category at once. Biological assessment methods were insufficiently developed for many quality elements. Because of this, there have been three phases of intercalibration so far, and work on this continues. Intercalibration also forced changes to be made to the concept of typology (**Section 4.4**), plus refinements to the definitions of the 'reference state' (**Section 5.7**).

Many Member States used intercalibration to help them develop their own national biological assessment procedures, together with the data provided by monitoring. As a result, many countries have modified their biological assessment methods following the first river basin management cycle. Intercalibrating new or revised methods to an existing intercalibration is much simpler, and procedures for doing this are explained in CIS Guidance N° 30 *Procedure to fit new or updated classification methods to the results of a completed intercalibration exercise*[23] **http://ec.europa.eu/environment/water/water-framework/facts_figures/guidance_docs_en.htm**

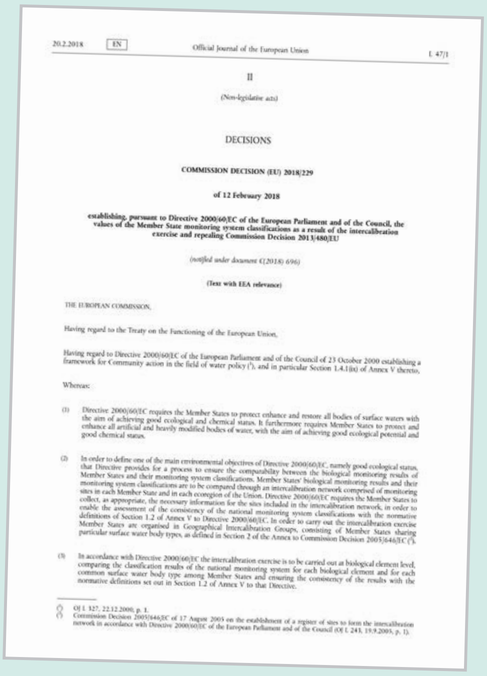
Figure 1.21

Common Implementation Strategy Guidance Documents for intercalibration. CIS Guidance No 14 was produced, detailing previous intercalibration phases and only the version for Phase 1 (2004–2006) is currently available from the Europa website **http://ec.europa.eu/environment/water/water-framework/facts_figures/guidance_docs_en.htm**

The results of intercalibration are described in 'Decisions' published in the Official Journal. These documents list the High-Good and Good-Moderate boundaries for each quality element in each common water body type that has been intercalibrated successfully by each Member State, with a summary of the common intercalibration types and the Member States that have intercalibrated them.

They are issued after each phase of intercalibration has been completed and include the results of previous intercalibration phases that remain in force (ie those that have not been replaced by intercalibrations for revised biological assessment methods). Despite their official nature, they provide a useful summary of the state of intercalibration.

The latest decision document is Commission Decision (EU) 2018/229 of 12 February 2018 (Figure 1.22) 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 Commission Decision 2013/480/EU [24] **http://eur-lex.europa.eu/legal-content/EN/TXT/?qid=1519131448747&uri=CELEX:32018D0229**


20.2.2018 EN Official Journal of the European Union L 47/1

II

(Non-legislative acts)

DECISIONS

COMMISSION DECISION (EU) 2018/229

of 12 February 2018

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 Commission Decision 2013/480/EU

(notified under document C(2018) 696)

(Text with EEA relevance)

THE EUROPEAN COMMISSION,

Having regard to the Treaty on the Functioning of the European Union,

Having regard to Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy (1), and in particular Section 1.4.1(ix) of Annex V thereto,

Whereas:

(1) Directive 2000/60/EC requires the Member States to protect enhance and restore all bodies of surface waters with the aim of achieving good ecological and chemical status. It furthermore requires Member States to protect and enhance all artificial and heavily modified bodies of water, with the aim of achieving good ecological potential and good chemical status.

(2) In order to define one of the main environmental objectives of Directive 2000/60/EC, namely good ecological status, that Directive provides for a process to ensure the comparability between the biological monitoring results of Member States and their monitoring system classifications. Member States' biological monitoring results and their monitoring system classifications are to be compared through an intercalibration network comprised of monitoring sites in each Member State and in each ecoregion of the Union. Directive 2000/60/EC requires the Member States to collect, as appropriate, the necessary information for the sites included in the intercalibration network, in order to enable the assessment of the consistency of the national monitoring system classifications with the normative definitions of Section 1.2 of Annex V to Directive 2000/60/EC. In order to carry out the intercalibration exercise Member States are organised in Geographical Intercalibration Groups, consisting of Member States sharing particular surface water body types, as defined in Section 2 of the Annex to Commission Decision 2005/646/EC (2).

(3) In accordance with Directive 2000/60/EC the intercalibration exercise is to be carried out at biological element level, comparing the classification results of the national monitoring system for each biological element and for each common surface water body type among Member States and ensuring the consistency of the results with the normative definitions set out in Section 1.2 of Annex V to that Directive.

(1) OJ L 327, 22.12.2000, p. 1.
(2) Commission Decision 2005/646/EC of 17 August 2005 on the establishment of a register of sites to form the intercalibration network in accordance with Directive 2000/60/EC of the European Parliament and of the Council (OJ L 243, 19.9.2005, p. 1).

Figure 1.22

The current official intercalibration decision document, covering intercalibration up to and including Phase 3, issued February 2018 [24]

To date there have been three phases of intercalibration within the EU river basins. Details about the specific methods used for intercalibration are provided in technical reports. For Phase 1, there was a technical report for each water body category: lakes, rivers, and transitional and coastal waters. These documents were accompanied by technical annexes that are, unfortunately, no longer available on the web.

For Phase 2, more detailed technical reports have been produced for each quality element in each water body category. All the technical reports for Phase 1 and Phase 2 are published on the EU Law and Publications web pages **https://publications.europa.eu/en/home, search for 'intercalibration technical report** Technical reports for Phase 3 are in preparation.

Figure 1.23

Intercalibration technical reports for Phase 1 intercalibration for rivers and for lakes [26]

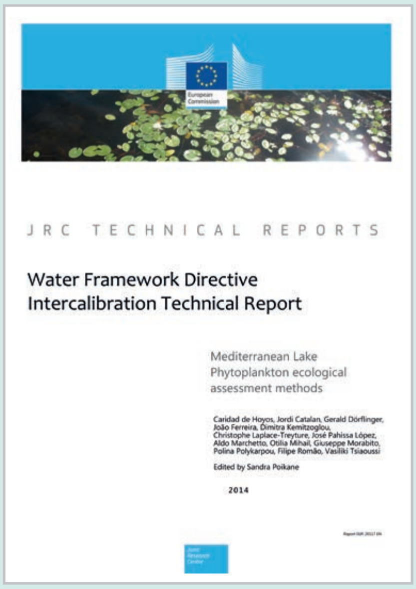


Figure 1.24

Example front cover for a Phase 2 intercalibration technical report [27]

Conclusion

This chapter provides an overview of the key elements of the Water Framework Directive approach to river basin management and the monitoring and assessment needed to provide the information needed for decision making. It also emphasises the technical capability and resource required to undertake this in a consistent way across the UK and the EU.

The WFD approach is a useful model for all river basins around the world. However, modifications to suit local situations will be required and are advocated positively. It is hoped that providing access to the considerable body of work undertaken by the EU and Member States to implement the WFD will speed up the development of new methods and applications. The EU has been promoting this and the **China Europe Water Platform** and the **India – EU Water Partnership** initiatives are two examples of this important knowledge exchange.

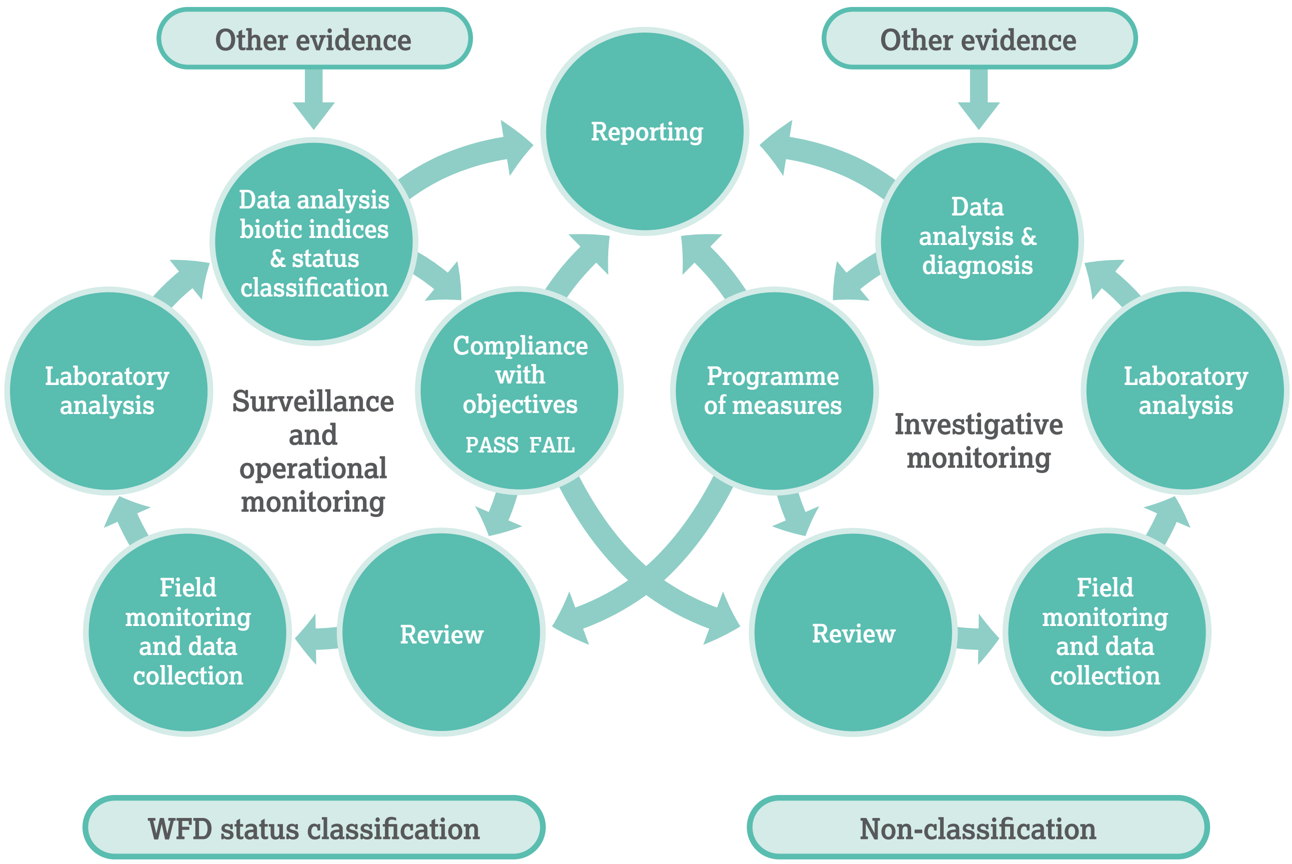
The use of biological and ecological monitoring and assessment has been well established through the development and implementation of the WFD. This has introduced new challenges and issues to overcome. Continued development is required to face the issues of climate change, water shortages and floods. However, changing the input parameters to simulate new scenarios will allow the WFD approach to be adapted to changing situations. Water planning requires a long-term approach. The 30-year horizon of the WFD takes us some way towards this.

To inform the overall structure of this handbook, Figure 1.24 shows a schematic diagram of the monitoring and assessment cycle that underpins the WFD and other monitoring programmes.

To recap, surveillance monitoring is focused at identifying long-term changes, and through this the state of the environment can be assessed against the objectives set: known as compliance assessment. Closely linked, the operational monitoring focuses on risk, to derive information relating to improvement programmes. Investigative monitoring is used to understand issues such as failures against objectives or accidental pollution impacts. Together they should make up a balanced monitoring and assessment programme that optimises scarce resource.

Schematic diagram of the monitoring and assessment cycle that underpins the WFD and other monitoring programmes

Figure 1.25
Schematic – Monitoring, Assessment and Reporting Cycle



Red circles drawn on each element of the diagram will assist in following the monitoring and assessment process throughout the handbook.

The information provided in this handbook is focused on river invertebrate monitoring and assessment. Other methods are important and complementary.

Fish, diatom and macrophyte techniques are well advanced. The underlying models are generally modified from the invertebrate assessment techniques described here. These methods are beyond the scope of this handbook at this time. It is hoped that additional chapters can be added later.

Chapter 2

MACROINVERTEBRATE MONITORING AND ASSESSMENT METHODS FOR WFD

1

FOCUS ON STANDARD INVERTEBRATE DATA COLLECTION THAT IS USED FOR RIVER WATER BODY STATUS CLASSIFICATION

Chapter 2 focuses on standard invertebrate data collection that is used for river water body status classification. It provides detailed information on methods for field and laboratory work. This includes sampling, laboratory assessment and enumeration methods, field data collection and storage.

It broadly follows the monitoring cycle below, focusing on the elements circled in red:

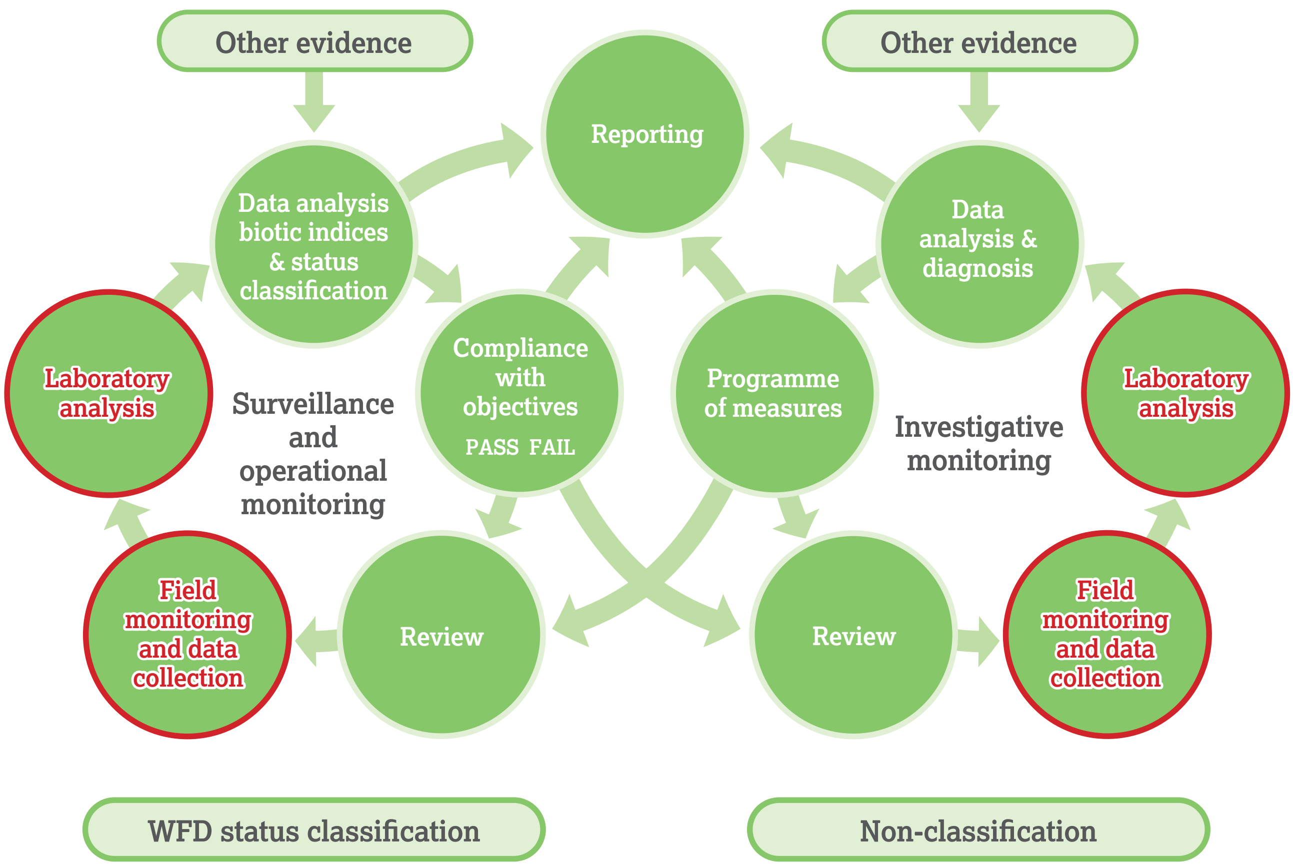


Figure 2.1
The monitoring and reporting cycles, showing the broad elements covered by Chapter 2, circled in red

All European aquatic monitoring programmes are based on common principles described in this chapter, but may vary to meet the needs of the individual country's approaches and methods. This chapter focuses on the UK's standard **RIVPACS (River Invertebrate Prediction and Classification System)** methods for invertebrate sampling, field data collection and laboratory analysis.

As an example of an international approach, a section has been included to cover the **STAR-AQEM method**. Both methods are widely used across Europe. (STAR is the **Sta**ndardisation of **R**iver Classifications. AQEM is **A**ssessment System for the Ecological **Q**uality of Streams and Rivers throughout **E**urope using Benthic **M**acroinvertebrates.)

Invertebrate status classification has a higher degree of refinement compared to other biological elements, having been in continuous development since the 1970s. This was partly as a result of the greater utility shown by invertebrates, and is also the reason for their greater prominence in monitoring compared to fish, algae and macrophytes.

Types of monitoring

WFD provides a useful terminology for the different types of monitoring undertaken for environmental management.

Surveillance monitoring

Surveillance monitoring is used to assess long term changes in the environment due to natural and widespread anthropogenic activity and to inform the efficient and effective design of future monitoring programmes and to validate the impact assessments used for characterisation. Surveillance monitoring provides an overall assessment of quality within whole catchments or sub-catchments and not individual water bodies.

Operational monitoring

Operational monitoring is used to confirm the status of water bodies at risk from known pressures and to assess the efficacy of programmes of measures.

Investigative monitoring

Investigative monitoring is used to identify the causes of poor environmental quality (diagnosis) and their timing and source so that an appropriate programme of measures can be implemented to restore quality.

Status classification is the main outcome from surveillance and operational monitoring, and also for some types of investigative monitoring. Status classification requires standardised methods to ensure consistency and reliability of reporting. The same sampling and data collection methods are used for many types of monitoring, but the frequency of monitoring and the data handling is different for each type.

Chapter 3 focuses on data handling and status classification methods, which will be considered in the field and laboratory methods and monitoring programme design. The data handling and classification is totally dependent on the quality and frequency of the field data. For this reason, monitoring methods must be tailored to the ultimate use of the information. In addition, quality assurance and staff training are essential to gain consistency in monitoring and this is addressed below.

This handbook does not provide species or genus identification or detailed taxonomic information. These can be found in numerous identification guides and keys. Most are specific to countries or eco-regions. We would expect field and biological laboratory staff to be trained to identify key invertebrates in their locality.

2

DESIGN OF INVERTEBRATE MONITORING PROGRAMMES

Most biological monitoring in the UK is based on the RIVPACS sampling and laboratory methods described here. It is provided as an example of a national invertebrate monitoring method. Similar biological monitoring methods are used across the EU and in many other countries. The principles are the same and can be adapted to suit the various aquatic habitats across the world.

The requirements for investigative, operational and surveillance monitoring are different (see Box above), so they should be planned as separate activities. Monitoring should be optimised to ensure efficient sampling and data collection, so samples may be used for more than one purpose, but data must be marked to identify each driver to prevent bias in classification. However, opportunities to collect samples for multiple purposes without biasing results are limited. For example, high frequency sampling during an investigative monitoring exercise in response to a short-term pollution event can significantly bias surveillance or operational monitoring results and result in an unrepresentative classification.

Because they rely on invertebrate status classification, surveillance and operational monitoring programmes, and some investigative monitoring programmes, involve the collection of samples in spring and autumn **(Section 4)**. New monitoring sites are also surveyed in summer, but only for the collection of environmental data **(Section 7.6)** so that the annual average values can be calculated for RIVPACS.

The location of sampling sites may be different for each type of monitoring. Sites that are representative of water bodies are selected for local surveillance, operational and some investigative monitoring, but must always comply with the more general requirements for RIVPACS **(Section 5)**. Operational and investigative monitoring sites are usually located at the downstream ends of water bodies to detect upstream pressures. Investigative or operational monitoring to monitor major discharges or abstractions usually involve both downstream and matching upstream control sites. However, in surveillance assessments, this bias can lead to an under-representation of smaller streams and the environmental pressures that they face.

Example of potential sampling bias – avoid poor quality bias

In 2020, the Environment Agency in England established a completely new River Surveillance Network, sampling for which started in 2021. It did this because its monitoring was increasingly targeted at investigative and operational monitoring to help its management activities. The resulting risk-based programme was inevitably biased to water bodies where there was poor quality. This bias led the organisation to conclude that it needed a completely separate and unbiased monitoring network to assess overall quality and changes in quality at a national and regional scale.

Example of potential sampling bias – under-representation of small water bodies

The Environment Agency 2021 River Surveillance Network in England includes small streams and streams that are intermittent (ie do not always flow), but not obviously artificial drainage ditches. These are important habitats that are usually beyond current monitoring. It also includes sites influenced by local pressures, such as bridges that are avoided when locating sites to represent whole water bodies. This corrects the potential bias identified above. It still excludes spring zones which require different sampling methods.

Surveillance networks are best designed using procedures that give a random design in order to avoid bias. The Environment Agency's River Surveillance Network is based on a truly random GRTS (Generalized Random Tessellation Stratified) design.

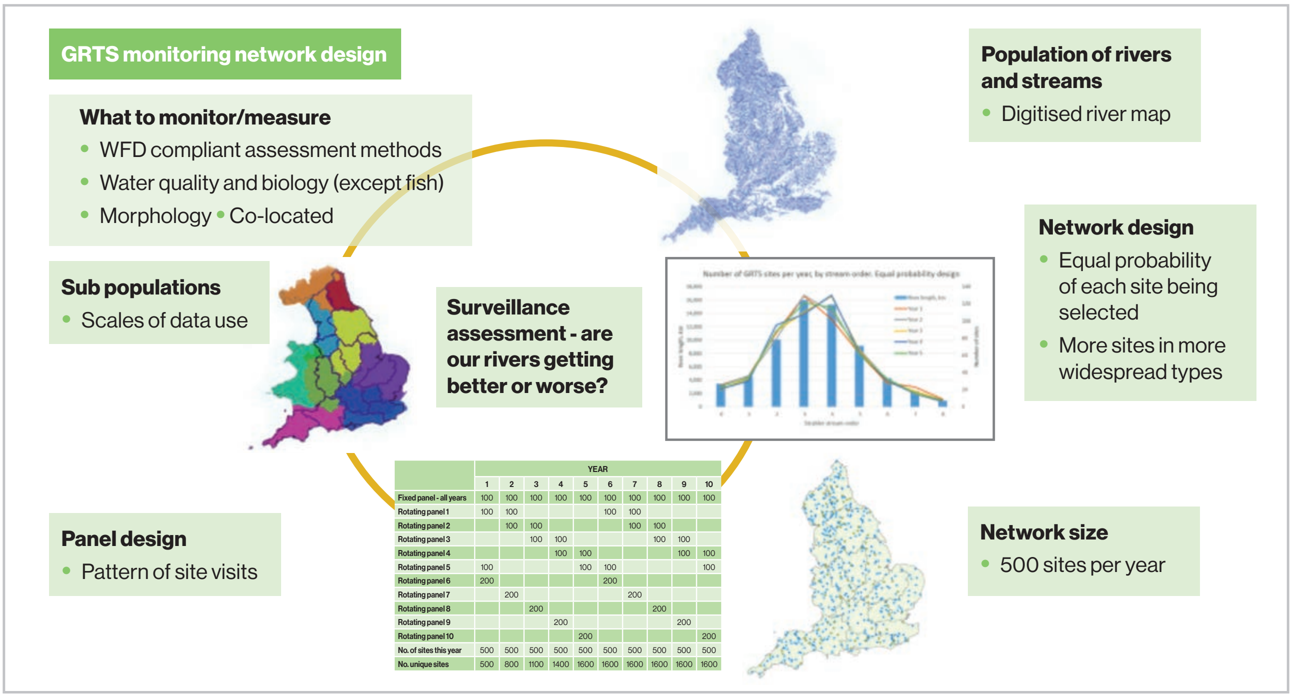


Figure 2.2

An overview of the GRTS monitoring network design of the Environment Agency's River Surveillance Network

Source: Environment Agency Training Slide (unpublished)

Table 2.1

The Environment Agency's River Surveillance Programme design: a fixed panel of sites are monitored every year; a rotating panel of sites are sampled for two consecutive years every 5 years and another rotating panel of sites is sampled once in every 5 years.

	YEAR																			
	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20
Fixed panel - all years	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100
Rotating panel 1	100	100				100	100				100	100				100	100			
Rotating panel 2		100	100				100	100				100	100				100	100		
Rotating panel 3			100	100				100	100				100	100				100	100	
Rotating panel 4				100	100				100	100				100	100				100	100
Rotating panel 5	100				100	100				100	100				100	100				100
Rotating panel 6	200					200					200					200				
Rotating panel 7		200					200					200					200			
Rotating panel 8			200					200					200					200		
Rotating panel 9				200					200					200					200	
Rotating panel 10					200					200					200					200
No. of sites per year	500	500	500	500	500	500	500	500	500	500	500	500	500	500	500	500	500	500	500	500
No. of unique sites	500	800	1100	1400	1600	1600	1600	1600	1600	1600	1600	1600	1600	1600	1600	1600	1600	1600	1600	1600

Investigative surveys that do not rely on status classification can happen at any time of the year and do not necessarily rely on samples collected in spring and autumn. Pollution, drought, floods, and other perturbations can happen at any time of the year so investigations to assess their impact cannot be planned. The operational and investigative monitoring programmes vary each year and sites are sampled depending on the requirements of investigations, or risk of water bodies failing their environmental quality objectives.

The Environment Agency's River Surveillance Network also changes from year to year, but in a pre-planned way that includes a combination of new and previously surveyed sites to balance the need to measure change and have a widespread coverage (see Table 2.1).

The Environment Agency's operational monitoring is managed by local Area teams responsible for managing individual catchments and water bodies, but its River Surveillance Network is managed nationally.

3

RIVPACS FIELD AND LABORATORY METHODS

The RIVPACS method is the standard sampling and assessment methodology for river invertebrate monitoring in the UK.

Variations of it are also used to collect samples from canals, ponds, and lake margins.

RIVPACS is an ecological model for predicting the invertebrate community that you would expect in a sample from any permanently flowing river or stream in the UK in its minimally impacted state.

RIVPACS is an integral part of the UK's river invertebrate status classification for the WFD.

Detailed sampling and laboratory methods have been developed to optimise its practicality, reliability, and performance.

It is an efficient method that provides sufficient precision for enforcing statutory biological quality objectives.

It is critical that any samples to be analysed by RIVPACS, including any used to determine river invertebrate status class, are collected in strict accordance with RIVPACS protocols.

Key documents describing RIVPACS sampling methodologies

For shallow rivers, the key methodology was published by the Freshwater Biological Association (FBA) and the UK Environment Agency. [28] It was issued with RIVPACS III software but has been out of print for some time although it is still cited widely.

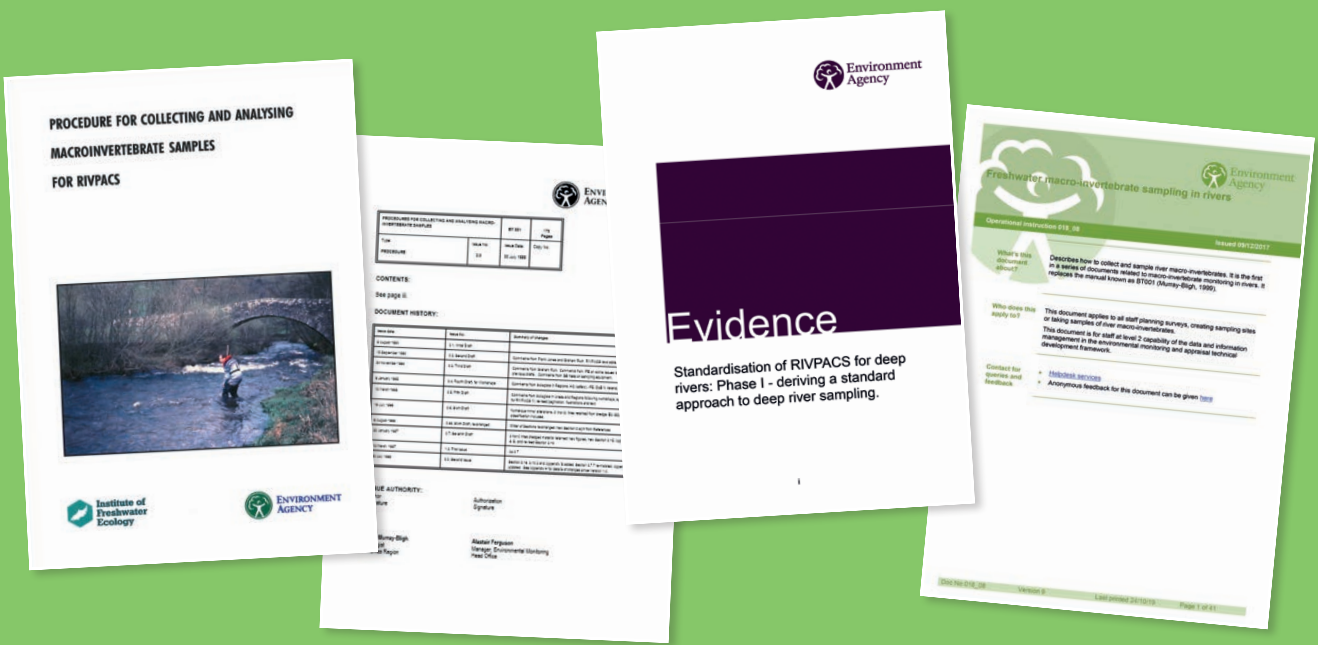
Updates of that document formed the basis of the Environment Agency's manual for collecting and analysing river invertebrate samples, commonly known as BT001, and that is still available. [29]

Deep water monitoring is more difficult because of shortcomings in the equipment available. As a result, RIVPACS sampling in deep water was fully standardised only recently, the Environment Agency adopting it in 2017. This followed a number of large research projects culminating in the final recommendations in a report by Davy-Bowker *et al.* (2014). [30]

An example of operational guidance for field staff can be found in the UK Environment Agency Operational Instruction documents. These are updated frequently and versions since 2017 cover the latest standard methods. It is the most recent detailed description of RIVPACS sampling, and it can be referred to if more detailed information is required. Copies are available from the user guide page of the RICT2 web pages: **https://www.fba.org.uk/rivpacs-and-rict/rict-rivpacs-user-guides**

Videos about RIVPACS sampling can be found in the user guide section of the RIVPACS/RICT2 web pages at: **https://www.fba.org.uk/rivpacs-and-rict/rict-rivpacs-user-guides**

RIVPACS - Key reference documents - front covers



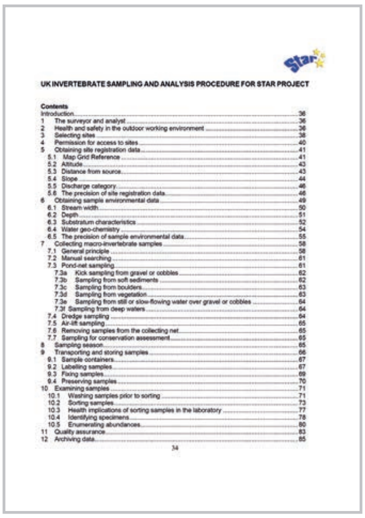
An extract from BT001, mentioned above, was published on the 'Protocols' page of the STAR-AQEM project website **http://www.eu-star.at/**. This only covers the shallow-water sampling protocols for RIVPACS.

Davy-Bowker *et al*. (2014) complements the shallow water method available from the STAR website [30] and is available from the FBA's website, together with a comprehensive review of deep water methods **https://www.fba.org.uk/rivpacs-and-rict/rivpacs-rict-resources**

The standard RIVPACS sampling method ensures that the results of surveys are comparable, even when RIVPACS itself is not used to analyse the results.

As well as being used for monitoring by the UK's statutory agencies and those of many other countries, the RIVPACS sampling protocol is used for complementary monitoring activities undertaken by other organisations and interested groups. Increasingly, citizen science and stakeholder contributions add to the overall riverine knowledge base. **Chapter 4** examines these and other initiatives in more detail.

The RIVPACS standardisation protocols have provided a useful baseline, allowing comparable samples to be taken from almost all riverine environments. It has helped the development of numerous biotic indices sensitive to different environmental pressures.



UK INVERTEBRATE SAMPLING AND ANALYSIS PROCEDURE FOR STAR PROJECT

Contents

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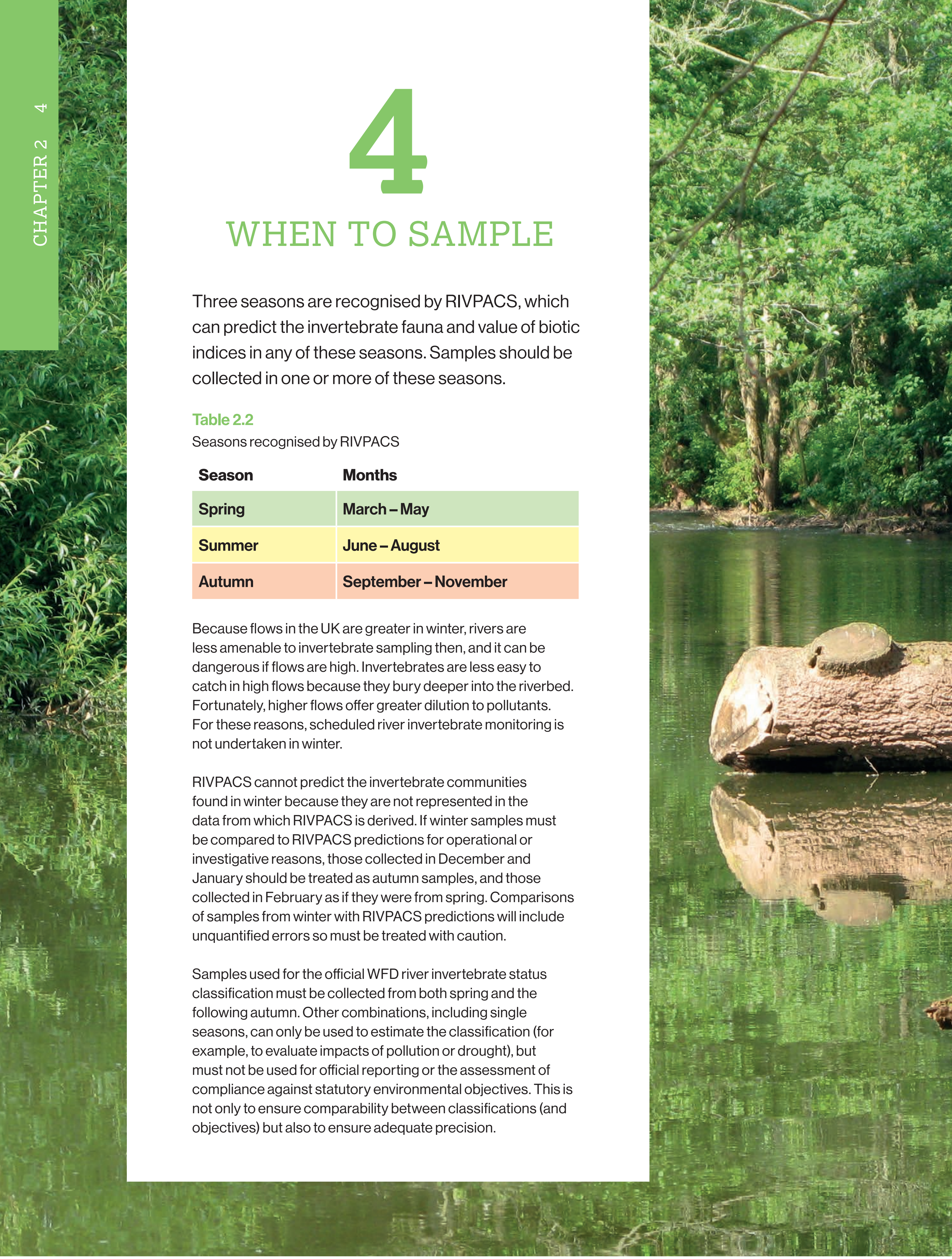


4

WHEN TO SAMPLE

Three seasons are recognised by RIVPACS, which can predict the invertebrate fauna and value of biotic indices in any of these seasons. Samples should be collected in one or more of these seasons.

Table 2.2
Seasons recognised by RIVPACS

Season	Months
Spring	**March – May**
Summer	**June – August**
Autumn	**September – November**

Because flows in the UK are greater in winter, rivers are less amenable to invertebrate sampling then, and it can be dangerous if flows are high. Invertebrates are less easy to catch in high flows because they bury deeper into the riverbed. Fortunately, higher flows offer greater dilution to pollutants. For these reasons, scheduled river invertebrate monitoring is not undertaken in winter.

RIVPACS cannot predict the invertebrate communities found in winter because they are not represented in the data from which RIVPACS is derived. If winter samples must be compared to RIVPACS predictions for operational or investigative reasons, those collected in December and January should be treated as autumn samples, and those collected in February as if they were from spring. Comparisons of samples from winter with RIVPACS predictions will include unquantified errors so must be treated with caution.

Samples used for the official WFD river invertebrate status classification must be collected from both spring and the following autumn. Other combinations, including single seasons, can only be used to estimate the classification (for example, to evaluate impacts of pollution or drought), but must not be used for official reporting or the assessment of compliance against statutory environmental objectives. This is not only to ensure comparability between classifications (and objectives) but also to ensure adequate precision.

5
SITE SELECTION

The following criteria must be considered when selecting monitoring sites:

Representative sites

It is important that monitoring sites are representative of the intended monitoring target. Sites for assessing and managing the quality of a whole reach or water body must be well away from the influence of local disturbances, particularly bridges, fords and other structures, and outside the mixing zones of discharges or tributaries, because the influence of these local pressures may mask the signal from the rest of the water body. However, where the site is one of a random set of sites for assessing quality over a wider scale, such as the Environment Agency's River Surveillance Network, it is important not to exclude such sites because that would bias the evaluation. The River Surveillance Network sites are not used individually but together to assess overall quality at a much wider scale. They must therefore cover all impacts, whether local or wide-scale. The network is designed to be representative of the whole river network, hence its truly random site selection.

Isolated habitats

Physically isolated habitats should be avoided because they support fewer taxa than extensive areas of the same habitat. Their reduced diversity is likely to be misinterpreted as an impact of environmental disturbance. For example, an isolated shallow gravelly riffle would be unsuitable, even though it is easier to sample and supports more taxa. If the river is mostly deep and wide with a silt bed, the sampling site should also be deep and wide with a silty bed. This can be another reason for avoiding some bridges because the stream bed near them may comprise rubble from previous structures, and because bridges are often built where rivers are particularly narrow or shallow. Our assessment methods, including RIVPACS, are not able to distinguish isolated from extensive habitats and isolated habitats are not included in the RIVPACS reference database.

Suitability for the RIVPACS model

If data from the site is to be analysed by RIVPACS, for example if it is to be classified, it must be aligned with the parameters of the original model. The average value of environmental predictor variables such as width and depth must be representative of the site as a whole.

RIVPACS is suitable only for permanently flowing streams that flow above ground and downwards to the sea. Standardised sampling methods for temporary and intermittent streams are still under development, particularly for the terrestrial phases. Methods for sampling from the dry regions of intermittent streams, very small streams, springs, and underground streams are given in **Chapter 4**. The RIVPACS sampling methods described here can be used for canals, ditches, drainage dykes and tidal streams but they are unsuitable for analysis by RIVPACS and therefore their WFD invertebrate status cannot be classified.

The survey area and sampling area

The sampling site should be in a survey area with consistent features to ensure that a minor error in re-locating a site does not upset the monitoring result. This also helps to ensure that the site is not in an isolated habitat.

The survey area must be seven channel widths, up to a maximum of 50 metres, either side of the sampling area (Figure 2.3). This reduces differences between samples taken on different occasions that may not be in the exact location sampled previously. Surveys that need more intensive sampling, for conservation or those which require replicate samples, can also use this larger area.

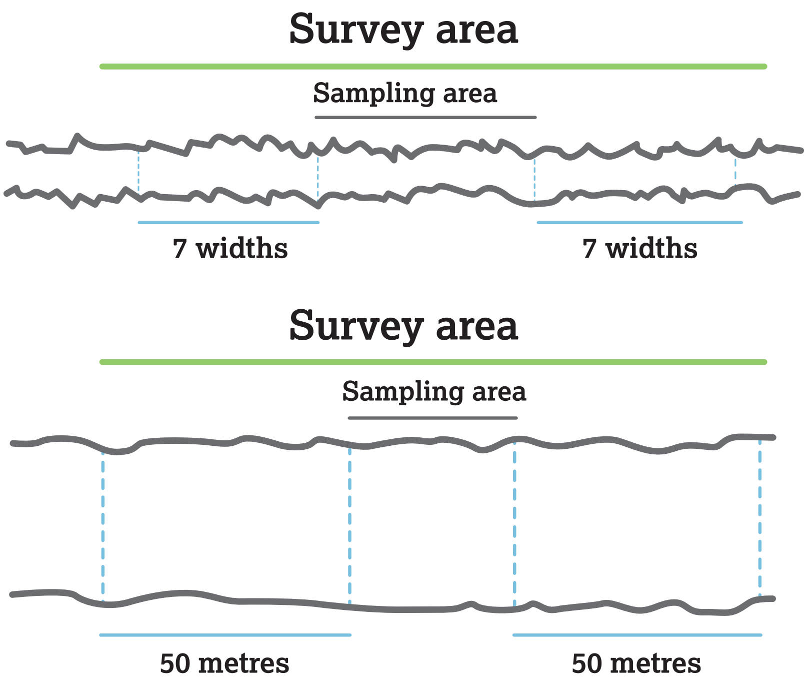


Figure 2.3
The sampling area should be centred in a survey area with the same characteristics and extending 7-widths (in a narrow stream) or up to 50 metres (in a wider stream)

6

GENERAL PRINCIPLES OF RIVPACS SAMPLE COLLECTION

For RIVPACS, all habitats in the sampling area should be sampled in proportion to their cover. This multi-habitat *pro rata* approach enables comparable samples to be collected from any type of river, regardless of the habitats present. Sampling methods used in other countries for WFD status assessment also follow this approach.

Table 2.3
Training requirements for RIVPACS sample collection

Knowledge required for RIVPACS sample collection
Identification of invertebrate habitats
Knowing how much material to discard in the field to reduce laboratory work without compromising the integrity of the sample – knowing what to look for in the net
Recognition of species collected in the search
Understanding when you have sampled a particular habitat sufficiently and effectively
Identifying rare species that are to be recorded in the field and returned to the stream
Understanding the protocols for RIVPACS sampling, including collection of environmental parameters


Ecdyonurus **sp. nymph**
John Davy-Bowker


Hydrometra stagnorum **female**
John Davy-Bowker

Freshwater Biological Association Training

RIVPACS sampling is based on catch-per-unit effort and is appropriate for semi-quantitative analysis. Although suitable for assessing environmental quality and community composition, it is not suitable for quantitative assessment of population sizes.

- Because this approach requires samplers to identify all the potential invertebrate habitats at any site, they must have a good understanding of invertebrate ecology. They also need to understand the practicalities of laboratory analysis and the RIVPACS model for which the samples are collected. The level of knowledge required for RIVPACS sample collection should not be underestimated because all other analysis depends on the quality of sample collection and its adherence to the strict protocols.

- For RIVPACS, the sample is standardised by the time spent actively sampling. In shallow streams this is 3 minutes, but in deep rivers it is 3 minutes in the main channel and 1 minute sweeping the margins. An additional minute is always spent searching for and collecting individual specimens of attached or surface-dwelling taxa. The 3 minutes for the main sample are allocated to different habitats in proportion to their cover. Where it is not possible to see different habitats, in deep or turbid water, samples are collected along transects across the river, covering shallower areas nearer the banks and the deeper mid channel.

- RIVPACS sampling is based on a 1 mm mesh net. The net is a critical component for standardisation that is common to all the sampling equipment. It collects invertebrates of a sufficient size for effective laboratory sorting and identification and is less prone to blockage than finer meshes, which helps its efficiency in the field. Most other European national methods use a 500 µm mesh. The net must be made of soft-woven multifilament polyester. This is much easier to repair than hard monofilament nylon nets, which are also difficult to empty. Because the size of the mesh is critical, the condition of the nets is important and damaged nets must not be used. Always take spare nets with you when collecting samples.

- RIVPACS sampling is quicker and therefore cheaper than many other methods. One of its main advantages is that its limitations are well known and documented. There is good quantitative information about its precision, with estimates of sampling variation based on extensive replicate sampling tests. There are also robust quality assurance procedures for laboratory analysis and estimates of its errors. This means that, although apparently imprecise sampling methods are used, statistically significant differences between samples from those caused by random error can be distinguished.

- Smaller, 1-minute samples are fine for detecting gross impacts. However, the benefits of the time saved in laboratory analysis must be balanced against their incompatibility with standard samples and their quantified estimates of precision, incompatibility of measures of taxonomic richness or other measures of diversity based on them, the inability to calculate indices that take abundances into account (**Chapter 3**), and their incompatibility with RIVPACS and therefore the official WFD status classification.

7

SELECTING THE APPROPRIATE RIVPACS SAMPLING METHOD

The three-minute hand-net sample is the preferred method for macroinvertebrate sampling in shallow water and is described in **Section 7.2**.

Frequently referred to as a 'kick sample', it also involves sweeping and poking.

Where it is too deep to wade in the water to collect a kick sample, use either:

- **a long-handled pond net (Section 7.3) + marginal sweep (Section 7.5)**
- **an airlift (Section 7.4) + marginal sweep (Section 7.5).**

Do not combine these methods in the same sample. Aim to always use the same method at a site to ensure comparability between samples.

All samples include an additional 1-minute manual search for surface dwellers and animals attached to rocks and other submerged objects **(Section 7.1)**.

Dinocras cephalotes
John Davy-Bowker

THE THREE-MINUTE HAND-NET SAMPLE

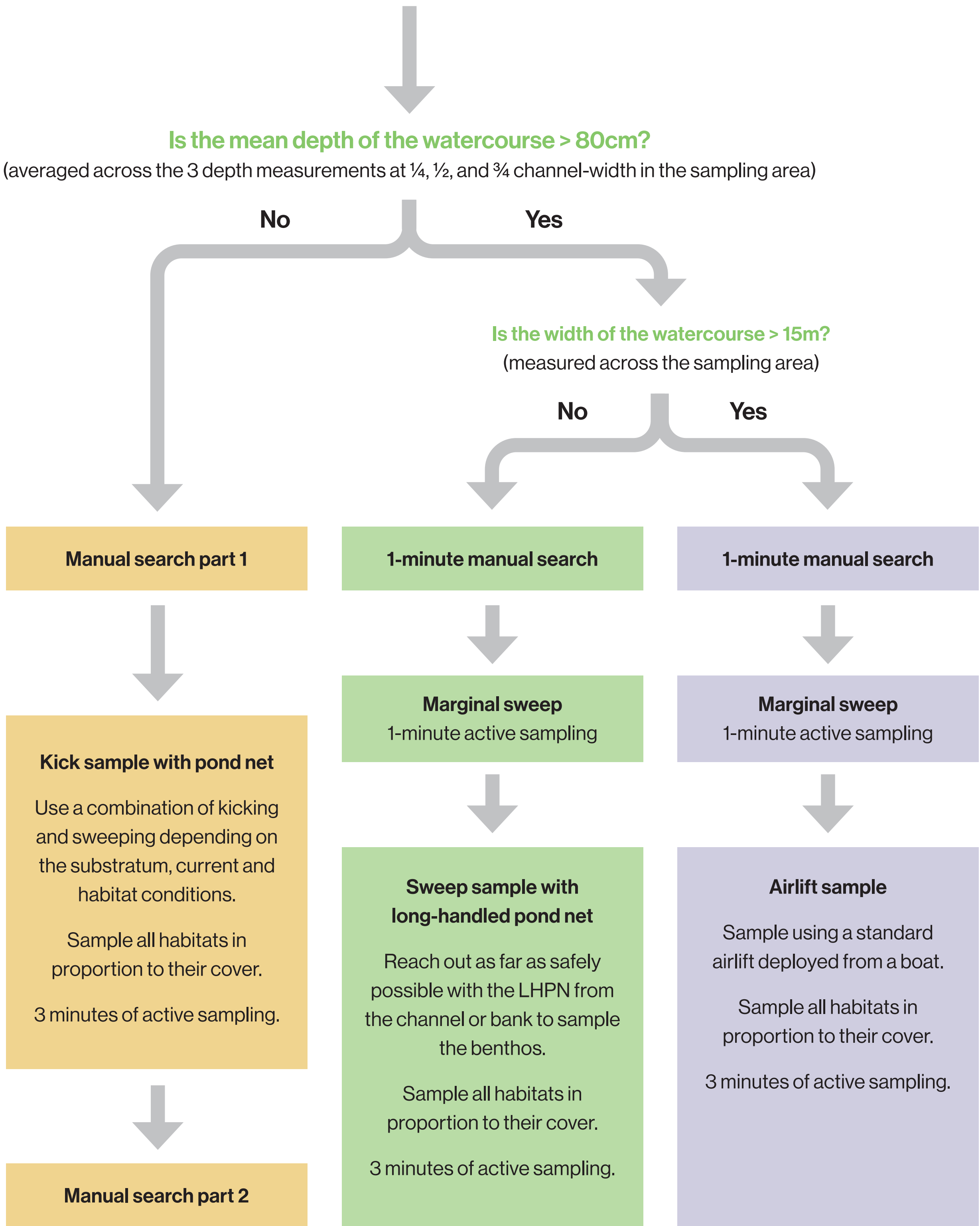


Figure 2.4
How to choose the appropriate invertebrate sampling method for RIVPACS

7.1 Manual search

The 1-minute manual search always accompanies the main sample.

It is done to collect individual specimens of species that are unlikely to be collected in the main sample – in particular, surface-dwelling animals such as pond skaters and water crickets which move away rapidly from any disturbance, and for animals attached to rocks and other firm surfaces – in particular, limpets, blackfly pupae and certain caddis larvae and pupae. The search will not always be fruitful but must always be undertaken.

The 1-minute search may be split into an initial search for surface-dwellers before you enter the stream, as these are very sensitive to disturbance of the water surface, and a subsequent search for attached animals on large stones and other objects from the riverbed.

7.2 3-minute kick sampling from shallow rivers

The aim of kick sampling is to collect as many animals as possible while minimising the removal of gravel and wood from the riverbed.

On stony or gravelly beds, kick the riverbed with the heel of your boot, holding the net a few centimetres away so that animals flow into the net but stones drop before reaching the net. Sweep the surface of mud and amongst vegetation. Wash detritus and vegetation in the net (do not include washing time in the 3 minutes).

Figure 2.5
Kick sampling © *Judy England*

Figure 2.6

Underwater view of kick sampling with net held a short distance downstream in the silt cloud but far enough away for stones to drop before reaching the net *Photo credit: Tim Flood*

The nets used for RIVPACS kick sampling are based on a standard Freshwater Biological Association (FBA)-pattern pond net.

As mentioned earlier, RIVPACS sampling is based on a 1 mm mesh net. The net is a critical component for sample standardisation. It collects invertebrates of a sufficient size for effective laboratory sorting and identification, and is less prone to blockage than finer meshes which helps its efficiency in the field. Most other European national methods use a 500 µm mesh.

The net must be made of soft-woven multifilament polyester. This is much easier to repair than hard monofilament nylon nets, which are also difficult to empty. Because the size of the mesh is critical, the condition of the nets is important and damaged nets must not be used. Always take spare nets with you when collecting samples.

Nets and frames vary slightly between manufacturers, but their basic features should not differ from those described below and in Figure 2.7:

- The frame must have a straight lower edge of 20–25 cm and straight, vertical sides of 19–22 cm.
- Use nets which are 50 cm in depth. They are less easily blocked because of their greater mesh surface.
- The pond net handle should be about 1.5 metres in length.
- Periodically check that the bottom edge of the frame is not bent because this reduces its sampling efficiency. Thin gauge aluminium frames are prone to this type of damage but are easily straightened.

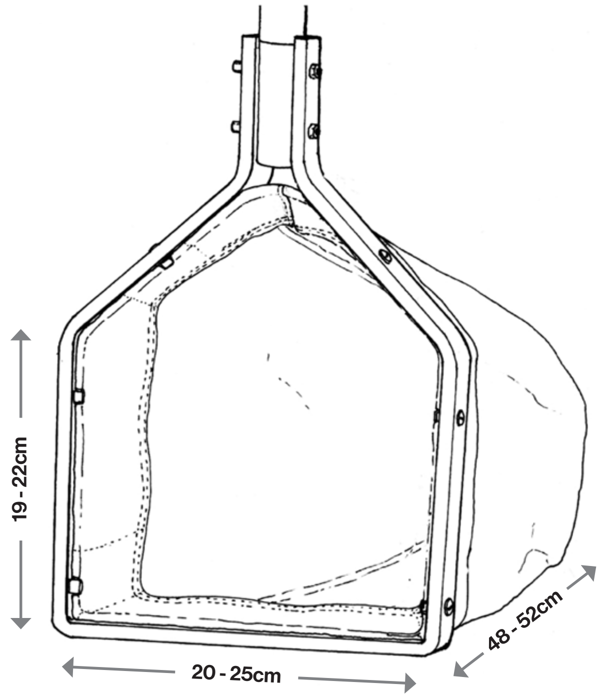


Figure 2.7

Standard pond net for invertebrate sampling

7.3 3-minute sweeping with long-handled pond net from deep narrow rivers

The long-handled pond net is essentially the same as the FBA-pattern pond net, but it has a much longer handle, up to 4 m long, usually in three screw-together sections.

From the bank, reach out into the main channel and sweep the surface while bringing the net back to you. Rotate the net before lifting it so that you do not lose any material that you have collected. Repeat as many times as you can for 3 minutes from different places along both banks.

In silty rivers it is particularly important to ensure good washing of silty samples. Working upstream may reduce silt contamination.

Supplement the long-handled pond net sweep of the main channel with a 1-minute marginal sweep (**Section 7.5**). The search will not always be fruitful but must always be undertaken.

7.4 3-minute airlift sampling from deep wide rivers

Only the Yorkshire-pattern airlift sampler is appropriate for this procedure.

Figure 2.9 shows different Yorkshire-pattern airlifts. The aluminium airlift in the centre is a newer, lightweight version that is easier and safer to use.

Figure 2.8
Long-handled pond net

Figure 2.9
Airlift samplers. The silver model is a lightweight aluminium version with low pressure air feed developed for use by the Environment Agency.

On loose riverbed substrates, leave the air flowing and move the airlift continuously across the riverbed.

On more compacted substrates, sample in a series of short bursts in different locations by turning the air supply on and off and bouncing the airlift to help disturb the riverbed. Whichever method you use, you must aim to sample the habitats present in proportion to their cover.

Supplement the airlift sample from the main channel with a 1-minute marginal sweep (**Section 7.5**).

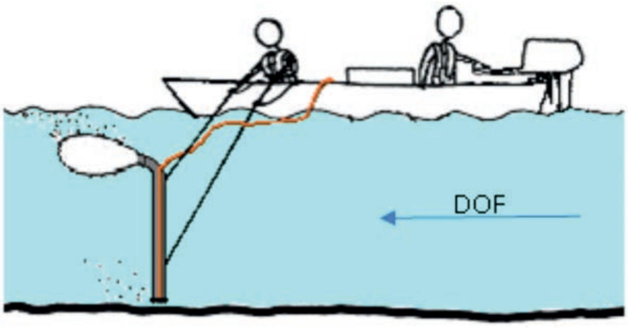


Figure 2.10

Using an airlift. DOF = direction of flow

Figure 2.11

Using an airlift in the field

7.5 Marginal sweep

A 1-minute marginal sweep using a standard pond net always accompanies a 3-minute sweep with a long-handled pond net or a 3-minute airlift from sites that are too deep to wade.

The marginal sweep is for sampling the shallow margins including emergent vegetation that is likely to be missed in the main sample, particularly from airlift samples. It can be done from the banks, but in large deep rivers where an airlift is used, it may be more effective to collect the sample from a boat. Ideally, the margins of both riverbanks should be sampled. The 1 minute should be divided between the different marginal habitats according to their cover.

7.6 Environmental measurements for RIVPACS

RIVPACS samples must always be accompanied by certain field measurements that are used by the predictive model, all of which are annual averages. These parameters must be measured with every sample as they are also useful more generally for interpretation.

Width	Average of three measurements taken across different places within the sampling area
Depth	Average of three measurements taken across the sampling area
Substrate	Estimate of the percentage cover of different size particles, excluding bedrock: **• Clay & silt • Sand • Gravel & pebbles • Cobbles & boulders**

Table 2.4
Substrate categories used for RIVPACS

Substrate Category	Width (millimetres)	Description
Clay	<0.06	Sticky and cohesive
Silt	<0.06	Soft in texture and not abrasive to the hands when rubbed. Not cohesive or sticky.
Sand	0.06–2	Smaller than instant coffee granules and, unlike silt/clay, abrasive to the hands when rubbed
Gravel	2–16	Instant coffee granule to broad bean
Pebble	16–64	Broad bean to half fist size
Cobbles	64–256	Half fist to head size
Boulders	>256	Head size and larger

Geochemistry	One of either alkalinity (preferred but requires laboratory analysis), hardness, calcium concentration or electrical conductivity
Velocity category	Estimate of surface velocity in the main flow, optional if an estimate of discharge category is not available

These parameters should be recorded with every biological sample. If the samples are to be analysed with RIVPACS, it may be necessary to collect these environmental measurements in other seasons in order to determine the annual averages that RIVPACS needs.

Table 2.5

Surface velocity categories used by RIVPACS

Velocity category	Current velocity (metres per second)
1	≤ 10
2	> 10–25
3	> 25–50
4	> 50–100
5	> 100

Table 2.6

Discharge categories used by RIVPACS

Discharge category	Mean annual discharge
1	< 0.31
2	0.31–0.62
3	0.62–1.25
4	1.25–2.50
5	2.50–5.00
6	5.00–10.00
7	10.00–20.00
8	20.00–40.00
9	40.00–80.00
10	> 80.00

RIVPACS also requires environmental data collected from maps, originally 1:50 000 Ordnance Survey, but now more commonly from GIS systems at similar scale:

Ordnance Survey grid reference

Altitude

Distance from source – (the furthest source along the longest tributary from the site)

Slope

Discharge category – from maps or hydrological systems

8

HEALTH AND SAFETY WHEN COLLECTING INVERTEBRATE SAMPLES

Being near water, particularly rivers, is potentially dangerous. Many organisations will have specific health and safety guidelines which should be followed and applied to each situation. The Environment Agency has developed specific guidelines for biological monitoring which may be useful for reference and adaptation. [31]

However, as an overview, when you are collecting samples your attention will be on collecting the sample and you will not be able to pay so much attention to other risks. Use your pond net to check the stability and depth of the bed before you enter the water and to help you keep your balance. Look around you when moving between sampling points at the site. Always wear suitable clothing, bring dry spares, and always wear a life jacket. Be aware of pollution and risks of waterborne diseases, so use bactericidal hand cleaners after every site visit.

> **When working alone always make sure that someone knows where you are and sign in with a home base when you start and sign off when you are finished.**

9

LABELLING

Labels should be written on the outside of sample containers, including bags, using a waterproof marker pen.

There is always a risk that labels on the outside of containers could get lost or damaged. To prevent this, waterproof labels written in pencil or alcohol-resistant waterproof ink must be placed inside every biological sample container. Note, very few inks are alcohol resistant.

River Piddle (Site P4) at Hyde SY 86475 90639 15 JUL 2015 *Austropotamobius pallipes* (Native Crayfish) (Specimen returned to river) *John Davy-Bowker*

10 COLLECTING FIELD DATA

Invertebrate samples should always be accompanied by observations made in the field to help interpret the invertebrate data, in addition to the field measurements necessary for RIVPACS (**Section 7.6**). This includes observations about the presence of rare or invasive species, the physical structure of the site and surrounding land use, the habitats present, any indicators of pollution or physical degradation and any difficulties collecting the sample, particularly if they could affect its quality.

Photographs are most useful and should always be taken when invertebrate samples are collected. Concentrate on the surroundings to put the site into context when photographing sites. Photographs that concentrate on the water surface without the surroundings do not convey much information. Underwater photographs taken with waterproof cameras on the end of a pole (such as a pond net handle) that show the condition of the riverbed are also particularly useful. These photographs should be stored electronically to help interpret any changes observed in the river invertebrate data. Photographs are always useful and are often vital in reports.

Larger rare species of invertebrates must be recorded in the field and returned to the site immediately. This includes animals such as the medicinal leech, native white-clawed crayfish, and pearl mussels. Amphibians and fish should also be recorded and returned immediately.

Water chemistry measurements taken by simple hand-held meters are also useful, particularly pH, conductivity, temperature, and oxygen concentration. Electronic field meters must be calibrated periodically, to ensure that they remain accurate.

This supplementary environmental information is useful for interpreting your results, even if it is not to be analysed using RIVPACS tools.

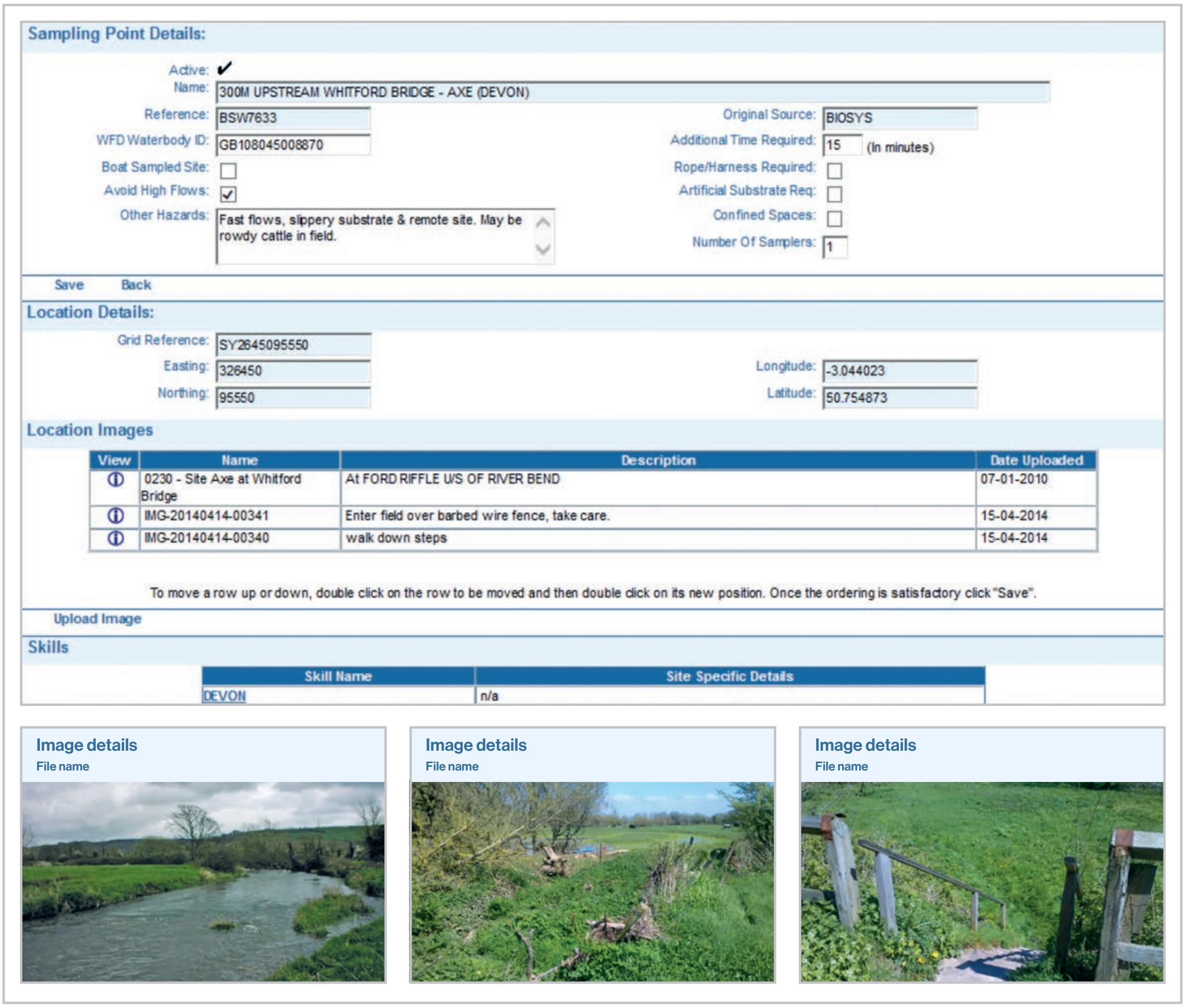
10.1 Site description form

Site description forms are useful for recording information and to help surveyors locate sampling sites.

Photographs are particularly useful for this, and if a site is hidden, for example, behind buildings, it is helpful to include photographs from the road or parking place showing the route to the site, even though they don't show the site itself. Photographs of the sampling site must include its surroundings.

The forms should also include information about health and safety issues at the site, who to contact to gain access and their contact details, what type of sampling or other equipment may be needed, and the presence of alien invasive species requiring additional biosecurity measures. Copies of these forms should be used when sampling, but ensure the original documents are safe in the laboratory if they are not stored electronically.

An example of a form for a biological site from the Environment Agency's electronic Monitoring Site Information System is shown in Figure 2.11, including three images stored on the system to help samplers find the site and to show its characteristics. Information about site contacts and parking is also provided but is not shown here in the figure for reasons of confidentiality. Sketch maps are also stored for some sites.



Sampling Point Details:

Active: ✔
Name: 300M UPSTREAM WHITFORD BRIDGE - AXE (DEVON)
Reference: BSW7633
WFD Waterbody ID: GB108045008870
Boat Sampled Site: ☐
Avoid High Flows: ☑
Other Hazards: Fast flows, slippery substrate & remote site. May be rowdy cattle in field.
Original Source: BIOSYS
Additional Time Required: 15 (In minutes)
Rope/Harness Required: ☐
Artificial Substrate Req: ☐
Confined Spaces: ☐
Number Of Samplers: 1

Save Back

Location Details:

Grid Reference: SY2645095550
Easting: 326450
Northing: 95550
Longitude: -3.044023
Latitude: 50.754873

Location Images

View	Name	Description	Date Uploaded
ⓘ	0230 - Site Axe at Whitford Bridge	At FORD RIFFLE U/S OF RIVER BEND	07-01-2010
ⓘ	IMG-20140414-00341	Enter field over barbed wire fence, take care.	15-04-2014
ⓘ	IMG-20140414-00340	walk down steps	15-04-2014

To move a row up or down, double click on the row to be moved and then double click on its new position. Once the ordering is satisfactory click "Save".

Upload Image

Skills

Skill Name	Site Specific Details
DEVON	n/a

Image details
File name

Image details
File name

Image details
File name

Figure 2.11
Example of site information held on the Environment Agency's Monitoring Site Information System (MSIS)

Brachycentrus montanus

10.2 Monitoring field data collection forms

Figure 2.12 is an example field data form used by the Environment Agency. This form includes information that will help the ecologists to interpret the invertebrate sample data and it includes all the field-measured environmental parameters needed for RIVPACS prediction.

ENVIRONMENT AGENCY
South West
Devon

BIOLOGICAL SURVEY – SITE DETAILS **Version 2018**

Watercourse: Site ID: Sample ID:
Site:

Date: Time: Sampler: Grid Ref:

SAMPLE METHOD
Routine ☐ Pollution ☐ Other

PHOTOS

HABITAT INFORMATION

Width (m): **Depth (cm)** R: M: L: Av: **Conductivity (uS/cm)** **DO%**

Flow:	**Detritus:**	**Shading:**	**Turbidity:**	**Odour:**
☐ Dry	☐ None	☐ None	☐ Clear	☐ None
☐ Low	☐ Local(<30%)	☐ Light (<25%)	☐ Slight	☐ Slight
☐ Normal	☐ Widespread (30-60%)	☐ Moderate (25-50%)	☐ Moderate	☐ Strong
☐ High	☐ Extensive (>60%)	☐ Heavy (>50%)	☐ High	

Sewage Litter: None ☐ Local(<30%) ☐ Widespread(30-60%) ☐ Gross(>60%) ☐

LAND USE (indicate Primary or Secondary)

Broadleaf / mixed woodland		Wetland		Rough / unimproved pasture	
Coniferous plantation		Moorland / Heath		Industrial	
Open Water		Scrub		Farm Buildings	
Suburban / Urban		Tall Herbs / Rank Vegetation		Roads & Railways	
Rock & Scree		Tilled land		Parkland & Gardens	
Orchard		Improved pasture			

SUBSTRATE (%)

Boulders (> 256 mm)		Sand (0.06 – 2 mm)	
Cobbles (64 – 256 mm)		Silt	
Pebbles (16 – 64 mm)		Clay	
Gravel (2 – 16 mm)		Bedrock	

Bed Stability Solid ☐ Stable ☐ Unstable ☐ Loose ☐ Soft ☐ Very Soft ☐

BIOLOGICAL INFORMATION (Where S.F / ochre is present record density as Trace, Thin, Thick or Massive)

S.F above stones: None ☐ Local(<30%)........ Widespread(30-60%)........ Extensive(>60%)........
S.F below stones: None ☐ Local(<30%)........ Widespread(30-60%)........ Extensive(>60%)........
Ochre: None ☐ Local(<30%)........ Widespread(30-60%)........ Extensive(>60%)........

MACROPHYTES	BRYOPHYTES	ALGAE
		Benthic Diatoms: Cladophora: Other:
Total % Cover –	Total % cover -	Total % cover

Marginal Channel Vegetation:

FIELD SIGHTINGS

FISH / BIRD / MAMMAL	INVASIVE SPECIES

Comments (pH, odours, additional information etc):

Time since last spate (DIATOMS)
>14 days ☐ <3 days ☐
7-14 days ☐ Not known ☐
3-7 days ☐

Figure 2.12

Example field data collection form. (SF = sewage fungus.)

10.3 Electronic data collection

Hand-held electronic devices are useful for completing site description and field data forms in the field and adding photographs to them. Transcription errors when transferring data from paper forms to databases are avoided, although typos can still happen. Point of entry data validation can warn of errors that are easy to correct while you are still on site. Connected to GPS, they can also record where field staff are, for safety, and can confirm to surveyors and others they have correctly located the sampling sites and the time of sampling. Results can be uploaded and sent back to base in real time. The Environment Agency also uses electronic systems for scheduling sampling runs, so that field staff can use the same devices to check what types of measurements and samples they need to collect at each site, what type of equipment they should take, and their travel route.

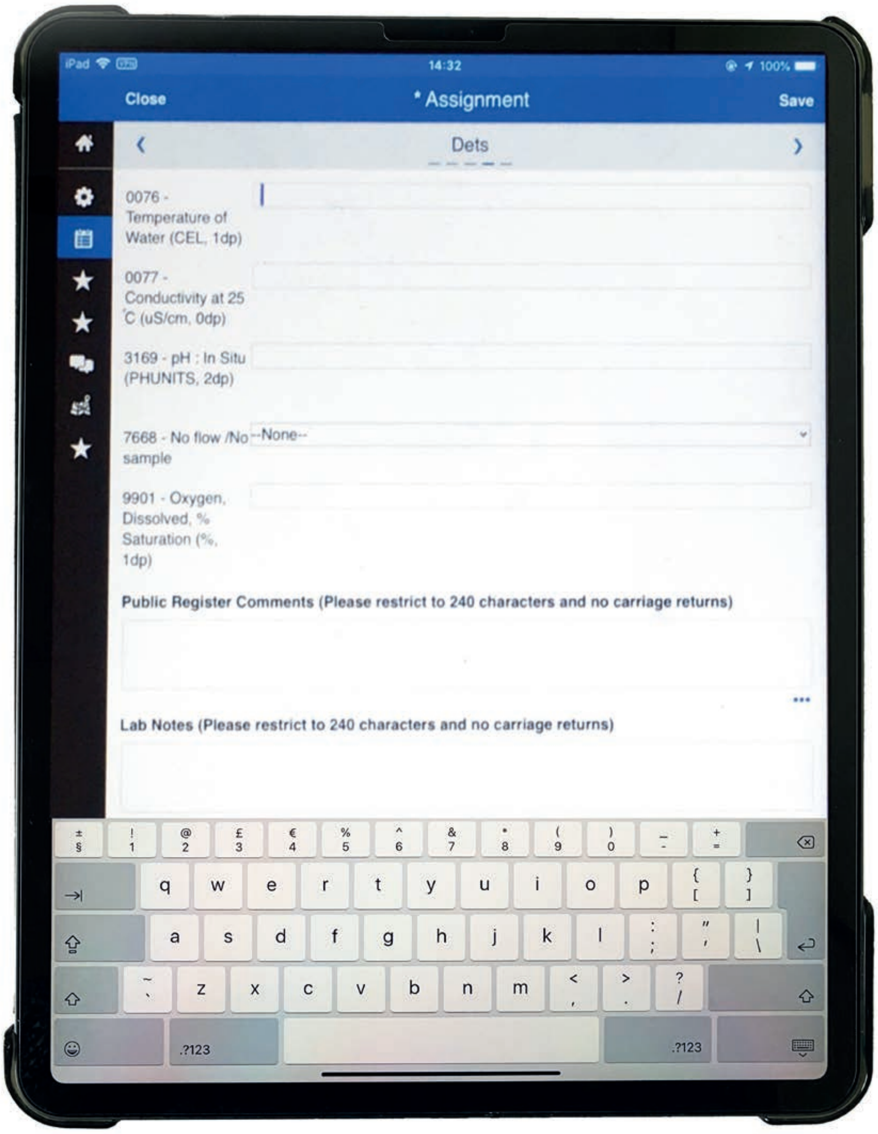


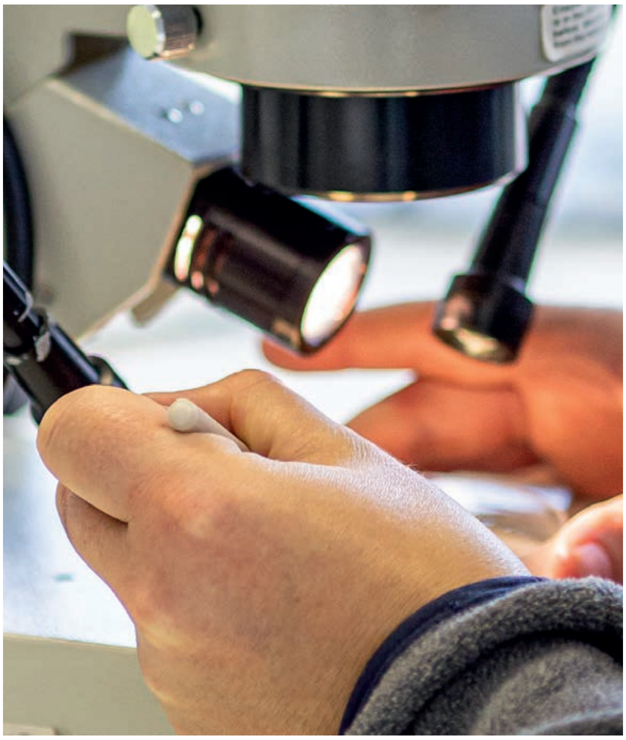
Figure 2.13

Electronic field recording form on iPad tablet computer, as used by the Environment Agency. This example shows a page for recording physico-chemical measurements taken with field meters.

11

LABORATORY VERSUS FIELD ANALYSIS OF INVERTEBRATE SAMPLES

Laboratory analysis includes sieving, sub-sampling, sorting, and identification of animals.



Poor weather and light will affect biological data recorded from field sorting. The main benefit of laboratory analysis is that its errors are well understood and have been quantified. Because of that, samples to be analysed by RIVPACS, particularly those used for WFD status classification, must be sorted and identified in the laboratory under controlled conditions, not in the field.

Measures of laboratory error for laboratory-analysed samples based on independent audit are incorporated in RICT (River Invertebrate Classification Tool). Audits of laboratory analyses of invertebrate samples by regulatory agencies in Great Britain also provide estimates of bias (the impact of non-random error on biotic indices, see **Section 13**) and quantitative information about error. Estimates of bias are incorporated in WFD status classification so that they can be accounted for in estimates of probability of class, and results can be adjusted to take account of variations in analytical quality.

Analysis in the field is less accurate and less precise. It is not suitable for WFD status classification because far more precision is needed to differentiate good from moderate status reliably than is possible with field analysis. However, field analysis is ideal where high precision is not needed, such as rapid screening for gross pollution. It also allows the proportion of living and dead animals to be recorded, which can be important evidence for assessing the impact of pollution. Because of that, samples collected for investigating major pollution incidents are often analysed in the field and again in the laboratory.

Field analysis is most effective when it is used with data analyses that are optimised for field data. These improve efficiency and limit errors by concentrating on key taxa and features that are suitable for field analysis (see, for example, **Chapter 5 Section 3.1.26** – Rapid Appraisal Key for detecting farm pollution). These methods tend to be accurate but less precise than laboratory methods. Biotic indices used for status classification, such as WHPT (Whalley Hawkes Paisley Trigg), are not designed for use with field data and they include taxa that cannot be identified reliably in the field.

Paraleptophlebia submarginata

12
LABORATORY ANALYSIS OF SAMPLES FOR RIVPACS

High-quality biological laboratories are important to ensure consistency of results and a safe and productive working environment for staff. Standards for chemical laboratories have been developed over many years with quality assurance and safe working conditions at the heart of their design and operation. Similar standards for biological laboratories have been established more recently. In general, requirements for biology laboratories are much simpler than for analytical chemistry laboratories because less dangerous reagents are used. Ergonomics, good lighting facilities for washing and sorting samples, and space for using microscopes and identification guides are the main requirements. An overview of biological laboratory methods and management follows.

12.1 Laboratory sample reception

All samples must be dated and recorded upon reception by the laboratory. All information from the sample container label should be included on the sample log. If more than one container has been used, the number of containers should be indicated as well.

12.2 Basic principles

The whole sample must be sorted, even if few animals are found in some samples. The exceptions to these rules are:

Larger rare species must be recorded in the field and returned to the site immediately. This includes medicinal leech, native white-clawed crayfish, and pearl mussels. Amphibians and fish should also be returned immediately.

A field search of discarded material when sub-sampling the contents of an airlift.

Samples collected to investigate pollution incidents may be examined in the field to check for the effects of pollution, in particular the presence of dead animals. Samples analysed in the field are particularly important as evidence in legal cases, usually supported by subsequent laboratory analysis, which benefits from quality assurance measures.

When an immediate, interim assessment of a standard sample is needed, it may be examined in the field, but it must not be altered in any way.

12.3 Preserving samples and specimens

Samples can be sorted and identified live as soon as possible after collection, ideally within 48 hours, including any re-analysis of live samples.

Samples must be stored between 1°C and 3°C. Any live samples not processed within this time or not kept at this temperature must be discarded and new samples taken. Be aware of the risk of predators eating prey, particularly if the sample is from a cold (upland) stream or if it is not kept very cold or is kept for longer periods. Many taxa such as flatworms and leeches are much easier to identify when alive.

Fixatives or preservatives may be used if samples need to be stored or analysis is delayed.

12.3.1 Fixative

The best fixative is 5% aqueous formalin solution. This makes specimens more robust by strengthening proteins so that limbs and other parts are less likely to detach from bodies. Fixation is only needed if specimens are to be kept for more than a couple of months before they are analysed.

NOTE

Formalin is hazardous and must only be used in a fume cupboard.

Many laboratories are not set up to use formalin and it is no longer used by most European environmental protection authorities.

12.3.2 Preservative

Preservatives are used if samples need to be stored. They stop specimens from decomposing by preventing the growth of microbes. An aqueous solution of 70% industrial methylated spirit (IMS) is often used as a preservative. Replace the alcohol a number of times to ensure that there is an adequate final concentration in the sample. 5% glycerol may also be added when storing individual specimens, to reduce the risk of damage should the alcohol dry out.

The organisms should be stored in glass vials filled with ethanol, and plugged with cotton swabs. Once any air bubbles inside the vials are removed, place the vials inside a larger glass container and cover with ethanol. The external container should be sealed tightly.

Specimens for genetic (DNA) analysis, which is not used for standard environmental monitoring assessment, should be fixed in 96% ethanol.

Preserved samples must be stored at cool temperatures, away from any heat source and preferably in the dark to minimise the loss of colour. Whole samples in preservative should be stored away from the laboratory.

12.3.3 Packing and dispatching

If samples or specimens are sent to outside analysts or taxonomists, ensure they are packed robustly to prevent any damage.

Record all sample information in a logbook before dispatching samples.

Figure 2.14

Poorly packaged sample delivered to a laboratory. Logistics companies may refuse to accept samples from laboratories that do not pack them securely and may charge for damages.

12.4 Sieving and sorting

Washing, sieving and sorting should be done in a well-ventilated space, ideally in a fume cupboard or under a fume extractor, particularly if the sample contains preservative or is from a polluted environment.

Samples that have been fixed in formalin must be washed in a fume cupboard.

Figure 2.15

Fume extractor over a sink for sieving invertebrate samples

Figure 2.16
Set of sieves for sorting an invertebrate sample. Left to right: mandatory 500 µm for sorting, advisable 250 µm for catching fine waste to prevent blocking the sink, and optional coarse sieve for separating larger debris and specimens to aid sorting.

12.4.1 Sieving

Before sorting, the sample must be passed through a set of sieves under running tap water to gently rinse out the fine silt. A 1 mm sieve with either a 4 mm or 8 mm sieve above it is recommended, depending on the nature of stream being sampled. The finest sieve must be 500 µm mesh size (half the aperture of the nets used for sample collection) and anything retained on it is considered to be a part of the sample. Coarser sieves are merely to help sorting but the 500 µm sieve is critical to the procedure. It is advisable to place a 250 µm mesh sieve below it, to prevent drains without silt traps from becoming blocked by silt, but material on this sieve is not part of the sample. The condition of the sieves is not as critical for washing invertebrate samples as it is for sediment particle size analysis. Holes in fine mesh sieves should be repaired by filling with solder.

After rinsing and removing the fine sediments, large organic material such as whole leaves, twigs, algal or macrophyte mats that were not removed in the field should be rinsed, inspected for attached animals, and discarded.

12.4.2 Sorting tray

This must be white and have a completely flat bottom surface. Figures 2.17 and 2.18 show sorting trays in use. Mark the tray with a grid of thin lines to divide it into 12 or 16 equal-sized areas. Gridlines help you to estimate abundances and sort methodically. Apply the lines with an indelible marker pen. Pale or mid-blue lines are better than black because dark coloured animals remain clearly visible on them.

Smaller trays of about 35 x 25 cm are recommended for general use. They focus attention better than larger trays. They are also more comfortable to use because it is not necessary to lean so far over them, improving staff posture and concentration. Larger trays, about 45 by 35 cm, are useful for sorting stones and larger fragments of debris. Some people prefer larger trays for all sorting.

Figure 2.17
Biologist sorting a sample in a white sampling tray. Good lighting is essential. Notice how little material is in the tray.

12.4.3 Sorting process

The same methods are used for sorting live and preserved samples.

Identification is much easier with live specimens, particularly flatworms and leeches, but they **must be analysed within 48 hours of collection**. Some animals are almost impossible to identify to species level when preserved. However, preserving samples makes it easier to balance workloads.

Preservative must be washed thoroughly from preserved samples with tap water before they are sorted.

Several trays may be needed to sort a sample. Place a small amount of material in each sorting tray. It is much quicker and far more accurate to be able to distinguish fragments and animals by eye against a largely white background than to have to move material around the tray to uncover the invertebrates (Figure 2.18).

Although the whole sample should be scanned by eye, it is not necessary to sort the whole of it in detail. The proportion of the sample sorted in detail will vary according to how many animals it contains but should include the first 1000 specimens.

A larger proportion of the samples should be sorted in detail if there are a few common taxa, but the rest are rare. If most taxa are common, you can sort a smaller proportion in detail. The commonest proportion is 25%.

Sort the whole of the first tray to assess what proportion you should aim to sort in detail, then work out the number of squares in the sorting tray that represents that proportion. Sort this number of squares in each remaining tray but select which squares to sort at random.

Take all the specimens out of these squares and place them in a Petri dish or vial for identification. You must scan the remaining squares for new taxa, but store these separately as you may find more in squares for detailed sorting in the rest of the sample. It is helpful if visually similar taxa are stored together.

Count common taxa using tally counters. You only need to remove about 50 specimens of each common taxon from the sample, after which you should continue to remove specimens from the square you are working on, then estimate its abundance in the whole sample by proportions. If a common taxon turns out to comprise more than one species, you should also estimate their abundances by proportion.

Identification is easier if the animals sorted in the laboratory are separated taxonomically. Some animals that are readily identified by eye can be counted in the tray.

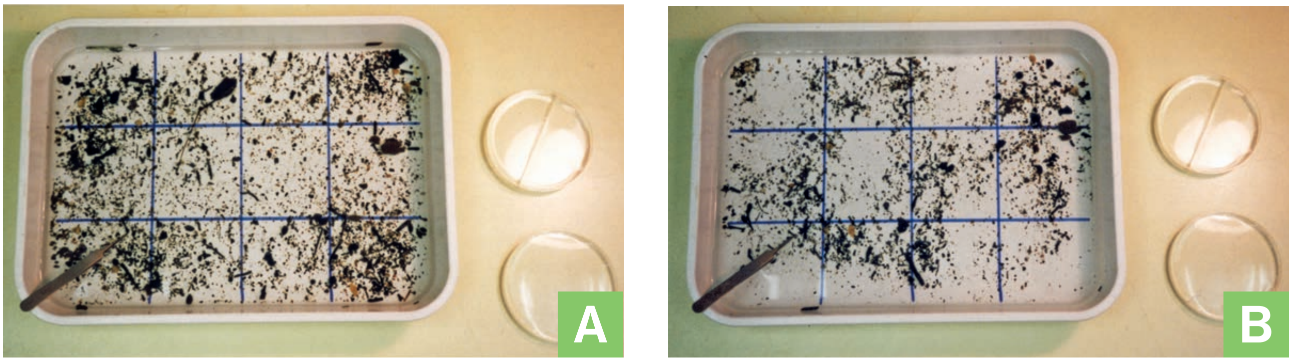


Figure 2.18

The amount of material to sort in a tray. Tray A contains too much material – some animals may be hidden by detritus. Tray B contains the maximum amount that we recommend.

Larger rare species such as crayfish, pearl mussels and medicinal leeches must be counted in the field and returned to the river.

Figure 2.19

Two designs of fume extractors for sorting invertebrate samples. Both designs take fumes away from the analyst towards the back of the tray.

Caution

Be aware of the health risks from poor posture and inadequate lighting. When spending protracted periods sorting and identifying invertebrate samples, pay attention to your chair and its height in relation to the laboratory bench. Take frequent breaks to exercise when sorting or identifying samples. Wash samples carefully because surface waters can be polluted and pose a risk to staff.

12.5 Identification

For environmental quality assessment, family is generally the minimum level of identification based on the taxa included in the WHPT index. That enables WFD quality status to be established and it is also sufficient for the calculation of a number of other biotic indices to help diagnose environmental pressures.

Species-level analysis provides more data from which to diagnose environmental degradation. This may provide greater accuracy necessary to detect emerging environmental pressures that have more subtle impacts, such as climate change, the impacts of some invasive species, and morphology. Species-level identification is generally needed for conservation analysis. However, its advantages for environmental assessment are often overstated. Precision is limited by sampling and the response of many species to common anthropogenic environmental pressures such as organic enrichment is similar to that of other species in the same family.

The main differences in sensitivities of species within a genus or family often involve the natural environmental pressures that determine their habitat, such as substrate and stream size. **To summarise: you need to decide on the diagnostic indices you want to apply, as different diagnostic indices require different levels of identification.**

Despite being particularly useful for environmental assessment, some species, including Oligochaeta and Chironomidae are difficult to identify. An operational compromise, known as 'mixed taxon analysis', is used to identify groups that are readily identifiable to species, and to identify other groups to a higher level. This is the standard level of analysis used by the Environment Agency. In addition to advantages already mentioned for species-level identification, it provides information from which to develop improved indices, and to assess the impacts of new pressures such as climate change and alien invasive species. This is summarised in Table 2.7.

Table 2.7
Taxonomic levels recognised by RIVPACS

Taxonomic level	Description	Notes
TL1	BMWP (Biological Monitoring Working Party) families	Taxa (families + Oligochaeta) recognised by BMWP indices
TL2	WHPT (Walley Hawkes Paisley Trigg) families	Taxa (families + Oligochaeta) recognised by WHPT indices
TL3	All RIVPACS families	Families recognised by RIVPACS
TL4	All RIVPACS species	Species recognized by RIVPACS; includes some composite species and higher taxa that are not differentiated
TL5	Mixed taxon level	Standard level of analysis used by environmental protection agencies

These taxonomic levels are described in more detail in a report by Davy-Bowker *et al.* (2010),[32] which includes appendices listing their constituent taxa, and is available from the Reports page of the RICT2 web pages **https://www.fba.org.uk/rivpacs-and-rict/rivpacs-rict-resources**

Further development of River Invertebrate Classification Tools can be found in this final report **https://www.sniffer.org.uk/wfd100**

Taxonomy poses many problems because it is constantly being refined, with our understanding being developed, and so it constantly changes. RIVPACS follows the nomenclature of the revised Furse-Maitland coded checklist, but it also includes other coding systems including the National Biodiversity Network codes. The taxonomy in RICT was reviewed in 2007, so presumably follows the 2007 update of the revised Furse-Maitland coded checklist. [33]

A description of the taxonomy adopted in RICT is provided in the following report which can be downloaded from the Reports page of the RICT2 website [34] **https://www.fba.org.uk/rivpacs-and-rict/rivpacs-rict-resources**

12.6 Enumeration

Numerical abundances are either counted or estimated. Numerical abundances allow much more flexibility in subsequent data analysis than the RIVPACS logarithmic abundance categories (Table 2.8). For standard RIVPACS samples, counting taxa present in low numbers (up to 50 individuals) and estimating the rest is recommended. There are a variety of methods for doing this, based on counting a proportion in each sorting tray and multiplying up by proportion.

Until about the year 2000, the environmental protection agencies in the UK only recorded RIVPACS abundance categories. Numerical abundance records facilitate many more types of analysis, including calculation of biotic indices using different abundance scales, and diversity indices.

Table 2.8
RIVPACS abundance categories

Abundance category	Numerical abundance
1	1–9
2	10–99
3	100–999
4	1000–9999
5	>9999

Key documents describing laboratory analysis for RIVPACS

Laboratory procedures for RIVPACS are described in Murray-Bligh *et al.* (1997) [28] and also in versions of this written for the Environment Agency (known as BT001) and in documents on the **STAR website**

Instructions for laboratory analysis of samples for RIVPACS are also included in the Environment Agency's Operational Instruction for analysing river invertebrate samples: Environment Agency (2014) Freshwater macro-invertebrate analysis of riverine samples.[35] This is available from the user guide page of the RICT2 web pages at: **https://www.fba.org.uk/rivpacs-and-rict/rict-rivpacs-user-guides**

Complementary and additional information linking to the STAR-AQEM project includes information from Furse and Gunn, (2002) RIVPACS sorting and recording.[36] **STAR website**

Environment Agency

Freshwater macro-invertebrate analysis of riverine samples

Operational instruction 024_08 Issued 28 January 2014

What's this document about?	Describes how to analyse samples of river macro-invertebrates. It is the fourth in a series of documents related to macro-invertebrate monitoring in rivers.
Who does this apply to?	• Environmental Monitoring (Analysis and Reporting) teams • Any staff or contractors analysing invertebrate samples
Contact for queries and feedback	• **For further information please contact** Helpdesk Services • **Anonymous feedback for this document can be given** here

Contents

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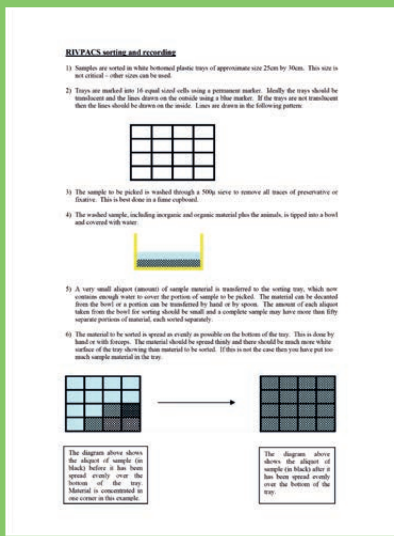


RIVPACS sorting and recording

1) Samples are sorted in white bottomed plastic trays of approximate size 25cm by 30cm. This size is not critical – other sizes can be used.

2) Trays are marked into 16 equal sized cells using a permanent marker. Ideally the trays should be translucent and the lines drawn on the outside using a blue marker. If the trays are not translucent then the lines should be drawn on the inside. Lines are drawn in the following pattern:

3) The sample to be picked is washed through a 500µ sieve to remove all traces of preservative or fixative. This is best done in a fume cupboard.

4) The washed sample, including inorganic and organic material plus the animals, is tipped into a bowl and covered with water.

5) A very small aliquot (amount) of sample material is transferred to the sorting tray, which now contains enough water to cover the portion of sample to be picked. The material can be decanted from the bowl or a portion can be transferred by hand or by spoon. The amount of each aliquot taken from the bowl for sorting should be small and a complete sample may have more than fifty separate portions of material, each sorted separately.

6) The material to be sorted is spread as evenly as possible on the bottom of the tray. This is done by hand or with forceps. The material should be spread thinly and there should be much more white surface of the tray showing than material to be sorted. If this is not the case then you have put too much sample material in the tray.

The diagram above shows the aliquot of sample (in black) before it has been spread evenly over the bottom of the tray. Material is concentrated in one corner in this example.

The diagram above shows the aliquot of sample (in black) after it has been spread evenly over the bottom of the tray.

13

QUALITY ASSURANCE OF LABORATORY ANALYSIS AND DATA HANDLING AND DATA QUALITY STANDARDS

Error is inevitable in ecological survey data, but it can be minimised, and also measured by numerical quality assurance procedures so that it can be considered in data analysis and interpretation. Measuring errors ensures that data is interpreted correctly by enabling real ecological differences to be distinguished from those caused by error.

Clear, unambiguous and comprehensive instructions are an essential step in quality assurance. The Environment Agency has produced a series of Operational Instructions that cover almost every aspect of its work. Those relevant to invertebrate sampling and analysis are mentioned here.

Unless an error is highlighted, it is often impossible for an individual to know that they are making mistakes. Active feedback is at the core of quality assurance procedures.

It is impossible for a human to sort samples without error. Small inconspicuous specimens are easily missed. To assess sorting error, the environment protection agencies in the UK undertook audits in which a number of samples are re-sorted by experts. For many years, these were undertaken annually by the River Communities team that analysed the RIVPACS reference samples. The audit measured analytical quality against the RIVPACS baseline. This enabled analytical quality to be adjusted to that used by the RIVPACS model, so that observed results were comparable to the expected results predicted by RIVPACS.

The first time that most analysts, including experienced ones, have their samples audited they are often surprised at the number of errors. Unless someone points out unintended errors to analysts, they may be unaware of them. This was demonstrated in the STAR project (Table 2.9), in which laboratories across Europe were audited (including the UK River Communities team). STAR/AQEM samples were collected according to the method described in **Section 15** and national samples were collected by a variety of methods according to different national protocols used in 2002. The results showed the considerable range in quality that is typical of laboratories that have not been audited before, and therefore their analysts are unaware of what they are missing. Experience from the UK audit showed that most errors stem from not noticing the presence of specimens in the sorting tray rather than from misidentifications. It also showed that quality tends to vary much more between laboratories than within them. As a result, the UK's environment protection agencies audited each of their laboratories by having 20 random samples re-analysed from each laboratory each year.

Table 2.9

Results of the family-level sorting audit undertaken in the STAR project

METHOD				
Partner	STAR/AQEM		NATIONAL	
	Mean	Range	Mean	Range
A	1.00	0 - 2	0.83*	0 - 3
B	0.83	0 - 3	0.83	0 - 1
C	4.17	0 - 11	3.50	1 - 7
D	1.83	0 - 3	4.83	2 - 11
E	1.67	1 - 3	-	-
F	3.50	0 - 9	3.17	0 - 6
G	0.33	0 - 1	0.83	0 - 2
H	4.25	3 - 6	8.25	5 - 12
I	1.00	0 - 2	2.75	2 - 3
J	1.00	0 - 3	8.83*	3 - 18
K	4.33	2 - 7	6.33	1 - 11
L	-	-	5.33*	2 - 11
M	1.33	0 - 3	1.50	0 - 3
N	2.67	0 - 4	5.67*	1 - 13
O	3.17	0 - 10	3.17	0 - 7

The audit also enabled an acceptable standard of analysis to be identified for standard RIVPACS samples – Figure 2.20. Based on what was achieved by some laboratories in the 1990 audit, an average of no more than 2 gains (families found by the auditor that the original analyst missed) was considered to be a realistic standard for samples analysed to the level required for BMWP indices, and it was adopted by the then regulatory agencies in Great Britain (the NRA (National Rivers Authority) in England and Wales, the River Purification Boards in Scotland and the Department of the Environment for Northern Ireland), and that standard is still used. Laboratories and individual analysts that do not meet this standard have to take action to restore their quality to this standard. In 2000, the number of families recorded in standard analyses was increased, which enabled them to be included in the current WHPT indices. In 2013, the Environment Agency adopted mixed taxon analysis. Despite the fact that many of the additional taxa were more difficult to recognise and caused more errors, the data standard remained at an average of no more than 2 gains at family-level.

Performance rapidly improves when a laboratory is audited, as analysts have errors brought to their attention and are therefore able to act. Over the years, the quality of Environment Agency laboratory analysis became more stable, so after 2011 it stopped auditing to reduce costs.

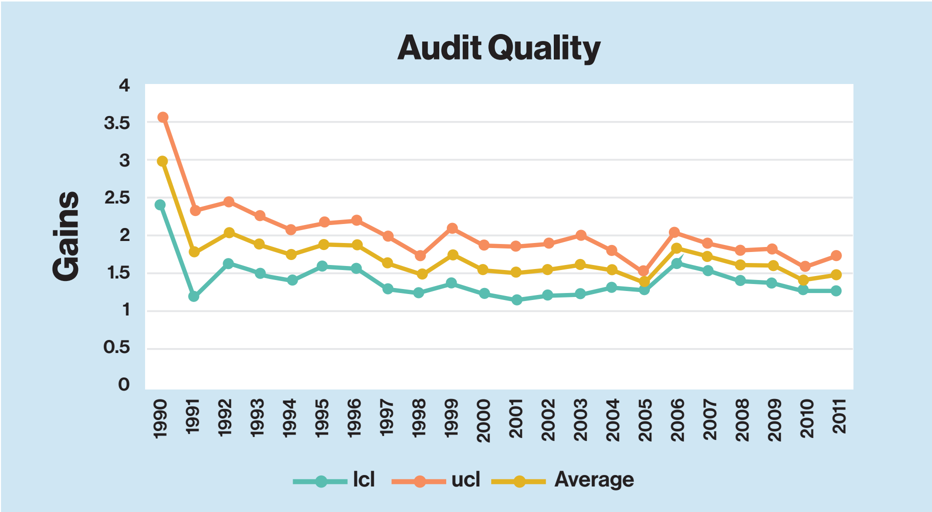


Figure 2.20

Results of the audits of the Environment Agency laboratory analyses of invertebrate samples. Gains are families identified by the auditor that the original analyst failed to record.

Although the Environment Agency has stopped the independent auditing of its laboratories every year, each of its laboratories continue to undertake internal checks for both sorting and identification. One in ten samples analysed are selected by a true random method and are re-analysed within the laboratory by another experienced analyst. In the past, internal checking was linked to a statistical quality control procedure to ensure that the laboratory achieved the analytical target of an annual average of no more than two gains. In more recent times the Environment Agency laboratories have not analysed a sufficient number of samples for the procedure to work effectively. To overcome this, invertebrate identification, which the Environment Agency undertakes to mixed taxon level, is checked by a ring test in which pre-identified samples are posted to the analysts to check their identification skills.

Methods for these quality assurance schemes are described in the Environment Agency's Operational Instruction for Quality Assurance.[37]


Environment Agency

Quality Assurance (AQC, Audit and Ring Test) programme for freshwater macro-invertebrate riverine sample analysis

What's this document about? Describes our quality assurance programme for freshwater macro-invertebrate samples including

- The internal analytical quality control (AQC) scheme;
- External audit process;
- Mixed taxon level ring test;
- Mixed taxon level reverse ring test.

The procedures in these schemes allow us to quantify and minimise errors in the sorting and identifying of freshwater macro-invertebrate samples

Who does this apply to? Environmental Monitoring teams (Analysis & Reporting (A & R) and Sampling & Collection, (S & C). External contractors undertaking invertebrate sample analysis.

Contact for queries and feedback

- For further information please contact Helpdesk Service
- Anonymous feedback for this document can be given here

Figure 2.21

Front cover of the Environment Agency's Operational Instruction on quality assurance for laboratory analysis of invertebrate samples[37]

14 TRAINING

Training is a critical part of quality assurance because it is the best way to ensure that procedures are followed. The best form of training for many activities, including sampling and sample analysis, is on the job, with mentoring from experienced ecologists who are familiar with the techniques. For many activities, experience is as important as theoretical knowledge. Some tasks are quite simple to explain but remarkably difficult to do well without experience. Sampling and sorting samples require particular visual awareness skills that take time to develop.

Comprehensive training on freshwater ecology survey methods is available in the UK from a number of MSc degree courses and the Field Studies Council (FSC, **https://www.field-studies-council.org/**). The Freshwater Biological Association (FBA) runs specialist two-day training courses every year about RIVPACS and RICT that cover every aspect including sampling, sample analysis (but not identification) and the use of RICT2 software. This is recommended for all users of RIVPACS and RICT. Information about this course is available from **https://www.fba.org.uk/rivpacs-and-rict/training-with-the-fba** Both the FBA and FSC run short courses in invertebrate identification, from general to specialist courses on particular groups of taxa. Some of these courses offer accreditation.

Even experts benefit from periodic refresher courses because it is easy to deviate unintentionally from the precise procedures, particularly sampling.

The Environment Agency has developed distance-learning courses on species-level identification and requires anyone analysing invertebrate samples to have passed an assessment on the basic species-level module. Other modules cover individual families in more detail.

Table 2.10

Environment Agency invertebrate identification distance learning courses

Module Number	Title	Author
1	**Basic Principles and Processes**	Richard Chadd
2	**Rapid Identification of Species (and Recognition of Distinctive Taxa)**	Richard Chadd
3	**Advanced Identification of Flatworms (Tricladida) And Leeches (Hirudinea)**	Richard Chadd
4	**Advanced Identification of Mayflies (Ephemeroptera) and Stoneflies (Plecoptera)**	Craig Macadam
5	**Advanced Identification of Caddisflies (Trichoptera)**	Ian Wallace
6	**Advanced Identification of Water Bugs (Hemiptera, Heteroptera)**	Sheila Brooke
7	**Advanced Identification of Water Beetles (Coleoptera) Part A (Gyrinidae, Haliplidae, Hygrobiidae), Part B (Noteridae, Dytiscidae), Part C (Polyphaga)**	Garth Foster
8	**Advanced Identification of Dragonflies and Damselflies (Odonata): Larvae and Exuviae**	Steve Brooks and Steve Cham
9	**Advanced Identification of Miscellaneous Taxa including Crustacea**	Terry Gledhill and Ian Wallace
10	**Identification of Molluscs (Gastropoda and Bivalvia)**	Ian Killeen
11	**Identification of Diptera (True Flies)**	Michael Dobson with the assistance of James Pretty and John Blackburn

These documents are available on **https://www.fba.org.uk**

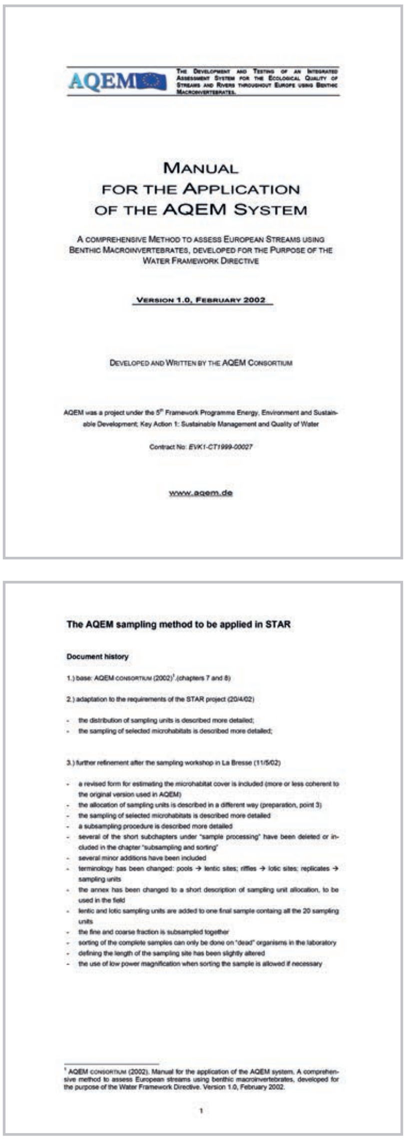
15

STAR-AQEM SAMPLING METHODOLOGIES AND SYNERGIES TO RIVPACS

RIVPACS and STAR-AQEM are the two methods most widely used across Europe for sampling and analysing river invertebrates for the Water Framework Directive. They share fundamental principles of pro rata multi-habitat sampling. The STAR-AQEM method is used mainly in central Europe. We have included it in this book for the benefit of readers who are standardising methods in their own countries. In Europe, we recommend using either RIVPACS or STAR-AQEM, depending on which is used in neighbouring states, especially those sharing transboundary rivers and therefore sharing the same River Basin Management Plans. This will ensure that data is comparable beyond intercalibration, not only for analysis and reporting, but also for sharing data from reference sites. STAR-AQEM sampling is described in the current European (CEN) standard (EN 16150:2012. Water Quality – Guidance on pro-rata multi-habitat sampling of benthic macro-invertebrates from wadeable rivers). (38)

Like RIVPACS, STAR-AQEM sampling is pro rata multi-habitat, in which all habitats at the site are sampled in proportion to their cover. (39) This approach provides a consistent method that can be used in all types of river, providing comparable samples for classification and assessment, whatever habitats are available.

A manual describing STAR-AQEM is available from the STAR website **http://www.eu-star.at/**, by following the path: enter > protocols > AQEM macro-invertebrate sampling protocol. The original, detailed method is described on the AQEM website. (16) The AQEM manual (**http://www.aqem.de/mains/products.php** > product > AQEM manual) is also very useful, but it does not include the refinements made in the STAR project – Figure 2.22.



AQEM — The Development and Testing of an Integrated Assessment System for the Ecological Quality of Streams and Rivers throughout Europe using Benthic Macroinvertebrates.

MANUAL FOR THE APPLICATION OF THE AQEM SYSTEM

A comprehensive Method to assess European Streams using Benthic Macroinvertebrates, developed for the Purpose of the Water Framework Directive

VERSION 1.0, FEBRUARY 2002

Developed and Written by the AQEM Consortium

AQEM was a project under the 5th Framework Programme Energy, Environment and Sustainable Development; Key Action 1: Sustainable Management and Quality of Water

Contract No: *EVK1-CT1999-00027*

www.aqem.de

The AQEM sampling method to be applied in STAR

Document history

1.) base: AQEM Consortium (2002)[1] (chapters 7 and 8)

2.) adaptation to the requirements of the STAR project (20/4/02)

- the distribution of sampling units is described more detailed;
- the sampling of selected microhabitats is described more detailed;

3.) further refinement after the sampling workshop in La Bresse (11/5/02)

- a revised form for estimating the microhabitat cover is included (more or less coherent to the original version used in AQEM)
- the allocation of sampling units is described in a different way (preparation, point 3)
- the sampling of selected microhabitats is described more detailed
- a subsampling procedure is described more detailed
- several of the short subchapters under "sample processing" have been deleted or included in the chapter "subsampling and sorting"
- several minor additions have been included
- terminology has been changed: pools → lentic sites; riffles → lotic sites; replicates → sampling units
- the annex has been changed to a short description of sampling unit allocation, to be used in the field
- lentic and lotic sampling units are added to one final sample containg all the 20 sampling units
- the fine and coarse fraction is subsampled together
- sorting of the complete samples can only be done on "dead" organisms in the laboratory
- defining the length of the sampling site has been slightly altered
- the use of low power magnification when sorting the sample is allowed if necessary

[1] AQEM Consortium (2002). Manual for the application of the AQEM system. A comprehensive method to assess European streams using benthic macroinvertebrates, developed for the purpose of the Water Framework Directive. Version 1.0, February 2002.

1

Figure 2.22

Front covers of AQEM manual (16) and STAR revision (39)

STAR-AQEM samples are standardised by sampling a fixed area of 1.25 m².

A sample consists of 20 sub-samples taken from all microhabitat types at the sampling site with at least 5% coverage.

A sub-sample is collected from a 0.25 x 0.25m square quadrat. The 20 sub-samples are allocated to microhabitats according to their proportional cover. For example, if the habitat in the sampling reach is 50% sand, 10 of the sub-samples must be taken from this microhabitat. See Figure 2.23.

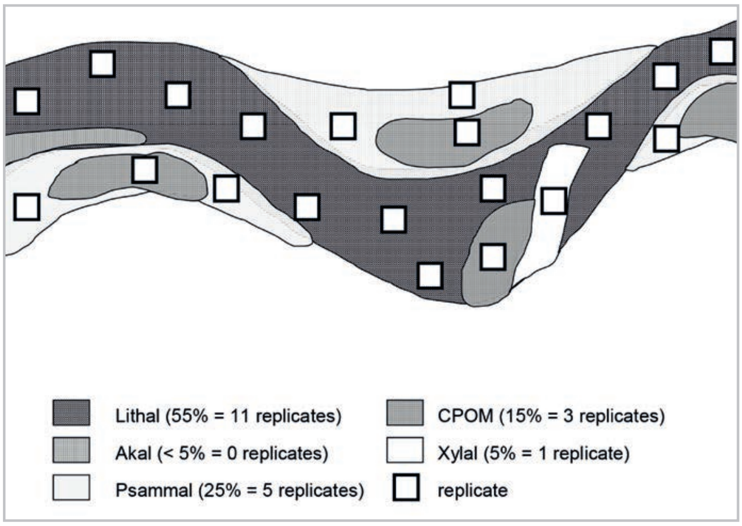


Figure 2.23

Example of the location of sub-samples in a theoretical sampling site according to the STAR-AQEM 'multi-habitat sampling' method [16] *(From AQEM manual, 2002)*

In contrast, a RIVPACS sample is standardised by sampling for a fixed amount of time: 3 minutes + 1-minute search, supplemented in deep waters by a 1-minute marginal sweep.

For STAR-AQEM, the sampling site is a 500m stretch. A RIVPACS site is much shorter. The nets used to collect STAR-AQEM samples are 500 µm mesh whereas nets for RIVPACS samples are 1 mm. In this aspect, STAR-AQEM may provide more differentiation between the habitats sampled, which may allow more detailed analysis in some cases.

Much of the rest of the STAR-AQEM procedure is the same as RIVPACS. Although more material is collected by the STAR-AQEM method than RIVPACS, the amount of material analysed in the laboratory and the number of animals identified is similar.

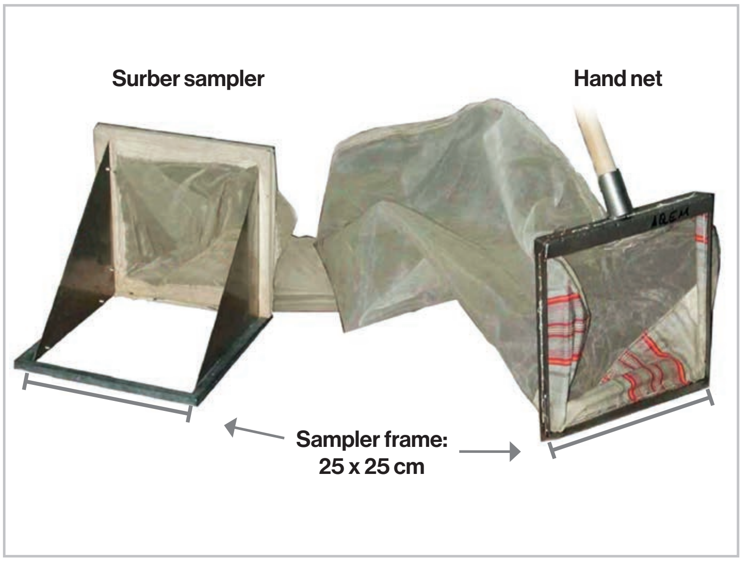


Figure 2.24

Surber sampler and hand net used to collect AQEM-STAR samples *(From AQEM manual, 2002)*

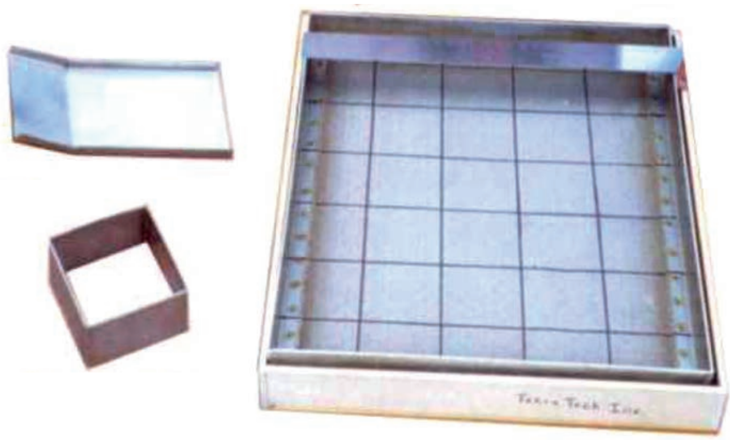


Figure 2.25

Sample splitter used to sub-sample a STAR-AQEM sample so that just over 700 animals are analysed *(From AQEM manual, 2002)*

site name	date	sample no.	investigator

Sample related information, to be recorded at each sampling date (copy if necessary)

23 MINERAL SUBSTRATES (5% steps, mark substrates <5% with ‘X’)	% of coverage (5% classes); sum of mineral and biotic microh. = 100%	no. of replicates for sample	x = artificial substrate ‘technolithal’
hygropetric sites water layer on solid substrates			☐
megalithal >40 cm large cobbles, boulders and blocks, bedrock			☐
macrolithal >20 cm to 40 cm coarse blocks, head-sized cobbles, with a variable percentages of cobble, gravel and sand			☐
mesolithal >6 cm to 20 cm fist to hand-sized cobbles with a variable percentage of gravel and sand			☐
microlithal >2 cm to 6 cm coarse gravel, (size of a pigeon egg to child's fist) with variable percentages of medium to fine gravel			☐
akal >0.2 cm to 2 cm fine to medium-sized gravel			
sand and mud			
silt, loam, clay (inorganic)			
phytal floating stands or mats of macrophytes, lawns of bacteria or fungi, and tufts, often with aggregations of detritus, moss or algal mats			
algae filamentous algae, algal tufts			
submerged macrophytes macrophytes, including moss and Characeae			
emergent macrophytes e.g. *Typha, Carex, Phragmites*			
living parts of terrestrial plants fine roots, floating riparian vegetation			
xylal (wood) tree trunks, dead wood, branches, roots			
CPOM deposits of coarse particulate organic matter, e.g. fallen leaves			
FPOM deposits of fine particulate organic matter			
organic mud mud and sludge (organic) = pelal			
debris organic and inorganic matter deposited within the splash zone area by wave motion and changing water levels, e.g. mussel shells, snail shells			
sewage bacteria, –fungi and sapropel sewage bacteria and –fungi, (*Sphaerotilus, Leptomitus*), sulphur bacteria (e.g. *Beggiatoa, Thiothrix*), sludge			
➔	**sum = 100%**	**sum = 20**	

- PAGE 2 - AQEM

Figure 2.26
The AQEM Site Protocol, page 2, listing the micro-habitats for recording their percentage cover and therefore the allocation of the 20 sub-samples to them

Steps in the STAR-AQEM method are as follows:

- Before entering the water, **complete the STAR-AQEM site protocol forms**, which cover site assessment and field data, available from Annex 2 of the AQEM manual.

  Based on the microhabitat list given on page 2 of the site protocol (Figure 2.26), the **coverage of all microhabitats with at least 5% cover** is recorded to the nearest 5%, from which the **number of replicates** in each of the individual habitats is determined.

- **Start sampling at the downstream end** of the reach and proceed upstream. Use a hand net either as a kick net, or for 'jabbing', 'dipping' or 'sweeping', or use a Surber sampler.

- After every three replicates (or more frequently if necessary) **rinse** the collected material in clean stream water two to three times. If clogging occurs, discard the material in the net and redo the replicates in the same habitat types but at different locations.

- **Remove large wood and stones** after rinsing and inspecting for clinging or sessile organisms, which should be placed into a sample container. Do not spend time inspecting small debris in the field, although larger and fragile organisms (eg Ephemeroptera) or species that cannot be preserved (eg Tricladida, Oligochaeta) should be partly sorted in the field. Store these organisms in a separate small container.

- **Remove large and rare organisms** that can be identified in the field, such as large mussels, and return them to the stream after recording their presence.

- Immediately after collection, transfer the sample from the net to sample container(s) and **preserve** with enough 95% ethanol to cover the sample after decanting any water from the sample, to prevent carnivores from eating other organisms. The final ethanol concentration should be about 70%.

  Close the sample container tightly. The samples should be stored cool. Alternatively, for live sorting in the laboratory, the samples must be kept in a minimum amount of liquid and transported immediately to the laboratory. They must be kept cool during transport.

- **Labelling:** Place a label (written in pencil, printed on a laser printer or photocopied) inside the sample container.

- **Refine the site protocol,** particularly the share of microhabitats, after sampling has been completed, when you will be able to provide a more accurate assessment.

- **Sample processing** is described in the STAR revision of the AQEM protocol. Only a portion of the sample needs to be analysed. Methods for splitting the sample are provided in the STAR document and the CEN standard on pro rata sampling, mentioned earlier. Only the first 700 organisms need to be analysed, so further sub-samples do not need to be taken after this number is reached.

You will note that there is significant synergy between the UK RIVPACS methodology and the STAR-AQEM methodology. In recent years, collaboration and exchange have continued, encouraging the development of methods for ecological assessment.

The Water Framework Directive implementation and the need to adapt existing methods and develop new methods was the catalyst needed to advance cooperation.

Resources were made available via the EU and Member States to advance the science and operational implementation of these ecological and biological methods.

We hope this partnership will continue.

The WFD implementation and projects such as STAR-AQEM are good examples of sharing expertise and demonstrate the potential for development and adaptation of methods to suit a wide range of riverine habitats.

Chapter 3

MACROINVERTEBRATE STATUS CLASSIFICATION METHODS

Biological information is increasingly being used as the key environmental indicator to direct investment in the water industry and agriculture to achieve environmental improvement and protection.

1
MACROINVERTEBRATE DATA ANALYSIS FOR SURVEILLANCE AND OPERATIONAL MONITORING INCLUDING STATUS CLASSIFICATION

1.1 Overview

This section provides an overview of data analysis for surveillance and operational monitoring including the general river quality status classification. Other methods, including those used for investigative monitoring, are covered in **Chapter 4** and reporting is covered in **Chapter 6**. This section is aimed at ecologists and river managers as a summary of river invertebrate methods and how to interpret their results. It is also aimed at ecologists in other countries who are setting up their own status classification schemes and want to know how this was done in the UK. It focuses on RIVPACS (River Invertebrate Prediction and Classification System), which is at the heart of the UK's approach – its underlying principles having been adopted in the Water Framework Directive for all biological status classifications across Europe.

The data analyses described in this part of the handbook depend on adherence to the field and laboratory methods described in **Chapter 2**. The accuracy and precision of results from the data analysis depend on the quality of the sampling and laboratory analyses. The emphasis of methods for monitoring must therefore be on reproducibility, quality assurance and the accountability of those responsible for undertaking each step.

See the monitoring cycles in Figure 3.1: the red circle shows where this fits into the overall monitoring cycle.

Hydrometridae (Water Measurer)

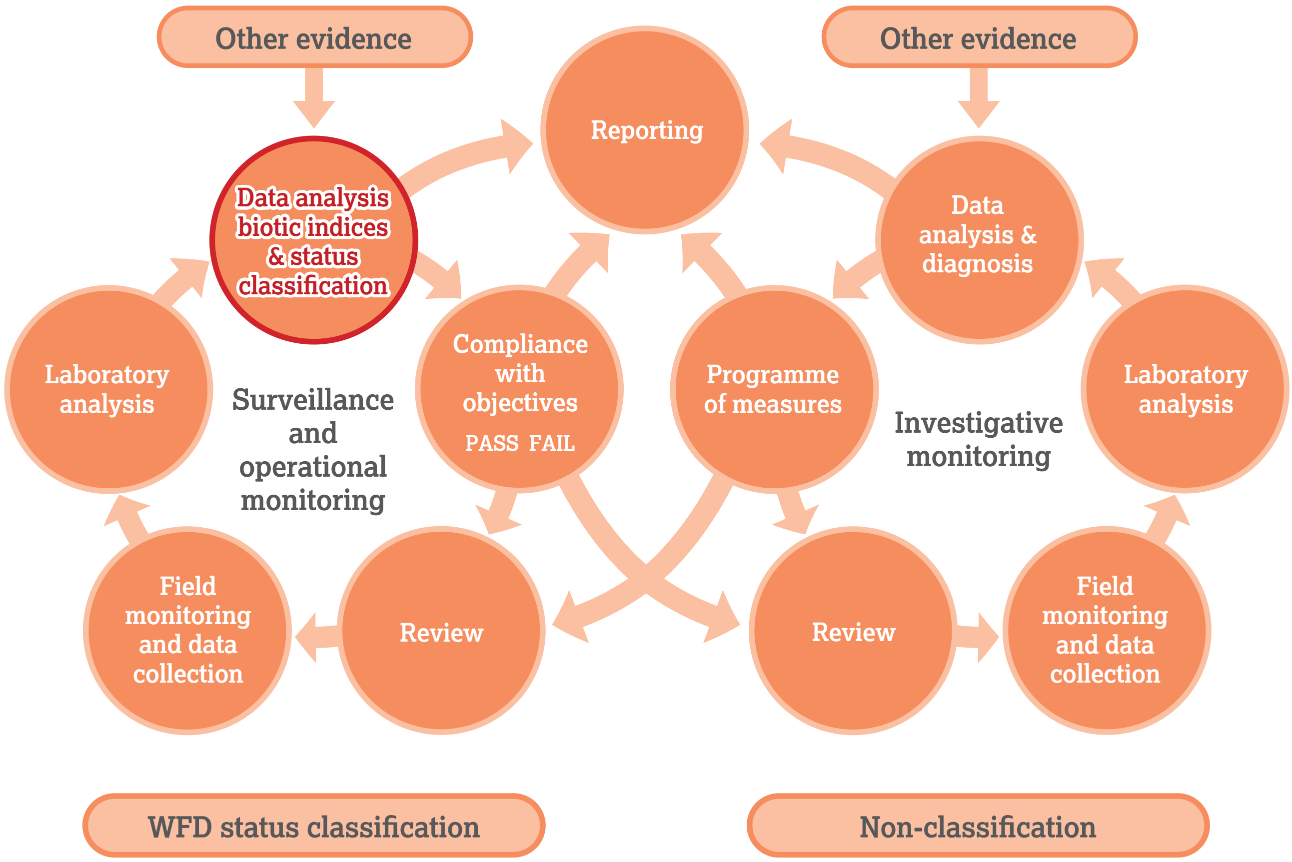


Figure 3.1

The monitoring cycles with topic covered by this part of the handbook highlighted in red

It is important to record on databases the monitoring programme for which the sample was collected, particularly the samples that should or should not be used for status classification. For example, surveillance monitoring should rely on an unbiased sampling programme to assess the background status of a river basin or country.

Investigative monitoring can add many additional samples to the database. If they were analysed with surveillance monitoring samples they could skew the results and misrepresent the overall status of the river basin.

Data from operational monitoring can also distort wider-scale analyses of long-term changes in river basins, because they are concentrated in water bodies where quality issues are suspected, where measures to restore quality are implemented or where the risks of failing quality standards are high, and because they are more numerous than surveillance sites. Conversely, operational monitoring can under-represent unsuspected issues because it is undertaken where issues are known.

Flagging each sample in the database as surveillance, operational or investigative allows the selection of data that is fit for particular purposes, so that inappropriate data does not skew the analyses.

Biological classification data can be utilised in a number of key reports:

- at international level, via the EU State of the Environment reports
- at national level, as key elements of the UK State of the Environment Reports
- at more local catchment level as part of River Basin Management Plans, stakeholder engagement and local reports.

Biological information is increasingly being used as the key environmental indicator to direct investment in the water industry and agriculture to achieve environmental improvement and protection. It also informs biodiversity initiatives.

1.2 Management context for river invertebrate classification

Classification is a means of explaining environmental quality in very simple terms that can be understood by anyone. We currently use a 5-point scale for this (Figure 3.2) and each class has a very simple description and is associated with a colour so that water bodies of different quality can be mapped (Figure 3.3).

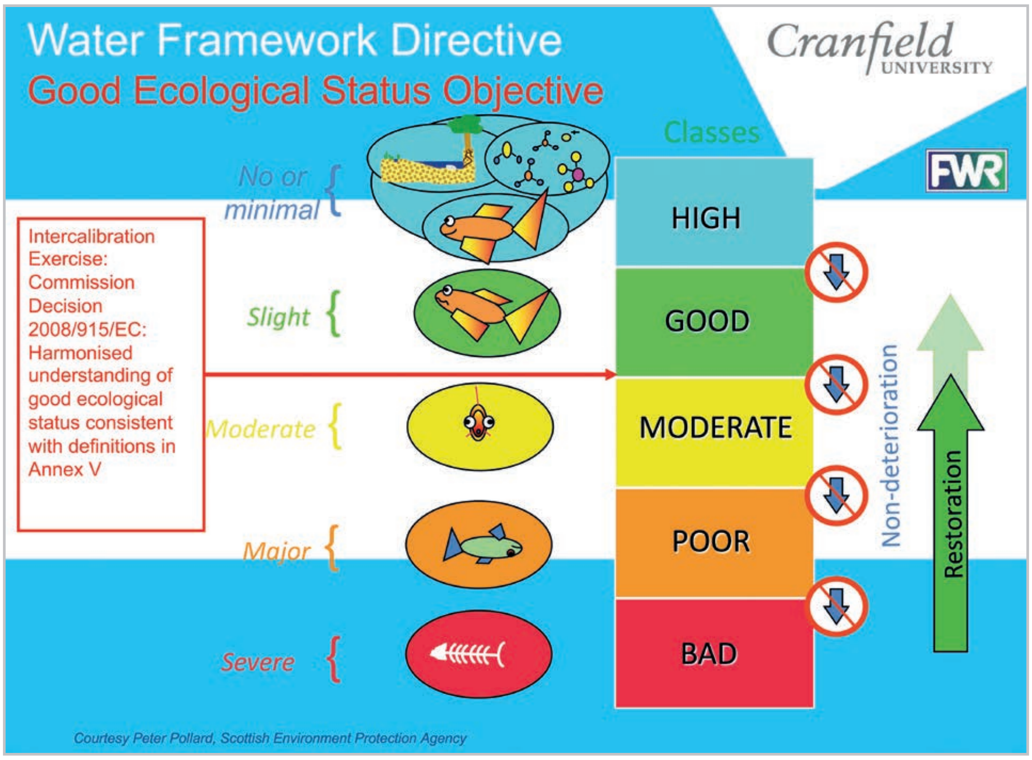


Figure 3.2

Correctly setting class boundaries is critical, helped by intercalibration *(Peter Pollard for UK TAG)*

Water chemistry has been classified for many years as the core element of water quality assessment and management. More recently, we have recognised that environmental quality may be better reflected as ecosystem quality, which is defined by the biota (together with indicators of human pathogens and physical parameters such as the amount of water). The WFD therefore gives prominence to biological parameters for classifying and reporting on the status of our water bodies in relation to their environmental quality objectives. Biology is also used to define boundaries for abiotic elements (including standards for physico-chemical elements, specific pollutants and flow). These remain vital for regulation, water quality improvement and abstraction management programmes to ensure that ecological quality objectives are achieved and maintained. Biological monitoring and assessment is now key to determining environmental quality and the following sections focus on this with regard to river invertebrates.

Biological status is defined in detail in the normative definitions in Annex V of the Water Framework Directive. Identifying the metrics that we use to quantify biological status and setting class boundaries correctly is critical for setting environmental quality objectives that fulfil their intended role, and for assessing the effectiveness of measures to restore or maintain that quality. It is also critical for reporting the state of the environment. The concept of reference condition and its accurate determination for all water bodies is also critical, because it enables the classifications to be comparable between different types of rivers that support different biotas. This comparison is important locally, nationally and internationally for identifying where we should target our limited resources for investigation and restoration.

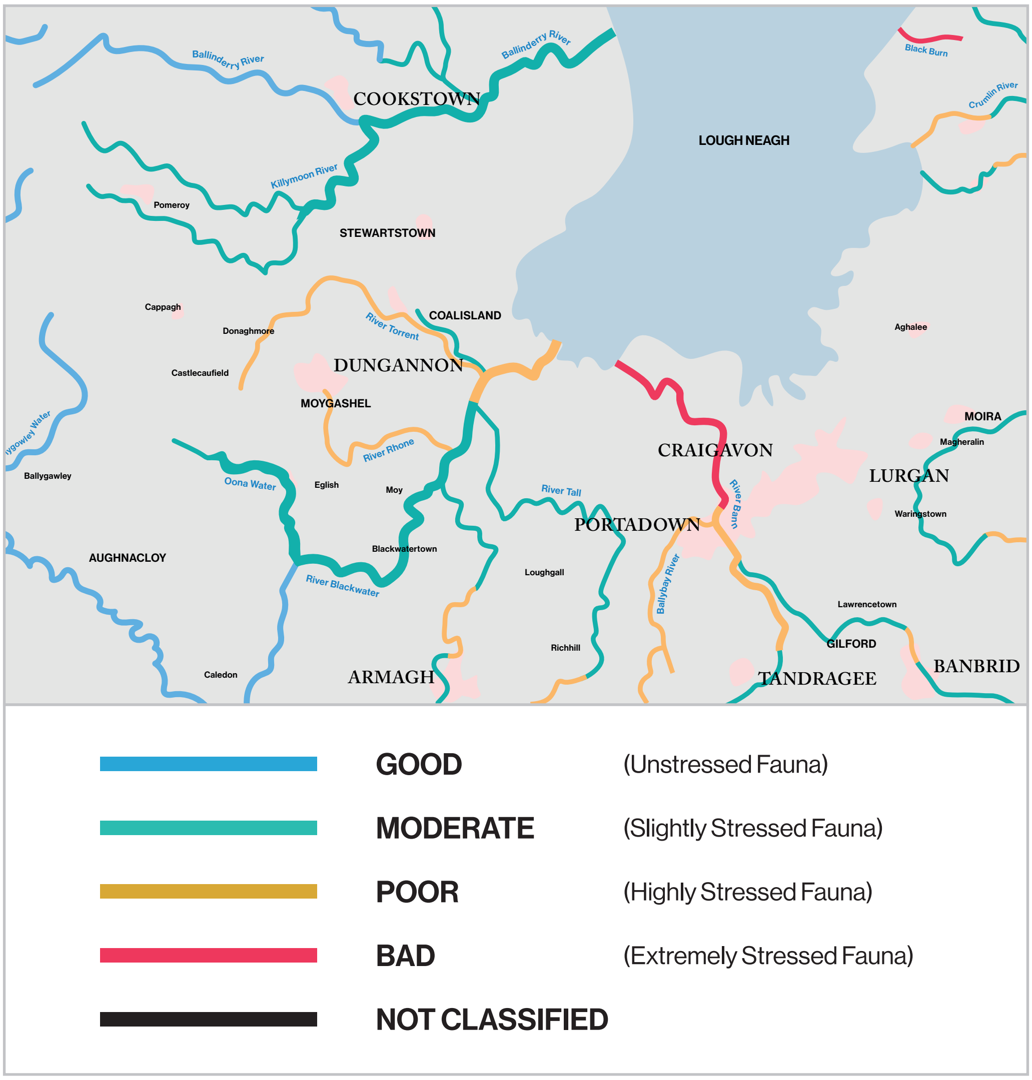


Figure 3.3

Map showing biological quality status – example is an extract from the 1991 survey of Northern Ireland (DoENI, 1993) based on the 5M classification with river stretches belonging to different classes in contrasting colours [40] – compare with Figure 3.6

The key objective of achieving Good Ecological Status places significant emphasis on ensuring that the class boundaries (particularly between Moderate and Good) are correct and equivalent across all Member States, which is ensured by intercalibration.

The criteria for choosing appropriate river invertebrate classification metrics and the classification metrics adopted in the UK are described in **Section 4.1** and **Section 2.2**. If class boundaries are set too low, the environmental improvements that the WFD aims to achieve will not be met. If class boundaries are set too high, the costs will be excessive. The emphasis on setting the Good/Moderate status boundary correctly is crucial because Good status is the ultimate objective for most water bodies, so it is the boundary that determines compliance or non-compliance with environmental objectives. Other boundaries are also important: lower standards may be set as interim standards and existing High status sites should not be allowed to deteriorate. The process of setting the river invertebrate status boundaries is described in **Section 4.2**.

Biological metrics are complex because they describe complex processes. The classification metrics model the normative definitions, but like all models they have their limits, and they cannot cover every unusual condition. An understanding of the classification metrics is needed to fully interpret the reliability of the classification results (**Section 5**). In cases beyond the limits of the model, the final classification has to be based on other information.

The classifications are inevitably simplifications that need to be understood by politicians and the general public, to allow programmes of measures to be chosen, funded and undertaken.

2
METRICS AND INDICES

2.1 Introduction

The field and laboratory analysis of invertebrate samples results in a long list of species or other taxa found in the samples together with their abundances. Whilst an experienced biologist can interpret much about the quality of the environment from this raw data, based on their knowledge of the ecology of each species and previous experience of finding them in samples from rivers of a particular quality, the data is impenetrable to others, including river managers and other customers of biological monitoring.

Biologists therefore convert their results into simple numerical values that summarise the complex raw data so that others can understand and interpret them.

A metric is any numerical parameter derived from biological monitoring data to summarise it, such as totals (number of families, number of individuals), diversity indices (see **Chapter 5**) and biotic indices. Information is inevitably lost in these gross summaries, so biologists still use the raw biological data for more detailed analysis and interpretation.

As biologists working on rivers gained more knowledge about the ecology and distribution of river invertebrates in relation to the environmental pressures that they were concerned with (which until the late 1980s was mainly water pollution from industry and domestic sewage), more reliable biotic indices were developed that were applicable across the whole country. The most successful are biotic indices that relate the biological data to the intensity of environmental pressures.

The Biological Monitoring Working Party score was the first index that was used widely across the whole of the UK.[10]

The BMWP-score was developed for the 1980 National River Quality Survey and the average BMWP score per taxon (BMWP ASPT) and the number of BMWP-scoring taxa (BMWP NTaxa) were used for status assessment until 2015.

BMWP is described in **Chapter 5 Section 3.1.4.** Although still used in England for operational assessment in relation to water resources, these indices have been replaced by a major revision known as Walley Hawkes Paisley & Trigg indices.[41]

2.2 WHPT (Walley Hawkes Paisley Trigg) indices

WHPT was developed as a more accurate and more precise replacement for BMWP for operational and surveillance assessment and status classification, made possible because of data available from monitoring between 1990 and 2005. Better accuracy and precision were needed because of the subtle but critical difference between Moderate and Good status, which is the boundary between achieving and failing environmental quality objectives for most water bodies.

The derivation of WHPT values from BMWP ASPT is described in Paisley *et al.* (2014).[41] WHPT indices replaced BMWP indices in time for the UK's second River Basin Management Plans, published in 2016.

The accuracy and precision of biotic indices depends on:

- the accuracy of the index value assigned to each taxon
- the number of taxa and individuals from which the index is calculated at a site
- the narrowness of the range of conditions in which that taxon is found.

WHPT values are far more accurate than BMWP values mainly because they were derived from an analysis of a huge set of field data from more than 100,000 standard quality-assured samples from across the UK, rather than on expert judgement based on the limited knowledge that was available in the late 1970s. WHPT makes use of more taxa than BMWP by using data from additional families included in monitoring from about the year 2000.

The inclusion of abundance data improved the index's ability to detect moderate changes in invertebrate quality associated with eutrophication, and it also improved compliance with the WFD's normative definitions. When deriving WHPT index values, different abundances of the same taxon were analysed as if they were different taxa. For most taxa, there is a different value of WHPT for each RIVPACS abundance category (Table 3.1 & Table 3.2).

Considering each abundance category of each taxon as if it was a different taxon effectively narrowed the range of conditions in which each 'taxon' was found. Taking abundance into account in this way gives much more precise results than simply applying a common abundance weighting factor.

To ensure the accuracy of WHPT index values, those based on less than 75 records were removed before the definitive WHPT index was finalised, as were values that caused a bimodal distribution across abundance categories. Both the index values calculated from the data and the definitive values used operationally are listed in Paisley *et al.* (2014).[3]

Table 3.1

RIVPACS Log_{10} abundance categories

Abundance category	AB1	AB2	AB3	AB4
Numerical abundance	1–9	10–99	100–999	1000+

Table 3.2

Examples of abundance-related WHPT index values. For tolerant families like Asellidae, WHPT values decrease with increasing abundance, but for sensitive families like Heptageniidae, they increase. The index values for WHPT can be less than 1 or greater than 10 to reflect sensitivity and ensuring that WHPT ASPT (WHPT average score per taxon) is on the same scale as BMWP ASPT.

Taxon	Abundance categories			
	AB1	AB2	AB3	AB4
Asellidae	4	2.3	0.8	-1.6
Heptageniidae (incl. Arthropleidae)	8.5	10.3	11.1	11.1

Because Chironomidae and Oligochaeta are found in almost all invertebrate samples, the index values derived for these taxa were essentially the average value across the whole quality spectrum. That would have elevated WHPT ASPT values at sites with strong organic pollution, when these taxa but few others are present. As a result, the definitive index values used for Oligochaeta were those derived for the most tolerant family, Tubificidae, and index values used for Chironomidae were the values for Chironomini, as they had been in the BMWP index to allow the most polluted sites to have very low WHPT ASPTs.

We considered developing species-level indices to increase the number of taxa from which the index is calculated and to narrow the range of conditions from which each taxon is found. When WHPT was developed, however, index values could be allocated to each family far more accurately than they could to species because a far larger data set was available from which to derive the index values. We did not base WHPT indices on families to save costs.

The regulatory agencies in Great Britain actually analyse invertebrate samples to a mixed taxonomic level that is mostly to species, so that monitoring data can be used to track invasive species as well as subtle effects from emerging pressures such as climate change. Because they share similar physiology, many species of river invertebrates belonging to the same family have similar tolerances to pollution and other pressures, with only a few notable exceptions. This causes species-level versions of indices that also have family-level versions to offer little additional precision or accuracy.

Like BMWP, WHPT is used in two complementary forms: the average WHPT score per taxon (WHPT ASPT) and the number of WHPT scoring taxa (WHPT NTaxa).

WHPT ASPT is the average sensitivity of invertebrates to organic and nutrient loads and other pressures associated with domestic or mixed waste discharges (fine sediment, ammonia toxicity) or that affect dissolved oxygen and productivity (reduced flow, reduced shading).

Status classification based on WHPT ASPT reflects these pressures, which are still the most widespread combination of environmental pressures that impact river invertebrate communities. Although nominally an index of saprobity (organic loading), WHPT ASPT is actually a multi-pressure index (as are all indices of organic pollution) because all these pressures interact and can co-exist. River invertebrates are particularly sensitive to dissolved oxygen and anything that impedes their respiration.

The wide range of tolerances of different taxa enables the strength of these pressures to be measured relatively accurately across a wide range of intensity. The precision of WHPT ASPT is enhanced by being an average and is relatively insensitive to sampling effort.

WHPT NTaxa is a measure of taxonomic richness and it (and status classification based on it) responds to all types of environmental pressures, including toxic chemicals.

We use this index because WHPT ASPT cannot detect certain pollutants, in particular those caused by acidification and metal pollution, which is common in mine water and industrial discharges. WHPT NTaxa is less precise than WHPT ASPT because it is not an average, and it is more sensitive to laboratory error and sampling variation. Beware that poorer NTaxa (ie fewer taxa) sometimes represents better quality: in oligotrophic upland streams, particularly with low alkalinity, mild organic pollution can increase NTaxa, although the composition of taxa may change and be reflected in poorer ASPT.

The combination of WHPT-ASPT and WHPT NTaxa responds to virtually all combinations of pressures. Using WHPT ASPT and WHPT NTaxa together not only provides a measure of ecological quality but also gives a limited diagnostic capability that enables sites affected predominantly by organic and related pressures to be differentiated from those affected by toxic pressures. High WHPT ASPT but low WHPT NTaxa indicates an absence of organic or related pressures (indicated by the presence of sensitive taxa) but the presence of another type of pressure, which must either be toxic metal/acidification or habitat degradation. When both WHPT ASPT and WHPT NTaxa are calculated, the WHPT score is redundant.

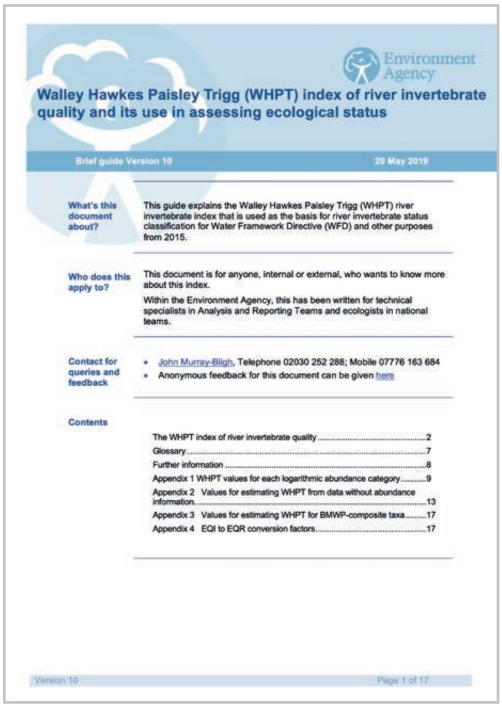
Score = ASPT x NTaxa

2.3 Comparison of BMWP and WHPT

WHPT ASPT is on the same scale as WHPT BMWP. The only difference, apart from improved accuracy and precision, is the greater sensitivity of WHPT ASPT to mild enrichment that affects abundances but not species composition. WHPT NTaxa responds to the same environmental pressures as BMWP NTaxa but it is not on the same scale because it is based on a greater number of taxa (106 taxa in WHPT and only 82 in BMWP). WHPT includes more families of Diptera, and the constituents of composite families in BMWP are treated as distinct families in WHPT.

2.4 Calculating WHPT indices

For status classification, WHPT must be calculated at RIVPACS Taxon Level 2 (ie including families not included in BMWP), with BMWP composite taxa separated into individual families and using abundance-related index values. This is erroneously termed 'WHPT abundance-weighted with distinct families' in RIVPACS (River Invertebrate Prediction and Classification System) and RICT (River Invertebrate Classification Tool): WHPT does not use abundance weighting but gives independently defined index values for different abundances of each taxon (**Section 2.2**). If you do not have abundance data, or only have data for BMWP taxa including composite families, you can still estimate WHPT, but not for status classification.



Environment Agency

Walley Hawkes Paisley Trigg (WHPT) index of river invertebrate quality and its use in assessing ecological status

Brief guide Version 10 — 29 May 2019

What's this document about? This guide explains the Walley Hawkes Paisley Trigg (WHPT) river invertebrate index that is used as the basis for river invertebrate status classification for Water Framework Directive (WFD) and other purposes from 2015.

Who does this apply to? This document is for anyone, internal or external, who wants to know more about this index.

Within the Environment Agency, this has been written for technical specialists in Analysis and Reporting Teams and ecologists in national teams.

Contact for queries and feedback

- John Murray-Bligh, Telephone 02030 252 288; Mobile 07776 163 684
- Anonymous feedback for this document can be given here

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Figure 3.4
Environment Agency guide to WHPT

A guide to WHPT (Figure 3.4) and an Excel spreadsheet for calculating it (Figure 3.5) can be downloaded from the RIVPACS/RICT methods web page **https://www.fba.org.uk/rivpacs-and-rict/rict-rivpacs-user-guides**

Beware that WHPT was known as 'revised BMWP' in early reports such as Davy-Bowker *et al.* (2008) on RIVPACS development,[34] but was changed to WHPT to avoid confusion with earlier revisions of BMWP described in Walley & Hawkes (1996 and 1997).[42][43]

	A	B	C	D	E F	G H	I J	K L	M	N–U
1–2	***Use this sheet to become familiar with the abundance-weighted WHPT ASPT and how it differs from BMWP indices and the presence/absence version of WHPT*** **Taxon**	**BMWP**	**Presence only (PO) WHPT**	***Enter numerical abundance here***	**WHPT for abundance category 1 (1-9)**	**WHPT for abundance category 2 (10-99)**	**WHPT for abundance category 3 (100-999)**	**WHPT for abundance category 4+ (1000 +)**	**Abundance related WHPT for this sample**	**Instructions and notes**
3	**Triclada (flatworms)**									Enter numerical abundance in column D
4	Planariidae	5	4.9		4.7	5.4	5.4	5.4	**0**	Before re-using this sheet, delete all values from Column B: there should be zeros in Column M
5	Dugesiidae	5	2.9		2.8	3.1	3.1	3.1	**0**	Non-scoring taxa are highlighted in green
6	Dendrocoelidae	5	3.0		3.0	2.6	2.6	2.6	**0**	
7	**Mollusca (snails, limpets and mussels)**									WHPT-ASPT and WHPT-Ntaxa are calculated at bottom of table
8	Neritidae	6	6.4		6.4	6.5	6.9	6.9	**0**	
9	Viviparidae	6	5.7		5.2	6.7	6.7	6.7	**0**	BMWP = Biological Monitoring Working Party index, superseded by WHPT
10	Valvatidae	3	3.2		3.3	3.1	2.7	2.7	**0**	Presence only WHPT is for estimating WHPT ASPT from old data without abundances
11	Hydrobiidae	3	4.2		4.1	4.2	4.6	3.7	**0**	
12	Bithyniidae	3	3.7		3.6	3.8	3.3	3.3	**0**	
13	Physidae	3	2.4		2.7	2.0	0.4	0.4	**0**	
14	Lymnaeidae	3	3.3		3.6	2.5	1.2	1.2	**0**	
15	Planorbidae (excl. *Ancylus* group)	3	3.1		3.2	3.0	2.4	2.4	**0**	
16	*Ancylus* group (= Ancylidae)	6	5.7		5.8	5.5	5.5	5.5	**0**	Comprises *Ancylus* and *Ferrissia* (*Pettancylus*)
17	Acroloxidae	6	3.6		3.6	3.8	3.8	3.8	**0**	
18	Unionidae	6	5.3		5.2	6.9	6.9	6.9	**0**	
19	Sphaeriidae (Pea mussels)	3	3.9		4.4	3.5	3.4	2.3	**0**	
20	Dreissenidae		3.7		3.7	3.7	3.7	3.7	**0**	Excluding *Quagga*
21	**Oligochaeta (worms)**									
22	Oligochaeta	1	2.7		3.6	2.3	1.4	-0.6	**0**	
23	**Hirudinea (leeches)**									
24	Piscicolidae	4	5.2		5.2	4.9	4.9	4.9	**0**	
25	Glossiphoniidae	3	3.2		3.4	2.5	0.8	0.8	**0**	
26	Hirudinidae	3	-0.8		-0.8	-0.8	-0.8	-0.8	**0**	
27	Erpobdellidae	3	3.1		3.6	2.0	-0.8	-0.8	**0**	

Figure 3.5
Screenshot of part of the Excel calculator tool for WHPT indices

2.5 Correcting bias caused by analytical error

Analytical quality affects our assessment of river invertebrate quality, particularly when we use taxonomic richness as a measure of quality, such as WHPT NTaxa (the number of WHPT-scoring taxa). We have measured these errors in independent audits. Audit 'gains' caused by taxa present in a sample in low numbers or as a single specimen that are not noticed in the sorting tray by the laboratory analyst, are much more common than 'losses', errors where a taxon is recorded that is not actually present in the sample. The effect of poorer analytical quality is therefore not random but biased, almost always resulting in fewer taxa being recorded and therefore poorer quality being deduced. To ensure that differences in analytical quality are not mistaken for differences in environmental quality, we make a 'bias correction' to the observed data, based on the result of audits in which sorting errors were measured independently, before using it to determine WFD status.

The bias correction for WHPT NTaxa is relatively simple. We add the net gains (gains minus losses) recorded in audits to WHPT NTaxa that the laboratory analysts record for every sample. The net gains are based on the annual average for a laboratory.

Observed WHPT NTaxa		Bias (= net gains)		Corrected WHPT NTaxa
20	+	1.68	=	21.68

For WHPT:
Bias = net gains (net additional taxa revealed by the audit)
= mean net effect of errors on WHPT NTaxa
= (mean gains of WHPT-scoring taxa) – (mean losses of WHPT-scoring taxa)

The status class boundaries for WHPT NTaxa (Table 3.7) assume that bias correction has been applied to the observed index values.

WHPT ASPT is also corrected for analytical error but the correction is far smaller because the effect of errors on this metric are less biased. The mean net effect of error (= bias) on ASPT arises because of the WHPT values of the taxa that are most prone to error. If only sensitive taxa that have high WHPT values were more prone to error than low value taxa, greater analytical error would lead to lower ASPT values, and vice versa. Fortunately, this is not the case and the taxa causing most errors do not all have high or low values. The bias correction for ASPT is calculated by RICT2 (River Invertebrate Classification Tool) software (see **Section 4.4**), using WHPT NTaxa and WHPT NTaxa bias for the samples.

Correcting for bias does not completely remove the effect of laboratory errors but it ensures that the bias in observed values match the bias in values predicted by RIVPACS. They cancel out in EQRs (**Chapter 1, Section 5.6 Biological Quality Elements**) because the audit was undertaken by the same team that analysed the RIVPACS reference samples, so after bias correction, the remaining bias in observed results caused by the auditors' error equals the bias in the reference values. RICT2 also uses bias results with information about sampling error to determine the overall imprecision in EQRs that are used in Monte Carlo simulations to determine probabilities of class (**Section 4.3**).

3

RIVPACS PREDICTIONS AND REFERENCE

3.1 The confounding influence of natural variation

3.1.1 The problem

Different natural invertebrate communities are found in different types of stream, and biotic indices vary as much between different natural communities as they do because of pollution and other forms of damage caused by human activity. A poor index value could be caused by human pressures or by natural conditions.

This is why biotic indices cannot be used directly to assess environmental quality across different rivers. They are fine for comparing similar sites on the same watercourse – for example, comparing conditions upstream and downstream of a discharge to assess its impact. See figure 3.6.

Figure 3.6

Results from the 1980 national river survey of England & Wales showing BMWP-scores from each monitoring site. There is no way to distinguish low scores caused by harsh natural conditions from those caused by poor environmental quality.

The following example explains why this is so. Figure 3.7 shows two streams providing very different natural habitats but of broadly similar environmental quality in terms of human impact.

Good quality mountain stream

WHPT ASPT 7.14
WHPT NTaxa 21.4

Good quality chalk stream

WHPT ASPT 4.17
WHPT NTaxa 41.7

Figure 3.7

Two different streams of similar environmental quality but supporting very different invertebrate faunas, resulting in different values of biotic indices

The mountain stream is a physically stressful environment with very fast and flashy flow, unstable rocky bed, and it is relatively cold. Its low productivity provides little food for invertebrates. Its hard geology causes greater acidity in which toxic metals from the rock are present in more bioavailable forms.

Few species are able to live in mountain streams and this is reflected in a relatively low value of WHPT NTaxa. The larger chalk stream is more benign with a more stable flow and riverbed, and its higher productivity provides plenty of food. It supports more ecological niches and therefore more taxa. WHPT NTaxa in the chalk stream is twice that in the mountain stream. The turbulent water in the mountain stream is well aerated and shallow, so oxygen is always saturated and taxa that are very intolerant of lower oxygen concentrations can survive. The chalk stream has smoother flow and is deeper, so re-oxygenation from the atmosphere is slower than in the mountain stream. Its luxuriant emergent and submerged vegetation causes oxygen to become super-saturated during the day, but when photosynthesis stops at night, the oxygen concentration falls dramatically because the plants continue to respire. Animals living in chalk streams therefore have to be able to survive with less oxygen and those taxa tend to have lower WHPT values because less oxygen is also associated with organic pollution. There is a greater organic loading in the chalk stream because of its higher productivity, and this supports a much larger community of species that can tolerate the low oxygen concentrations. As a result, the value of WHPT ASPT is substantially lower in a chalk stream than it is in a mountain stream of similar quality.

The example above is extreme, but even within the same river catchment, natural differences in invertebrate communities can cause significant variations in the values of biotic indices independent of the impacts of human activity, reflecting the natural change in character of a river from headwaters to the sea or between catchments on different geologies. Even small differences in index values caused by natural environmental conditions can be important, because the differences between Good and Moderate WFD status, ie between communities passing and failing environmental quality objectives, are relatively small.

This phenomenon affects all biotic indices and metrics. Table 3.3 shows the range of values for a selection of biotic indices from the RIVPACS reference database from High to Good (best available) quality sites.

Table 3.3

Variation of a range of indices between stream types, illustrated by the minimum and maximum values in the RIVPACS reference database, all sites being high or best available quality

Index	Min	Max
TL1 NTaxa	3	46
TL1 ASPT	3.6	8
TL2 WHPT NTaxa (AbW,DistFam)	3	56
TL2 WHPT ASPT (AbW,DistFam)	3.1	9.4
TL5 WFD AWIC(Sp)	0	14
TL4 MetTol	36.3	58.1
TL3 LIFE(Fam) (DistFam)	4.6	9.5
TL4 LIFE(Sp)	4.8	9.7
TL3 PSI(Fam)	0	100
TL4 PSI(Sp)	0	100
TL3 E-PSI(fam69)	0	100
TL4 E-PSI(mixed level)	0	100
TL2 08 Group ARMI Score	1	26

Coenagrionidae sp. juvenile

3.1.2 The solution: RIVPACS community prediction

In October 1977 the River Communities Project started to investigate whether the natural invertebrate communities in different streams were each unique or whether the same communities were found in any river where the same environmental conditions existed.

Invertebrates were sampled from a wide range of unpolluted rivers across the UK and at these sites a wide range of environmental parameters were measured. To ensure comparability, standard methods for sampling and analysis were devised and these are the standard methods described in **Chapter 2**. The invertebrate communities were characterised by their species composition and abundances. Because the communities change throughout the year, samples were collected in spring, summer and autumn. All macroinvertebrate taxa in the samples were identified to species or as far as was practicable at the time, and their abundances recorded.

The key conclusion was that similar communities existed wherever environmental conditions were the same. If you know the environmental conditions, you can predict the invertebrate community.

After identifying the combination of environmental parameters that most closely distinguished the invertebrate communities, a predictive model was developed, known as **RIVPACS (River Invertebrate Prediction and Classification System)**.

Early developments of RIVPACS are described in Wright *et al.* (2000). [44] Using the current version of RIVPACS to predict river communities, ie species composition, is described in **Chapter 5 Section 3.2.1**.

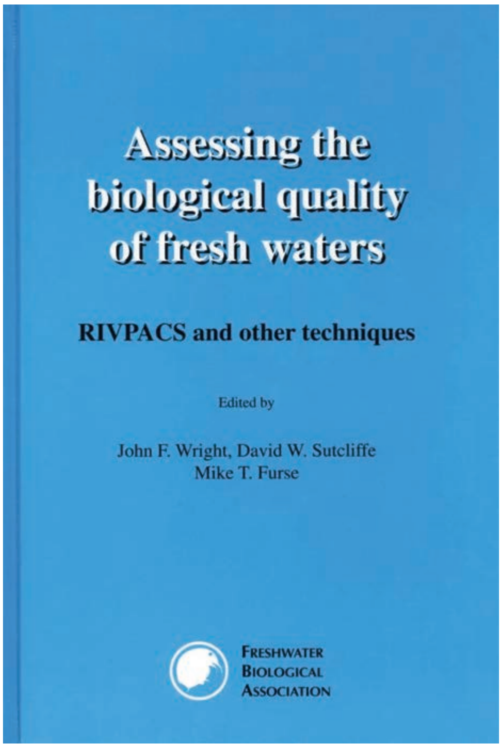


Figure 3.8
Front cover of Wright *et al.* (2000), the key reference for RIVPACS [44]

3.1.3 RIVPACS prediction of indices

If you can predict the invertebrate community (ie species and their abundances) using RIVPACS, you can predict the value of biotic indices. Indeed, early versions of RIVPACS did that literally by calculating the predicted value of indices from the predicted list of species (or the families that they belonged to) and their abundances. The current version of RIVPACS does this directly, see **Section 3.2**.

In 1979, the Biological Monitoring Working Party concluded that predictions from RIVPACS could be used to overcome the problem of reporting invertebrate quality in different rivers in a comparable way for the national river quality surveys. A biotic index expressed as the fraction of the value that it should have under natural conditions would be comparable regardless of the impact of river type on the actual value of the index. The same proportional reduction in index value represents the same level of damage, whatever the actual index value or river type. Because the River Communities Project was based on unpolluted 'natural' rivers, chosen to be the best examples of their type, RIVPACS would be capable of predicting the value of biotic indices expected under natural, un-impacted, conditions. Because the environmental data that was collected described the natural abiotic conditions of the site, predictions from RIVPACS are site-specific.

Biotic indices expressed as the proportional fraction or ratio of observed value to predicted value are known as O/E values or EQIs (Environmental Quality Indices):

$$\text{EQI} = \frac{\text{observed value of index}}{\text{value of index predicted by RIVPACS}}$$

The same value of EQI represents the same degree of degradation in the biological community, regardless of stream type.

Unfortunately, RIVPACS was not sufficiently developed for it to be used for the 1980 national river quality survey, the results of which were reported as plain BMWP-scores (Figure 3.6). It was not until the 1990 survey that RIVPACS could be used.

3.1.4 River invertebrate classifications

Having devised the EQI format for biotic indices that is comparable across all sites regardless of location or stream type, EQIs could be used to devise a simple classification that was also comparable between site and stream types. Classifications are even simpler ways to communicate the results of river monitoring surveys to river managers and the general public. River quality classification was used to report the state of the river environment in the 5-yearly national river quality surveys, and each quality was represented by a different colour on the map.

Biological quality was split into 4 grades in the 5M classification (5M = 5th model tested based on medians, Figure 3.9) used from 1990 to 1995 (National Rivers Authority, 1994).[45] The median value of the EQI amongst RIVPACS reference sites (EQI = 1.0) was used as an anchor point. The first national class boundaries were then set at equal distances representing 5% of the total distribution for BMWP ASTP, but 10% for BMWP NTaxa and score, to reflect the greater imprecision in those metrics: BMWP ASPT and BMWP NTaxa being analogous to their WHPT derivatives described in **Section 2.2**.

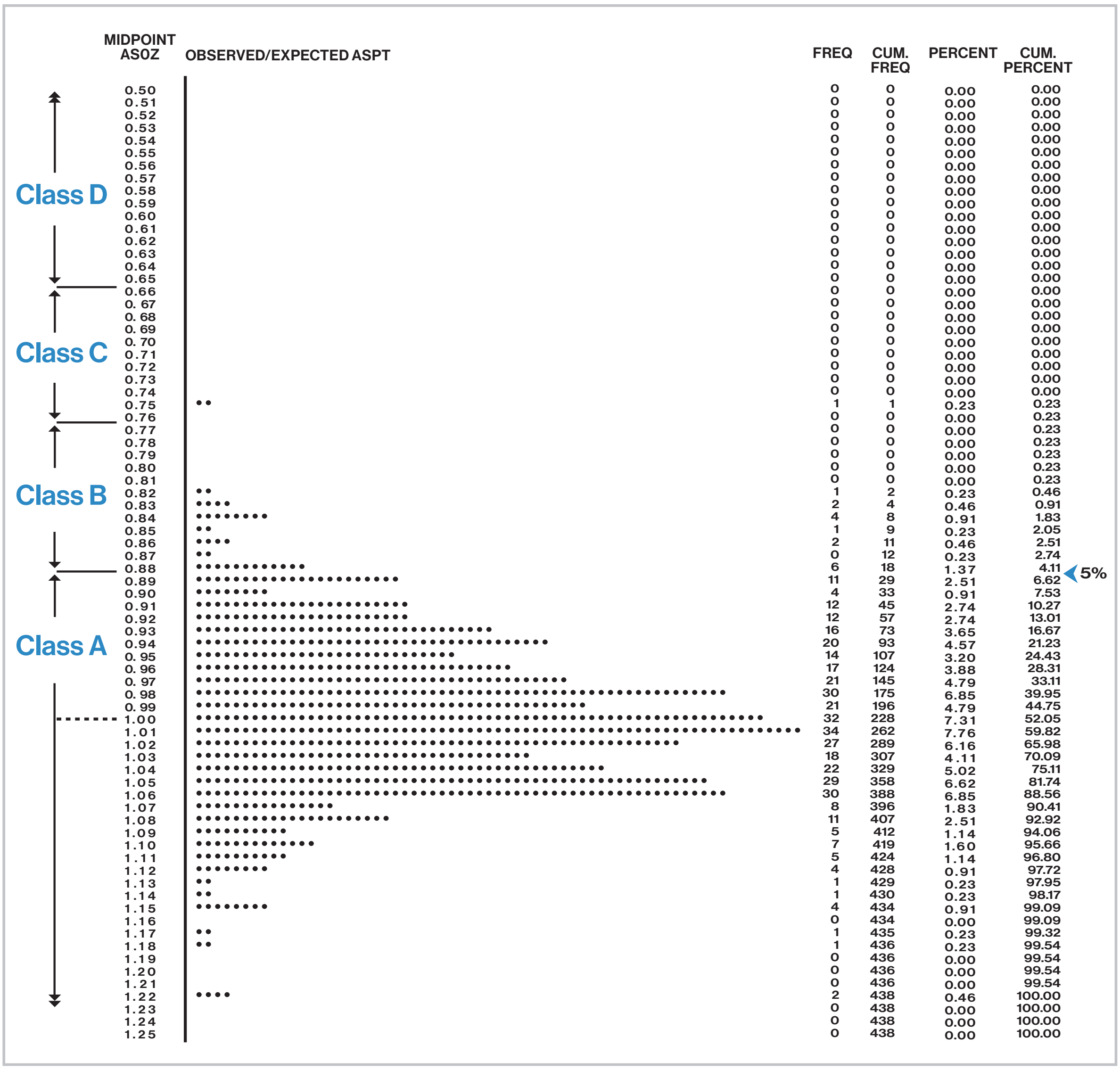


Figure 3.9

5M classification showing location of class boundaries as equal distances from the median, in this case for BMWP ASPT [45]

The 5M classification (also known as National Rivers Authority, NRA, classification) was revised in 1995 to become the biological General Quality Assessment (GQA) classification. This took advantage of an improved version of RIVPACS (RIVPACS III) based on an enhanced reference dataset with many more reference samples, particularly from headwaters. The GQA classification comprised 6 classes to give better resolution. The class boundaries were initially set using the 5M approach, but then adjusted to better reflect operational requirements across the country. The class boundaries were now based on BMWP ASPT and BMWP NTaxa only, as it was realised that BMWP-score was redundant. The development of this classification is described in Wright *et al.* (2000) [44] and it was used throughout the UK until 2006.

The next major refinement of river invertebrate assessment was the result of a new version of RIVPACS (RIVPACS IV) (Figure 3.12) and refinements of the concept of reference, and these changes were instigated with the introduction of the Water Framework Directive (WFD) in 2006. [46] The 5M approach for setting the initial class boundaries was used again, with the median value representing the boundary between Good and High status. These boundaries had proved to be robust and practical in the previous 5M (NRA) and GQA classifications. However, the boundaries were also checked against pressures, with the Good/Moderate boundary representing the point where sensitive indicators become more numerous than tolerant indicators. The final class limits were modified to suit operational experience – in this case some of the class boundaries for NTaxa were adjusted to reflect operational experience in the Scottish Highlands. RIVPACS IV involved a modest refinement of the reference dataset.

Separate RIVPACS models for the Scottish Highlands and Islands were incorporated in the GB model and a few reference sites representing sites that were of a poorer environmental quality were removed. Because the quality of RIVPACS reference sites varied from relatively pristine in the Scottish Highlands to heavily used and populated in the English lowlands, the environmental quality that RIVPACS predictions represent also varies.

For WFD status classification, the predictions were standardised to a particular environmental quality referred to as 'reference' (**Chapter 1 Section 5.7**). Standardising the predictions also standardises the EQIs that are derived from them, and when they are standardised to reference state they are known as EQRs (Environmental Quality Ratios) – see also **Chapter 1 Section 5.6** Biological Quality Elements. This refinement improves the comparability between classifications of sites from different river types.

$$\text{EQR} = \frac{\text{value of biotic index observed in samples}}{\text{reference value of biotic index}}$$

Agabus sp. larva

Corixa punctata adult

Goera pilosa larva

Figure 3.10

Front cover of Davy-Bowker *et al.* (2008) describing the development of RIVPACS IV and RICT (34)

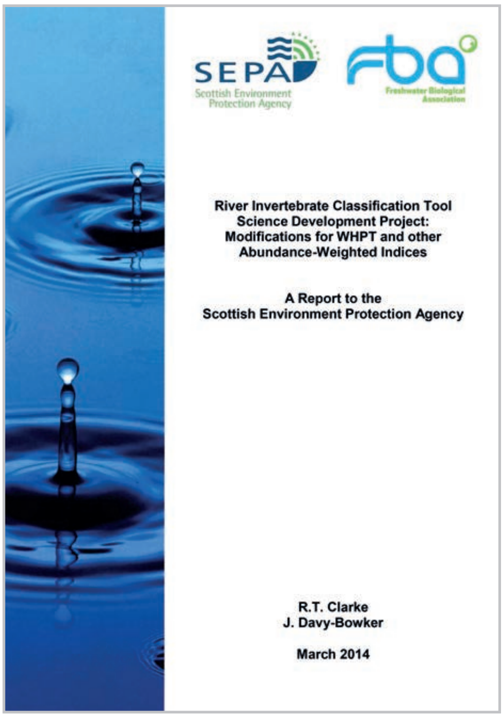


Figure 3.11

SNIFFER Project WFD72c final report, Clarke & Davy-Bowker(2014) covering development of RICT to incorporate WHPT status classification (47)

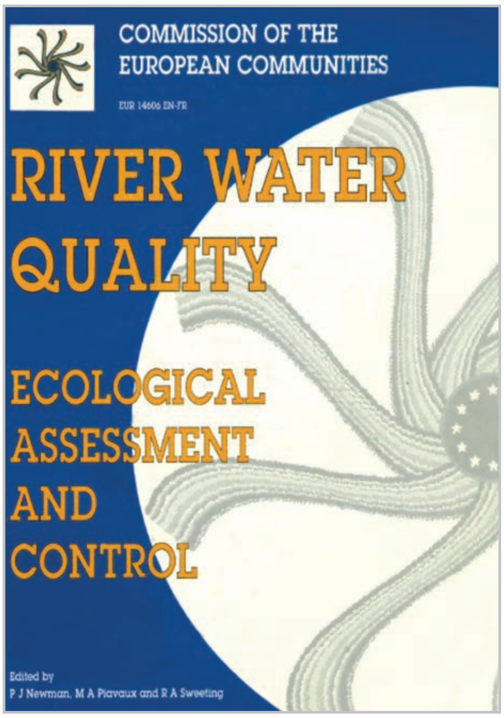


Figure 3.12

Front cover of the proceedings of a conference held in Brussels in December 1991 at which the RIVPACS approach was introduced for the proposed European directive that became WFD (46)

The most recent refinement to the river invertebrate classification occured in 2016 and replaced the classification metrics based on BMWP indices by equivalent metrics based on WHPT indices (Figure 3.11). The 5M approach to setting initial class boundaries was used, with boundaries for NTaxa again slightly adjusted. The remaining sections of this chapter describe in more detail the steps to deriving the current classification.

RIVPACS does far more than predict reference values for river status classification. It can predict the natural invertebrate community (the species composition and the abundance of each species) found in any permanently flowing stream or river in the UK, so it is a far more adaptable tool than those developed for other quality elements, particularly plants and algae. It recognises the different natural communities in terms of their species composition (its original aim) and not just the intensity of one or more environmental pressures to which the classification metric is supposed to respond. It is therefore relatively easy to get RIVPACS to predict the un-impacted or reference value of any index. This 'over-engineering' means that it is a far more capable predictor than tools designed only to predict reference values for particular indices or the single pressures to which they are assumed to respond.

The success of RIVPACS and the reference approach to classification in the UK led to its adoption in the Water Framework Directive for all biological quality elements in all water body categories across Europe (Figure 3.12).

> The success of RIVPACS and the reference approach to classification in the UK led to its adoption in the Water Framework Directive for all biological quality elements in all water body categories across Europe.

3.2 How RIVPACS predicts an index

RIVPACS can predict the value of a wide range of biotic indices and the abundance and probability of occurrence of each species and family of invertebrates, from the 12 environmental parameters listed in Table 3.4.

Table 3. 4

Environmental variables used by RIVPACS for prediction. RICT (River Invertebrate Classification Tool) converts many to logarithms and converts alternative variables to discharge or alkalinity. Note that Northern Ireland models don't use mean air temperature or temperature range.

Map data	Sample data	
OS grid reference	**Width**	
Altitude	**Depth**	
Distance from source	**Substrate**	% clay/silt
Slope		% sand
Discharge or velocity from sample data		% gravel/pebbles
		% cobbles/ boulders
	Velocity (if discharge is not available from map data)	
	RICT will calculate mean particle size from the substrate data	
RICT will calculate the following internally from the OS grid reference	**Geochemistry**	
	One of:	
→ mean air temperature	alkalinity	total hardness
→ air temperature range	calcium	conductivity
→ latitude		
→ longitude		

The current RIVPACS IV model for Great Britain (GB) is derived from invertebrate samples from 685 sites and the Northern Ireland (NI) model on samples from 110 sites. These reference sites cover the full range of flowing waters and were chosen to be the best available (most natural). Their invertebrate communities were classified into different types (known in RIVPACS as *end groups*) based on similarities in their composition: 43 end groups in GB and 11 in NI. These end groups are associated with different environmental conditions. To make a prediction for a new site, RIVPACS first determines the probability of the site belonging to each of the end groups, based on the similarity of the values of the 12 environmental parameters at the new site with the average value between the reference sites belonging to each end group. RIVPACS uses multivariate ordination to predict the probability of a new site belonging to each end group, explained in more detail in Wright *et al.* (2000). [44]

It then multiplies the average value in each end group of whatever is being predicted (the value of a biotic index, the proportion of sites where the taxon is present, or the abundance of a taxon) by the probability of the site belonging to that end group. The sum of these products across all end groups is the prediction of the value of that metric (biotic index, probability of the taxon occurring or its abundance) at the new site. The process is illustrated by a simplified example in Table 3.5. Average values of abundances or biotic indices in each end group are listed in the RIVPACS database.

Table 3.5

Explanation of how RIVPACS calculates a prediction (in this case, ASPT index). The example is a simplified model with only 5 end groups, A-E. RIVPACS IV GB model has 43 end groups; NI model has 11 end groups.

End group	Probability of site being in end group (p)	Average of values of ASPT in reference sites in the end group ×	Contribution of end group to prediction of ASPT =
A	0.3	6.3	1.89
B	0.2	8.2	1.64
C	0.4	6.0	2.4
D	0.08	5.4	0.432
E	0.02	5.3	0.106
	$\Sigma = 1$		
	Predicted value of ASPT at site =		**Σ = 6.47**

3.3 Predicting reference values for WHPT indices

Not all RIVPACS reference sites are in WFD reference state, which is somewhere in High status. The High/Good boundary is the median quality of RIVPACS III+ reference sites, so half of them must be in Good status or worse (Figure 3.9, Figure 3.15 and Figure 3.16). The environmental quality that RIVPACS predictions reflect therefore varies according to the environmental quality of the reference samples on which each prediction is based. To ensure comparability between predictions, and between classifications based on them across different river types, we must standardise the predictions to ensure that they always relate to the same quality (a process that we call *adjustment*) and then ensure that standardised quality is the WFD reference state (in a step we call *conversion*).

RICT2 can transform RIVPACS predictions for WHPT ASPT and WHPT NTaxa into reference values that are needed for WFD status classification. If you want to create WFD-compatible class boundaries for other indices, you will need to follow the process of adjustment and conversion described below.

3.3.1 Adjustment of predictions

In adjustment, we standardise the predictions at the environmental quality represented by High/Good status boundary, defined as the median value of the index or metric across all RIVPACS III+ reference sites (GB + NI + Highlands and Islands), as explained in **Section 4.2.** This median is the environmental quality that RIVPACS predicts most reliably. The adjustment acknowledges that WFD reference state is a specific environmental quality but not what that quality is. The next step, conversion (**Section 3.3.2**), converts this standardised RIVPACS prediction to the environmental quality defined by reference.

Predictions for individual sites are adjusted by adding or subtracting an amount that depends on how far the environmental quality of their predictions deviate from the High/Good boundary. We know the value of EQRs representing each class boundary (see **Section 4.2**), and therefore we can interpolate, for any observed value, the amount by which a prediction will vary for any particular deviation in quality.

We can calculate the environmental quality to which a prediction relates from biologists' assessment values (Figure 3.14). These quantify the environmental quality for each RIVPACS reference site when the reference samples were collected in terms of their deviation from the High/Good boundary quality, in intervals of half-a-status class (see key in Figure 3.14). Biologists' assessment values were estimated for every RIVPACS reference site using contemporary environmental data (mostly chemical) and assessments by the biologists who collected the reference samples, based on descriptive definitions covering water quality, land use and, as a backstop, biological indicators. They were also screened against either the North Europe or the Central-Baltic Geographical Intercalibration Group (GIG) type-specific chemical, flow and land cover criteria for defining WFD reference state, depending on their hydro-ecoregion (Wasson *et al.* 2006 [48] and Figure 3.13). The biologists' assessment value for every RIVPACS reference site is recorded on the RIVPACS reference database that can be downloaded from the RIVPACS/RICT web pages at **https://www.fba.org.uk/rivpacs-and-rict/rivpacs-rict-resources**

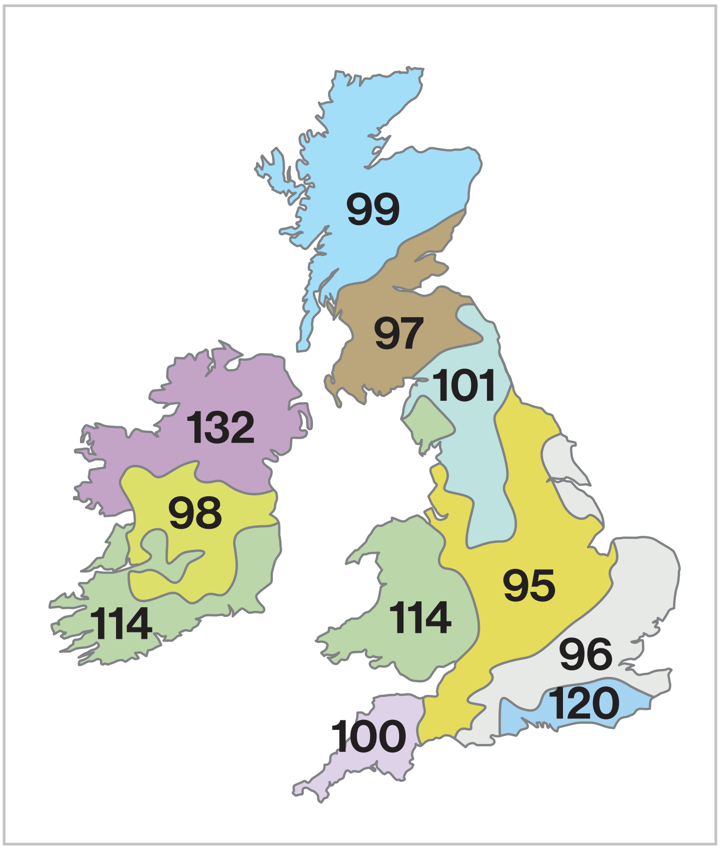
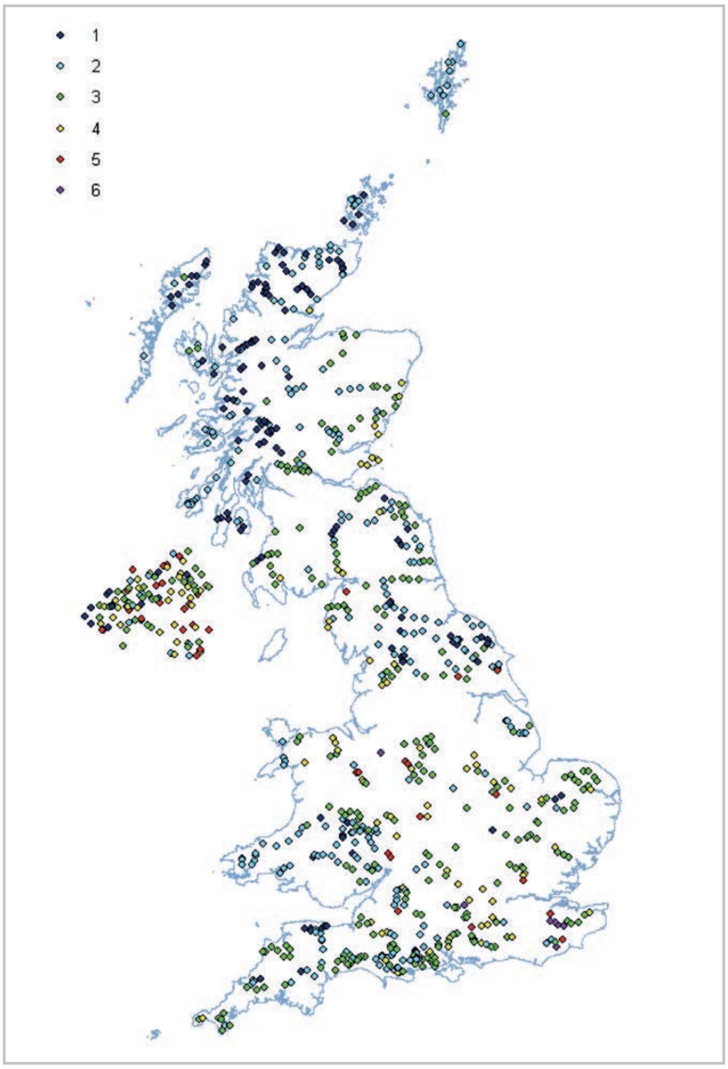


Figure 3.13

Hydro-ecoregions of the British Isles. Hydro-ecoregions 95, 96 and 120 belong to Central-Baltic GIG and 97, 98, 99, 100, 101, 114 and 132 belong to Northern GIG, from Wasson *et al.* (2006). [48]



Key	Biologists' assessment value	WFD quality
◆	1	top of High
◆	2	mid-High
◆	3	High/Good
◆	4	mid-Good
◆	5	Good/Moderate
◆	6	worse

Figure 3.14
Biologists' assessment values

The environmental quality which any RIVPACS predictions for a site relates to is the sum of the products of biologists' average assessment values in each RIVPACS end group and the probability of the site belonging to that end group. The calculation is analogous to that used for prediction in Table 3.5. This 'predicted' biologists' assessment value indicates the environmental quality of the prediction in terms of its deviation from the High/Good status boundary. We can therefore adjust the predicted value of the index by adding an amount representing its deviation from High/Good (its predicted biologists' assessment value), scaled according to the status class interval. This is shown in Figure 3.15.

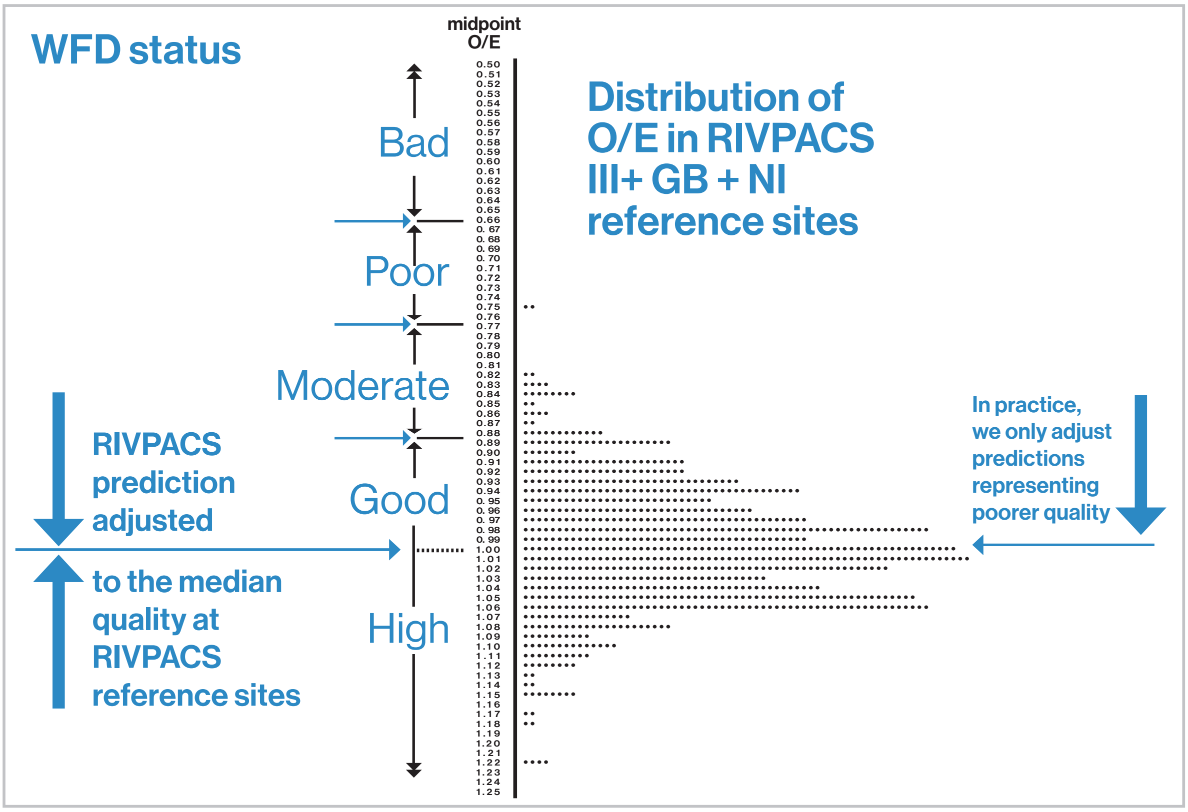


Figure 3.15

Adjustment to remove the effect of variation in the environmental quality of RIVPACS predictions by adjusting them to the High/Good boundary. In practice, we only adjust poor quality predictions because it is not possible to differentiate differences in environmental quality from the natural variations in invertebrate communities in High status.

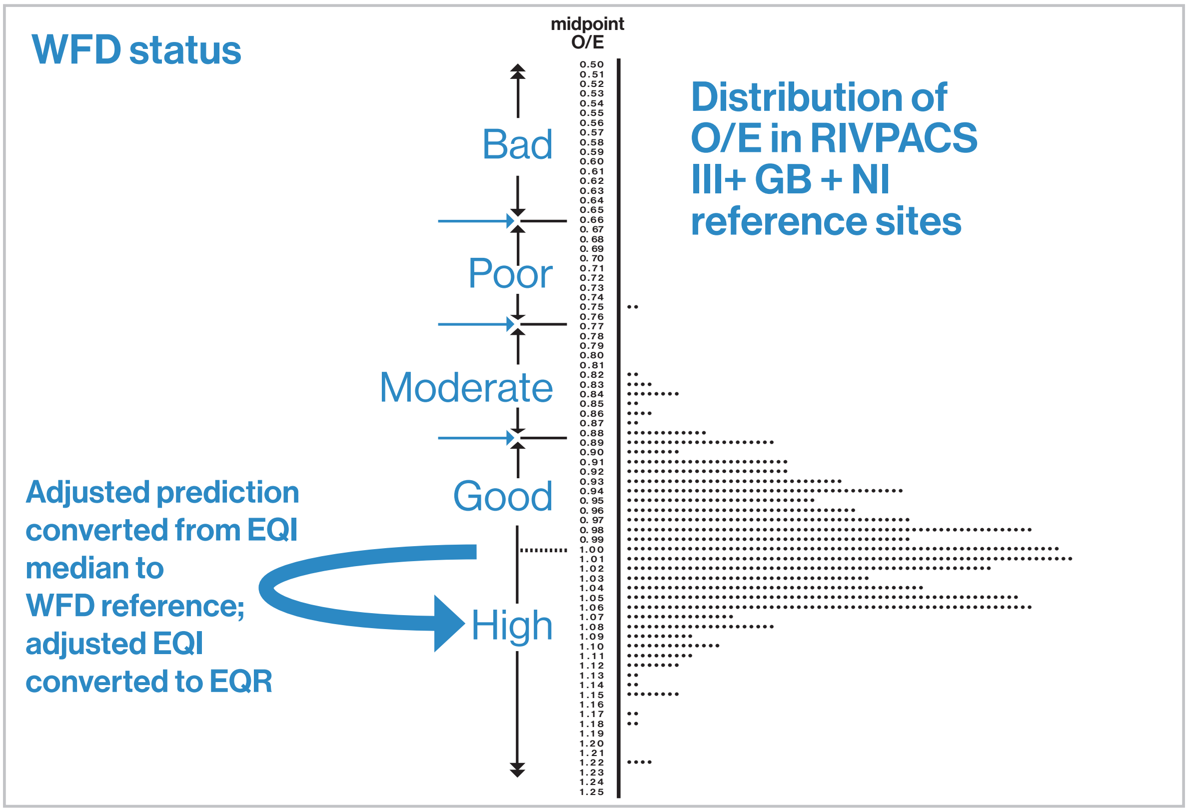


Figure 3.16

Conversion of adjusted prediction from High/Good boundary to WFD reference value

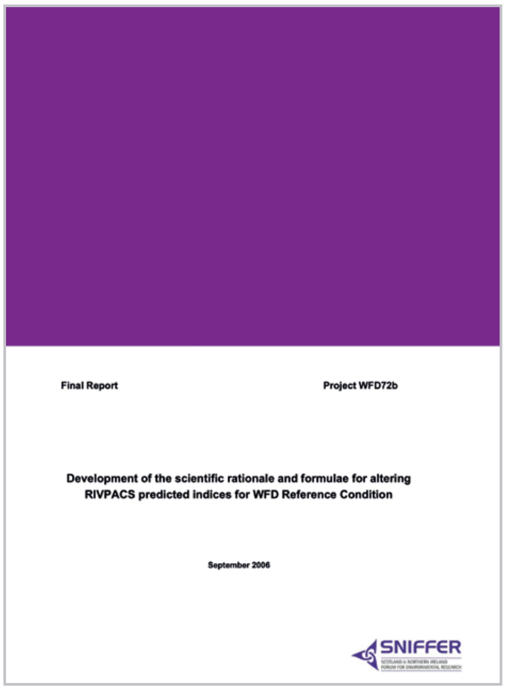
3.3.2 Conversion of adjusted predictions to reference values

This step is to convert the adjusted prediction from a value representing the High/Good boundary to a value representing the WFD reference condition – ie to a WFD reference value (Figure 3.16). The WFD reference value of an index is defined as its median value at WFD reference sites (sites that comply with WFD reference criteria) and it is somewhere in High status. The conversion factor is the ratio of the median value of the index across all RIVPACS reference sites (both GB and NI models) to the median of the value from RIVPACS reference sites that are also in WFD reference condition.

Ideally, we would have derived separate conversion factors for each end group, but that was not possible because, in some end groups, none of the reference sites were in reference condition. The effect of conversion is relatively small, but it does ensure that, as a whole, the UK's river invertebrate classification is equivalent to those used in other countries.

The adjustment is described in a report by Clarke and Davy-Bowker J (2006). [49] This report can be downloaded from the SNIFFER website **https://www.sniffer.org.uk**/ (search for WFD72b).

RICT2 can only predict reference values for the few indices used for the UK's general degradation status assessment. To help derive adjustment and conversion parameters, Clarke & Davy-Bowker also produced an adjustment calculator in Excel and a Minitab macro called FitM4. The spreadsheet can be downloaded from the same web address as the report. The Minitab macro enables the Excel prediction adjustment spreadsheet to be used to adjust any biotic index or metric by calculating new coefficients for it. The original Minitab Macro was based on RIVPACS III, which has a different number of end groups, so it has been modified for RIVPACS IV. The modified macros and instructions for using it are available from John Murray-Bligh.



Final Report Project WFD72b

Development of the scientific rationale and formulae for altering RIVPACS predicted indices for WFD Reference Condition

September 2006

SNIFFER

Figure 3.17

Front cover of Clarke & Davy-Bowker, SNIFFER project WFD72b [49]

Paraleptophlebia submarginata

4

RIVER INVERTEBRATE STATUS CLASSIFICATION IN THE UK

4.1 Overview

There are currently two statutory river invertebrate quality classifications in the UK: the general quality assessment described here and another specifically for acidification.

The acidification classification is only used to assess river status in Scotland and Wales, although it is used for investigations in England. Reference values for this classification are not based on RIVPACS but on the susceptibility of sites to acidification from acid deposition. It is described in **Section 6.**

The River Invertebrate Classification Tool Version 2 (RICT2) software, released in 2020, not only implements the current version of RIVPACS (RIVPACS IV) but also calculates the general river invertebrate status classification. The software, user guides, reference database and development reports are available from the RICT website at: **https://www.fba.org.uk/other-scientific-collaborations/rivpacs-and-rict** The software is written in R and can be downloaded from the website or run as a web application.

You can download the RIVPACS reference database from: **https://www.fba.org.uk/rivpacs-and-rict/rivpacs-rict-resources** This lists the species found in all the RIVPACS reference samples and their abundances, biotic indices derived from them, and the averages of these vaues across sites in each 'end group' (known as end-group means), which RIVPACS uses in its agorithms to predict them.

The flow diagram in Figure 3.18 describes the classification process from sample collection to data analysis.

Leuctra geniculata nymph

Dinocras cephalotes

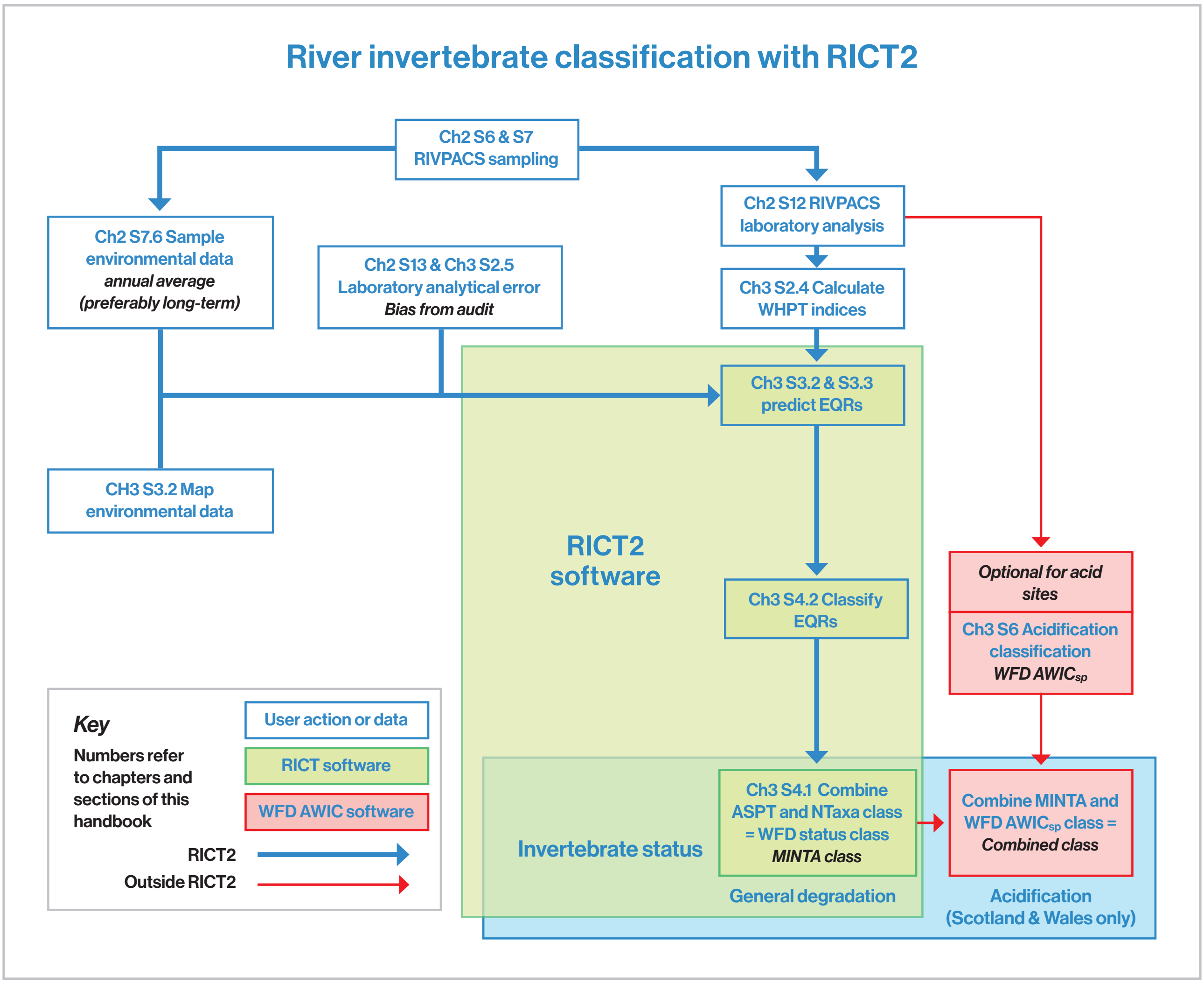


Figure 3.18

The UK's river invertebrate status classification and the Chapters (Ch) and Sections (S) in which each step is described in this book

Planorbis albus

Gammarus pulex

Caenis pusilla

The properties of the biological communities representing High, Good and Moderate status in rivers according to its benthic invertebrate fauna are described in the normative definitions in Annex V of WFD (see Table 3.6 below).

The metrics that we use to calculate the classification must reflect all aspects of those definitions with sufficient precision to differentiate the classes.

Table 3.6

Normative definitions for High, Good and Moderate invertebrate status in rivers (from WFD Annex V 1.2.1)

Status	Normative definition
High	The taxonomic composition and abundance correspond totally or nearly totally to undisturbed conditions. The ratio of disturbance-sensitive taxa to insensitive taxa shows no signs of alteration from undisturbed levels. The level of diversity of invertebrate taxa shows no sign of alteration from undisturbed levels.
Good	There are slight changes in the composition and abundance of invertebrate taxa from the type-specific communities. The ratio of disturbance-sensitive taxa to insensitive taxa shows slight alteration from type-specific levels. The level of diversity of invertebrate taxa shows slight signs of alteration from type-specific levels.
Moderate	The composition and abundance of invertebrate taxa differ moderately from the type-specific communities. Major taxonomic groups of the type-specific community are absent. The ratio of disturbance-sensitive taxa to insensitive taxa, and the level of diversity, are substantially lower than the type-specific level and significantly lower than for Good status.

These class descriptions provide the anchors for the boundary-setting protocol and links to the concept of reference and type-specific communities (**Chapter 1 Section 5.7**).

The normative definitions do not relate to particular types of pressures, only to the invertebrate communities. The classification metrics could be based simply on the degree to which the invertebrate community deviates from the reference community in terms of its composition and abundance, using a similarity index. In the author's view, that approach may be the best. However, when the Water Framework Directive was implemented, a more iterative approach was taken and all Member States, including the UK, were allowed to retain continuity with their existing and established systems by modifying them to comply with the main concepts demanded by the directive. This is why we continue to use biotic indices that relate to the sensitivity of invertebrates to pressure. The strength of large existing data sets helped smooth the transition to WFD, but at the cost of not allowing for the development of methods to recognise new environmental challenges.

In the UK, we use a combination of WHPT ASPT and WHPT NTaxa that together respond to almost all pressures (see **Section 2.2**). Invertebrate communities respond to the integrated effect of all pressures, both natural and anthropogenic. The result of this is that all biotic indices co-vary to an extent and all respond to pressures other than those that they are designed to reflect.

The UK government has set the class boundaries (Table 3.7) in statutory directions in *The Water Framework Directive (Standards and Classification) Directions (England and Wales) 2015* (download from **https://www.legislation.gov.uk/uksi/2015/1623/resources**), and the Scottish Government in *The Scotland River Basin District (Standards) Directions 2014* and for the Solway and Tweed, which have separate legislation as cross-border river basins, in *The Solway Tweed River Basin District (Standards) (Scotland) Directions 2014*, both of which can be downloaded from **https://www.gov.scot/publications/water-environment-legislation/**

The official definitions of each of these classifications are in UK TAG's (UK Technical Advisory Group) method statements, which can be downloaded from the UK TAG website: **http://www.wfduk.org/resources/category/biological-standard-methods-201**

Each river invertebrate classification metric is classified separately, and the overall river invertebrate class is determined by the 'one out, all out' principle (**Chapter 1, Section 5.1** Introduction).

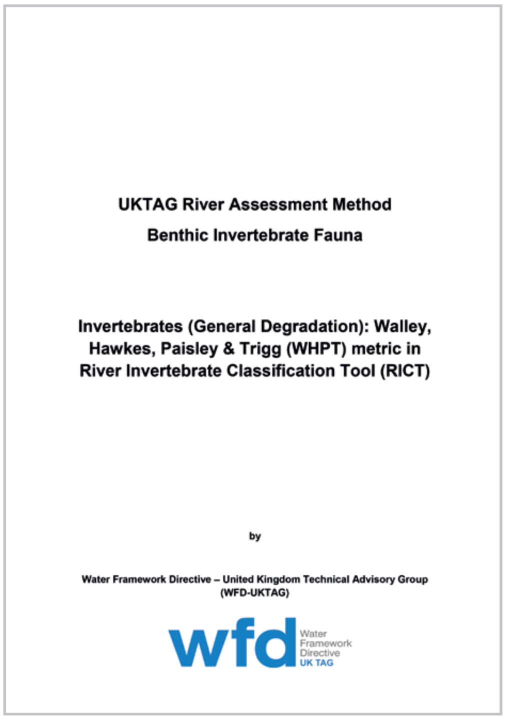
Table 3.7

The UK's official river invertebrate class boundaries (from government directions)

Class boundary	WHPT ASPT (EQR)	WHPT NTaxa (EQR)
High/Good	0.97	0.80
Good/Moderate	0.86	0.68
Moderate/Poor	0.72	0.56
Poor/Bad	0.59	0.47

Understanding how these indices work is important for interpreting the WFD river invertebrate status classification. Similarly, an understanding of how the classification metrics of all the biological quality elements relate to each other enables a much better interpretation of overall biological status. WHPT ASPT and WHPT NTaxa are described in more detail in **Section 2.2**.

Technical guidance for calculating these classifications is described in documents on UK TAG's website **http://www.wfduk.org/** (follow the link 'Biological Standard Methods', 'Rivers – Invertebrates (General Degradation)', then select 'River Invertebrates WHPT UKTAG Method Statement Dec2014.pdf' (see Figure 3.19 below).


UKTAG River Assessment Method

Benthic Invertebrate Fauna

Invertebrates (General Degradation): Walley, Hawkes, Paisley & Trigg (WHPT) metric in River Invertebrate Classification Tool (RICT)

by

Water Framework Directive – United Kingdom Technical Advisory Group (WFD-UKTAG)

wfd Water Framework Directive UK TAG

Figure 3.19

Front cover UK TAG River Assessment Methods – Benthic Invertebrate Fauna, which describes the UK's WFD status classification (50)

4.2 Setting status class boundaries for WHPT indices using RIVPACS

This section explains how the class boundaries of the river invertebrate (general degradation) classification shown in Table 3.7 were set and what you need to do to set equivalent boundaries based on other indices and metrics.

Initially, at least, we base the general degradation status class boundaries on the distribution of quality across the RIVPACS III+ reference dataset (combining the GB, NI, Highlands and Islands data sets). This is included in the RIVPACS reference database that you can download from the RIVPACS/RICT web pages at: **https://www.fba.org.uk/rivpacs-and-rict/rivpacs-rict-resources** We always use the old RIVPACS III+ data set and not that of the most recent version of RIVPACS, which we use to determine the classification, so that the boundaries always relate to the same biological quality and don't change whenever we alter the sites in the RIVPACS reference data set to improve its predictive ability, the stream types that it covers, or its geographical coverage.

Following 5M principles (see **Section 3.1.4**) the median value of O/E (observed value / predicted value) for whatever index or metric we use for classification (WHPT ASPT or WHPT NTaxa for WFD general degradation) across the RIVPACS III+ reference sites defines the High/Good boundary, with seasons combined by whatever method is used for classification: for WHPT indices, this is a mean. The remaining class boundaries are set at equal intervals away from this O/E ratio. The interval depends on the variability of the metric across the RIVPACS III+ reference sites. For ASPT, we use a 5% interval (Figure 3.20), but for NTaxa a 10% interval is used to take account of its greater imprecision. The interval that you should use for other indices depends on the standard deviation of O/E ratios across the RIVPACS III+ reference sites using Figure 3.21. The intervals are whole number percentages (ie 0 decimal places) to reflect their precision.

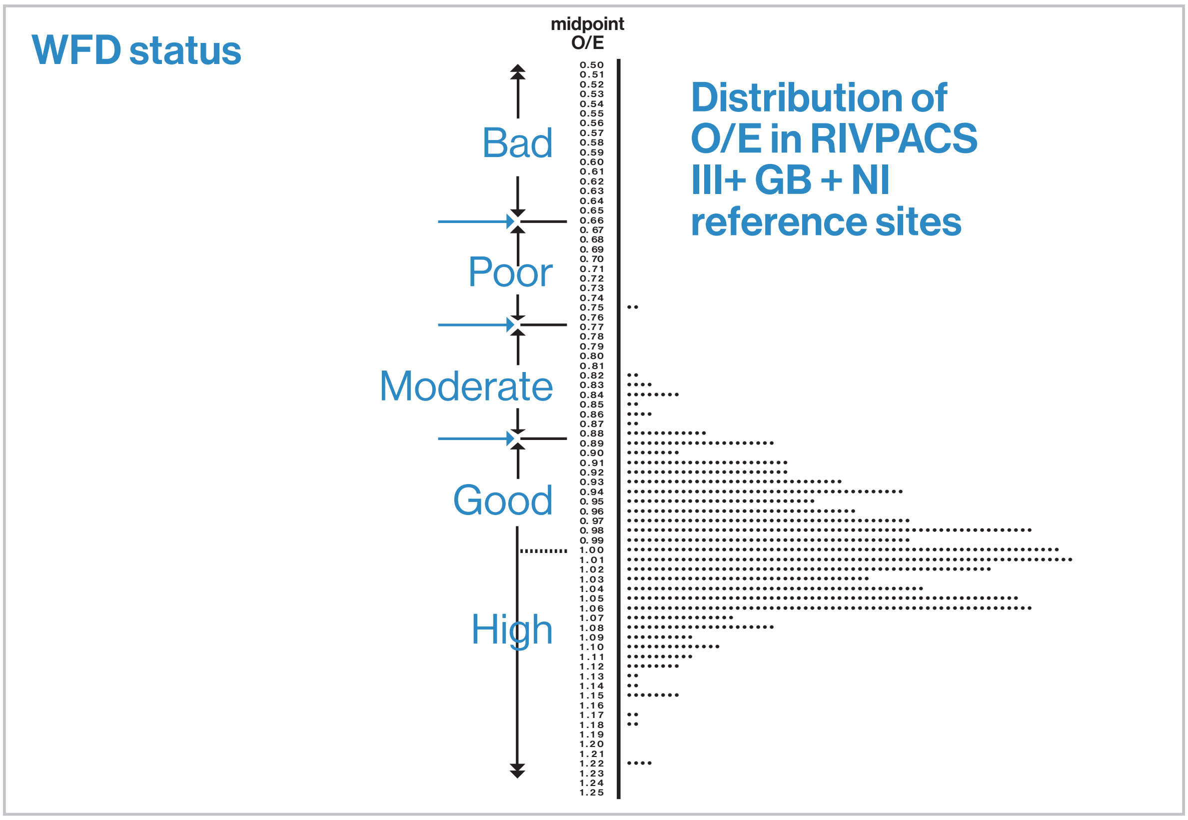


Figure 3.20

Basic model for initial WFD status class boundaries based on ASPT

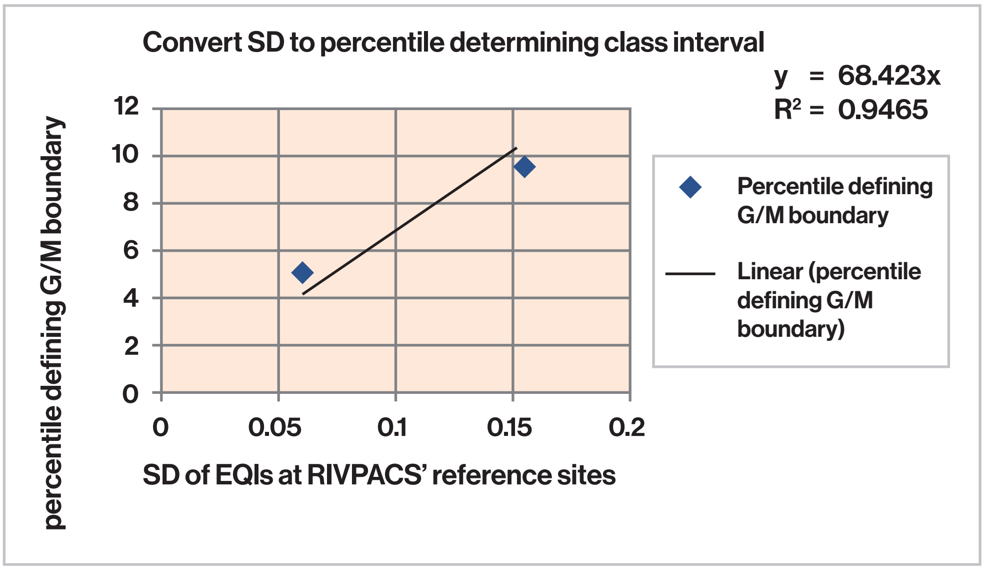


Figure 3.21

Trend line for determining the whole number percentile for determining status class intervals for an index from the standard deviation of O/E values at RIVPACS III+ GB and NI reference sites

The mathematical calculation of initial class boundaries following 5M principles ensured that the WFD class boundaries were equivalent to the GQA (General Quality Assessment) class boundaries that preceded them. The environment protection agencies, through UK TAG, amended some of the initial boundaries for NTaxa to reduce the number of sites that this index downgraded in Scotland. This process of initial mathematical calculation followed by amendment to reflect operational experience or requirements has been used to set the UK's official river invertebrate classification boundaries since 1990. The move from BMWP Ntaxa to WHPT Ntaxa caused a slight tightening of standards in Scotland because SEPA continued to apply bias values for BMWP NTaxa to WHPT NTaxa.

These class boundaries expressed as observed/expected (O/E) ratios are also known as EQIs (Environmental Quality Indices). For WFD, we must express them as Environmental Quality ratios (EQRs), which is O/reference-value. The predicted values need to be adjusted (**Section 3.3.1**) and converted to reference values (**Section 3.3.2**).

$$\text{EQR} = \frac{\text{value of metric observed in sample(s)}}{\text{value of metric at reference condition}}$$

A WFD reference value is the value of an index that we would expect at that type of site in its near natural, but not pristine, reference state (see **Chapter 1 Section 5.7 Reference Conditions**). Reference state is somewhere in High status. Reference values are usually type-specific and based on the average value of an index calculated from WFD reference sites belonging to the same type. In the UK, we predict site-specific reference values based on weighted averages of RIVPACS reference sites in different end groups (see **Section 3.2**). RIVPACS end groups are essentially abiotic river types defined by the biological communities that they support, and the weighted average reflects the fact that the types are not discrete.

Before these class boundaries could become official, they had to be agreed by all UK environmental protection agencies. This was done through WFD UK TAG. It was at this stage that minor changes were made to the boundaries for WHPT NTaxa, based on practical experience. The boundaries then had to be intercalibrated, to ensure that the High and Good status classes covered the same range of quality as in other Member States (see **Chapter 1 Section 5.8 Intercalibration**).

4.3 Probabilistic classification

The WFD requires that a high level of confidence and precision of the classifications should be achieved. This is done by taking error into account. RICT2 does this in two ways:

- RICT2 indicates the suitability of the RIVPACS predictive model to the site in question and indicates when the site is beyond the model's capability because it is not covered adequately by the typologies included in the reference samples, ie the combination of environmental parameters is different to that of any of the sites in the reference database.
- RICT2 also takes error into account to indicate the relative probability of the site being in each of the 5-status classes. This provides information about the confidence of class. Where confidence is low, you will need more evidence before taking expensive remedial action. RICT is able to provide a probabilistic classification because it uses Monte Carlo simulation to take errors into account (Figure 3.22). The resulting probabilistic classification enables RICT2 to compare two classifications statistically in order to indicate the statistical certainty that a classification has changed. These outputs are described in **Section 4.4** and information about interpreting them is included in **Section 5**.

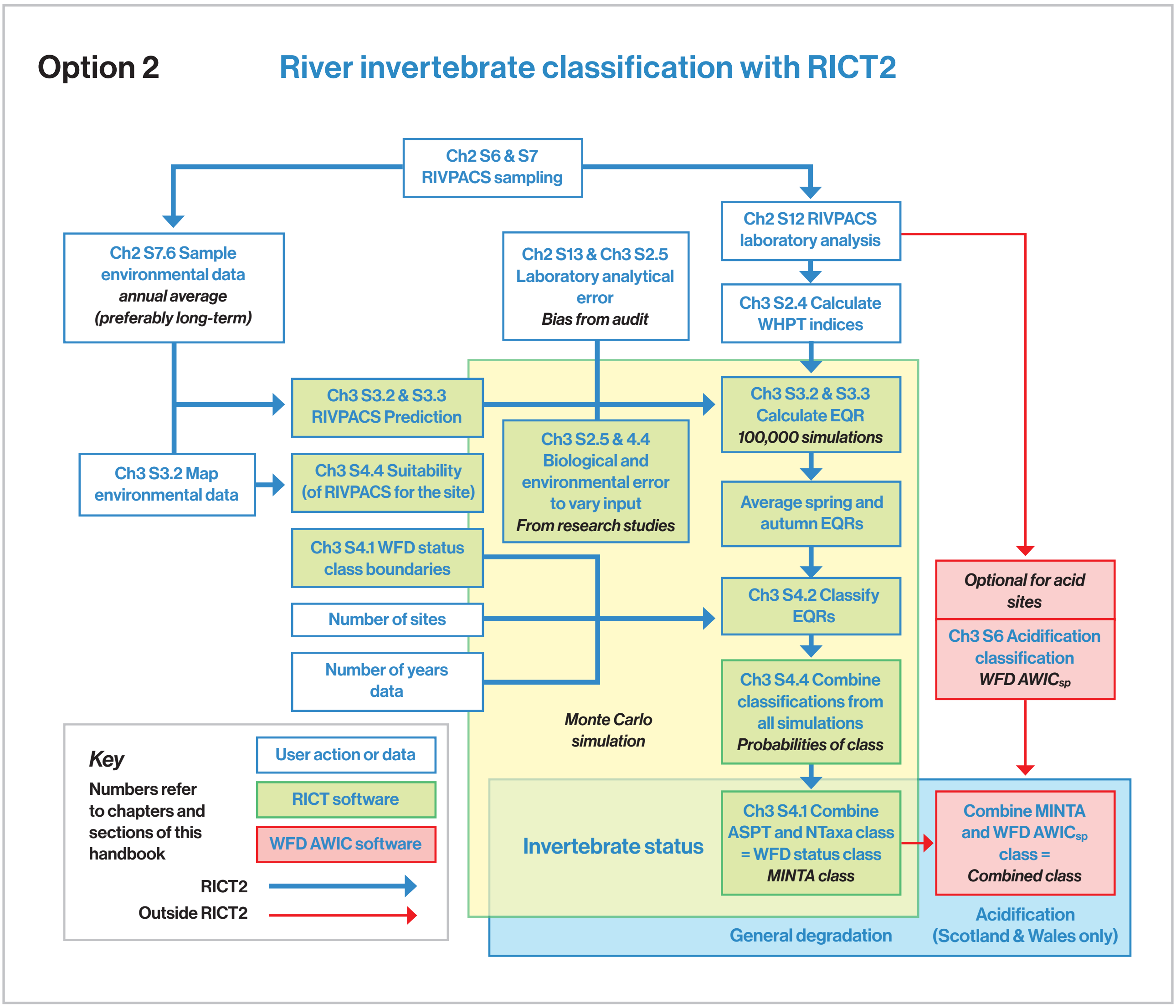


Figure 3.22

A more detailed outline of the classification process than that provided in Figure 3.18, indicating the role of Monte Carlo simulation to consider error in calculating a probabilistic classification

Gerridae

Sialis lutaria larva

Sericostoma personatum

Molanna angustata from lane at Whisby Nature Park
John Davy-Bowker

4.4 Using RICT2 software for classification in the UK

The software used for the general river invertebrate classification in the UK is River Invertebrate Classification Tool Version 2 (RICT2).

RICT2 software, which implements RIVPACS IV, is available as a web tool at: **https://www.fba.org.uk/rivpacs-and-rict/river-invertebrate-classification-tool** Figure 3.23. From here you can also download the RIVPACS database, user and technical guides and development reports, as well as the software programs themselves. These are written in R so that researchers can modify them for their own purposes.

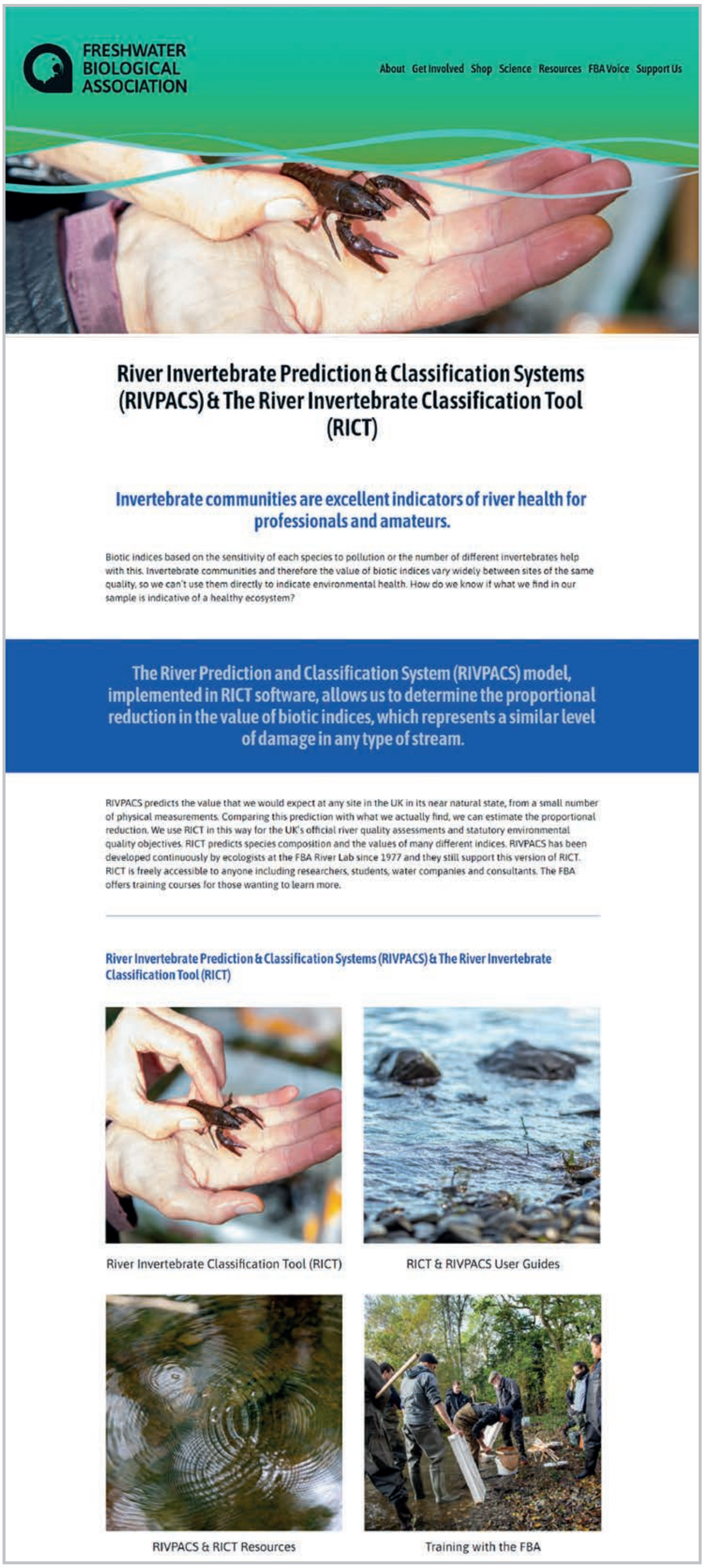


Figure 3.23

RICT2 website landing page. Note the four links at the bottom of the page to applications (the software), user guides, reports and the RIVPACS reference database, and information about training provided by the Freshwater Biological Association.

For the official status classification, you must use spring/autumn data.

RICT2 is used across Great Britain and Northern Ireland and can:

- calculate the official general degradation status for river invertebrates
- predict the value of a wide range of biotic indices
- predict the probability of occurrence of species or higher taxa and their numerical abundance and abundance category
- compare two status classifications and indicate the statistical significance of any difference, described below.

Comprehensive user guides are available from the website, together with a template for entering data and test data sets that you can use to try out the programs.

To determine the river invertebrate status, you will need the following information:

1. Biological data from your site, which must be suitable for RIVPACS, in the form of WHPT ASPT and WHPT NTaxa from spring and autumn samples collected according to RIVPACS methods.
2. The environmental data needed by RIVPACS to predict the reference value of WHPT ASPT and WHPT NTaxa, listed in Table 3.4; see also **Chapter 2 Section 7.6**. Bias value for single-season WHPT NTaxa, see **Section 2.5.**
3. Details of the methods for collecting this data are described in the Environment Agency's guides *Freshwater macro-invertebrate sampling in rivers* and *Freshwater macro-invertebrate analysis of riverine samples,* both of which can be downloaded from the user guides web page of the RICT2 website: **https://www.fba.org.uk/rivpacs-and-rict/rict-rivpacs-user-guides**

RICT2 classification programmes will produce results for the predictions of WHPT ASPT and WHPT NTaxa and for the status classification. The prediction results include the probability of belonging to each RIVPACS end group. The end groups represent the river invertebrate community types recognised by RIVPACS (43 end groups for GB and 11 end groups for NI). The suitability information relates to how similar your site is to those included in the RIVPACS database – if it is not similar, the suitability indicates that the prediction results, and therefore also the classification, are likely to be unreliable for that site. This is useful when evaluating the weight of evidence provided by the classification.

The classification results include the probability of belonging to each of the status classes, the most probable class and the EQR (observed/reference value) for WHPT NTaxa, for each season and for the average between them (spring and autumn results are combined by averaging the EQRs). These results are repeated for WHPT ASPT and then for MINTA, which is the definitive classification based on whichever index indicates the poorest quality status class (MINTA = minimum of NTaxa and ASPT).

The probabilities of class are based on the frequency of each class from 100,000 Monte Carlo simulations, each simulation varying according to the known distribution of sampling error and information from the bias value entered by the user to represent laboratory error. The final class is based on the most probable class indicated by the Monte Carlo simulations. MINTA results can seem counterintuitive – it is possible for the MINTA class to be worse than either of the classes indicated by WHPT ASPT and WHPT NTaxa. This happens occasionally, particularly when the probabilities of belonging to two classes are not very different.

Pairs of classification results can be compared and the statistical significance of any differences estimated using RICT2's Compare program. Classifications from the same site in different years or from different sites in the same (or different) years can be compared. This is useful for checking that differences in class between river basin management plans have or have not changed, or for comparing results upstream and downstream from a discharge or other activity to determine its impact.

5

INTERPRETING THE RESULTS OF WFD RIVER INVERTEBRATE STATUS CLASSIFICATION

This section provides some reminders for anyone interpreting the UK's WFD river invertebrate status classification beyond face value, giving a few tips to check before engaging in expensive remediation work, plus some extra diagnostic clues to pressures.

The classification can be imprecise. Always check the relative probabilities of belonging to each general degradation status class as well as the probability of the face value class. That will also show you the likelihood of being better or worse than the face value class or the environmental quality objective. You should not implement an expensive restoration programme on the basis of a face value class or likelihood of failure to meet the objective class that has comparatively low probability.

Always check the RIVPACS suitability code. That tells you how well the site fits the RIVPACS model (according to its environmental predictor data) and therefore how reliable the classification is, regardless of the probabilities of class. Never implement a restoration programme solely on the basis of a river invertebrate classification with low suitability. Always check the environmental input data for errors if you get a poor suitability code.

Check the error messages produced by RICT2. Exceeding a warning threshold can indicate an error in your environmental input data. Warnings caused by location data can simply mean that you are at the far north, south, east or west of the country, beyond the range of RIVPACS reference sites, so they are generally not a concern. Warnings that other limits have been exceeded indicates that those environmental parameters are beyond the conditions covered by the RIVPACS model. Exceeding the failure limits is more serious because it means that your environmental input data exceeds the values found anywhere in the UK, so it always indicates an error.

Comparing the classifications from WHPT ASPT and WHPT NTaxa provides some basic diagnostic information. If WHPT ASPT class is Good or High but WHPT NTaxa class is poorer, there may be toxic pollution or habitat degradation. If WHPT ASPT class is not Good, there may be organic pollution (still the most common environmental pressure) but it could also be caused by any of the pressures that WHPT ASPT responds to, including any that affect the oxygen concentration or the ability of animals to breathe, such as fine silt. More information about the diagnostic ability of classifications from the individual indices is provided in **Section 2.2.** When WHPT NTaxa is very small, the precision of WHPT ASPT and many other biotic indices will also be low. The diagnosis provided by the different classifications is indicative and you must undertake a more detailed investigation to identify pressures more reliably before implementing measures to restore quality.

6
ACIDIFICATION CLASSIFICATION

In acid waters with pH <7 and Ca <4 mgl^{-1} an additional component is used for WFD river invertebrate status classification in Scotland and Wales to reflect impacts from acid deposition – see Section 2.4. It is based on WFD AWICsp (see Chapter 5 Section 3.1.8).

WFD AWIC$_{sp}$ is an index of acid deposition. It was developed from previous versions of AWIC (Acid Water Indicator Community). Unlike the WHPT indices, its reference values are not derived from RIVPACS (**Section 3.1.2 and 3.1.3**) but are based on the likelihood of acid deposition. This is because acidification (acid deposition) is not the only source of increased acidity: effluents from existing and historical mining, particularly for metals, and some industrial effluents can also increase acidity.

Measures of confidence of class are not provided for this classification, which hinders its interpretation. The acidification classification is used to report WFD status in Scotland and Wales only, although the method can be applied (with caution) to rivers elsewhere in the UK.

The WFD AWIC$_{sp}$ index, typology and classification are calculated using simple algorithms in a spreadsheet that can be downloaded from UK TAG's website: **http://www.wfduk.org/sites/default/files/Media/Characterisation%20of%20the%20water%20environment/Biological%20Method%20Statements/WFD-AWIC%20Calculation%20sheet.xls**

The class boundaries are given in Table 3.8. At pH and Ca levels in excess of those stated, WFD AWIC ceases to respond to the primary environmental gradient and will give erroneous results.

Table 3.8

The UK's official river invertebrate acidification class boundaries (from government directions)

Class boundary	WFD AWIC (EQR) Humic water	WFD AWIC (EQR) Clear water, England & Wales	WFD AWIC (EQR) Clear water, Scotland
High/Good	0.93	1.00	0.91
Good/Moderate	0.83	0.89	0.83
Moderate/Poor	0.77	0.78	0.72
Poor/Bad	0.73	0.67	0.66

The UK government has set these acidification class boundaries in statutory directions in *The Water Framework Directive (Standards and Classification) Directions (England and Wales) 2015* (download from link at **https://www.legislation.gov.uk/uksi/2015/1623/resources**), and the Scottish Government in *The Scotland River Basin District (Standards) Directions 2014* and for the Solway and Tweed, which have separate legislation as cross-border river basins, in *The Solway Tweed River Basin District (Standards) (Scotland) Directions 2014,* both of which can be downloaded from **https://www.gov.scot/publications/water-environment-legislation/**. No status acidification class boundaries have been developed explicitly for English sites.

Dissolved Organic Carbon (DOC) data is needed to differentiate humic (coloured) from clear waters. Lack of data made it impossible to test whether there was also a difference in sensitivity between these two types of environment in Wales (and England), so there are further grounds for using WFD AWIC with caution in England if the water is humic. DOC is not monitored widely in England, so data is not always available.

McFarland (2010)[51] also indicated that Cantrell Acid Neutralising Capacity (ANC, Cantrell *et al.* 1990[52]) was needed for the typology, but this requirement was removed from the definitive method. However, ANC is still useful in acidification studies because there is a UK chemical standard based on that parameter: download link to the standards from **https://www.legislation.gov.uk/uksi/2015/1623/resources**

Poor WFD AWIC$_{sp}$ class indicates that acid deposition may be causing an impact, but it cannot confirm it. WFD AWIC$_{sp}$ will also respond to other pressures including toxic metal pollution, which is common in mine drainage. When WHPT NTaxa is very small, the precision WFD AWIC$_{sp}$ will be low.

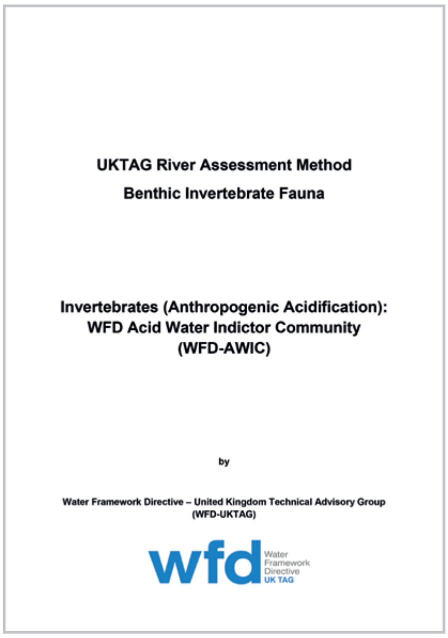
The official definition of the acidification classification can be found in UK TAG's method statements, which can be downloaded from UK TAG website **http://www.wfduk.org/resources/category/biological-standard-methods-201**

Technical guidance for calculating the acidification classification is described in documents on UK TAG's website **http://www.wfduk.org/** follow the link 'Biological Standard Methods: for Rivers – Invertebrates (General Degradation)' select **River Invertebrates AWIC UKTAG Method Statement.pdf** and WFD-AWIC Calculation sheet.xls.

Although acid deposition is a problem in some parts of England (the Pennines and Dartmoor), the method has not been optimised for use in England because no English data was used in its development. It was therefore not possible to check the acidification typology for England. The few data from England that were available were closest structurally and functionally to those from Welsh sites.

It is recommended that users in England use the reference typology developed for Wales, and this is reflected in the statutory directions in *The Water Framework Directive (Standards and Classification) Directions (England and Wales) 2015,* download from link at **http://www.legislation.gov.uk/uksi/2015/1623/resources** English uplands are in a different hydro-ecoregion (Figure 3.13), so the acidification classification based on Welsh reference values should be used with caution in England.

The High/Good and Good/Moderate status class boundaries for WFD AWIC$_{sp}$ have been intercalibrated, but only with Norway: Commission Decision (EU) 2018/229[54] at **https://www.legislation.gov.uk/eudn/2018/229/annex/division/1/division/5/division/2/adopted**. Intercalibration data for North GIG rivers at: **http://www.freshwatermetadata.eu/metadb/pdf/BFE_50-Northern_GIG_Rivers_Macoinvertebrates_acidification_data_WFD_Intercalibration.pdf**. One outcome from that work was a wider comparison of methods by Moe *et al.* (2010).[55]


UKTAG River Assessment Method

Benthic Invertebrate Fauna

Invertebrates (Anthropogenic Acidification): WFD Acid Water Indictor Community (WFD-AWIC)

by

Water Framework Directive – United Kingdom Technical Advisory Group (WFD-UKTAG)

wfd Water Framework Directive UK TAG

Figure 3.24

Front cover of UK TAG method statement for River Invertebrate (Anthropogenic Acidification) classification[53]

7

COMPLIANCE ASSESSMENT AND REPORTING

Compliance assessment and its use in reporting is an important step in determining management actions and also in informing the River Basin Planning Process.

This element of the monitoring cycle is described in Chapter 6.

Chapter 4

OTHER MACROINVERTEBRATE MONITORING AND ASSESSMENT METHODS FOR INVESTIGATIVE MONITORING

A number of UK experts have contributed to this chapter, sharing their specialist expertise and experience.

1
INTRODUCTION

The standard methods leading to status classification described in **Chapters 2 and 3** are not always appropriate for investigative monitoring and it is these other methods that are described in this chapter.

This chapter provides a number of monitoring methods for rivers and streams that cannot be assessed using our current classification methods. The methods are generally used for Specialist Investigative Monitoring surveys. These may relate to a specific pollution incident impacting on a particular riverine habitat, gathering background information within a complex catchment, or for scientific investigation.

We aim for consistency and reproducibility within the methods, so that comparisons can be made. This information is not generally suitable for classification, but may complement formal RIVPACS or other monitoring programmes.

The methods shown are a sub-set of those available and can be used to investigate a wide range of river morphologies, including some specialist niche habitats. A number of UK experts have contributed to this chapter, sharing their specialist expertise and experience.

Variations on these approaches can be developed to meet specific needs and to address issues of interest in a wide range of riverine habitats.

Figure 4.1 shows where these methods fit into the monitoring and assessment cycle.

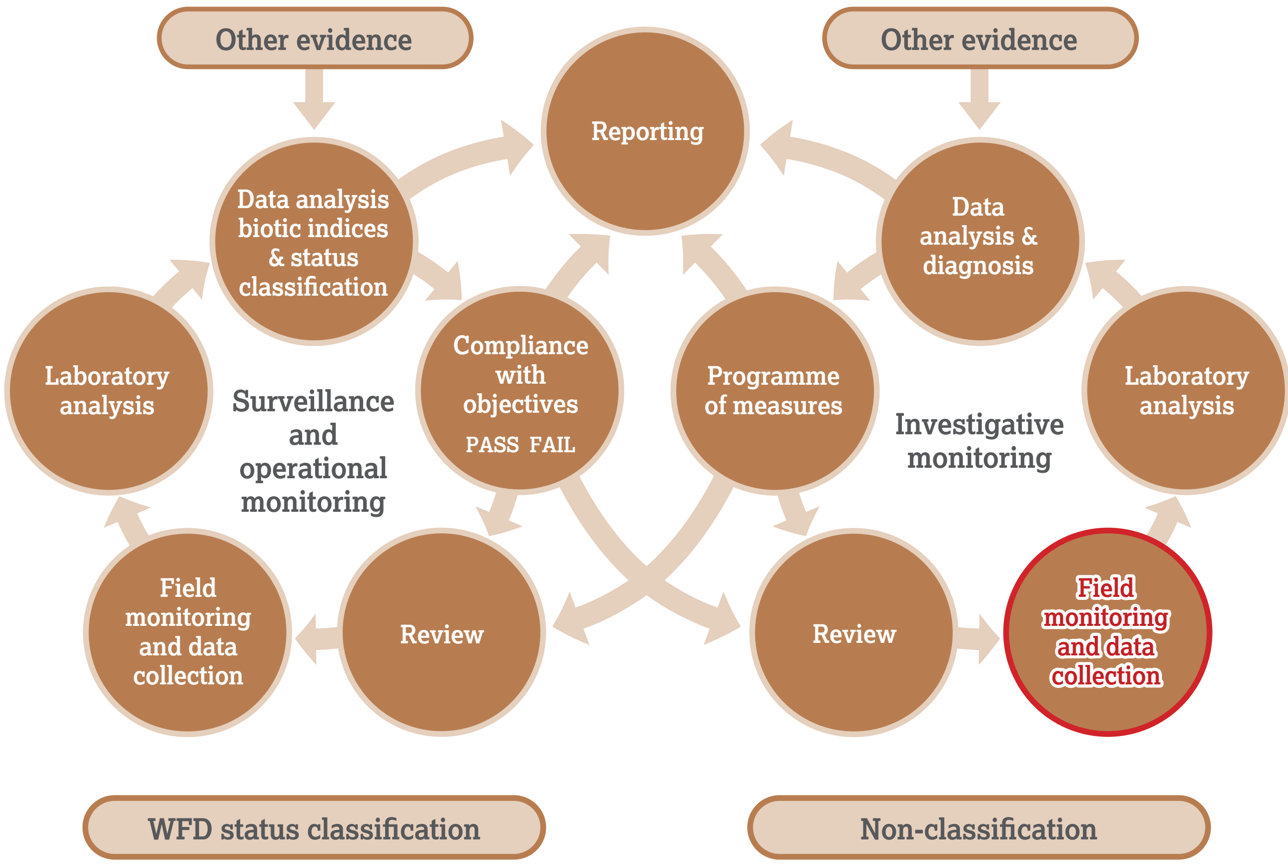
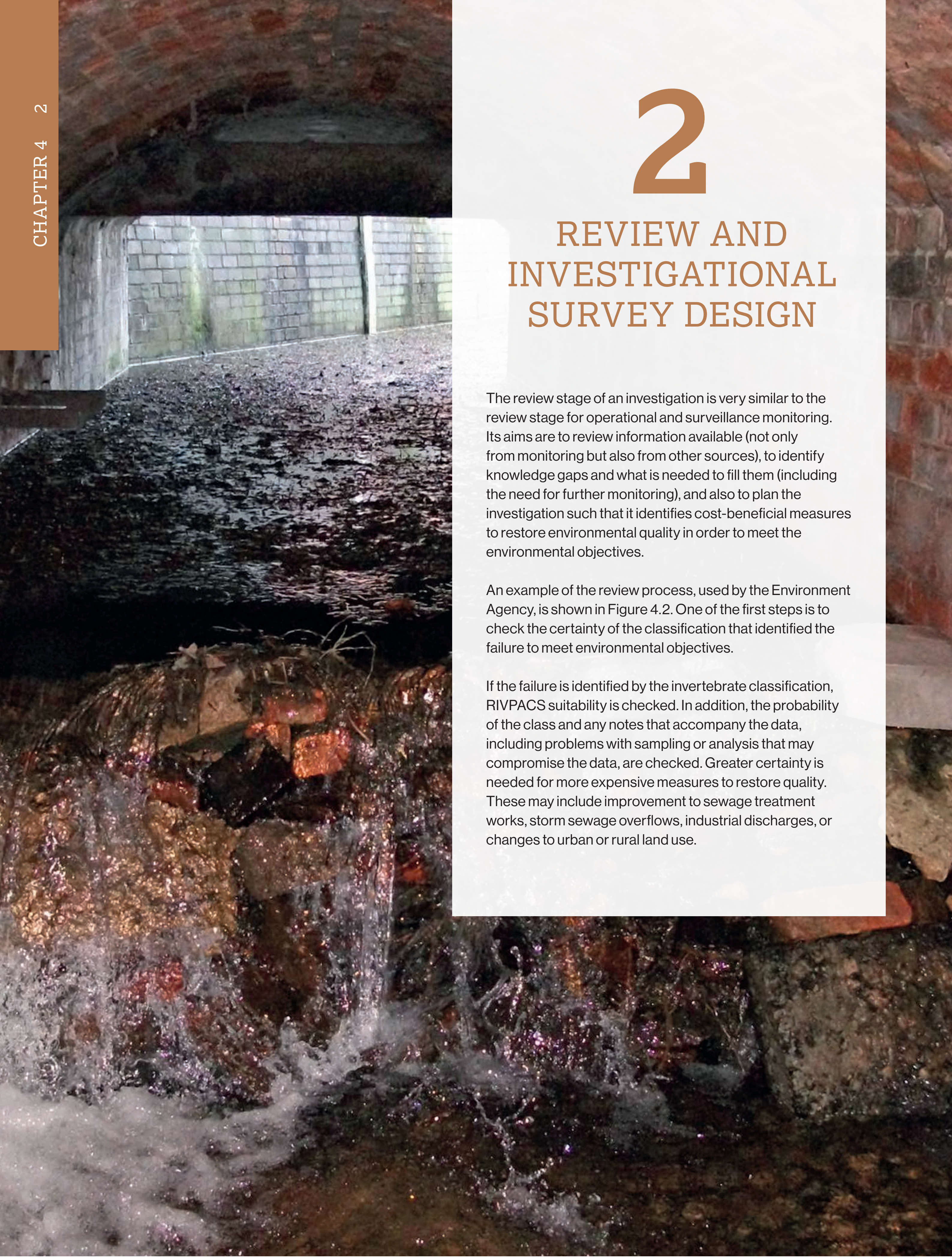


Figure 4.1
Other investigative methods – the investigative monitoring cycle and its relation to the surveillance and operational monitoring cycle

The aim of investigative monitoring is to identify the causes of failures to meet environmental objectives, including their location, and to help develop appropriate programmes of measures to restore quality. Investigations are also used to measure the impacts of pollution and other incidents so that they can be stopped or controlled, and this includes collecting evidence for legal action where environmental laws have been broken.

In addition, investigations are used to assess the quality of environments or biotas for which formal classification methods have not been developed and to help us to understand the mechanisms by which environmental pressures affect ecology, including ecosystem function, responses to particular pressures and effects on population sizes.

When a programme of measures has been identified, the site will enter operational monitoring, to check that the management measures have the desired effect and that compliance with the environmental quality objectives has been achieved. This may run parallel with the investigational monitoring described here.



2

REVIEW AND INVESTIGATIONAL SURVEY DESIGN

The review stage of an investigation is very similar to the review stage for operational and surveillance monitoring. Its aims are to review information available (not only from monitoring but also from other sources), to identify knowledge gaps and what is needed to fill them (including the need for further monitoring), and also to plan the investigation such that it identifies cost-beneficial measures to restore environmental quality in order to meet the environmental objectives.

An example of the review process, used by the Environment Agency, is shown in Figure 4.2. One of the first steps is to check the certainty of the classification that identified the failure to meet environmental objectives.

If the failure is identified by the invertebrate classification, RIVPACS suitability is checked. In addition, the probability of the class and any notes that accompany the data, including problems with sampling or analysis that may compromise the data, are checked. Greater certainty is needed for more expensive measures to restore quality. These may include improvement to sewage treatment works, storm sewage overflows, industrial discharges, or changes to urban or rural land use.

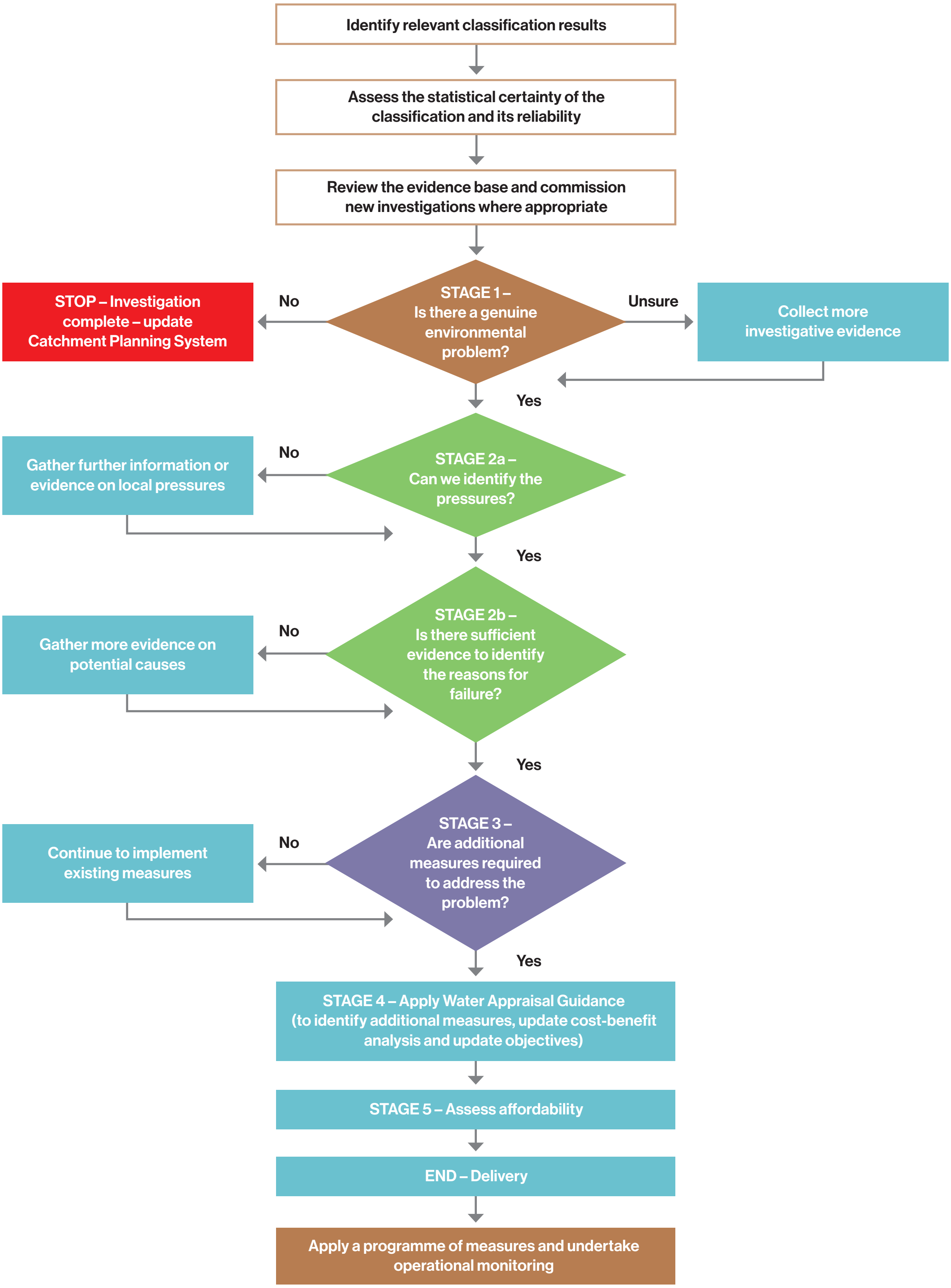


Figure 4.2
The investigation process for catchment management – from Environment Agency

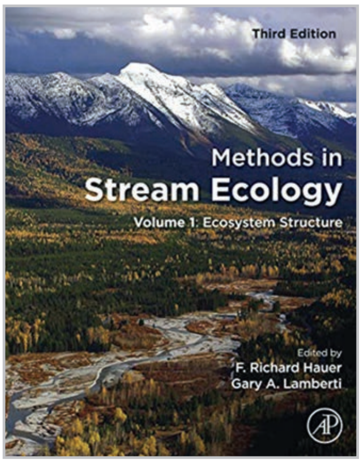
Investigative monitoring only needs to address the environmental pressures suspected of causing the water body to fail to achieve its environmental quality objectives.

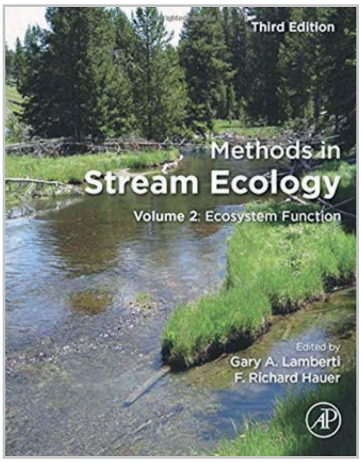
Investigations usually require different methods to those used for status classification. Sometimes the same sampling and laboratory analysis is used but with a different type of data analysis; often a different design is needed.

If it is only necessary to identify the source of pollution, a one-off survey using rapid appraisal methods may be sufficient. If the causes of poor ecological status are not understood, a more detailed survey and wider research may be needed.

Descriptions and explanations of a wide range of methods for biological surveys of rivers are included in John Hellawell's 1978 book *Biological Surveillance of Rivers.* [56] The book reviews and explains (often with worked examples) the sampling and data analysis methods developed and used up to that date and it is still strongly recommended.

In addition, more recent books describe a wide range of methods useful for investigations to improve scientific understanding. For example, Hauer & Lamberti (2017) *Methods in Stream Ecology*, in two volumes. [57]





Hauer, R & G.A. Lamberti (2017)
Methods in Stream Ecology, 3rd edn.
Volume 1: Ecosystem Structure
Volume 2: Ecosystem Function

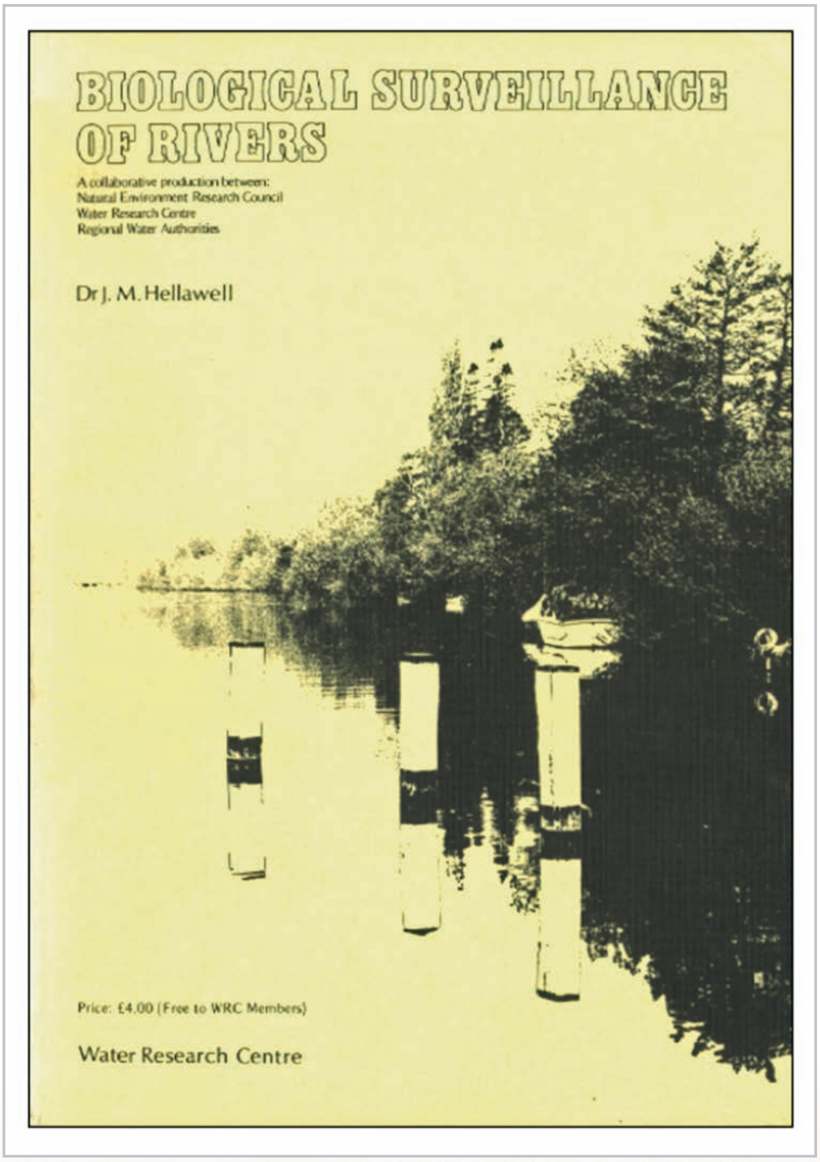


Figure 4.3

Front cover of John Hellawell's book *Biological Surveillance of Rivers* [56]
On left, the two volumes by Hauer and Lambretti, *Methods in Stream Ecology* [57]

A range of sampling devices are also described in the British and European standard BS ISO 10870:2012 (British Standards Institution, 2012). [58]

Green (1979) provides sound advice about designing ecological investigations and the endpapers provide a useful checklist, including the 'Ten Principles', shown in Figure 4.4, which remain current today. [59]

TEN PRINCIPLES

1. Be able to state concisely to someone else what question you are asking. Your results will be as coherent and as comprehensible as your initial conception of the problem.
2. Take replicate samples within each combination of time, location, and any other controlled variable. Differences among can only be demonstrated by comparison to differences within.
3. Take an equal number of randomly allocated replicate samples for each combination of controlled variables. Putting samples in "representative" or "typical" places is *not* random sampling.
4. To test whether a condition has an effect, collect samples both where the condition is present and where the condition is absent but all else is the same. An effect can only be demonstrated by comparison with a control.
5. Carry out some preliminary sampling to provide a basis for evaluation of sampling design and statistical analysis options. Those who skip this step because they do not have enough time usually end up losing time.
6. Verify that your sampling device or method is sampling the population you think you are sampling, and with equal and adequate efficiency over the entire range of sampling conditions to be encountered. Variation in efficiency of sampling from area to area biases among-area comparisons.
7. If the area to be sampled has a large-scale environmental pattern, break the area up into relatively homogeneous subareas and allocate samples to each in proportion to the size of the subarea. If it is an estimate of total abundance over the entire area that is desired, make the allocation proportional to the number of organisms in the subarea.
8. Verify that your sample unit size is appropriate to the size, densities, and spatial distributions of the organisms you are sampling. Then estimate the number of replicate samples required to obtain the precision you want.
9. Test your data to determine whether the error variation is homogeneous, normally distributed, and independent of the mean. If it is not, as will be the case for most field data, then (*a*) appropriately transform the data, (*b*) use a distribution-free (nonparametric) procedure, (*c*) use an appropriate sequential sampling design, or (*d*) test against simulated H_0 data.
10. Having chosen the best statistical method to test your hypothesis, stick with the result. An unexpected or undesired result is *not* a valid reason for rejecting the method and hunting for a "better" one.

Figure 4.4

Ten Principles, from the front endpaper of Green (1979), described in detail in the main text of that book

3

OTHER MACROINVERTEBRATE SAMPLING METHODS FOR INVESTIGATIONS

The Standard RIVPACS sampling and analysis protocol, described in **Chapter 2,** is recommended for most monitoring purposes, including investigations, for a number of reasons: it provides samples comparable with those from most other surveys; its behaviour and errors are understood and quantified; and it responds to almost all environmental pressures.

However, there are times when other methods are needed:

1. **Where it is difficult to use RIVPACS methods**
2. **Where different habitats are to be investigated**
3. **Where quantitative analysis is needed**

The methods described below provide alternatives to complement or replace RIVPACS, for specialist aquatic environments, or where additional information is required.

3.1 Artificial substrates

The 'artificial substrates' methods are useful in deep and fast flowing sites where there is no access for conventional sampling, or where a more quantitative analysis is needed. Two types of sampling device are recommended for collecting invertebrate samples for environmental assessment.

The first is the Standard Aufwuchs Units (SAufU) (Figure 4.5), and the second is stone-filled mesh bags, using cleaned stones from nearby on the same river. Both work well. SAufU are better standardised; stone bags provide samples that are more representative of the benthic fauna at the site.

Artificial substrate methods sample a different invertebrate community (the fouling community) to that sampled directly by RIVPACS methods. Therefore it is necessary to sample both upstream and downstream of RIVPACS (and if possible, before and after) using the same artificial substrates method to observe an impact. Upstream and downstream sites should be as similar as possible. SAufU are best deployed by mounting them on two bricks, to help anchor them and prevent them from silting-up when on sand or clay river beds. Ideally, they should be tethered to the bank (perhaps from a tree) using black rope (which is less visible than lighter colours) so that they can be retrieved, but care should be taken to avoid areas where boats or people may go. They should be left in situ for 3–4 weeks, so they need to be hidden well. It is recommended that more devices than needed are deployed in order to allow for loss, damage or disturbance. Indices such as WHPT ASPT and Ntaxa may be used but not RIVPACS, as results from artificial substrates methods are unsuitable for status classification.

Prichard samplers are similar to stone bags, but they are much larger, filled with bed material from the site and embedded in it, but open at the surface until the sample is retrieved.[60] They have been designed as alternatives to traps for sampling crayfish.

Figure 4.5
SAufU artificial substrate

3.2

Sampling interstitial habitats (by John Davy-Bowker)

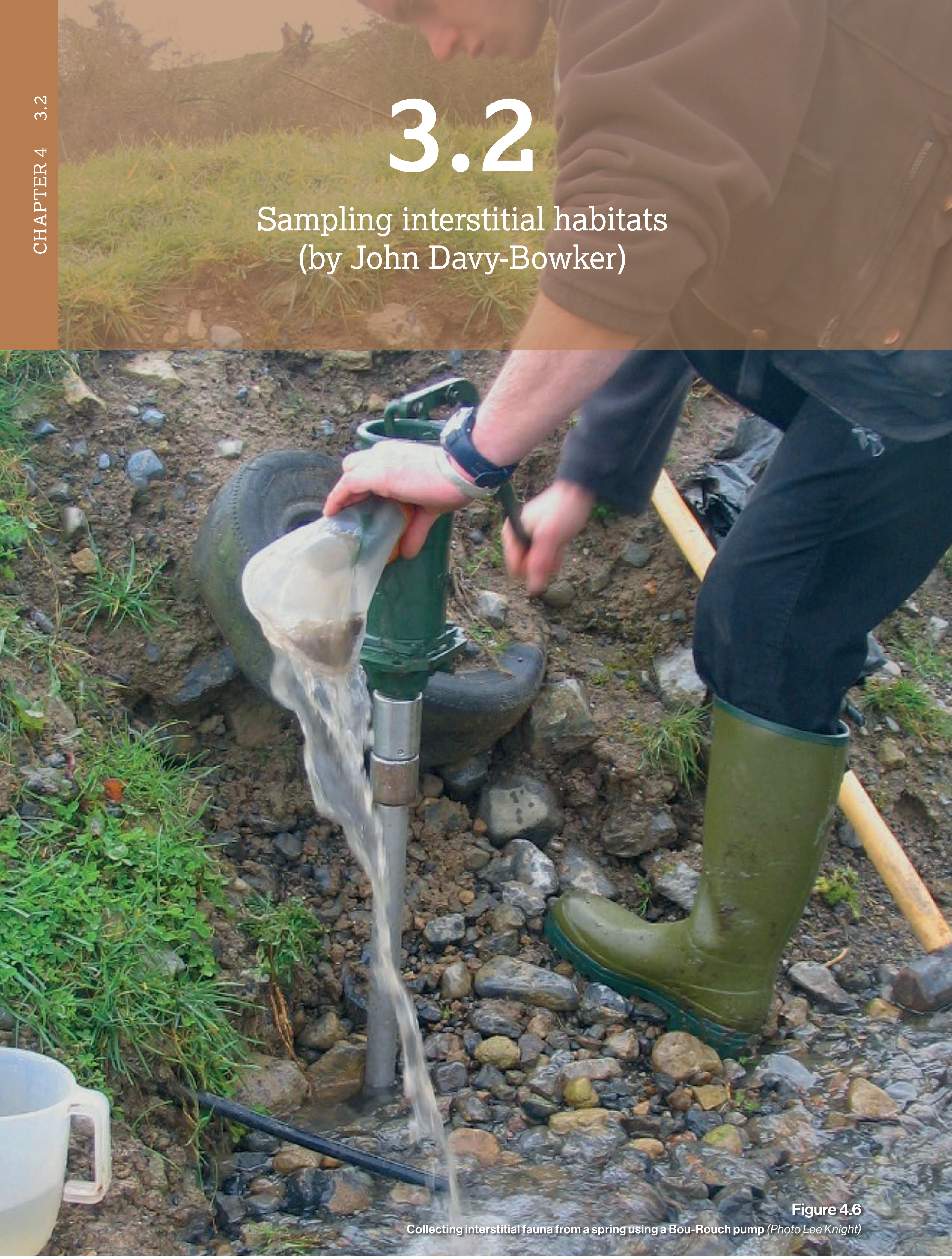


Figure 4.6
Collecting interstitial fauna from a spring using a Bou-Rouch pump *(Photo Lee Knight)*

Figure 4.7
Bou-Rouch pump and standpipe

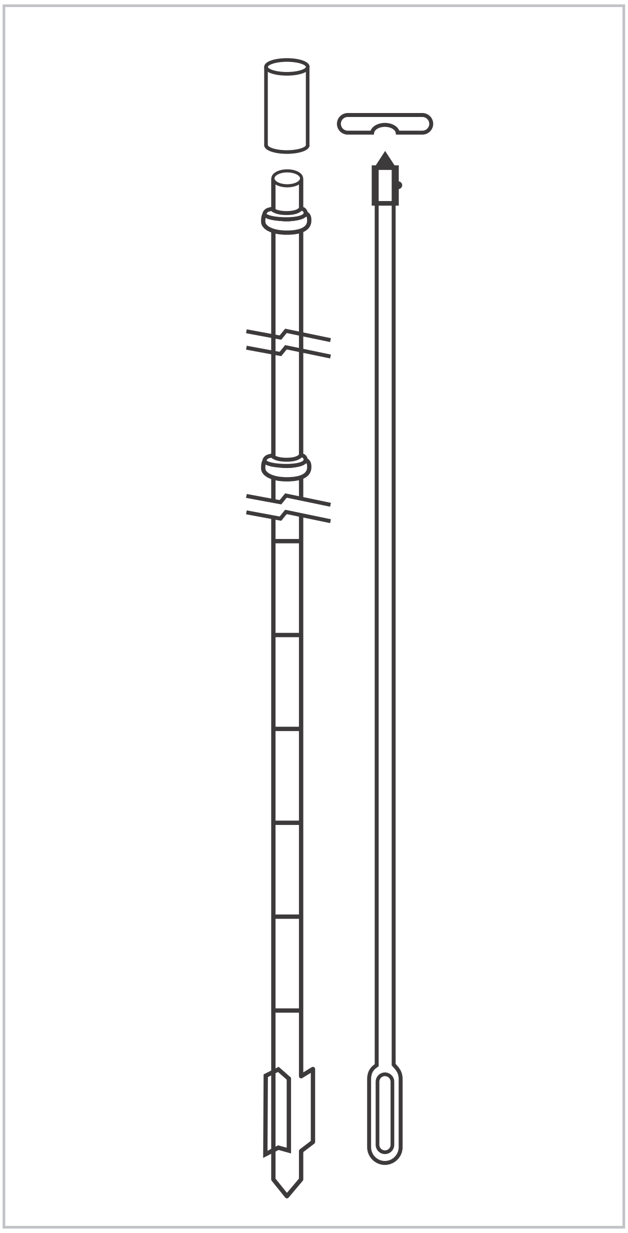


Figure 4.8
Williams standpipe corer

Sometimes it is helpful or even necessary to obtain samples from the hyporheic zone: the deep interstitial gravels beneath the surface benthic layer. Several specialised sampling devices have been developed to obtain hyporheic macroinvertebrate samples.

The Bou-Rouch Pump (Figure 4.7) is a hand-operated pump that sucks interstitial water up a standpipe hammered into river gravels and into a collecting net (Figure 4.6). This is probably the quickest way to obtain a hyporheic sample, although pumping can filter out the larger freshwater invertebrates.

The Williams Standpipe Corer (Figure 4.8) has a similar pipe, although instead of pumping water upwards, the Williams corer has a sampling aperture that is opened at the desired gravel depth, rotated to scoop a sample inside the standpipe, and then closed before extraction. This provides a small but non-filtered sample from deep within river gravels.

Finally, there are freeze-coring techniques. These involve passing either liquid nitrogen (Figure 4.9) or pressurised carbon dioxide gas (Figure 4.10) down a sealed standpipe such that a freezing front advances outwards into the river gravel. The frozen sample attached to the outside of the standpipe is then winched free from the surrounding gravels and defrosted into compartments. This sophisticated technique gives a large, vertically intact sample comprising undisturbed gravels and freshwater hyporheic macroinvertebrates (Figure 4.11). However, although useful for research investigations, it is unlikely to be practical for most monitoring purposes because of its cost and the precautions required for health and safety.

Information about the hyporheic invertebrate communities in Britain is included in the review by Robertson *et al.* (2008) [61] and their relevance to wider environmental management in Buss *et al.* (2009). [62]

Figure 4.9
Liquid nitrogen freeze corer *(Photo John Davy-Bowker)*

Figure 4.10
Carbon dioxide freeze corer *(Photo John Davy-Bowker)*

Figure 4.11
Material collected on a liquid nitrogen freeze corer *(Photo John Davy-Bowker)*

3.3

Sampling subterranean streams (by Lee Knight)

Figure 4.12
***Niphargus aquilex*, a common and widespread subterranean amphipod** *(Photo Chris Proctor)*


Subterranean streams are subject to similar environmental pressures as surface streams, including pollution, particularly from agriculture. However, these impacts are rarely investigated, despite subterranean faunas being at risk and having the same intrinsic value as surface faunas. The subterranean fauna include rare and endemic macroinvertebrates and widespread species such as *Niphargus aquilex* (Figure 4.12). However, even the common, widespread species are rarely encountered by most freshwater biologists.

Epikarst is the layer of weathered carbonate bedrock between the soil and bedrock and it can form the roof of caves. It is usually sampled using funnels to collect dripping water in a cave. It can also be sampled indirectly from drip-fed pools, but drip-fed pools can have different faunas and be affected by predation (Pipan & Culver, 2005). [63]

Subterranean streams can be fed by sinking surface streams (allogenic), infiltration from the epikarst (autogenic), or a combination of both. Subterranean streams originating from surface streams vary in size, chemistry, and flow patterns and can form underground rivers. They rely on nutrients and often biota too from the surface, outside the cave system. Subterranean streams fed by epikarst tend to be smaller, poor in nutrients and contain more specialised biotas. Subterranean streams emerge on the surface as relatively small springs or larger resurgences, sometimes far from the location of the cave where they are studied. The location of springs is determined by local geology, with water being forced to the surface as it comes into contact with less permeable strata beneath the karst. The deployment of a drift net at a spring source or resurgence for 24 to 48 hours can be a useful surrogate method to assess the diversity of a local aquifer if subterranean sampling is not viable. Springs that rise through gravels can be similarly sampled using the Bou-Rouch pump at the issue point (Figure 4.7). Alternatively, invertebrates can be sampled from wells and boreholes with suitable nets. Macroinvertebrates from larger cave streams are sampled by similar methods to surface streams, with the same equipment including standard hand nets (Figure 4.13 and 4.14). The method of deploying them will depend on the size of the stream and the nature of the substrate, which can be a shallow layer over bedrock or much deeper. One of the key considerations in method selection at subterranean locations is the logistics of transporting the equipment through cave systems or mines. This can sometimes involve long arduous journeys to the sampling sites with a variety of obstacles to traverse, ranging from narrow and very low passages to deep pitches and even flooded sections requiring scuba equipment to pass.

The bulk and encumbrance of any sampling equipment therefore has to be kept to a minimum, especially when one considers that other equipment including rope, electron ladders, karabiners and scuba cylinders might also need to be carried as well.

Although cave streams support a range of different sub-habitats, meso-habitat heterogeneity is restricted compared to surface watercourses; diversity of subterranean streams is generally much less than that of surface streams.

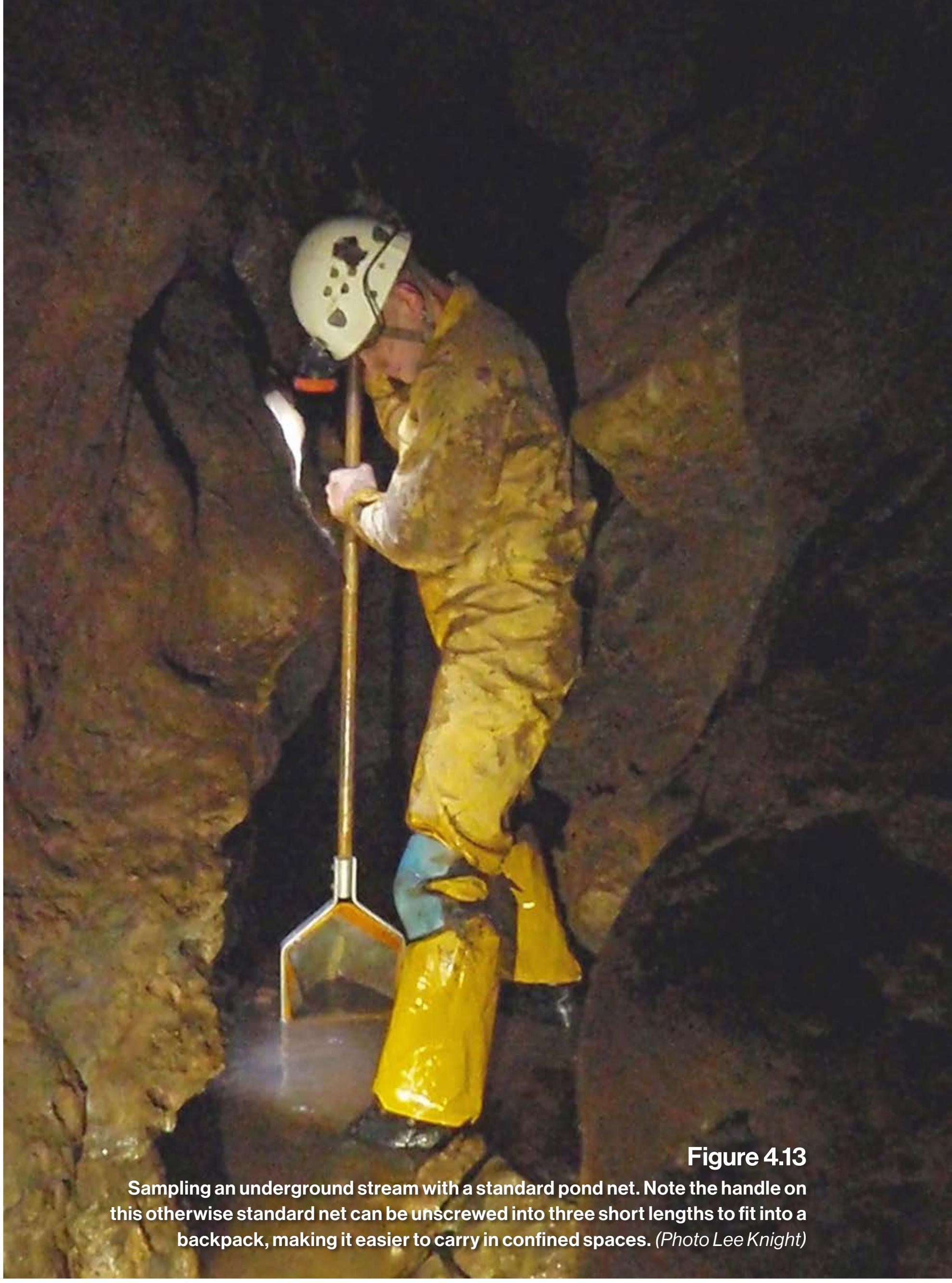


Figure 4.13
Sampling an underground stream with a standard pond net. Note the handle on this otherwise standard net can be unscrewed into three short lengths to fit into a backpack, making it easier to carry in confined spaces. *(Photo Lee Knight)*

Figure 4.14
Sampling an underground stream – emptying the net into a sample container *(Photo Lee Knight)*

Figure 4.15
A range of nets for collecting invertebrates from underground streams and pools *(Photo Lee Knight)*

In smaller underground streams, or those deep into cave systems, smaller equipment is more appropriate.

The standard FBA-pattern pond net comes in a version in which the handle can be broken down into three sections, making it easier for transportation.

There is also a smaller frame-sized version that is useful for sampling narrow diameter streams. Small pools and underground 'lakes' can be sampled using various aquarist hand nets or trawl nets (Figure 4.15).

Details of sampling methods for underground waters are described in Knight *et al.* (in press) [64] and more general information about subterranean aquatic ecology can be found in Robertson *et al.* (2008). [61]

3.4

Sampling exposed river sediments and riparian zones (by Jon Webb, Natural England and Nick Mott, Staffordshire Wildlife Trust)

Invertebrates associated with exposed riverine sediments (ERS) and riparian zones are known to be an important conservation resource. Beetle surveys are a useful approach to assess the overall quality of the habitats along watercourse corridors. Beetles are considered to be one of the best overall 'indicators of habitat quality' because of their incredible diversity and individual species' fidelity to particular niche habitats within these riparian zones.

3.4.1 Best time for surveys

Although many species are active throughout spring, summer and autumn, between the first week in May and the last week in June is optimal. This is when adult beetle activity is often at its peak.

Surveys should be carried out during dry weather when river levels are normal or low. It is also crucial to check the long-range weather forecast before setting pitfall traps.

3.4.2 What is a standard riparian sample?

There are a variety of techniques with no single optimum way to survey. Some techniques are rapid and do not require repeat visits whereas other methods collect more individuals but can be resource intensive. One approach uses a series of techniques with the aim of collecting a large diversity of species in a relatively short time period (Natural England, 2017); see also Webb *et al.* (2022). [65]

On any given stretch of river, four sample stations are selected in close proximity to each other (c.200 m). These are usually identified beforehand and tend to each be located on a sand or shingle bank and its associated habitats. These are often situated along a series of bends in the river.

A Riparian Sample is defined by Natural England as:

For each of four stations

- one hour hand searching
- an additional 20 minutes excavation (on sand and shingle bars only)

At one station, usually the one with the largest area of open sand or shingle

- ten pitfall traps in place for a two-week period

The combined results can be used to provide a species list for quality assessment, but the results should also be retained for the individual stations for two reasons:

a. it allows for specific interest features on the site to be identified (eg *Atheta basicornis* found under riparian tree bark at grid ref X)
b. pitfall trapping was excluded from some earlier surveys; keeping separate records of the results (pitfall and hand searching) allows comparisons with the results of earlier surveys.

3.4.3 Hand searching

Hand searching can be a very effective method for sampling riparian invertebrates, particularly in terms of recording the smaller, cryptic species and those which are subterranean for the majority of the time (Drake *et al.* 2007). [66]

Each sample station consists of a stretch of river and its associated riparian habitat, often based around ERS but will also include other adjacent riparian habitat. Sampling may therefore cover a variety of habitats, eg eroding banks, vegetated sand and shingle, riparian woodland, stretches of emergent vegetation, woody debris, etc.

Each sample consists of the combined catches of six separate 10-minute searches within a one hour period at each sample station. The aim of these separate searches is to target the specific habitat types present.

This searching includes the time involved in transferring specimens to collecting tubes, preparing equipment, etc. So the actual time spent searching tends to be in the range of 5 to 8 minutes per search.

At each sample station one or more of the following techniques are used to find animals, depending on the habitats present:

- Soft sediments are trampled or patted, and surface-active insects pooted up directly from the ground (Figure 4.16 and 4.17).
- Next to water margins, exposed sediment is splashed with water. This works best on steeper banks where a plastic kitchen sieve can be used to catch insects washed into the water, or beetles can simply be pooted as they run back up the slope. The basal parts of plants are examined or pulled apart; tussocks can be dissected over a sheet or tray using a small hand-saw and sieve and insects then pooted.
- Litter and dense mats of fallen vegetation are sieved over a plastic sheet or tray, using a sieve with a mesh size of 4 to 8 mm.
- Emergent vegetation is submerged and the insects that float to the surface are scooped up with a plastic kitchen sieve.
- Large stones can be lifted and species pooted from the underside and woody debris can be broken apart and actively searched before pooting.

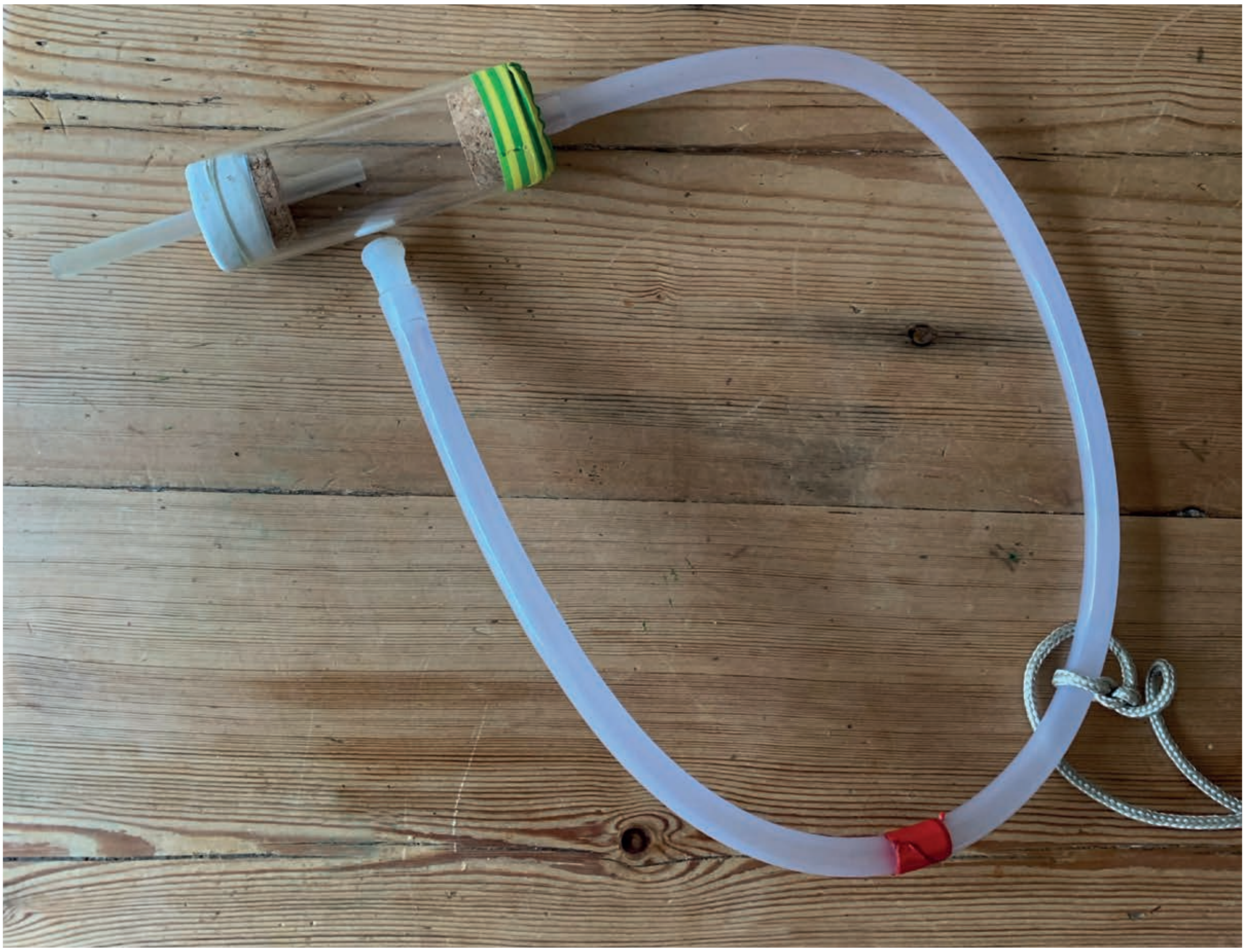


Figure 4.16

A pooter or aspirator; a fine mesh at the end of the long flexible tube, hidden by the cork, prevents the inhalation of a mouthful of insects! *(Photo Nicolas Button)*

Figure 4.17
Using a pooter or aspirator to collect invertebrates from the dry phase of an intermittent stream *(Photo: Tim Sykes)*

Figure 4.18
Pitfall trapping and hand searching, River Dove *(Photo: Nick Mott)*

3.4.4 Pitfall trapping

Based on Sadler and Bell (2000),[67] ten small plastic cups, c.10 cm diameter, are dug into the sediment so that the rim is flush with the surface. These are filled one-third full of a 50:50 mixture of propylene glycol and water, with a small amount of detergent added to break the surface tension. Propylene glycol assists in sample preservation and reduces evaporation.

At each site, these pitfalls are placed sufficiently high up the bank (but still in the riparian zone) to lessen the risk of flooding and/or hidden away to avoid detection. In large and diverse areas, pitfalls can be set at least 2 m apart and placed in each of the main habitat types detected (eg coarse shingle, bare sand, vegetated sand, etc). It is also important to cover each pitfall with an elevated plastic grid guard (20 mm mesh size). This will allow beetles to be caught, but will reduce the incidence of accidental by-catch of small mammals, reptiles and amphibians.

Pitfalls are left at each site for at least two weeks, and not more than four weeks, before collection and storage in 50% ethanol; for species that are collected for analysis via DNA meta-barcoding, the current advice is to transfer them to 90% ethanol as soon as possible.

3.4.5 Excavation

This technique is described by Sadler & Petts (2000). [68] At a distance of 1 to 2 m from the river's edge, a garden trowel is used to dig out an area of approximately one square metre down to the water table. The sides of the excavation are then collapsed down into the ponded water so that animals trapped in the sediment float to the surface where they can be scooped up with a tea-strainer. The time allotted for the excavation of shingle should be 15 to 20 minutes. Excavation works best on sand and shingle but cannot be used for finer grade silts or on very coarse boulders.

It takes some practice to work out where such excavations are best situated. On shingle they are often more productive in the finer grade material at the leading edge of a shingle bar.

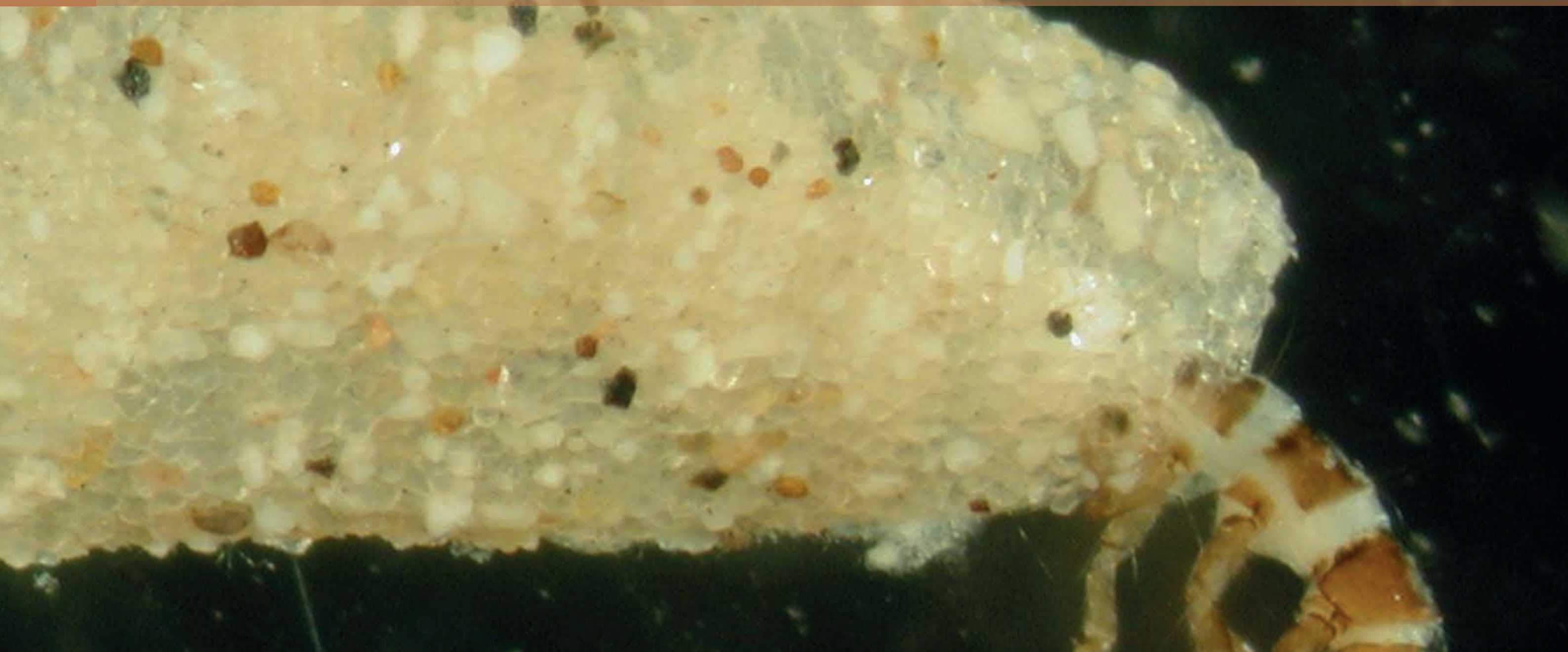
3.4.6 Reporting on the riparian fauna

Important aspects of the fauna to analyse should include the number of species with a high fidelity to riparian and floodplain habitats, as well as rarity and other scoring metrics. Reports should emphasize on-site features of interest and use comparative analysis to help classify a site's importance. Selected elements from Pantheon (an online tool to analyse invertebrate species samples) can be used to help interpret and understand riparian interest. [69] (see **Chapter 5 Section 3.2.6** Pantheon).

Useful references include Drake *et al.* (2007), [66] Webb *et al.* (2017), [69] and Webb *et al.* (2022). [65]

3.5

Sampling from intermittent rivers and ephemeral streams (by Judy England, Environment Agency)



Methods for sampling intermittent rivers and ephemeral streams (also known as temporary rivers) are still in development.

Three phases have been recognised in temporary rivers: flowing, ponded and dry.

Wet phases (flowing and ponded) are usually sampled using the methods used for permanently flowing streams. The standard RIVPACS sampling only includes parts of the channel covered by water, despite being a pro rata multi-habitat method in which all habitats are sampled in proportion to their cover.

So, in temporary rivers and perennial streams during drought, dry habitats are not sampled. That would make taking representative samples from drying perennial streams and temporary rivers during the dry phase difficult to incorporate into existing monitoring networks for invertebrates. Alternative methods are needed for dry areas and the dry phase.

One way to incorporate drying and temporary streams into monitoring programmes would be to take separate samples for wet and dry areas. That would enable current analytical methods such as RIVPACS to be applied, although that will need some development because currently it only covers permanently flowing streams. Dry phases can be sampled using methods described in **Section 3.4**. The most commonly used invertebrate groups for assessing the dry phase of temporary and intermittent streams are beetles, ants and spiders. There is no equivalent to RIVPACS for predicting the natural fauna of the dry phase or habitats and therefore no way to assess the degree to which they might be damaged by anthropogenic pressures. Research is underway to address this gap.

The most useful reference is Magland *et al.* (2020).[(70)] See Figure 4.19.

Drought

- Less flow
- Less turbulence: smooth flow, stagnant pools
- Less surface area between air and water (flow underground)

January

May

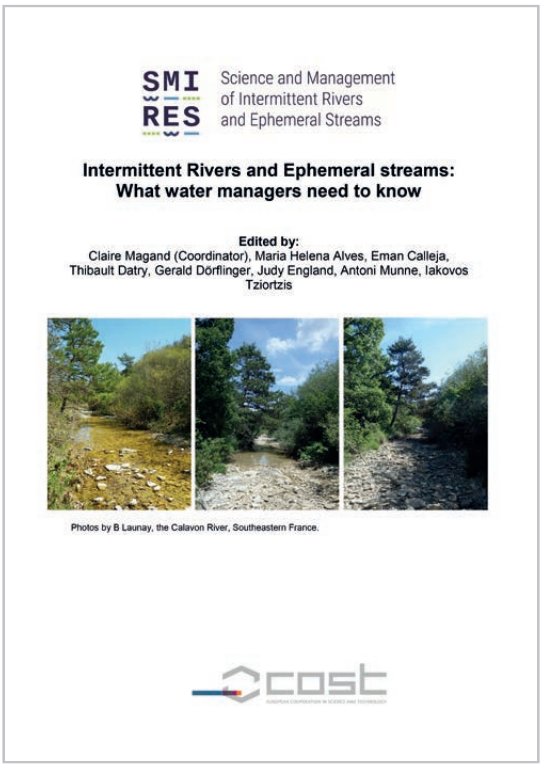
August

3.6 Quantitative sampling

Most quantitative samplers are suitable only for shallow gravelly riffles. They include Surber samplers and cylinder samplers in which a known area of substrate is sampled, to about 5 cm depth, so providing information about the numbers per unit area of this habitat.

Both are described in Hellawell (1978), together with other quantitative devices.[56] In most cases, they cannot provide an estimate of population size or density because most invertebrates are not restricted to this particular habitat (many adult insects are not even aquatic), and because small juvenile instars may pass through the net.

In deeper waters, artificial substrates (**Section 3.1**) enable quantitative sampling of the fouling or benthic community, particularly SAufU because of their standardised design.



SMIRES Science and Management of Intermittent Rivers and Ephemeral Streams

Intermittent Rivers and Ephemeral streams: What water managers need to know

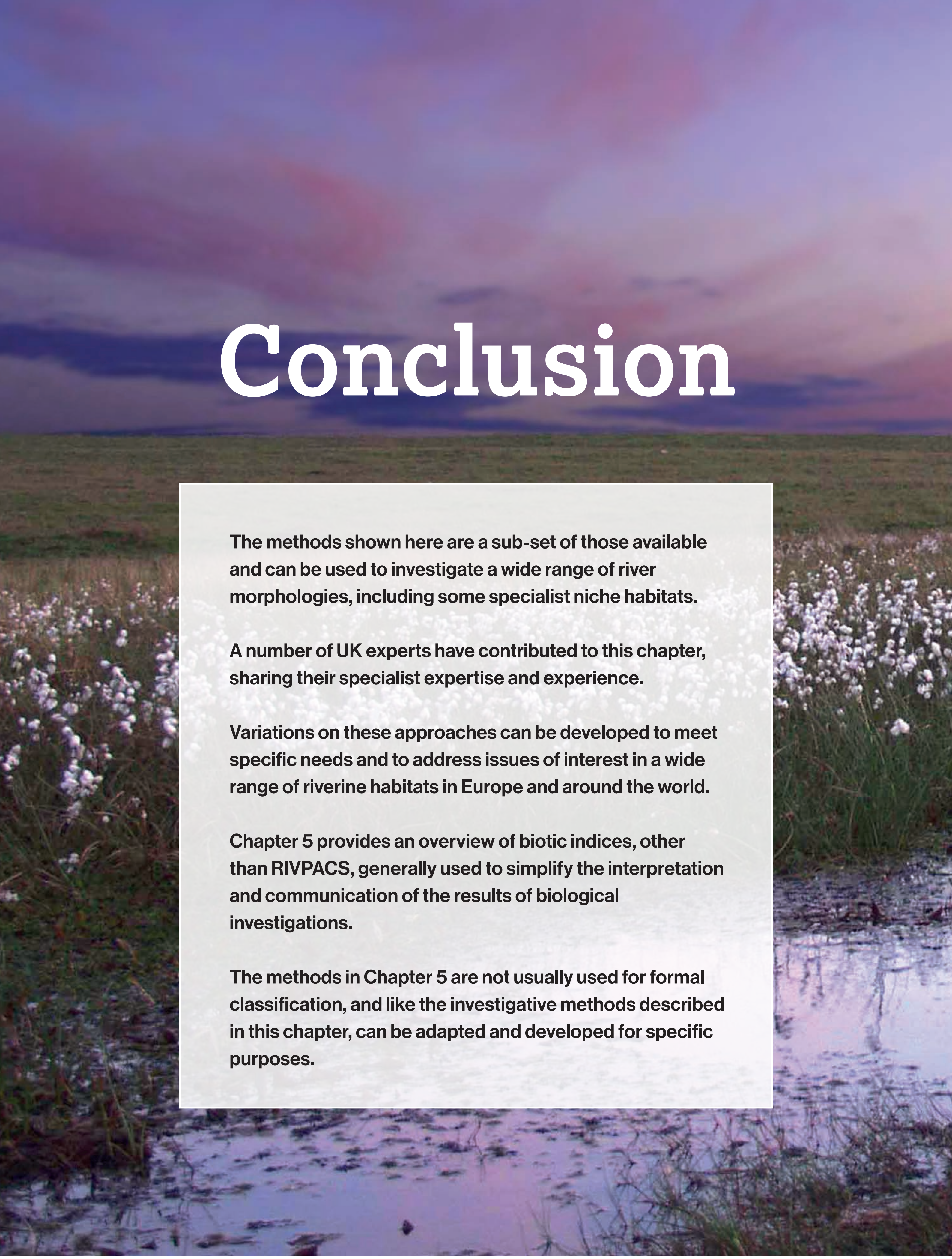
Edited by:
Claire Magand (Coordinator), Maria Helena Alves, Eman Calleja, Thibault Datry, Gerald Dörflinger, Judy England, Antoni Munne, Iakovos Tziortzis

Photos by B Launay, the Calavon River, Southeastern France.

cost

Figure 4.19

Front cover of Magand *et al.* (2020), a key reference for assessing and managing intermittent rivers and ephemeral streams [70]



Conclusion

The methods shown here are a sub-set of those available and can be used to investigate a wide range of river morphologies, including some specialist niche habitats.

A number of UK experts have contributed to this chapter, sharing their specialist expertise and experience.

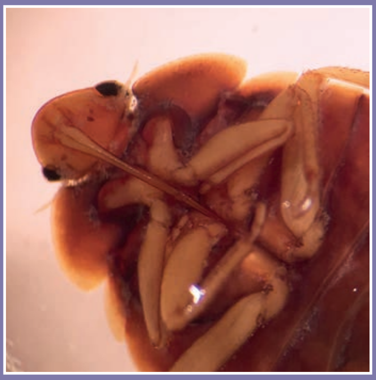
Variations on these approaches can be developed to meet specific needs and to address issues of interest in a wide range of riverine habitats in Europe and around the world.

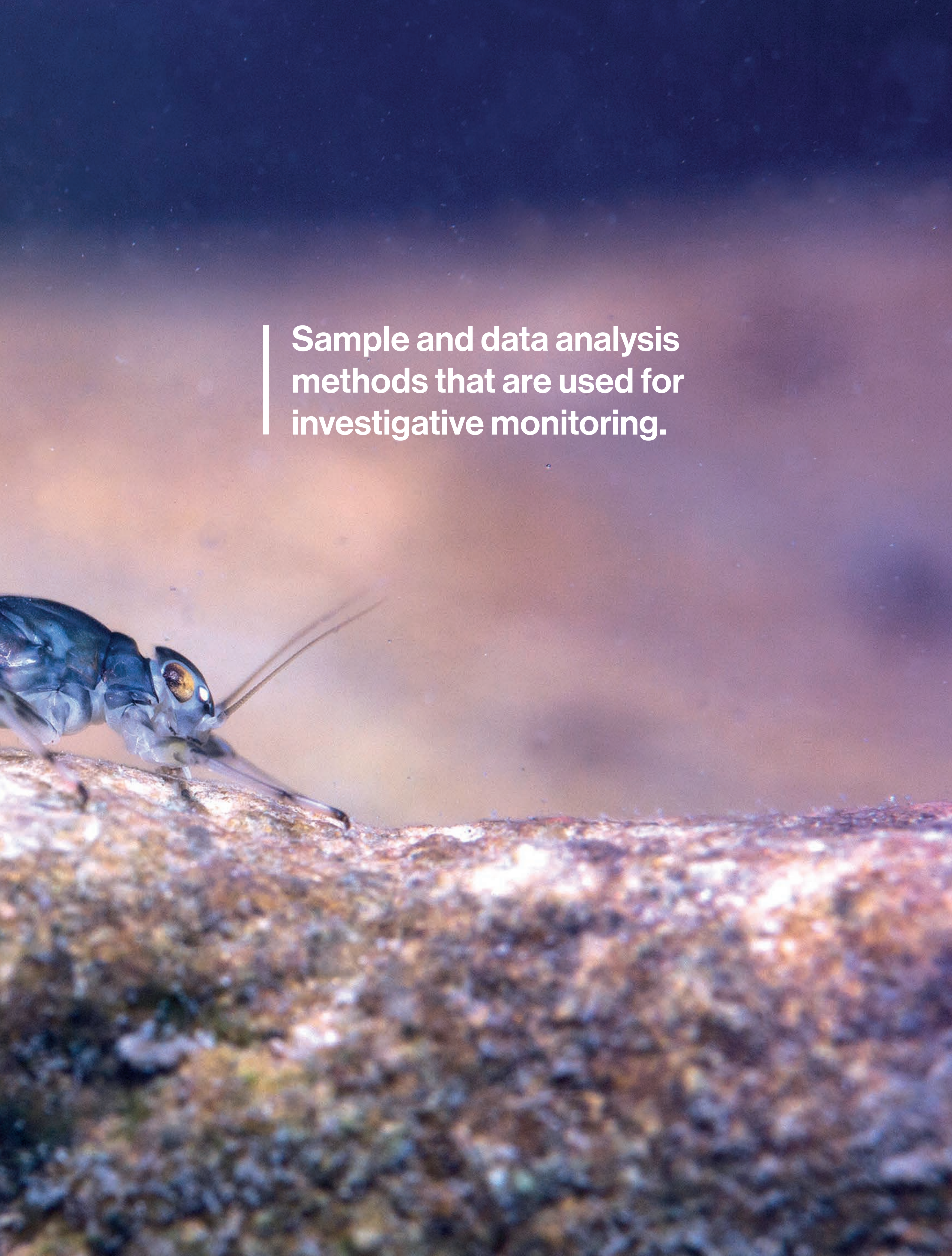
Chapter 5 provides an overview of biotic indices, other than RIVPACS, generally used to simplify the interpretation and communication of the results of biological investigations.

The methods in Chapter 5 are not usually used for formal classification, and like the investigative methods described in this chapter, can be adapted and developed for specific purposes.

Chapter 5

SAMPLE AND DATA ANALYSIS FOR INVESTIGATIONS





Sample and data analysis methods that are used for investigative monitoring.

1
INTRODUCTION

This chapter covers sample and data analysis methods that are used for investigative monitoring. Often, the standard methods leading to status classification, as described in **Chapters 2 and 3** are suitable, but sometimes other methods are more appropriate. An introduction to investigative monitoring and its aims are described in **Chapter 4.**

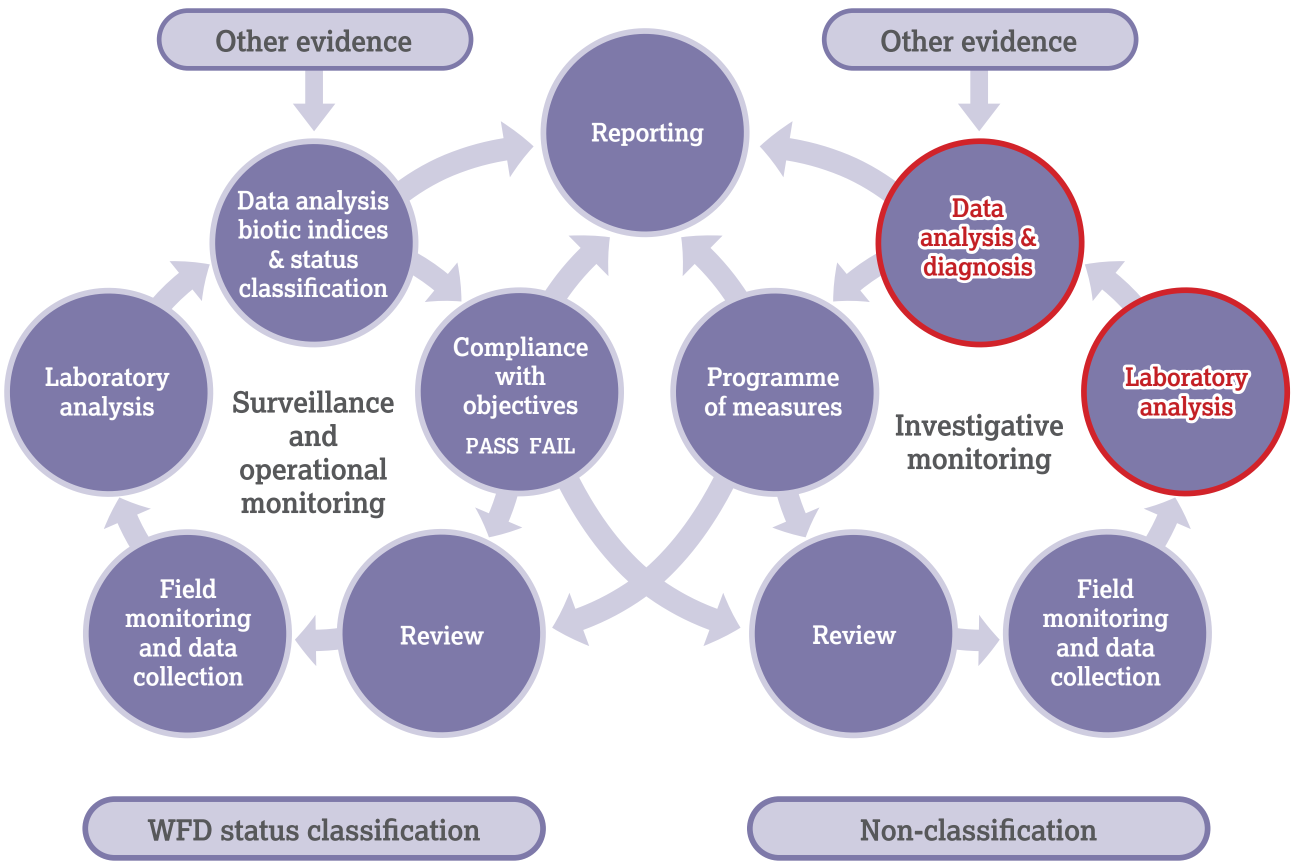


Figure 5.1
The subject of this chapter (shown in red) in relation to the surveillance and operational monitoring cycle

As a reminder, investigative monitoring is usefully defined by the EU WFD as:

Investigative monitoring is designed to identify the causes of poor environmental quality (diagnosis) and their timing and source so that an appropriate programme of measures can be implemented to restore quality.

It is undertaken in close association with **surveillance monitoring** which assesses long-term changes in the environment due to natural and widespread anthropogenic activity. Surveillance monitoring is the basis for formal classification and reporting of water quality and drives the infrastructure investment programmes.

Operational monitoring is used to confirm the status of water bodies at risk from known pressures and to assess the efficacy of improvement programmes.

As previously stated, it is important to differentiate the data sets from these investigative monitoring activities, as the data from investigative monitoring may bias classification toward a specific short-term event, such as a transitory pollution incident.

Care must be taken in sample programme design to optimise these complementary activities and yet ensure high quality information for decision making and reporting.

2
SAMPLE ANALYSIS

The standard laboratory analysis described in **Chapter 2 Section 12,** is appropriate for most investigative and operational purposes, but where the degree of precision necessary to distinguish between high and good status is not needed, field sorting may be appropriate.

For pollution investigations, field sorting on the bankside immediately after sampling provides important information about the numbers of invertebrates in the sample that were already dead, and may have been killed by the pollution event (see **Chapter 2 Section 11**). This can supplement information provided by standard laboratory analyses, which must be subject to full quality control and therefore have known precision.

3
DATA ANALYSIS

3.1 Biotic indices

Biotic indices are numerical values that relate the presence of taxa to environmental pressures. Their role is to simplify complex biological data so that ecologists can explain their results to environmental managers who may have little knowledge of ecology. Since the turn of the 21st century, indices have also been used to define ecological quality objectives and compliance with them, notably for the European Water Framework Directive.

Biotic indices

Biotic indices are numerical values that relate the presence of taxa to environmental pressures.
Their role is to simplify complex biological data so that ecologists can explain their results to environmental managers who may have little knowledge of ecology.

Although their format is intentionally very simple (usually a single number or letter), biotic indices are actually very complex, and most do not behave as parameters on a continuous scale of equal intervals. **Biotic indices should not be used as a basis for statistical analysis.**

Users should be wary of using biotic indices, particularly without understanding the extent of the data from which they were derived: comprehensiveness, reliability, geographical and stream type coverage; the statistical properties of their format (average, score, percentage); the impact of sampling and analytical error on the index and the magnitude of that error; the way in which the environmental pressure that the index is designed to respond affects invertebrates; other environmental pressures that have the same effect, cause the same environmental pressures or interact with each other (for example, reducing the availability of oxygen, increasing siltation, altering the availability of metal ions or nutrients); and other environmental pressures that co-occur at sites where the pressure of interest occurs. The fact that an index sensitive to a particular pressure indicates an impact does not necessarily mean that pressure is present at a site.

Finally, invertebrates respond to the integrated effect of all environmental pressures (both natural and anthropogenic) and it is impossible to apportion impact to individual pressures unless an environmental pressure is so severe that it is overwhelming. This will increasingly be the case as gross pressures are eliminated by environmental regulation. The corollary, that the impact of one environmental pressure can be mitigated by reducing other unrelated pressures, is true, but not widely recognised.

Hellawell's 1978 book [56] gives a comprehensive and detailed overview of biotic indices and other metrics used for river management up to that date. A critical review of biotic indices (including diversity, sensitivity and similarity indices), was written by Washington (1984).[71]

There are two types of biotic indices:

- parameters that describe the biological community as a whole
- those based on the sensitivity of taxa to environmental pressures.

Most indices that describe the community as a whole are measures of diversity, of which taxonomic richness is the simplest index.

3.1.1 Taxonomic richness

The most straightforward measure of taxonomic richness is the number of species, genera or taxa. This is the simplest measure of diversity to interpret, but care is still needed. There is usually an assumption that greater taxonomic richness indicates better environmental quality, but this is not always so.

For example, in base-poor rivers that are naturally oligotrophic, mild organic pollution can increase taxonomic richness by increasing the availability of nutrients directly and by reducing the bioavailability of toxic metals that may be present naturally; but the increased richness is at the expense of naturally occurring species that are intolerant of such conditions. Higher than normal taxonomic richness can indicate enrichment, but it can also indicate unusually high habitat diversity which is characteristic of some of the best sites, including nature reserves.

An issue with measures of taxonomic richness is knowing what taxa are included. Not every species or genus of invertebrate is readily identifiable in the aquatic stage. WHPT NTaxa is a standardised measure of richness at family level as the taxa included are already pre-defined.

Indices of taxonomic richness are sensitive to the sampling method because larger samples will contain more invertebrate taxa and some habitats support greater richness than others. It is therefore most important that any comparison of differences in taxonomic richness are based on samples collected in the same way. Measures of taxonomic richness generally have lower precision than other indices because they are more sensitive to sampling variation caused by sampler variation and the patchy distribution of invertebrates.

3.1.2 Diversity indices

Diversity indices were popular in the 1970s. Diversity indices are a more complex refinement of measures of taxonomic richness that also take account of the pattern of distribution of abundances across the taxa, principally the evenness of abundances. They were originally developed in the search for parameters that encapsulated the properties of ecological communities. Pollution and other environmental pressures are usually associated with less even distributions: in stressed environments, many taxa cannot cope so are absent, but the few tolerant taxa that can occur do so in greater abundance because of the reduced competition. The classic example is severe organic pollution in which the few species that can tolerate very low oxygen concentrations and silt are found in very great abundance, supported by the nutrition provided by the organic matter and the bacteria that also thrive on it.

Asellus aquaticus

Because of their more complex derivation, diversity indices are more difficult to interpret – in simplest terms because they confound richness and evenness, but more fundamentally because the properties of biological communities that they attempt to describe are themselves complex and only partially understood (Green 1979). [72] It is much easier to interpret separate measures of richness (such as number of taxa) and evenness – of which the most widely used is that by Pielou (1969) [73] – in the same way that it is easier to interpret WHPT ASPT and WHPT NTaxa separately rather than combined as a score. Green (1979) [72] (Figure 5.2) provides a critique of diversity indices, which continue to be used. The Shannon-Weiner index is the most widely used diversity index for water quality assessment and it is a component of the Intercalibration Common Metric index ICMi (see **Section 3.1.7**).

Other fundamental ecological mathematical and ecological diversity analysis methods can be found in publications by Pielou (1969), [73] Pielou (1975) [74] and Pielou (1977). [75]

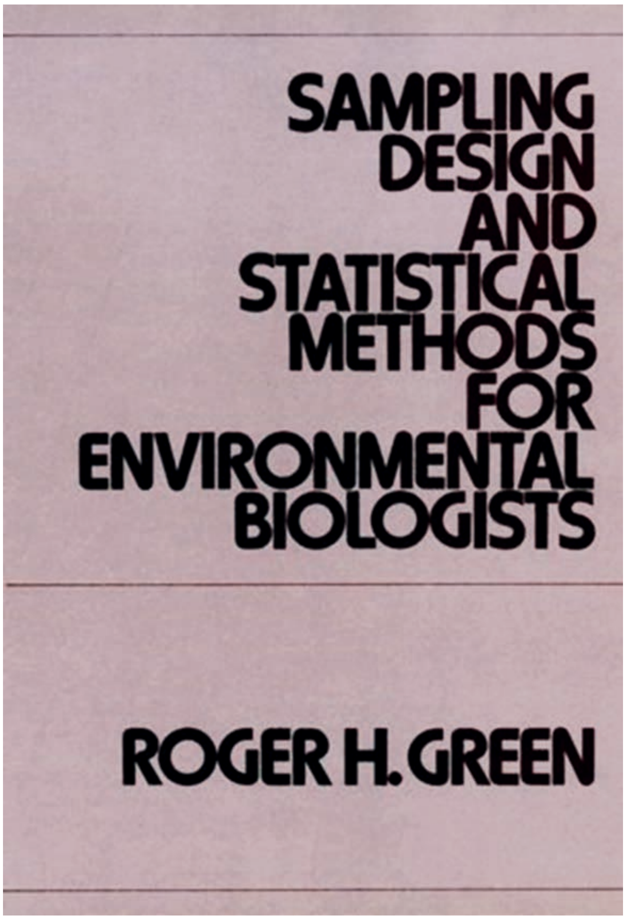


Figure 5.2

Front cover of Roger Green's book *Sampling design and statistical methods for environmental biologists.* [72] A highly recommended introduction to survey design. *(Illustration from Wiley)*

***Brychius elevatus* adult**

3.1.3 Sensitivity indices

The earliest and still the most widely used sensitivity indices respond to organic pollution from sewage. These indices therefore respond to organic loading, but also to siltation and the toxic effects of ammonia, although the main influence for animals is low oxygen concentration, and for plants it is more abundant nutrients and shading. Because they respond to multiple pressures, indices of organic pollution are widely used as general quality indices. However, they are insensitive to acidification and toxic metal pollution because many of the taxa most intolerant of low oxygen concentration and siltation are tolerant of metals and acidity. Indices have also been developed specifically for other environmental pressures including metal pollution, acidification, siltation and low flow. Sládeček (1973) [76] describes how indices can be derived for particular pressures. The most common types of sensitivity indices are calculated either as scores (the sum of sensitivity values), which therefore vary not only according to sensitivity but also to taxonomic richness, or average scores: average score per taxon, such as WHPT ASPT and LIFE, or per individual organism, eg saprobic indices (see **Section 3.1.6**).

Some sensitivity indices are based not only on values relating to sensitivity but also on weightings. Weightings are factors, usually multipliers, used to adjust the sensitivity values for individual taxa. Weightings can relate to the narrowness of response and therefore the ability of the taxon to indicate a particular band of quality (for example, the indicator weightings used in saprobic indices), or to abundance. Washington (1984) [77] describes different categories of biotic indices in more detail. Many authors do not differentiate scores, weightings and index values but treat them synonymously.

Because biotic indices simplify complex data, much information that is useful for interpreting biological data is lost. Many indices are designed to relate to a particular environmental pressure, but all indices will respond to other pressures, including natural environmental pressures, so a poor value of a particular index does not necessarily mean that there is an environmental problem or that it is caused by the pressure that the index is designed to respond to.

Some indices are more accurate than others. The accuracy of many depends on the type of water body – most river quality indices work best in shallow streams with gravel beds. The precision of indices can also depend on how they are derived: those based on average values across many taxa are more precise that those based on one or a few indicator taxa. This is a common property of averages and it is therefore recommended that such indices are accompanied by the number of taxa on which the index was based to provide an indication of precision. However, number of taxa is not a good measure of precision of averaged indices that use weighting factors. Index values derived from relatively small data sets or the expert opinion of a few experts will be less reliable than those based on the experience of many experts over a long period, or very large data sets covering the full spectrum of water bodies in which the indices are used. It is therefore still necessary for the final interpretation to be made by an ecologist, particularly where they are used to make expensive environmental management decisions or as evidence for legal prosecutions.

Sensitivity indices are generally less prone to error variation caused by sampler variation and the sampling method. Indices like BMWP-ASPT and presence-only WHPT-ASPT are particularly robust and, being averages, they have relatively low error variation. However, sensitivity indices in which the index values vary according to abundance are sensitive to sampling method. Indices such as WHPT-ASPT, LIFE and PSI should only be calculated from standard RIVPACS samples. The presence-only version of WHPT-ASPT described in **Section 3.1.5** can be calculated from other types of sample (including Surber and artificial substrates) because it does not depend on abundance data.

Terminology

Score	Index value	Average score per taxon
An index based on the sum of index values, much like the score of a game: football score, cricket score. Examples include BMWP-score, Chandler-score. In the same way that I don't recommend using diversity indices because they confuse richness and evenness, I don't recommend using scores because they confuse richness and sensitivity – it's far better to use number of taxa, and average score per taxon.	A value relating to the sensitivity of a taxon to an environmental pressure. Sometimes called a sensitivity score.	The value of an index expressed as a score divided by the number of taxa – examples include ASPT (average BMWP-score per taxon), WHPT ASPT, and LIFE. If you call an index value a sensitivity score, the term ASPT still makes sense as the average sensitivity score per taxon.

Proportional indices	Weighting	Abundance related index values
The value of an index expressed as a proportion or percentage of total: examples include PSI, SPEAR and TRPI.	A multiplier used to increase the influence of a taxon. *Indicator weightings* increase the influence of taxa with narrower distributions across the environmental gradient – for example, the indicator values used in saprobic indices. In *abundance weightings,* a greater weighting factor is usually applied to taxa with greater abundance. Abundance weightings are sometimes confused with abundance related index values.	Take abundance into account by assigning different index values to different abundances of each taxon. Values for each abundance level of each taxon are derived independently, as if they were different taxa – examples include WHPT.

3.1.4 BMWP (Biological Monitoring Working Party) indices

The Biological Monitoring Working Party score (BMWP-score), and its derivations, Average BMWP-score per taxon (ASPT or BMWP ASPT), and number of BMWP-scoring taxa (Ntaxa or BMWP Ntaxa) have been superseded in the UK by substantially revised versions known as WHPT (Walley Hawkes Paisley Trigg) indices (Hawkes 1998) [(78)] (**Chapter 3 Section 2.2**, **Chapter 5 Section 3.1.5**). BMWP is still used by the Environment Agency for Hydroecological Validation (HEV) for managing water resources (**Section 3.2.5**). The BMWP indices are still used outside the UK (Herman & Nejadhashemi 2015) [(79)] and as components of multi-metric indices, such as ICMi (**Section 3.1.7**).

BMWP – Biological Monitoring Working Party indices

BMWP indices (score, ASPT and Ntaxa) are precursors of the current WHPT indices, but are still used to assess environmental quality, principally for organic pollution.

BMWP has been used successfully around the world in both temperate and tropical countries. This is because species within families tend to have the same respiratory physiology and because most families are pandemic.

Beware, there are different versions of BMWP indices. The original had separate index values for eroding and depositing streams, but the versions used for the 1980 and 1990 national river quality surveys had one set of values for all stream types. From 1995, the regulatory agencies excluded Clambidae, Chrysomelidae and Curculionidae because they are very rare or absent from running waters.

BMWP indices follow the taxonomy described in the coded checklist of freshwater animals by Maitland (1977) [(80)] which helps us to continue to use it despite taxonomic changes since then. Some families that we recognise now as distinct families are combined as BMWP-composite taxa.

3.1.5 WHPT (Walley Hawkes Paisley Trigg) indices for older data sets

WHPT is a development of BMWP. Like BMWP, it is expressed as an ASPT and Ntaxa. The main differences are that it includes more families, particularly of Diptera. BMWP-composite families are considered as separate families, each with their own index values, and different abundances of each taxon also have their own index values. This not only improves its accuracy, but allows it to respond to subtle pressures around the Good-Moderate status boundary that affect abundances before the taxonomic composition changes.

WHPT was derived from an analysis of invertebrate data and BMWP-ASPT values from a very large data set of approximately 100,000 samples collected and analysed by standard RIVPACS procedures (see **Chapter 2**). In its standard form, it is the basis for the UK's river invertebrate (general degradation) status classification. Both the standard form of WHPT and the classification are described in detail in **Chapter 3**.

WHPT – Walley Hawkes Paisley Trigg indices

WHPT indices were developed from BMWP indices, which they replace. The most common form of WHPT uses abundance data but versions of WHPT have been devised for use with older data sets without abundance data or identified to the level used for BMWP indices. WHPT indices are more precise and more accurate than BMWP and are therefore better able to detect nutrient and other mild pressures.

Dikerogammarus haemobaphes

Variations of the standard WHPT indices have been devised so that it can be estimated from older data analysed to the level required for BMWP indices (ie RIVPACS Taxon level TL1), without records for abundance or the additional families used by WHPT but not by BMWP. Whereas the standard abundance-related version of WHPT ASPT can only be calculated from standard RIVPACS samples, the presence-only version described below is suitable for samples collected by other methods.

If abundance data is not available or the sample has not been collected by standard RIVPACS methods you can estimate WHPT ASPT by using index values for the presence of taxa termed 'presence only' or in RICT2 'non Ab' index values. Index values have also been devised for BMWP composite families (termed 'CompFam' in RICT2). These are listed in the paper by Paisley *et al.* (2014) [41] and included on the WHPT calculator spreadsheet. These forms of WHPT ASPT are more accurate than BMWP ASPT, but they are not as accurate as the abundance-related version of WHPT using distinct (rather than composite) taxa and they must not be used to determine WFD status class. Being on the same scale, these forms of WHPT ASPT are comparable with the standard form of WHPT ASPT, but WHPT NTaxa is only comparable if it is based on the same group of taxa.

3.1.6 Saprobic indices

The saprobic index is the principal metric used to determine river invertebrate status in central Europe. It is used in the UK as a component of the SmartRivers biometric analysis (**Section 3.1.24**) because it is based on species-level data. It is a measure of sensitivity to organic enrichment (saprobity) and is often used in combination with other metrics to determine status, in the same way that WHPT ASPT is used with WHPT NTaxa in the UK.

Saprobic indices

Saprobic indices are used mainly in central Europe as an index of organic enrichment and are used in many countries as a basis for WFD river invertebrate status classification. There are many versions of saprobic indices but they are generally based on species-level analyses.

Saprobity is the effect of decaying organic nutrients, analogous to eutrophy which is the effect of inorganic nutrients. The saprobic index relates directly to measures of putrescible organic matter, in particular biochemical oxygen demand (BOD), and saprobic values for individual species are determined by their relation to this measurement (Table 5.1). The monograph by Sládeček (1973) [76] explains how this can be done, and the same guidance can be applied to the derivation of biotic indices relating to other chemicals and pressures.

The saprobic index is the earliest biotic index of water pollution, first devised by Zelinka and Marvan in 1908–1909, and its basic principles are followed by almost all subsequent biotic indices of organic pollution. The index has undergone many revisions since then, and there are now many regional variations. The index has been extended to cover not only natural waters, but treated water for domestic and other uses, through to conditions found in waste (Table 5.1 and Figure 5.3). Saprobic indices exist for all types of biological quality elements, from protists to fish, birds and mammals, but they are most often applied to macroinvertebrates.

Table 5.1

Degrees of saprobity: red text indicates extended scale beyond that found in surface waters and used for waste waters; green text indicates extended scale into water treated for supply (drinking water).

	Index Value	BOD_5
Xenosaprobity	0	0
Oligosaprobity	1	<1
β-mesosaprobity	2	<5
α-mesosaprobity	3	<13
Polysaprobity	4	>20
Isosaprobity	5	Ciliates
Metasaprobity	6	Flagellates
Hypersaprobity	7	Bacteria
Ultrasaprobity	8	Abiotic

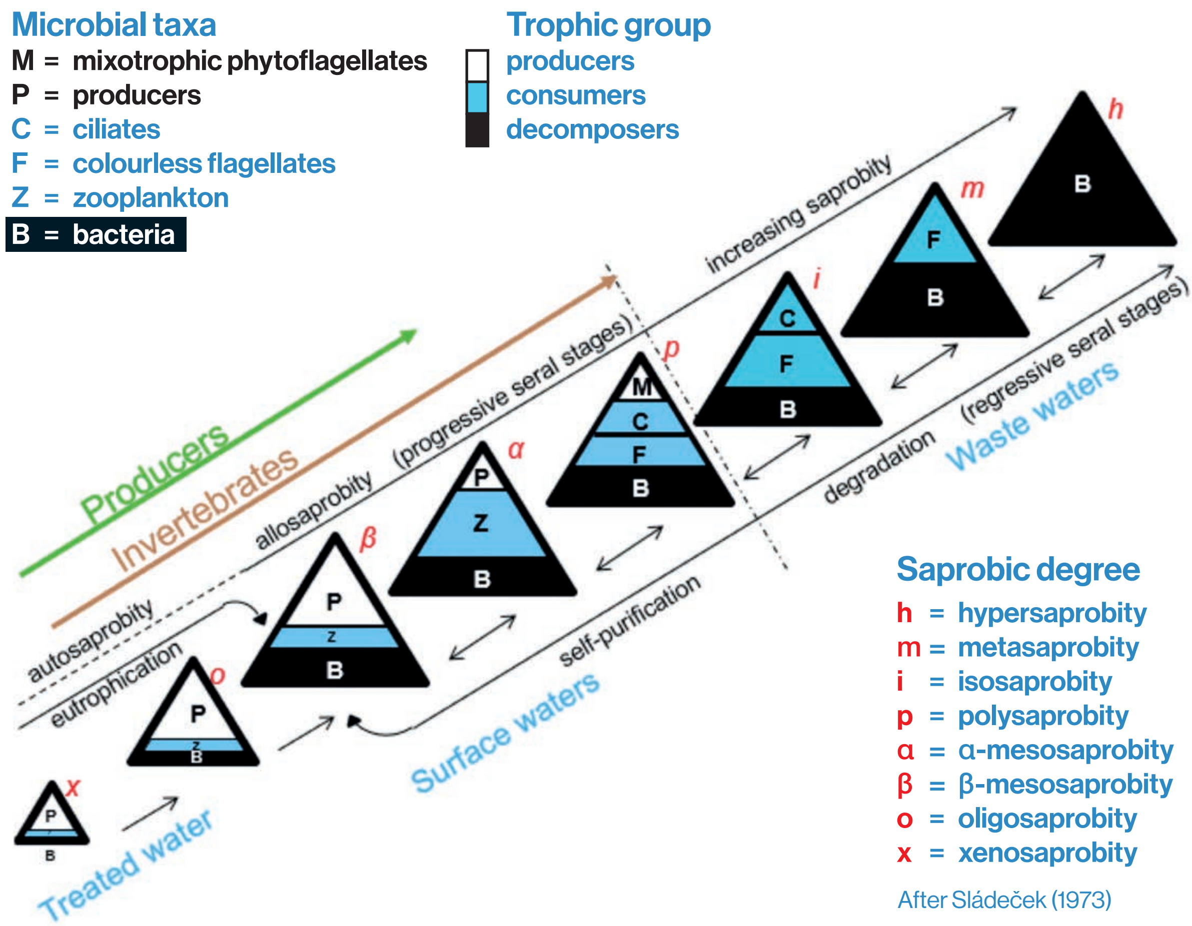


Figure 5.3

Saprobic scale indicating biological communities associated with different degrees of saprobity *(after Sládeček 1973).* [76]

Saprobic indices are average scores per individual organism (in contrast to ASPTs, which are average scores per taxon). However, most include an indicator weighting according to the narrowness of the range of saprobity in which an organism is found. The highest weighting is given to organisms that are found in a narrow range of conditions and which are therefore good indicators of that particular quality.

A general formula is given below:

$$\text{Saprobic index} = \frac{\Sigma\,(\text{weighting} \times \text{value} \times \text{abundance})}{\text{total abundance}}$$

Organic loading and hence its breakdown (saprobity) in surface waters can be generated by autotrophic production by aquatic plants and algae – a process known as autosaprobity – and the input of allochthonous organic matter from leaves, run-off and waste discharges is termed allosaprobity. Because nitrogen and phosphorus compounds are released by the breakdown of organic matter, the saprobic scale is intimately related to the eutrophic scale and therefore to the chemical indicators of eutrophication, though the concentrations vary between different types of surface waters, hence differing nutrient standards in rivers and lakes. However, this direct relationship is sometimes forgotten, and nutrient standards can be misaligned to standards for organic loading.

Table 5. 2

The relationship between indicators of saprobity and eutrophy (Wegl 1983) [81]

Saprobity	BOD_5 (mg/l)	Index Value	NH_4 (mg/l)	O_2 (mg/l)	Trophy	Total P (mg/m³)	Chlorophyll-a (mg/m³)	Summer Secchi (m)	Biomass (mg/m³)	Total N (mg/m³)
Oligosaprobic	<1	1	<0.1	>8	**Oligotrophy**	<13	<3	>5	<2,000	<300
β-mesosaprobic	<5	2	<0.5	>6	**Mesotrophy**	<40	<10	5–1	<7,000	<400
α-mesosaprobic	<13	3	<13	>2	**Eutrophy**	<100	<40	1–0.5	<10,000	<1,000
Polysaprobic	>20	4	>20	<1	**Hypertrophy**	>100	>40	<0.5	>10,000	>1,000

The official Austrian river invertebrate status classification based on saprobic indices follows ÖNORM M6232, and is calculated using ECOPROF version 5.0 software, available since February 2019 at: **https://www.ecoprof.at/index.php/Allgemeines.html**

The official German river invertebrate status classification based on saprobic indices following DIN 38 410, uses the Deutscher Saprobienindex (neue Version), which can be calculated using desktop ASTERICS Version 4.0.4 or the online PERLODES software, available from: **https://www.gewaesser-bewertung-berechnung.de/index.php/home.html**

Ecdyonurus sp nymph

3.1.7 ICMi (Intercalibration Common Metric index)

ICMi [82] was developed as a fully WFD compliant metric for classifying river invertebrate status that could be applied to data from all countries, to facilitate the intercalibration of river invertebrate classifications throughout Europe, including the UK (see **Chapter 1 Section 5.8**). It was adopted as the national classification metric in Italy, and I recommend it for new Member States that have not developed their own national classification metrics for WFD status classification, at least for their first cycle of river basin planning.

This can give new Member States time to develop their own methods based on the monitoring data that they collect in their first cycles of WFD river basin planning.

ICMi does not need intercalibration other than to check the local reference definitions and it will already have been compared to national classification metrics used by other Member States sharing transboundary rivers and therefore river basin plans. It is truly compatible with WFD. Its class boundaries have been defined already and it is easy to compute.

ICMi is a multi-metric invertebrate index based on family-level analysis so that every Member State can calculate it. The index covers the three main components of quality defined in the WFD's normative definitions (Annex V) with equal weightings, although the weights given to individual metrics were not equal but based on their precision and reliability (Table 5.3). ICMi is always expressed as an Environmental Quality Ratio (EQR).

ICMi – Intercalibration common metric index

This index is used to intercalibrate invertebrate status classifications for the Water Framework Directive across central Europe, including the UK, with variations used in other parts of Europe. It is a multi-metric index based on a group of biotic indices that together cover all the invertebrate responses recognised by the normative definitions in the directive.

Table 5. 3

Weightings used to combine intercalibration common metric EQRs in the Intercalibration Common Metric index (ICMi) [82]

Component of quality	Weighting factor	Intercalibration common metric	Weighting factor
Sensitivity/tolerance	0.333	ASPT	0.333
Abundance/habitat	0.333	Log$_{10}$(sel EPTD+1)	0.266
		1-GOLD	0.067
Richness/diversity	0.333	N-families	0.167
		EPT taxa	0.083
		Shannon-Weiner diversity	0.083

ASPT = Biological Monitoring Working Party Average Score per Taxon.

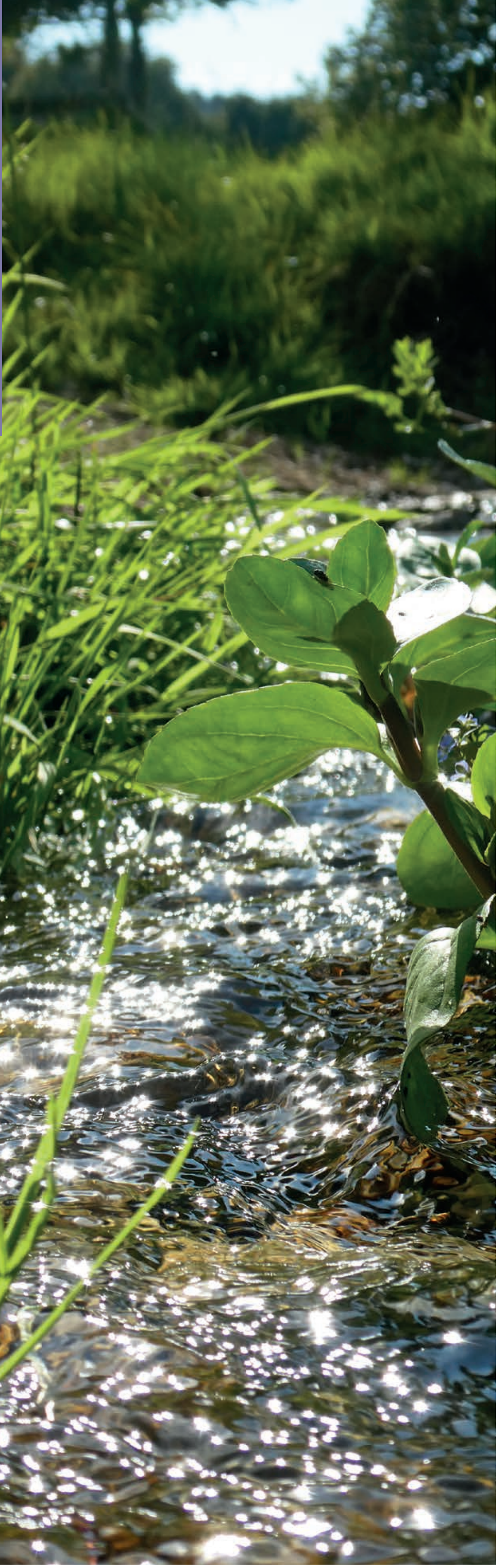
Log$_{10}$(sel EPTD+1) = logarithm of the sum of abundance of selected Ephemeroptera, Plecoptera, Trichoptera and Diptera +1. The selected EPTD are Heptageniidae, Ephemeridae, Leptophlebiidae, Brachycentridae, Goeridae, Polycentropodidae, Limnephilidae, Odontoceridae, Dolichopodidae, Stratiomyidae, Dixidae, Empididae, Athericidae & Nemouridae.

1-GOLD = 1- abundance of Gastropoda, Oligochaeta and Diptera.

EPT taxa = number of families of Ephemeroptera, Plecoptera and Trichoptera.

The intercalibration common metrics are expressed as EQRs by dividing by their reference values. They are multiplied by the weighting factors and summed before the final ICMi is again expressed as an EQR.

ICMi can be calculated with ASTERICS software, available at:
https://gewaesser-bewertung.de/index.php?article_id=419&clang=0



3.1.8 AWIC (Acid Water Indicator Community) indices

AWIC is a diagnostic index sensitive to acidification in British rivers. It responds to base-flow and storm-flow pH and acid neutralising capacity (ANC). WFD $AWIC_{sp}$ is a particular form of AWIC that is used for status classification and is described in **Chapter 3 Section 6.**

AWIC – Acid Water Indicator Community index

This index is used to assess the impacts of acidification and is the basis for the UK's status classification for streams subject to acidification from acid deposition.

The first version of this index was the family-level AWICfam: Davy-Bowker *et al.* (2005), [83] Ormerod *et al.* (2006). [84] For species-level data with abundance data, use WFD AWICsp – see **Chapter 3 Section 6** (McFarland 2010) [85] and WFD-UKTAG (2014) [86] – which is also used for determining acidification status class. WFD AWICsp was preceded by $AWIC_{sp}$ (Murphy 2009, [87] Murphy *et al.* 2013, [88] and McFarland 2010 [85]) which is based on presence-only data and you can use it if you do not have abundance data. However, it must not be used for status classification. AWIC responds to acidification from both acid deposition and acid mine drainage, and it can be impossible to differentiate them.

The species-level $AWIC_{sp}$ index values were derived empirically from a multivariate ordination based on a data set of 197 sites to quantify variation in macroinvertebrate assemblages, and to identify which environmental variables best described the ordination. Taxa were ranked along an acid–base gradient, having first considered the merits of factoring out confounding variation from natural environmental factors.

Index values for each species (integer values from 1–9) were based on the percentage distance along this canonical correspondence analysis (CCA) axis.

The resulting $AWIC_{sp}$ was tested using an independent data set. This was similar to the approach used to derive the CoFSI index of fine sediment (**Section 3.1.14**) and the MetTol index (**Section 3.1.17**). WFD $AWIC_{sp}$ (**Chapter 3 Section 6**) was derived from this index.

3.1.9 LIFE (Lotic-invertebrate Index for Flow Evaluation)

The Lotic-invertebrate Index for Flow Evaluation was developed by Chris Extence and his colleagues in Anglian Region of the Environment Agency to assess the potential impact of flow-related stresses on river invertebrate communities (Extence *et al.* 1999).[89] LIFE is used by the Environment Agency for Hydroecological Validation (HEV) to help manage water resources.

LIFE – Lotic-invertebrate Index for Flow Evaluation

LIFE index relates to the flow velocity preferences of invertebrates and is used to evaluate flow pressures.

Each species or family is assigned to one of six flow groups according to their perceived association with different flow conditions, although many taxa can be found in a range of habitats and flow types (Table 5.4). LIFE can be calculated for families LIFE(F) or for species LIFE(S). LIFE includes some estuarine families that are not found in purely freshwaters because saline intrusion is a common effect of reduced flows in the most downstream reaches of watercourses.

Different index values (flow scores) are given to each taxon depending on its flow group and abundance and the index is expressed as an average (flow score) per taxon. LIFE uses the same $\log_{10}$ abundance categories as RIVPACS, but designated A–E rather than 1–5 to avoid confusion with Flow Groups. LIFE should only be calculated from standard RIVPACS samples (described in **Chapter 2**).

LIFE has been incorporated in RIVPACS (Clarke *et al.* 2003) [90] so that it can be standardised across river types as O/E ratios. This is the form in which it is used for Hydroecological Validation (see **Section 3.2.5**). LIFE can vary between seasons, so spring and autumn samples are analysed independently, and impact is generally based on the season indicating poorest quality.

A LIFE index has been developed for New Zealand (Greenwood *et al.* 2016) [91] using the same principles as the UK version.

Table 5. 4

Flow associations used in LIFE

Group	Primary flow association	Typical mean current velocity
I	Rapid	>100 cm.s^{-1}
II	Moderate to fast	20–100 cm.s^{-1}
III	Slow to sluggish	< 20 cm.s^{-1}
IV	Flowing (usually slow) and standing waters	
V	Standing waters	
VI	Drying or drought-impacted sites	

Asellus aquaticus mating pair

3.1.10 DEHLI (Drought Effect of Habitat Loss on Invertebrates)

DEHLI quantifies the impact of drought on in-stream macroinvertebrate communities by assigning values to taxa according to their likely association with key stages of channel drying. It was devised to complement the LIFE index, which responds to flow velocity.

DEHLI Index – Drought Effect of Habitat Loss on Invertebrates

DELHI identifies areas where river restoration or revised abstraction licenses may be needed – to increase resilience to the effect of anthropogenic activities exacerbated by over-abstraction and climate change.

Although DEHLI has been designed to operate at family level, some genera need to be identified because a few families include genera with starkly differing habitat requirements: for example, Leptophlebiidae and Taeniopterygidae.

Drought Intolerance Score (DIS) values between 1 and 10 are allocated to each family, based on their intolerance to drying.

DEHLI is an average score per taxon based on DIS using the following equation:

$$\text{DEHLI} = \Sigma\ \text{DIS}/n$$

Where n = the number of scoring taxa in the invertebrate sample

DIS = Drought Intolerance Score

DEHLI index values of around 10 imply little or no evidence of an ecological impact from drought. Values towards 1 imply significant impacts associated with the advanced stages of a drought.

Chadd *et al.* (2017) [92] provides a useful reference to this work.

3.1.11 MIS-index (Monitoring Intermittent Streams index)

The MIS-index describes the total invertebrate community response to intermittent flow (England *et al.* 2019). [93] It incorporates invertebrate taxa from fully aquatic to terrestrial, all of which are collected during standard biomonitoring surveys undertaken by regulatory agencies (standard RIVPACS methods, sampling all wet habitats in proportion to their occurrence).

MIS-index – Monitoring Intermittent Streams index

The MIS-index describes the total invertebrate community response to intermittent flow in rivers.

Early indications are that the MIS-index complements existing indices used to assess aquatic invertebrate community responses to drought (ie DEHLI; Chadd *et al.* (2017) [92]) and to changes in flow (LIFE; Extence *et al.* (1999) [89]), by characterizing responses to flow intermittence and changes in flow state. Developed for lowland groundwater-fed streams in southern England, the MIS-index requires testing to see how applicable the taxa-habitat associations and weightings are across different regions and different types of intermittent rivers and ephemeral streams.

Invertebrate taxa (family, genera and species) are assigned to one of six MIS-groups based on their association with lotic (fast), lotic, generalist, lentic, semi-aquatic, and terrestrial habitats. Weighting factors are applied to the richness of each group to give a single score, with different weighting factors used in spring and autumn.

Serratella ignita nymph

Goera pilosa larva

Table 5. 5

Weighting factors for each MIS-group, with different values for autumn and spring

MIS-group	Autumn	Spring
Lotic (fast)	13	13
Lotic	7	11
Generalist	-2	2
Lentic	-10	-3
Semi-aquatic	-10	-7
Terrestrial	-17	-21.0

MIS-index scores are calculated as an ASPT using the formula:

$$MIS\_index_j = \frac{\sum^{i=1..6} M_{si} . T_{ij}}{\sum T_j}$$

$i=1..6$ 1..6 denotes the six MIS-groups
s is the season (separate weighting factors for spring and autumn samples)
M_{si} is the weighting factor for MIS-group i in season s
T_{ij} is the number of taxa in group M_{si} in sample j
T_j is the total number of taxa in sample j

Higher MIS-index values indicate a dominance of flowing water conditions. Lower MIS-index scores indicate drier conditions. Freshwater ecologists are encouraged to identify semi-aquatic and terrestrial taxa collected in their surveys, to provide data to test the index in intermittent rivers and ephemeral streams across Europe.

Recognizing and understanding responses to natural intermittence will inform our understanding of the biodiversity value of these systems and their responses to human pressures that alter ecological quality.

3.1.12 SAGI (Salinity Association Group Index)

The Salinity Association Group Index (SAGI) can be calculated using data at family or mixed (species) taxonomic level. The index works in a similar way to LIFE (**Section 3.1.9**) Taxa are classified into five groups, termed Salinity Association Groups (SAGs). The assignment of groups was based primarily on the results of a literature search, followed by expert opinion, to sense-check the results of the literature search [94] and to fill gaps in the published material.

SAGI (Salinity Association Group Index)

This index is used to assess the impact of saline intrusion on invertebrate communities.

Table 5. 6

Definition of Salinity Association Groups, from Pickwell (2012) [26]

Salinity Association Group (SAG)	Group definition*
I	Macroinvertebrate taxa which tolerate only salinities below 2.5gL^{-1}, approximately 1.73PSU. *Typically freshwater taxa; may be tolerant of slightly brackish conditions, or completely intolerant.*
II	Macroinvertebrate taxa which can tolerate salinities over 2.5gL^{-1} (1.73PSU) up to a salinity of 10gL^{-1} (7.63PSU). Taxa may be present at slightly higher salinities, but only in small numbers. *Freshwater taxa tolerant of mild brackish conditions.*
III	Macroinvertebrate taxa which are characterised by the largest abundance occurring in the salinity range 8–20gL^{-1} (5.99–16.22PSU). Taxa are tolerant of the salinity range 4–25gL^{-1} (2.85–20.73PSU), but may also be recorded at salinities greater, or less, than those specified in this range. *Characteristic brackish water taxa, tolerant of a wide range of salinity conditions from long-term brackish to near freshwater.*
IV	Macroinvertebrate taxa which tolerate salinities below 20gL-1 (16.22PSU) down to 14gL^{-1} (14.99PSU). Taxa may be present at slightly lower salinities, but only in small numbers. *Long-term brackish taxa tolerant of lower salinities, ie transition zones.*
V	Macroinvertebrate taxa which tolerate only salinities greater than 20gL^{-1}, approximately 16.22PSU. *Full coastal seawater taxa rarely moving into nominally freshwater habitats.*

***Definitions using salinity concentrations are in regular font style, whilst the descriptive definitions of the groups are in italic font style.**

The index value for each taxon depends not only on its SAG but also its RIVPACS Log_{10} abundance category. SAGI should only be calculated from samples collected by standard RIVPACS methods, described in **Chapter 2**.

The index has been used in the south-east of England where there is saline intrusion. The index is likely to detect salinization from industrial discharges and salt mining, but it has not been tested on these yet.

3.1.13 PSI (Proportion of Sediment-sensitive Invertebrates) and E-PSI (Empirically-weighted PSI)

The Proportion of Sediment-sensitive Invertebrates (PSI) – Extence *et al.* (2011) [95] – is a proxy to describe the extent to which riverbeds are composed of, or covered by, fine sediments. It is probably more accurately used as a measure of impact of fine sediment, either natural or anthropogenic. It is not intended to assess the ecological quality of the sediment.

PSI – Proportion of Sediment-sensitive Invertebrates

This index is used as a measure of the impact of fine sediments on invertebrate communities, either natural or anthropogenically derived, for example, from soil erosion.

PSI can be calculated for species (mixed taxonomic level) or family data. Invertebrate taxa are assigned to one of four groups indicating their adaptation to fine sediment deposition (Table 5.7). PSI index values (scores) for each taxon depend on their $\log_{10}$ abundance category.

PSI is calculated as the sum of abundance-related values (scores) for sensitive taxa as a percentage of the abundance-related values for all taxa.

Because the sensitivity ratings depend on abundance, PSI should only be calculated for samples collected by the standard RIVPAC methods described in **Chapter 2.**

Table 5.7

Fine sediment sensitivity rating definitions and abundance-related values (scores) for PSI – from Extence *et al.* (2011). [95]

Group	Fine sediment sensitivity rating (FSSR)	Abundance			
		1–9	10–99	100–999	1000 +
A	Highly sensitive	2	3	4	5
B	Moderately sensitive	1	2	3	4
C	Moderately insensitive	1	2	3	4
D	Highly insensitive	2	3	4	5

PSI is currently used in the Environment Agency's Hydroecological Validation (See **Section 3.2.5**).

The empirically-weighted E-PSI is a relatively minor refinement of PSI that can be used for the same purposes, principally to assess the impacts of sediment pressure on freshwater invertebrate communities. E-PSI can be calculated for species (mixed taxonomic level), (Turley *et al.* 2015 [96]) or family data (Turley *et al.* 2016 [97]).

The main difference between PSI and E-PSI is that in E-PSI weights are given to taxa belonging to the same FSSR (Fine sediment sensitivity rating) (Table 5.7) that vary according to empirical data, although the FSSRs are unchanged to retain the biological basis used in the original PSI. Taxa originally identified as moderately to highly sensitive are assigned a range between 0.5 and 1.0 whereas taxa identified as moderately to highly insensitive are assigned a range between 0.0 and 0.49. Sensitive species are assigned higher weightings because they are considered to be more important for identifying sediment pressures.

Table 5.8

Interpretation of PSI – from Extence *et al.* (2011). [95]

PSI	Riverbed condition
81–100	Minimally sedimented/un-sedimented
61–80	Slightly sedimented
41–60	Moderately sedimented
21–40	Sedimented
0–21	Heavily sedimented

Like PSI, E-PSI should only be calculated from samples collected by the standard RIVPACS methods because its sensitivity values depend on abundance.

3.1.14 CoFSI (Combined Fine Sediment Index)

$CoFSI_{sp}$ (Combined Fine Sediment Index species) reflects fine sediment stress. See Murphy *et al.* (2015) [98] and Wilkes *et al.* (2017). [99]

It comprises two distinct components:

- $ToFSI_{sp}$ = Total Fine Sediment Index of the inorganic component of fine sediment
- $OFSI_{sp}$ = Organic Fine Sediment Index of the organic component of fine sediment

CoFSI – Combined Fine Sediment Index

This index reflects fine sediment stress on invertebrates. It has inorganic and organic components and can help in identifying causes of sediment impact.

The index values are based on the position of species along Canonical Correspondence Analysis (pCCA) axes describing gradients of total fine sediment (ToFSI) and organic matter (OFSI). These indices are combined to form CoFSI, although they are also useful individually for diagnosing causes of impact. $ToFSI_{sp}$ and $OFSI_{sp}$ are expressed as ASPTs.

$$CoFSI_{sp} = 0.349\ OFSI_{sp} + 0.569\ ToFSI_{sp} - 6.80$$

It was designed from the outset to be used with RIVPACS and to be expressed as an EQI.

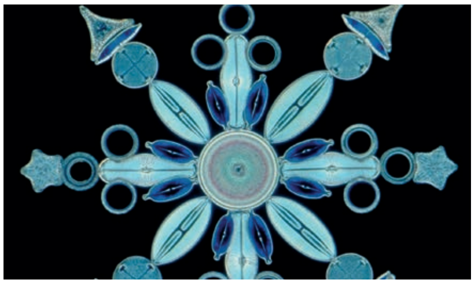
3.1.15 SPEAR (species at risk) indices overview

SPEAR (SPEcies At Risk) indices are based on various life-cycle and physiological traits of the constituent taxa to which species are defined either as being at risk or not at risk. The biotic index for a sample is expressed as the proportion (as a percentage) of taxa or individuals (depending on the pressure) of species at risk. SPEAR indices therefore address directly the ratio of disturbance-sensitive taxa to insensitive taxa demanded by WFD normative definitions (WFD, Annex V).

SPEAR – Species at risk indices

This group of Species at Risk indices are based on various life-cycle and physiological traits of the constituent taxa. They have been developed to assess different types of pressures. They can be regarded as an alternative to the saprobic approach.

SPEAR is an alternative to the saprobic approach to developing and calculating biotic indices. Different SPEAR indices have been developed to assess different types of pressures:



SPEAR$_{\text{habitat}}$ – Species at risk from habitat
SPEAR$_{\text{herbicides}}$ – Species at risk from herbicides, based on diatoms
SPEAR$_{\text{mesocosm}}$ – Species at risk from pesticides in mesocosm studies
SPEAR$_{\text{metals}}$ – Species at risk from dissolved metals
SPEAR$_{\text{pesticides}}$ – Species at risk from pesticides
SPEAR$_{\text{salinity}}$ – Species at risk from salinity (Australia)
SPEAR$_{\text{refuge}}$ – Species characteristic of refuge areas

Apart from **SPEAR**$_{\text{herbicides}}$ (Wood *et al.* 2019) [100] which is based on diatoms, SPEAR indices have been developed for invertebrates.

The next section, about **SPEAR**$_{\text{pesticides,}}$ provides an overview of principles of the SPEAR method.

3.1.16 SPEAR$_{\text{pesticides}}$ (Species at risk from pesticides)

The most widely used version of SPEAR$_{\text{pesticides}}$ in the UK is a version devised in 2008 specifically for the UK. It is used by the Environment Agency and by Salmon and Trout Conservation. However, SPEAR$_{\text{pesticides}}$ has undergone several refinements since then; the current version no longer includes different versions for different regions of Europe.

Simuliidae sp.

The UK version of SPEAR is no longer available in the official SPEAR software, which is now a web app **https://www.systemecology.de/indicate/**, but it was included in older versions that users installed locally.

SPEAR$_{\text{pesticides}}$ – Species at risk from pesticides

This index is used to assess contamination from pesticides and other organic chemicals and is based on toxicity data; it is the most widely used SPEAR index.

Ranatra linearis nymph

Both the old UK version and the current version of SPEAR$_{\text{pesticides}}$ are described in this section.

SPEAR$_{\text{pesticides}}$ is the only trait-based approach to pesticide contamination. However, it responds to a wider range of organic chemicals because it takes account of the toxicity of other organic chemicals.

For SPEAR$_{\text{pesticides}}$, a species is considered to be at risk if:

- its physiological sensitivity to toxicants is equal to or higher than *Daphnia* sensitivity (S_{organic} value above -0.36, see Von der Ohe, Carsten & Liess 2004) [101]
- it produces two or less generations a year (ie its generation time is half a year or longer)
- it is fully aquatic or does not emerge before the main period of agrochemical application (ie it has aquatic stages during May–June).

Gyrinus natator adult female

SPEAR$_{\text{pesticides}}$ is the proportion of individual invertebrates that are at risk and is calculated according to the equation:

$$\text{SPEAR}_{\textit{pesticides}} = \frac{\sum_{i=1}^{n} \log(x_i + 1) \cdot y}{\sum_{i=1}^{n} \log(x_i + 1)} \cdot 10$$

Elodes sp. larva

where n is the number of taxa
x_i is the abundance of the taxon i
and y is 1 if taxon i is classified as 'at risk', otherwise 0

SPEAR$_{\text{pesticides}}$ was originally developed in Central Germany (Liess and Von der Ohe 2005). [102] It was adapted for use in Britain following a review of biological methods for monitoring pesticides for the Environment Agency (Schriever *et al.* 2008). [103]

To adapt the SPEAR$_{\text{pesticides}}$ for England and Wales, 38 new taxa were added to the SPEAR database as well as UK-specific ecological data including life-cycle traits and, in particular, emergence times (Beketov *et al.* 2008). [104]

Notonecta sp. nymph

$SPEAR_{pesticides}$ was based on species-level data, but in the mid-2000s much of the monitoring data in the UK was from family-level analyses, so a family-level version of the index was developed $SPEAR(fm)_{pesticides}$, the species-level index being designated $SPEAR(sp)_{pesticides}$.

Definitions of family at risk or not at risk for $SPEAR(fm)_{pesticides}$ were derived from data in the database for the majority of the species comprising the family (above 50 per cent), so all species from a particular family have the same family-level SPEAR definition. As for species, defining the 'families at risk' was performed automatically by the algorithm in the database. SPEAR definitions for 152 families were created including 66 families at risk, 83 not at risk, and three that were not found in the UK.

The performance of $SPEAR_{pesticides}$ was validated in Europe by comparing it with an index derived from chemical monitoring data. This index is the maximum chemical Toxic Unit (TU_{max}), based on the maximum peak water concentration of all the pesticide data collected on a sampling visit. A tool for calculating this index is included in the SPEAR Calculator.

TU_{max} uses 48hr LC50 toxicity data for each pesticide (based on *Daphnia magna*). The 48hr LC50 for *Daphnia magna* is the concentration of the chemical under question that kills 50% of the test animal in 48 hours. This toxicity data is included in the SPEAR Calculator. TU_{max} is calculated according to the equation:

$$\mathrm{TU}_{(Daphnia\ magna)} = \max_{i=1}^{n} \left(\log\left(\mathrm{C}_i / LC50_i\right)\right)$$

Where TU is the maximum number of toxic units of the n pesticides detected at the considered site, C_i is the concentration (µg/l) of pesticide i, and $LC50_i$ is the 48-hr $LC50$ of pesticide i for *Daphnia magna* (µg/l).

Beketov *et al.* (2008) [104] recommended that the UK adaptation of $SPEAR_{pesticides}$ should be validated in the field. This was attempted by Graham & Gavin (2010) [105] using Environment Agency biological and chemical monitoring data to calculate the statistical relationship between the UK adaptation of $SPEAR_{pesticides}$ and TU_{max}. To do this, they developed a tool that calculated $SPEAR_{pesticides}$ and TU_{max} from biological and chemical data from Environment Agency databases and calculated the statistical relationship between them. The regressions that they produced were weak. Pesticides were rarely monitored because of the cost, and where they were analysed, most results were below the limit of detection. This is still true and still hinders testing the index's performance against pesticide concentrations in the field.

The current version of $SPEAR_{pesticides}$ includes several refinements. It underwent a major change in 2018, in particular the reclassification of 11 common taxa (mostly families) as being invulnerable to pesticides (Knillmann *et al.* 2018) [106] to remove the influence of refuge areas, which confounded the original version. A log(4x+1) transformation for abundance was introduced, as suggested by Knillmann [106] to decrease the influence of populations with mass developments:

$$\mathrm{SPEAR}_{pesticides} = \frac{\sum_{i=1}^{n} \log\left(4\mathrm{x}_i + 1\right) . y}{\sum_{i=1}^{n} \log\left(4\mathrm{x}_i + 1\right)}$$

At the same time, $SPEAR_{pesticides}$ was expressed as a decimal fraction, as in the formula above, instead of a percentage. Versions of $SPEAR_{pesticides}$ for different countries were replaced by a single version for Europe.

$SPEAR_{pesticides}$ was revised again in 2019 and most recently in February 2021.[107]

Software for calculating $SPEAR_{pesticides}$ is included in Indicate (Figure 5.4), which is available from **http://www.systemecology.eu/indicate/** Until late 2018, Indicate was called the SPEAR Calculator, and from 2021 it is a web app rather than a program for downloading onto a local drive.

Indicate

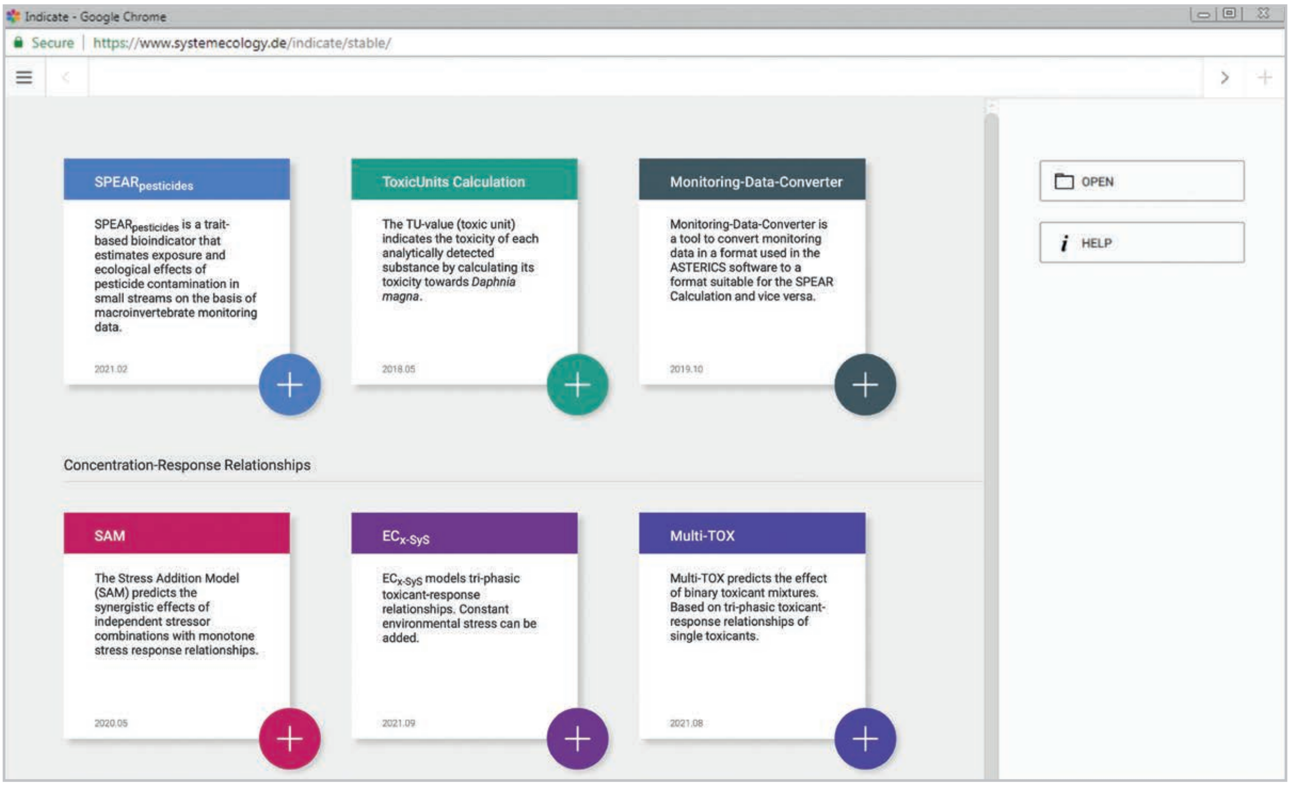


Figure 5.4
Screenshot of Indicate version 2.0.0, with link to the SPEAR$_{pesticides}$ Calculator

You can view the SPEAR$_{pesticides}$ database from the SPEAR$_{pesticides}$ Calculator in Indicate. It contains information about each of the ecological traits that define whether an invertebrate taxon is at risk or not at risk. In addition to ecological traits, the database contains references to the information sources used. The database no longer has separate information for each country.

More information about SPEAR$_{pesticides}$ is included on the official landing page for SPEAR **https://www.ufz.de/index.php?en=38122** and in Reiber *et al.* 2020.[107] This includes a link to Indicate **https://www.systemecology.de/indicate/**, from where you can access a helpful change log and documentation for SPEAR$_{pesticides}$, which includes information about the changes.

One of SPEAR's authors recommends that we use the new version in the UK because the benefits of the new version in reducing the influence of uncontaminated refuges are substantial (Liess, pers. comm.). The differences between British and continental European species are not substantial for these kinds of calculations.

3.1.17 MetTol (Metal Tolerance) index

The Metal Tolerance (MetTol) index is used to evaluate ecological damage caused by metal contamination. At ecologically degraded sites, it can be used to assess the degree to which metal contamination is a contributory cause.

MetTol – Metal Tolerance index

The Metal Tolerance index is for evaluating ecological damage caused by metal contamination.

MetTol uses species-level data and is expressed as an ASPT. It was designed to be compatible with current monitoring methods and for use with RIVPACS as an EQR (Environmental Quality Ratio).

MetTol = sum of MetTol value for each taxon / number of MetTol-scoring taxa

MetTol was derived from a survey of sites in 20 spatially independent river catchments in areas affected by metal mine facilities.[108] Ordination techniques were used to determine the influence of the metal stress gradient on macroinvertebrate community composition, whilst accounting for natural background variation among stream types. Using this approach, the relative sensitivity of a range of macroinvertebrates to metals stress was quantified and used as the empirical basis of MetTol.

Tests confirmed that the MetTol index is sensitive to bioavailable metal contamination, whether it is determined from analysis of tissue concentrations in selected biomonitoring taxa, predicted from stream water chemistry, or inferred from metal concentrations in the bed sediment.

3.1.18 TRPI (Total Reactive Phosphorus Index)

The Total Reactive Phosphorus Index (TRPI) – Everall (2010) [109] and Everall *et al.* (2019) [110] – relates river invertebrates to the concentration of total reactive phosphorus (TRP) in rivers. Phosphorus is usually the limiting nutrient for plant growth in rivers, so its enrichment is a key driver of eutrophication. The main artificial sources in freshwaters are sewage discharges and agricultural run-off.

TRPI – Total Reactive Phosphorus Index

The Total Reactive Phosphorus Index relates river invertebrates to the concentration of total reactive phosphorus in rivers.

The index is based on an analysis of the relationship between TRP and river invertebrates by Paisley *et al.* (2003, 2011) [111] [112] using information theory and neural networks to analyse data from the Environment Agency's national general quality assessment survey of river quality in England and Wales in 1995. This data, and therefore TRPI, is based on 76 families of invertebrates analysed to the level used in BMWP (**Section 3.1.4,** ie RIVPACS Taxon Level TL1). Only sites in GQA classes **a** (very good biological quality) and **b** (good biological quality) were analysed, but sites with total ammoniacal nitrogen concentrations greater than 0.15 mg/l or 5-day biochemical oxygen demand greater than 2.25 mg/l were excluded, to remove sites suffering from organic pollution, as well as some other outliers. To reduce the influence of natural environmental pressures and season, the data was split into 5 river types as well as keeping RIVPACS spring and autumn data separate. The river types were differentiated using neural network analysis, which identified altitude, alkalinity and substrate composition as the key controls to the macroinvertebrate community response to TRP. [44]

TRPI is based on the mutual information between the occurrence of a taxon and TRP concentrations for each type/season. Mutual information (MI) is a measure of the amount of information one random variable contains about another. The sensitivity groups do not relate to concentrations of TRP but to the strength of the association between the taxon and TRP. The sensitivity group to which a BMWP family belongs and its RIVPACS log_{10} abundance category are used to identify a nutrient score (Table 5.9).

Table 5.9

Relationship between significance, indicator, sensitivity group. Compiled from information in Paisley *et al.* (2011) [112] and Everall *et al.* (2019), [110] with tolerance definitions from Supplementary Information A from Everall *et al.* (2019). [110]

Significance of MI	Indicator	Sensitivity Group	TRP tolerance definition	Nutrient score according to log abundance categories			
				1–9	10–99	100–999	1000 +
1%	-	A	Very sensitive	2	3	4	5
5%	-	B	Sensitive	1	2	3	4
5%	+	C	Tolerant	1	2	3	4
1%	+	D	Very tolerant	2	3	4	5
>5%		E	Indifferent or excluded for other reasons	-	-	-	-

TRPI is calculated as the percentage of phosphorus sensitive indicators.

$$TRPI = \frac{\sum \textit{Nutrient scores for Groups A\&B}}{\sum \textit{Nutrient scores for Groups A,B,C,D}} \times 100$$

To determine TRPI you will have to identify the site type and then refer to tables in Supplementary Information from Everall *et al.* (2019) [110] to determine the sensitivity groups, because families can belong to a different sensitivity group in a different site type or season. Although Everall *et al.* (2019) gives a table defining each site type it cannot identify the site type in all cases. This is because they were originally based on the overall pattern of six variables (altitude, alkalinity, and substrate percentage of boulders, pebbles, sand and silt) identified by a standard back-propagation neural network, not on discrete bands of the variables (Walley and Fontama 1998). [113]

Because TRPI index values depend on abundance, it can only be calculated from samples collected by standard RIVPACS methods.

3.1.19 CCI (Community Conservation Index)

CCI is an index of the conservation value of a site based on the macroinvertebrates present. It therefore complements other biotic indices, which relate to environmental quality.

CCI – Community Conservation Index

The Community Conservation Index is an index of the conservation value of a site based on macroinvertebrates. It emphasises the rarity of individual species and Red Data Book status.

The CCI for a site is the product of the Community Score and the average Conservation Score. Conservation Scores of between 1 and 10 have been assigned to each macroinvertebrate species based on its rarity, according to its Red Data Book category or lesser conservation status. The Community Score (value from 1 to 15) is based on the BMWP-score or the species in the sample with the highest Conservation Score.

Theoretically, CCI is appropriate for any wet habitat in which nominally freshwater species can occur – ie for anything inland (rivers, lakes, wetlands and even damp mud or cattle troughs). Towards the coast, it has been used in saltmarshes (including hypersaline lagoons), but it is not appropriate for fully marine habitats (benthic sub-tidal) – Chadd (2004) [114] and Chadd (2015) pers. comm.

3.1.20 Anglers' Riverfly Monitoring Initiative and ARMI index

The anglers' riverfly monitoring initiative (ARMI) means that amateur volunteers can undertake an ecological assessment of river quality based on invertebrates, so that they can take action to help protect their local river environment. Originally aimed at anglers, monitoring groups have also been set up by some River Trusts and other associations.

https://www.riverflies.org/anglers-riverfly-monitoring-initiative-armi also **https://www.fba.org.uk/volunteer/riverfly-partnership**

ARMI - Anglers' Riverfly Monitoring Initiative

The Anglers' Riverfly Monitoring Initiative allows amateur volunteers to undertake an ecological assessment of river quality based on invertebrates, so that they can take action to help protect their local river environment.

The scheme is designed for use by amateurs, so its methods are simple but are capable of detecting severe degradation in river water quality. It enables volunteer monitoring groups to provide warnings to the statutory agencies (Environment Agency, Scottish Environment Protection Agency, National Resources Wales or Northern Ireland Environment Agency). ARMI monitoring is usually monthly throughout the year, so it complements the less frequent monitoring by regulatory authorities, which is usually no more than twice a year every 3 years.

The Anglers' Riverfly Monitoring Initiative is organised by the Riverfly Partnership and its database is managed by the Freshwater Biological Association. The initiative is supported by training courses, guides and other events. There is an accreditation scheme for trained surveyors.

Samples are collected by the standard RIVPACS method for wadeable streams and rivers (**Chapter 2**) but they are analysed on the bankside. The abundance of 8 easily identified invertebrate indicators are recorded: cased caddisflies, caseless caddisflies, Ephemeridae, Ephemerellidae, Heptageniidae, Baetidae, stoneflies and *Gammarus*. Abundances are recorded on a log scale, similar to WHPT. Because ARMI index values depend on abundance (Table 5.10), it is not suitable for samples collected by other methods. The overall index value for a sample is the sum of these index values, ie an ARMI score.

Table 5. 10

Abundance categories used for ARMI

Abundance	Category	Score
1–9	A	1
10–99	B	2
100–999	C	3
>1000	D	4

Regulatory authorities such as the Environment Agency set trigger levels for each site, and if it falls below this value the riverfly group notifies the regulatory authority so that it can investigate the cause and take measures to restore quality.

A guide to sampling, sorting and identification, together with a recording sheet, has been produced by the Riverfly Partnership (2017) [115] (Figure 5.5). As a quality assurance measure, this is only available to people who have completed a special training course, so that only the people who have the correct training undertake the monitoring. Results are uploaded to the ARMI database and warnings of samples indicating poor quality are sent to the local statutory agency ecologists. Brooks *et al.* (2019) [116] is a useful published reference.

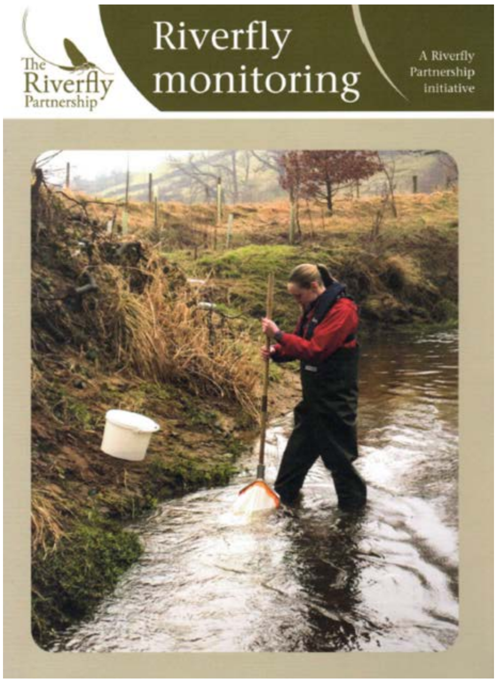


Figure 5.5

Cover of the guide to ARMI methods

The Riverfly Partnership

3.1.21 Riverfly Plus

Riverfly Plus is a group of citizen science methods and surveys that go beyond the basic Anglers' Riverfly Monitoring Initiative.

- Extended Riverfly Index (**Section 3.1.22**)
- Urban Riverfly Index (**Section 3.1.23**)
- SmartRivers (**Section 3.1.24**)
- Ecosystem Function Assessment (**Section 3.1.25**)

Riverfly Plus includes abiotic methods including MoRPh (Modular River Physical survey), Outfall Safari and Freshwater Watch.

See **https://www.riverflies.org**

3.1.22 Extended Riverfly indices

Extended Riverfly [117] is a development of the Anglers' Riverfly Monitoring Initiative based on 33 taxa compared to ARMI's 8 – hence it is sometimes referred to as ARMI 33-group. The additional indicators are flatworms, molluscs, annelids, crustaceans, alderflies, and true bugs.

It includes tolerant as well as sensitive indicators, making it suitable for detecting a wider range of environmental issues including pollution, low flow, and siltation. Samples must be collected in the same way as for ARMI 8-group (the standard RIVPACS method for wadeable streams and rivers) and they are analysed on the bankside.

Figure 5.6
Chart for identifying invertebrates used for the Extended Riverfly index [117]

Extended Riverfly index

This Riverfly Plus citizen science index is based on a greater number of families than ARMI to provide a more precise assessment of river quality.

Two indices are included in the Extended Riverfly scheme: a water quality score, and a silt and flow score. Unlike ARMI 8-group, the index values for different taxa vary: from 4 for the most sensitive indicators to -4 for the most tolerant indicators (water quality index values) and 5 to -5 (silt and flow index values). Different index values are also allocated to each taxon according to its ARMI abundance categories (Table 5.10). However, although crayfish and non-native shrimps are included in the scheme and recorded, they are not included in the water quality score or the silt and flow score. See **https://www.riverflies.org**

Extended Riverfly Groups Field Recording Form

The Riverfly Partnership

Site name:

Grid reference:

Recorder name(s):

The ARMI 8 groups

Survey dates:						
Season:	April	May	June	July	August	September
1 Flatworms						
2 Freshwater Snails						
3 Limpets						
4 Bivalves						
5 Aquatic Worms						
6 Leeches						
7 Crayfish						
8 Water Hoglouse						
9 Freshwater Shrimps						
10 Invasive Non-Native Shrimps						
11 Mayflies - Angler's Curse						
12 Mayflies - Olives						
13 Mayflies - Prong Gilled						
14 Mayflies - Flatbodied						
15 Mayflies - Green Drake						
16 Mayflies - Blue-Winged Olive						
17 Stoneflies						
18 Dragonflies and Damselflies						
19 Water Boatmen						
20 Other Water Bugs						
21 Water Beetles (Adults and Larvae)						
22 Alderflies						
23 Caseless Caddisflies - Green Sedge						
24 Caseless Caddisflies - Net Spinners						
25 Caseless Caddisflies - Non-Gilled						
26 Cased Caddisflies - Hood Case-Maker						
27 Cased Caddisflies - Weighted Case-Maker						
28 Cased Caddisflies - Bush Tailed Caddisfly						
29 Cased Caddisflies - Other						
30 Craneflies						
31 Blackflies						
32 Non-Biting Midges						
33 Water Snipe Flies						
ARMI 8 group trigger: / ARMI 8 group score:						
Survey notes:						

Information about the Riverfly database and scoring systems is given overleaf

Recording Form version 2.6 Aug 2021

Figure 5.7
Recording form for Extended Riverfly monitoring

3.1.23 Urban Riverfly index (by Nicola Edgar, Environment Agency)

Urban Riverfly is a Riverfly Plus scheme that uses an extended list of taxa compared to the original Anglers' Riverfly Monitoring Initiative index, but not so many as Riverfly Plus. It can be used across several different river systems. The addition of 6 more invertebrate groups to the original 8 provides greater sensitivity to help with the evaluation of pressures. The extra groups comprise worms, snails, beetles, leeches, blackfly larvae (Simuliidae) and freshwater hoglouse (Asellidae). They are all easy to identify (Figure 5.8) and are more pollution tolerant invertebrates that are commonly recorded in urban and modified rivers.

Urban Riverfly

Urban Riverfly is a Riverfly Plus citizen science index that uses an extended list of invertebrate families to detect pressures in urban rivers where many of the invertebrates used in ARMI are likely to be absent

The index values (Table 5.11) are impacted both positively and negatively by abundance, so they are more reflective of WHPT which is the standard index used by the UK's environmental protection agencies. Pollution tolerant groups such as leeches, worms and the freshwater hoglouse have lower index values as abundances increase.

The survey method remains the same as the traditional ARMI with a 3-minute kick sample and 1-minute hand search. Because its index values depend on abundance, it is only suitable for samples collected by the standard RIVPACS methods.

Table 5.11
Index values for the urban riverfly index

	Abundance			
	1–9	10–99	100–999	1000 +
Cased caddis	1	2	3	4
Caseless caddis	1	2	3	4
Stoneflies	1	2	3	4
Mayfly (Ephemeridae)	1	2	3	4
Mayfly (Ephemerellidae)	1	2	3	4
Mayfly (Heptageniidae)	1	2	3	4
Mayfly (Baetidae)	1	2	3	4
Freshwater shrimp – *Gammarus*	1	2	3	2
Blackfly larvae (Simuliidae)	1	2	2	0
Beetles	1	2	3	4
Snails	1	1	1	0
Leeches	1	1	0	-2
Freshwater hoglouse (Asellidae)	1	1	0	-2
Worms (Oligochaeta)	1	1	0	-3

The urban riverfly index is calculated in the same way as the ARMI index. However, the index values for the additional indicators are not all the same. Survey results are currently stored on Cartographer.

A pilot of this scheme was started in the Midlands (England) where both new and existing volunteers were provided with one day's training.

The scheme was launched nationally at the Riverfly Conference in March 2020.

Limited information is available from **https://www.riverflies.org/urban-riverfly**

Figure 5.8
Pictorial identification guide used in training for urban riverfly monitoring

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3.1.24 SmartRivers biometric analysis

This is not a biotic index, but a system for analysing data from multiple biotic indices. SmartRivers **https://salmon-trout.org/smart-rivers/** is the most complex of the Riverfly Plus schemes as it is based on species-level analysis in the belief that this is better than family-level analysis.

It is a volunteer extension of the Riverfly Census and uses a similar biometric analysis. Both schemes are organised by Salmon and Trout Conservation. Earlier versions were known as River Invertebrate Identification and Monitoring (RIIM).

SmartRivers

SmartRivers is a riverfly plus citizen science scheme based on species-level analysis.

Sites are sampled twice a year using the standard RIVPACS method for wadeable streams and rivers. Volunteers identify specimens in their samples using the 'River Invertebrate Larvae App' software with high-quality annotated photographic illustrations by Cyril Bennett. Alternatively, they can be analysed by a verified laboratory.

SmartRivers biometric analysis provides an assessment of the likely impacts on the site with a 5-class scale using a range of indices (Table 5.12). Because these are based on the raw values of the biotic indices, they do not take account of the effect of the natural typology on them.

Table 5.12
Impact thresholds for SmartRivers indices (from 2015 Riverfly Census report)

Traffic light measure of respective biological stress signatures					
Stress level descriptor	**Colour code**	**PSI (sediment) value**	**TRPI (nutrient total reactive P value)**	**Organic (Saprobic value)**	**Flow (LIFE value)**
Heavily impacted		**0–20**	**0–20**	**3.2–4**	**<6**
Impacted		**21–40**	**21–40**	**2.7–3.19**	**6–6.49**
Moderately impacted		**41–60**	**41–60**	**2.3–2.69**	**6.5–6.99**
Slightly impacted		**61–80**	**61–80**	**1.81–2.29**	**7.0–7.99**
Un-impacted		**81–100**	**81–100**	**1.0–1.80**	**>8.0**

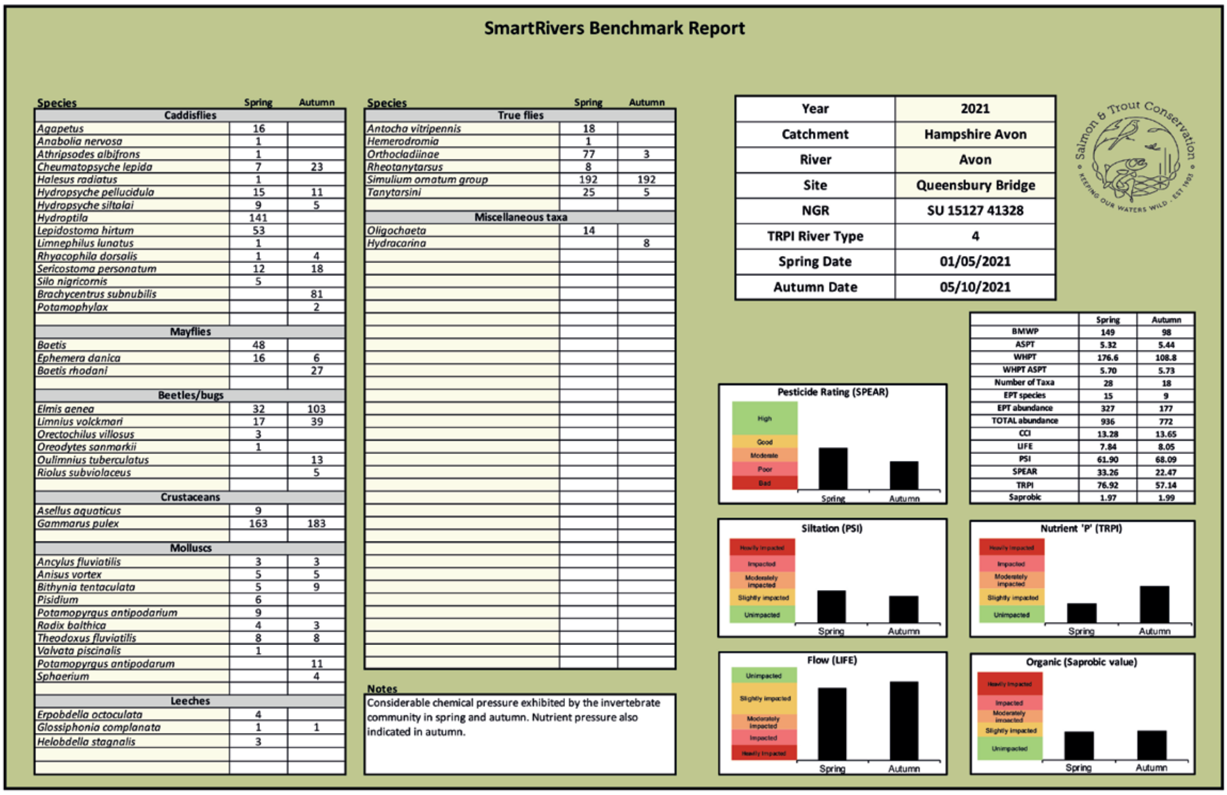
Although not included in the analysis, thresholds are available for BMWP-score and ASPT **https://salmon-trout.org/2021/04/07/smartrivers-other-metrics/**

Table 5.13

SmartRivers threshold values for BMWP-score and ASPT (Salmon & Trout Conservation website, 8 Sept 2021)

BMWP and ASPT scores/gradings:		
ASPT	BMWP	QUALITY DESCRIPTION
≥6	≥85	Excellent
5.0–5.9	70–84	Good
4.2–4.9	50–69	Fair
3.0–4.1	15–49	Poor
<3	<15	Seriously Polluted

Data generated by SmartRivers hubs are available on an open-access database (**https://salmon-trout.org/smart-rivers-resources/**) where it can be viewed on maps or in a standard report that includes a few, mostly family-level, biotic indices (Figure 5.9).



SmartRivers Benchmark Report

Species	Spring	Autumn
Caddisflies		
Agapetus	16	
Anabolia nervosa	1	
Athripsodes albifrons	1	
Cheumatopsyche lepida	7	23
Halesus radiatus	1	
Hydropsyche pellucidula	15	11
Hydropsyche siltalai	9	5
Hydroptila	141	
Lepidostoma hirtum	53	
Limnephilus lunatus	1	
Rhyacophila dorsalis	1	4
Sericostoma personatum	12	18
Silo nigricornis	5	
Brachycentrus subnubilis		81
Potamophylax		2
Mayflies		
Baetis	48	
Ephemera danica	16	6
Baetis rhodani		27
Beetles/bugs		
Elmis aenea	32	103
Limnius volckmari	17	39
Orectochilus villosus	3	
Oreodytes sanmarkii	1	
Oulimnius tuberculatus		13
Riolus subviolaceus		5
Crustaceans		
Asellus aquaticus	9	
Gammarus pulex	163	183
Molluscs		
Ancylus fluviatilis	3	3
Anisus vortex	5	5
Bithynia tentaculata	5	9
Pisidium	6	
Potamopyrgus antipodarium	9	
Radix balthica	4	3
Theodoxus fluviatilis	8	8
Valvata piscinalis	1	
Potamopyrgus antipodarum		11
Sphaerium		4
Leeches		
Erpobdella octoculata	4	
Glossiphonia complanata	1	1
Helobdella stagnalis	3	

Species	Spring	Autumn
True flies		
Antocha vitripennis	18	
Hemerodromia	1	
Orthocladiinae	77	3
Rheotanytarsus	8	
Simulium ornatum group	192	192
Tanytarsini	25	5
Miscellaneous taxa		
Oligochaeta	14	
Hydracarina		8

Notes

Considerable chemical pressure exhibited by the invertebrate community in spring and autumn. Nutrient pressure also indicated in autumn.

Year	2021
Catchment	Hampshire Avon
River	Avon
Site	Queensbury Bridge
NGR	SU 15127 41328
TRPI River Type	4
Spring Date	01/05/2021
Autumn Date	05/10/2021

	Spring	Autumn
BMWP	149	98
ASPT	5.32	5.44
WHPT	176.6	108.8
WHPT ASPT	5.70	5.73
Number of Taxa	28	18
EPT species	15	9
EPT abundance	327	177
TOTAL abundance	936	772
CCI	13.28	13.65
LIFE	7.84	8.05
PSI	61.90	68.09
SPEAR	33.26	22.47
TRPI	76.92	57.14
Saprobic	1.97	1.99

Figure 5.9

SmartRivers database with panel showing a summary biometric analysis *(illustration from Salmon & Trout Conservation)*

3.1.25 Ecosystem function assessment

Another Riverfly Plus citizen science scheme, Ecosystem Function Assessment, assesses the ecological impact of river restoration, but it can also be used to monitor the effects of invasive species and pollution events on ecosystem function.

Ecosystem function assessment

Ecosystem Function Assessment is used to assess the ecological impact of river restoration, and can also be used to monitor the effects of invasive species, and pollution events on ecosystem function.

The method is still in development but is based around an extension of ARMI using simple colonisation traps.

A short section of drainpipe, divided internally in two, is placed on the riverbed. Inside each half of the pipe a known weight of cloth paper is placed. A fine mesh placed over the entrance to one half of the pipe excludes macroinvertebrates but allows access by microbes; macroinvertebrates and microbes can both gain access to the other half.

After 2–4 weeks, the drainpipe is removed, and the paper reweighed. The reduction in weight of paper gives the microbial and macroinvertebrate decomposition rate and a measure of ecosystem function. The greater the decomposition rate, the more energy is released back into the ecosystem. For more information, see **https://www.riverflies.org/Ecosystem_Function_Assessment**

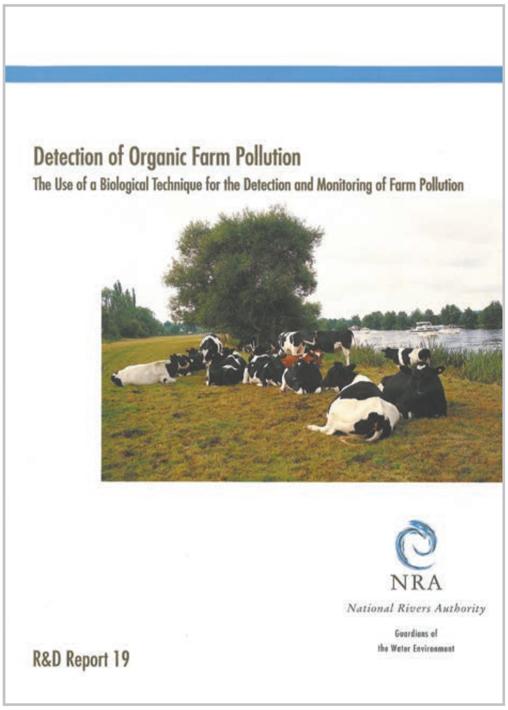
3.1.26 Rapid Biological Appraisal Key (RBAK) for farm pollution

This technique helps to screen shallow streams across whole catchments for farm pollution. It was developed in Wales but is likely to be applicable to the whole of upland England and Wales (Hydro-ecoregions 100, 101 and 114, see Figure 3.13). It has been tested successfully in the south west of England.

The method involves the identification in the field of only four readily identifiable and common invertebrate taxa, plus sewage fungus. No sampling method is provided in the document, but a 1-minute kick should be sufficient for the invertebrates.

RBAK – Rapid Biological Appraisal Key for farm pollution

This technique helps to screen shallow streams across whole catchments for farm pollution.



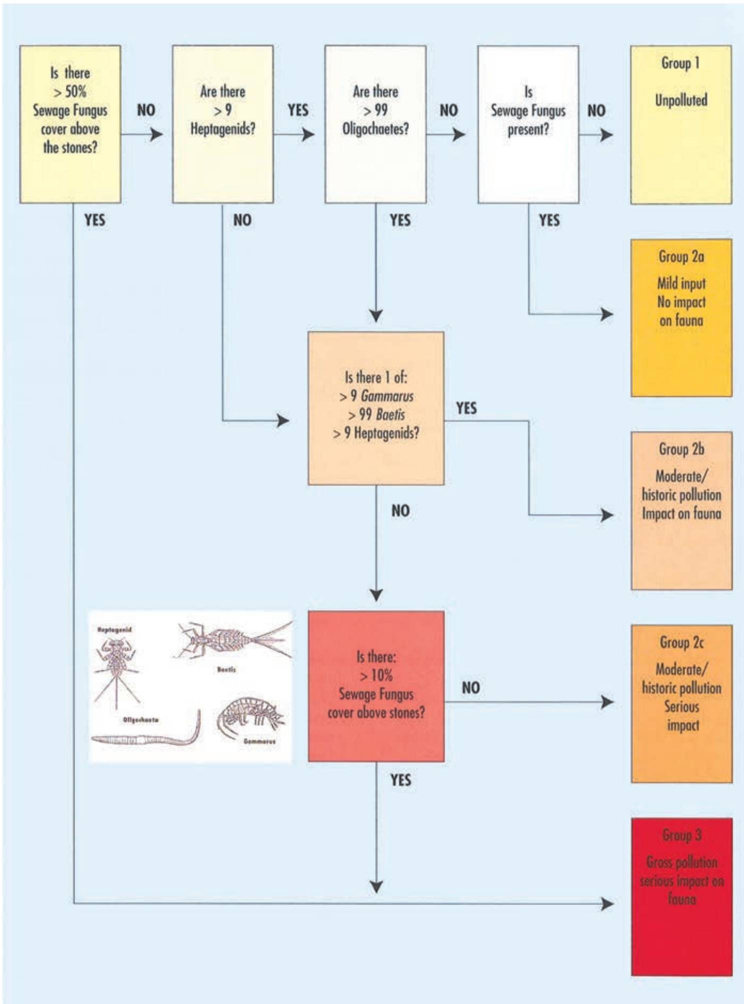


Figure 5.10

Cover of the report describing the Rapid Biological Appraisal Key for detecting farm pollution and a diagrammatic representation of the method

3.1.27 Simple observational methods for river wardens

Because the main river invertebrate indicators of pollution are so readily identified by non-specialists, it is easy for non-specialists to use them with sufficient accuracy to be useful for alerting that a water body may be polluted. Several simple tools have been devised to help train river wardens and environment officers who are not trained ecologists. Figure 5.11 is a recent example, devised by the Environment Agency to help field staff to recognise pollution.

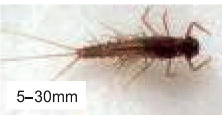
Similar schemes have also been developed for educational purposes.

Biological assessment following a pollution incident

Organism size
For most taxa there is a considerable range in length; however, individuals at the top end of the scales will be encountered rarely.

Pollution tolerance key
1 Very tolerant
2 Tolerant
3 Average sensitivity
4 Sensitive
5 Very sensitive

Insecta: Mayfly larva



- Most water bodies
- Actively swims/darting motion
- Three thread-like tails
- Obvious gills

4 Oil | 5 Organic | 5 Pesticide | 5 Ammonia

- Mostly in rivers with good water quality
- Very high value to anglers

Insecta: Stonefly larva

- Fast flow
- Crawls or darting motion
- Two thread-like tails

4 Oil | 5 Organic | 5 Pesticide | 5 Ammonia

- In rivers with very good water quality
- High value to anglers

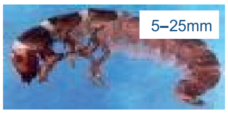
Insecta: Cased Caddis larva

- Most water bodies
- Crawls or swims, dragging case
- Sand/gravel cases often attached to surfaces

4 Oil | 5 Organic | 5 Pesticide | 4 Ammonia

- Generally in rivers with good water quality
- High value to anglers

Insecta: Caseless Caddis larva



- Most water bodies
- Crawls or wriggles
- May have tufts of gills

3 Oil | 4 Organic | 5 Pesticide | 4 Ammonia

- Generally in rivers with good water quality
- High value to anglers

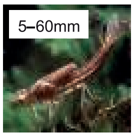
Crustacea: Freshwater shrimp

- Most water bodies
- Active swimmer
- Flattened from side to side
- Orange colour when moribund or dead

4 Oil | 3 Organic | 5 Pesticide | 4 Ammonia

- In waters of moderate to good quality
- High value to anglers

Insecta: Damselfly and dragonfly larva



- Slow flow
- Crawls or laboured swimming
- Large eyes – three spines or plates at tail

4 Oil | 4 Organic | 5 Pesticide | 5 Ammonia

- In waters of moderate to good quality
- Moderate value to anglers

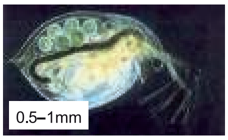
Crustacea: Water hog louse

- Most water bodies
- Crawls
- Looks like a wood louse

3 Oil | 2 Organic | 5 Pesticide | 4 Ammonia

- In waters of bad to good quality. Numbers increase with organic enrichment
- Low value to anglers

Crustacea: Water flea

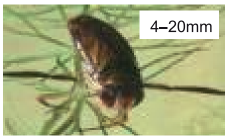


- Most water bodies
- Actively swims
- Transparent, many appendages

4 Oil | 3 Organic | 5 Pesticide | 3 Ammonia

- In waters with bad to good quality
- Moderate value to anglers

Insecta: Water boatmen

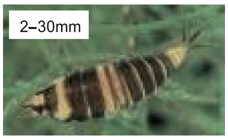


- Slow flow
- Use legs like oars; adults can fly
- One pair of long hairy legs

5 Oil | 3 Organic | 5 Pesticide | 4 Ammonia

- In waters of moderate to good quality
- Low value to anglers

Insecta: Beetle larva

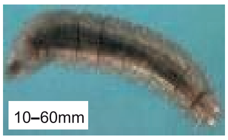


- Most water bodies
- Crawls or laboured swimming
- May have large jaws

4 Oil | 3 Organic | 5 Pesticide | 5 Ammonia

- In waters of moderate to good quality
- Low value to anglers

Insecta: Crane fly larva



- Most water bodies
- Crawls or wriggles
- Retractable head

3 Oil | 3 Organic | 5 Pesticide | 4 Ammonia

- In waters of moderate to good quality
- Moderate value to anglers

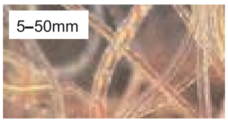
Mollusca: Pond snail

- Most water bodies
- Crawls using foot
- Pointed spiral shell

1 Oil | 2 Organic | 3 Pesticide | 4 Ammonia

- In waters of poor to good quality
- Low value to anglers

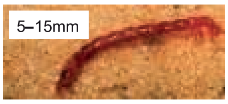
Annelida: True worms



2 Oil | 1 Organic | 3 Pesticide | 4 Ammonia

- All waterbodies, but especially abundant when organically polluted
- Wriggles and twists body
- No appendages or obvious head
- Major food source for course fish

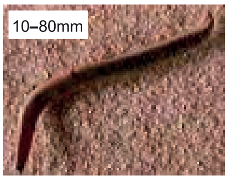
Insecta: Midge larva



1 Oil | 1 Organic | 4 Pesticide | 1 Ammonia

- All waterbodies, red types especially abundant when organically polluted
- Wriggling or crawling
- Head and legs not obvious
- Red, green or cream in colour
- Major food source for course fish

Annelida: Leech



3 Oil | 2 Organic | 3 Pesticide | 5 Ammonia

- Most water bodies from poor to good quality
- Swims or stuck on surfaces
- A sucker at each end

Figure 5.11
Simple guide to recognising pollution for non-specialist field staff, from the Environment Agency (2017) [118]

Baetis scambus nymph

3.2 Analytical systems

3.2.1 River Invertebrate Prediction and Classification Scheme (RIVPACS)

RIVPACS – River Invertebrate Prediction and Classification Scheme
and
RICT – River Invertebrate Classification Tool

RIVPACS, implemented in the River Invertebrate Classification Tool (RICT) software, does much more than determine river invertebrate status classification. It is also useful for a wide range of investigations.

In addition to status classification, it can:

- **predict the presence and abundance of species or families of invertebrates that we should find in any stream in the UK in its near natural state**
- **predict the value of a wide range of biotic indices based on family or species-level analysis that we should expect in any stream in the UK in its near natural state.**

An overview of RIVPACS is given in **Chapter 3 Section 3.1.2** and for an outline of how RIVPACS predicts a biotic index see **Chapter 3 Section 3.1.3.** RIVPACS must only be used for samples collected and analysed by the standard methods described in **Chapter 2**.

Taxonomic predictions

RIVPACS can provide taxonomic predictions at five taxonomic levels:

- TL1 = taxa recognised by BMWP indices
- TL2 = taxa recognised by WHPT indices
- TL3 = all families included in RIVPACS
- TL4 = all species included in RIVPACS species
- TL5 = mixed taxon level (a practical, mainly species level of analysis but with less easily recognised species identified to higher taxonomic levels)

And for each season:

- Spring = March, April and May
- Summer = June, July and August
- Autumn = September, October and November

For every taxon at every taxon level in each season, RIVPACS predicts the following:

- Numerical abundance
- Log_{10} abundance category: 1–9, 10–99, 100–999, 1000–9999, >10,000
- Probability of occurrence
- Probability of each log_{10} abundance category

Follow guidance in the RICT2 User Guide, which you can download from the User Guides page of the RICT/RIVPACS web page **https://www.fba.org.uk/rivpacs-and-rict/rict-rivpacs-user-guides**

Compare

RICT Compare calculates the significance and direction of differences between two classifications. This is useful for comparing whether the status of a site has changed between surveys or if there is a significant difference in class between two sites, such as control and impact sites in an investigation (eg upstream and downstream from suspected pollution). Compare provides information about the statistical significance of differences in Ecological Quality Ratios, taking into account sampling and analytical error. It also gives the probabilities of each combination of class for each pair of samples being compared.

All indices prediction

RIVPACS currently predicts a wide range of biotic indices, including most of the indices described in this handbook.

RICT software cannot currently adjust and convert RIVPACS predictions into predictions of WFD reference values (**Chapter 3 Section 3.3**) except for the forms of WHPT ASPT and WHPT Ntaxa that are used for classification.

Downloading and editing RICT2 programs

RICT2 programs and internal data files can be viewed, modified and downloaded from the RICT2 website. The programs have been written in R so that they can be edited by researchers.

Further advice is given in the RICT2 User Guide and in the Build Guidance which can be downloaded from the RICT/RIVPACS user guides web page **https://www.fba.org.uk/rivpacs-and-rict/rict-rivpacs-user-guides**. The RIVPACS reference database used to build RIVPACS has been uploaded to the Reports page of the RIVPACS/RICT website **https://www.fba.org.uk/rivpacs-and-rict/rivpacs-rict-resources**. Development of the current taxonomic prediction and prediction of other biotic indices are described in Davy-Bowker *et al.* (2010). [32]

3.2.2 RIVPACS Model 44

The width, depth and nature of the substrate measured at a site may not represent the natural state of the river because many have been re-sectioned, particularly over-deepened for land drainage, flood conveyance or navigation. The flow in many rivers may also be very different to the natural state because of abstraction or discharges. As a result, when measured values are used in RIVPACS, the fauna that it predicts may not be the natural fauna. To overcome this, a new predictive model, RIVPACS Model 44 (Clark & Davy-Bowker 2017, 2018) [119] [120] has been developed in which these environmental parameters have been replaced by parameters from Geographic Information Systems (GIS-systems) that are not affected by human interventions. Model 44 is potentially very useful for water resource assessment; but before it can be used operationally, it needs to be tested, because substrate, width and depth are strong predictors. The loss of predictive power caused by their omission could balance any improvements that Model 44 provides.

Currently, RIVPACS Model 44 exists only as an experimental system for Great Britain, so that its performance can be tested. GIS data has been compiled by CEH UK (Centre for Ecology and Hydrology) for all the environmental parameters used by Model 44 (except for alkalinity) but based around a proprietary GIS river network, the licencing requirements for which do not allow it to be released to the public domain other than one site at a time.

Until the new model has been tested and its performance is better that the original RIVPACS model, it will not be released for operational use. An advantage of obtaining environmental parameters from GIS is that users will not have to compile the environmental input data themselves, other than the Ordnance Survey grid reference and alkalinity. Obtaining the correct environmental input data relies on the grid reference being correct.

RIVPACS Model 44

An experimental, predictive model utilising GIS to substitute environmental information from pristine environments. This enables the natural state of the river to be simulated into the predictions.

Comparison of environmental predictors for RIVPACS **Model 1 and Model 44**

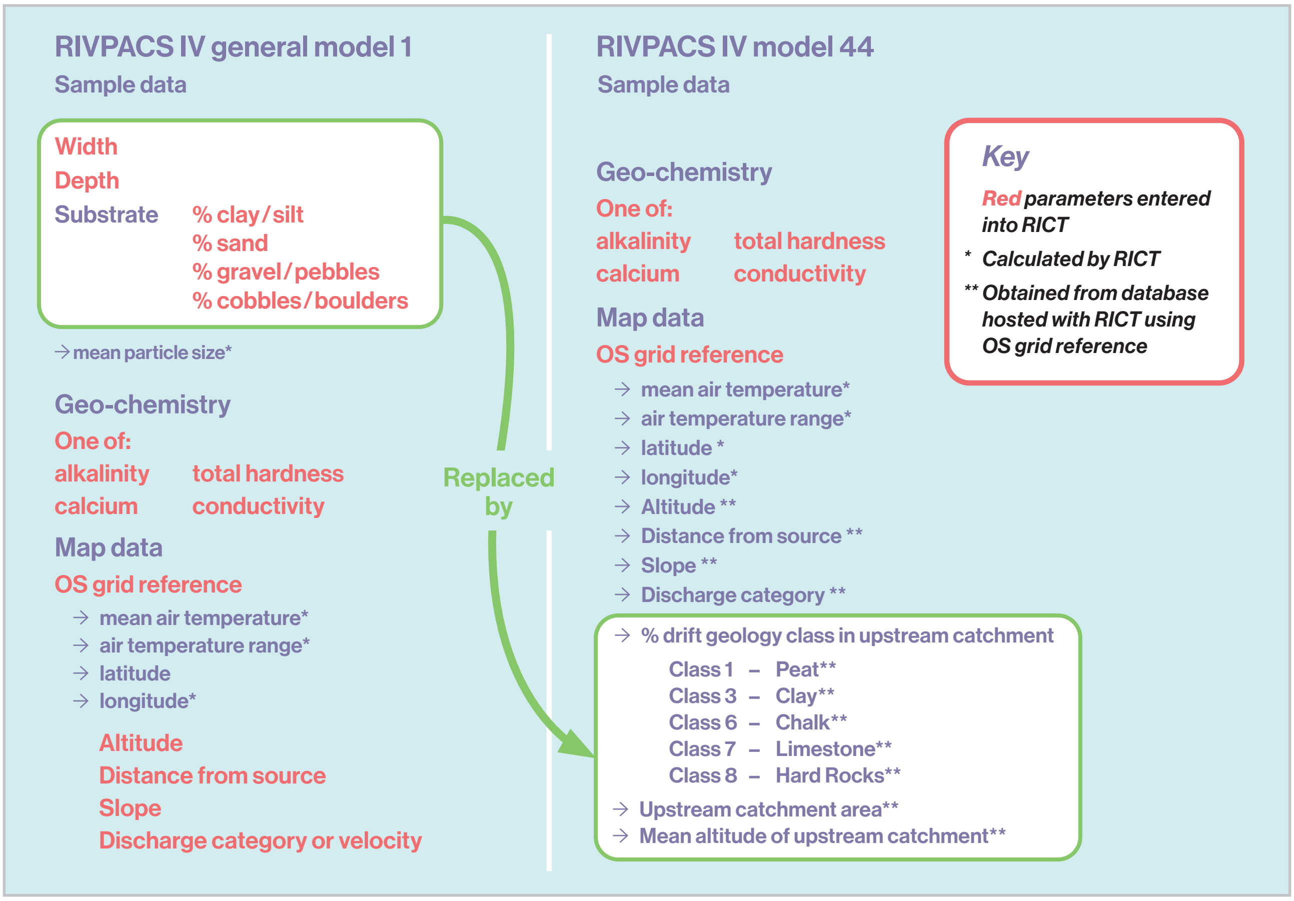


Figure 5.12

Anything in red font can be entered into RICT. You can either enter alkalinity, calcium concentration, total hardness or electrical conductivity. RIVPACS converts these to alkalinity which is the predictor variable. Likewise, you can either enter discharge category or flow velocity. If you enter the latter, RIVPACS will estimate discharge category, which it uses as the predictor variable. Model 44 is a flow and sediment independent model.

3.2.3 Invertebrate Impact Assessment Tool

This tool has been produced to help assess the impact of high frequency discharges from combined sewer overflows (CSOs) on receiving waters. It does this by assessing the degree of difference between abundance weighted WHPT NTAXA and WHPT ASPT from samples collected immediately upstream and downstream of an outfall. It uses estimates of replicate sampling uncertainty (the variation in observed values of WHPT NTAXA and WHT ASPT between replicate samples taken at the same site on the same day), in conjunction with a Monte Carlo approach to simulate a distribution of possible observed values for NTAXA and ASPT.

Invertebrate Impact Assessment Tool

This allows the impact of high frequency discharges from combined sewer overflows (CSOs) on receiving waters to be assessed.

Instructions, which include more details, can be downloaded from the user guide page of the RICT2 website **RICT2 SOAF CSO Impact Assessment Instructions**.

This tool is similar in aim to RIVPACS Compare, but instead of comparing EQRs and status class, it compares the value of biotic indices. This is described in a report to the Environment Agency by Clarke and Davy-Bowker (2017). [121]

3.2.4 Trait analysis

Ecological traits are the basis for several biotic indices, including PSI (Proportion of Sediment-sensitive Invertebrates) and SPEAR (Species at Risk). However, analysis of data in trait databases have so far failed to explain the behaviour of these indices, eg Wilkes *et al.* (2017).[122]

Trait analysis

Trait-based databases contain a wealth of data that can be useful for helping ecologists to understand the ecological requirements of freshwater biotas.

Trait-based databases, in particular the Freshwater Information Database (**www.freshwaterecology.info**, Schmidt-Kloiber and Hering (2015)[123] and Figure 5.13) contain a wealth of data that can be useful for helping ecologists to understand the ecological requirements of freshwater biotas and the natural biological variations that can interfere with environmental pressure responses.

From Version 7.0, October 2016, the Freshwater Information Database includes trait information from Tachet *et al.* (2010),[124] with brief explanations for each of the traits, resulting in Europe's most comprehensive trait collection for macroinvertebrates.

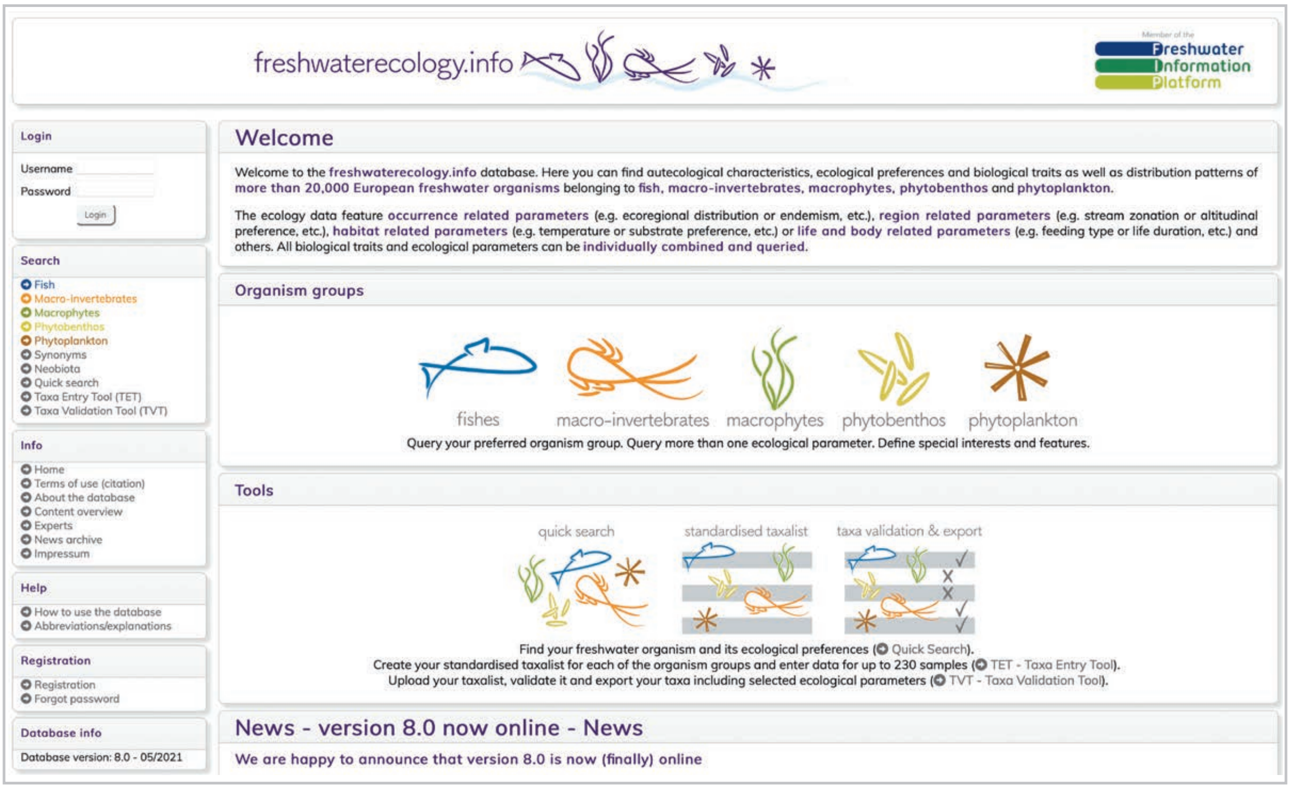


Figure 5.13
Home page of **www.freshwaterecology.info**

3.2.5 HEV (Hydroecological Validation)

Hydroecological validation (HEV) is based on a graphical comparison of ecological and hydrological data to help assess the ecological response at a site to river flow. A range of biotic indices expressed as Ecological Quality Indices are plotted with flow so that patterns in measured flow and the condition of the ecological community can be seen. (Note that EQIs are the observed value divided by the value predicted by RIVPACS for un-impacted conditions.) HEV is also used to infer the effect of other pressures, such as water quality and fine sediment.

HEV – Hydroecological Validation

Hydroecological Validation is based on a graphical comparison of ecological and hydrological data to help assess the ecological response at a site to river flow.

The Environment Agency uses HEV to provide evidence of where water resources activities, such as abstraction, might be contributing to an ecological problem. In turn, this helps indicate where further investigation and possible action is needed to protect river ecology to ensure that water bodies meet their environmental quality objectives.

Specifically, HEV is used for:

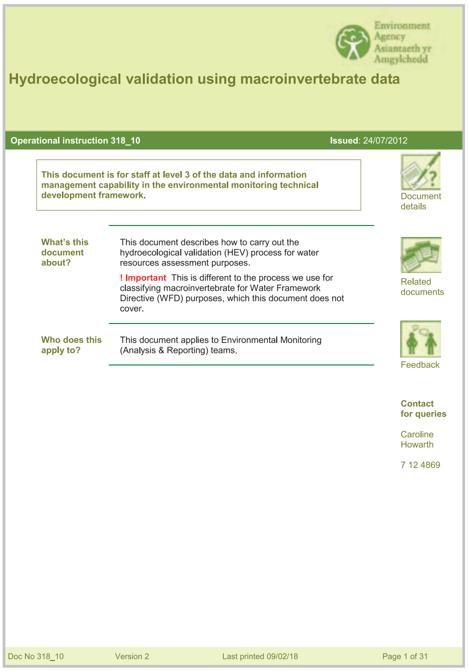
- reviewing and assessing EA Area drought plan monitoring data
- water resources investigations for river basin planning (normal and heavily modified water bodies) and the legacy Restoring Sustainable Abstraction (RSA) programme
- reviewing the Environmental Flow Indicator using the now defunct Catchment Abstraction Management Strategies (CAMS) invertebrate monitoring network
- national drought monitoring and reporting
- water resource licensing (where existing data are available).

HEV uses the following invertebrate biotic indices:

- Lotic Index for Flow Evaluation (LIFE)
- Proportion of Sediment-sensitive Invertebrates (PSI)
- BMWP average score per taxon (BMWP-ASPT)
- Number of BMWP-scoring taxa (BMWP-NTAXA)

An EQI (O:E ratio) LIFE 10-percentile of less than the guideline threshold of 0.94 (from a data set with at least 9 samples) is used as an indicator of possible flow stress. In chalk streams and naturally sandy rivers, a guideline threshold of 1.0 is commonly used because RIVPACS is known to under-predict the expected condition in these rivers. The Good/Moderate boundaries are used as guide values for BMWP indices. These thresholds are used only as a guide that must be supported by other evidence.

A refined version of HEV is in development.


Environment Agency
Asiantaeth yr Amgylchedd

Hydroecological validation using macroinvertebrate data

Operational instruction 318_10 Issued: 24/07/2012

This document is for staff at level 3 of the data and information management capability in the environmental monitoring technical development framework.

Document details

What's this document about? This document describes how to carry out the hydroecological validation (HEV) process for water resources assessment purposes.
! Important This is different to the process we use for classifying macroinvertebrate for Water Framework Directive (WFD) purposes, which this document does not cover.

Related documents

Who does this apply to? This document applies to Environmental Monitoring (Analysis & Reporting) teams.

Feedback

Contact for queries
Caroline Howarth
7 12 4869

Doc No 318_10 Version 2 Last printed 09/02/18 Page 1 of 31

Figure 5.14
Front cover of the Environment Agency's Operational Instruction for HEV

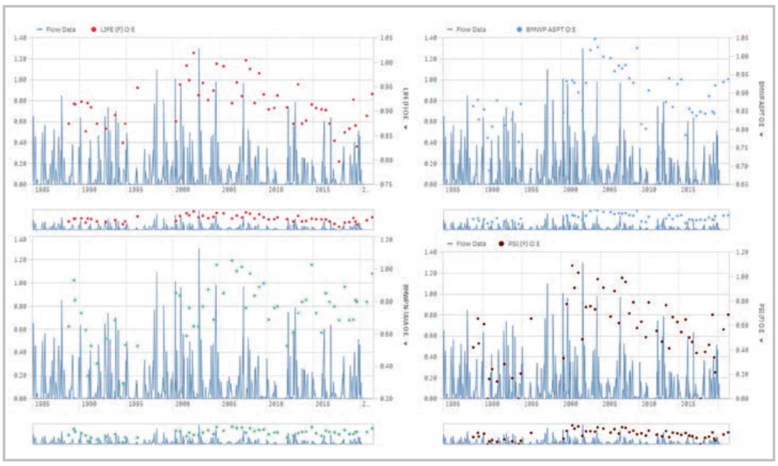


Figure 5.15
Example of an HEV plot; horizontal lines are guidance thresholds

3.2.6 Pantheon

Pantheon is an online tool to analyse invertebrate species samples. This analytical tool was developed by Natural England and the Centre for Ecology & Hydrology to assist invertebrate nature conservation in England – see Heaver *et al.* (2017). [125]

PANTHEON

Pantheon is an online tool to analyse invertebrate species samples.

The lists of invertebrates ('samples') are imported into Pantheon, which matches the species to the preferred name in the UK Species Inventory before analysing the sample, attaching to them associated habitats and resources, assemblage types – adapted from the Invertebrate Species-habitat Information System (ISIS; Webb and Lott 2006) [126] – together with habitat fidelity scores and other information. The analysis then displays much of this data as numerical scores. This information can be used to determine site quality by revealing whether the species list is indicative of good quality habitat, to inform on species ecology, and to assist in management decisions by revealing the key ecological resources.

Pantheon aids in establishing a shared terminology for describing invertebrate interest which will greatly augment invertebrate nature conservation.

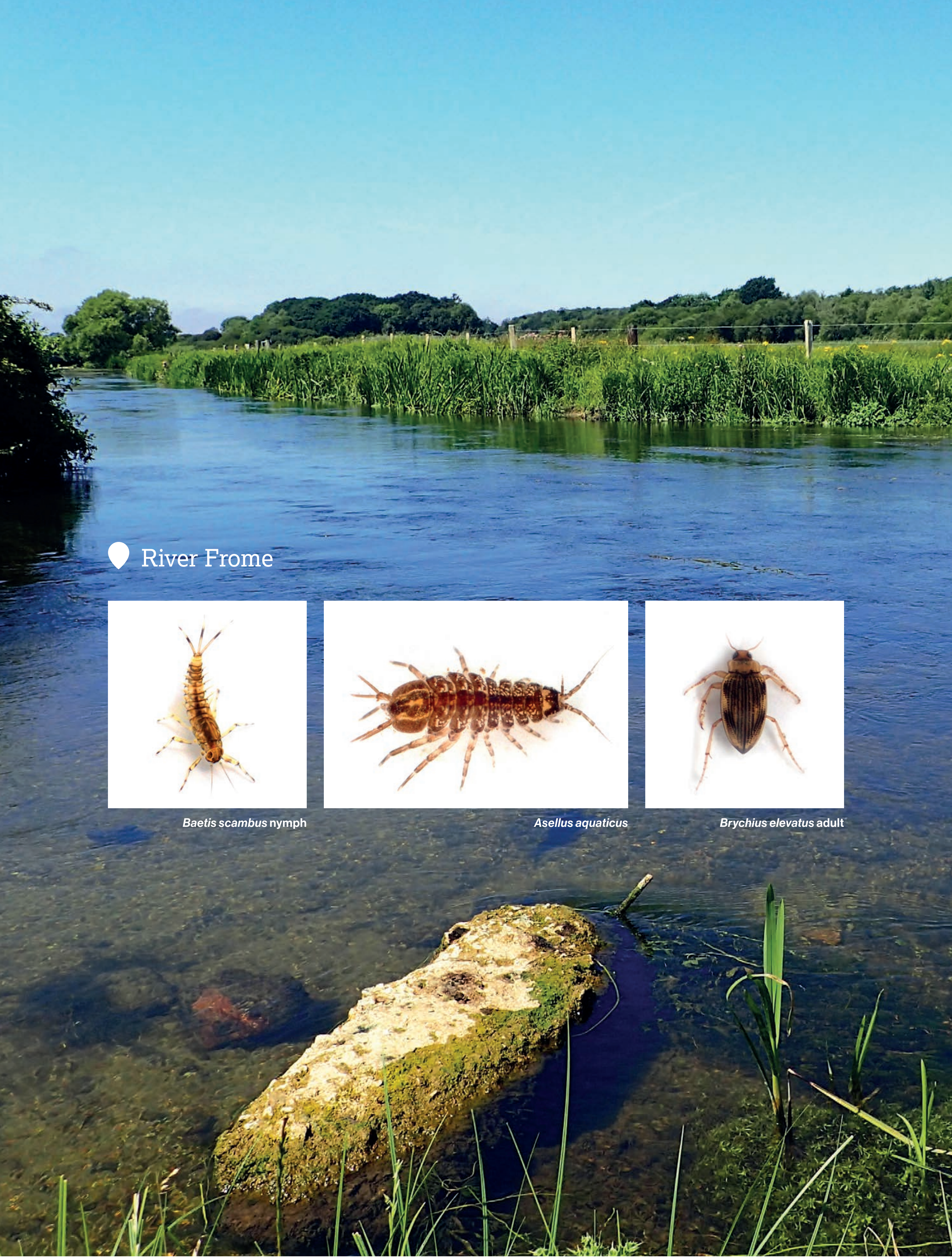
To date, more than 12,000 species have been typed and included in Pantheon, but this is about a quarter of the total macroinvertebrate fauna (estimated at 37,000). It remains limited to those taxa and families where there is enough ecological information to give a fair level of coding accuracy. These include species such as beetles, flies, true bugs, moths, bees and many more. Pantheon focuses on species primarily found in England.

http://www.brc.ac.uk/pantheon/ For more information contact **pantheon@ceh.ac.uk**

Planorbis albus

Gammarus pulex

River Frome



Baetis scambus nymph

Asellus aquaticus

Brychius elevatus adult

3.2.7 RPDS (River Pressure Diagnostic System)

This is one of the few systems or tools described in this handbook that is not currently available outside the regulatory agencies, although it is hoped that it can be reprogrammed into a web application to make it more widely available. It is one of the few tools that assumes that macroinvertebrate communities respond to combinations of environmental pressures, in contrast to most biotic indices which aim to identify the impacts of individual pressures.

RPDS – River Pressure Diagnostic System

A system for diagnosing environmental pressures that assumes that macroinvertebrate communities respond to combinations of pressures. This is in contrast to most biotic indices which aim to identify the impacts of individual pressures.

For River Basin Management under the Water Framework Directive, we must not only assess the status of ecological quality, for which we use the classification described in **Chapter 3**, but we must also discover the reasons for failure in order to identify an effective programme of measures to restore quality, which is the aim of investigations described in this chapter.

The River Pressure Diagnostic System (RPDS, previously referred to as River Pollution Diagnostic System) helps us diagnose which environmental pressures are influencing the biological quality. It is particularly useful when we don't know what may be causing poor quality, or to give objective evidence to support our own diagnosis – a second opinion.

RPDS mimics one of the main thought processes that ecologists use to interpret biological survey data: pattern recognition. RPDS recognises patterns of composition and abundance and associates them with the environmental conditions at other sites showing similar patterns. To use RPDS, you input results from a standard (RIVPACS) river invertebrate sample (a list of BMWP families and their abundances, together with RIVPACS environmental predictors). RPDS classifies your biological sample with the group of samples in its database that has the most similar composition. The average values of environmental parameters recorded at those sites provides a diagnosis of the environmental conditions that are affecting invertebrates at your site. Whereas RIVPACS GB recognises 64 natural types of invertebrate communities, RPDS recognises about 250, covering not only natural types but those associated with different combinations of environmental pressures.

The current version of RPDS (Version 3.5) holds river invertebrate data from more than 13,000 sites across England, Scotland and Wales, and data from 86,000 samples collected between 1995 and 2004, identified to BMWP-scoring families (RIVPACS Taxon Level 1).

These are matched to 13 environmental parameters including RIVPACS prediction variables, the concentrations of 42 chemicals from 5,600 chemical monitoring sites and expressed as 3-year percentiles (as used for chemical standards), flow data (whether the flows were higher or lower than average), information from local ecologists on the environmental stresses that they think or know affect the sites (perceived stresses), and river invertebrate status information. Recent data sets for RPDS include more biological and chemical data from 1995–2012, land cover, geology, and morphological information from River Habitat Surveys (RHS).

The RPDS display can also help users characterise the environmental conditions at their site by displaying the average values of any of the environmental parameters or abundance of any macroinvertebrate family as colours on the circles representing each community type (Figure 5.16). Diagnoses can be made for many sites together, for which RPDS produces results in a spreadsheet showing pressures in order of deviation from global average condition, to highlight environmental parameters with particularly high or low values. Using RPDS, it becomes clear that pressures do not occur randomly but in characteristic combinations and that different invertebrate communities are associated with these combinations of pressures. It prevents ecologists from falling into the trap of assuming that a particular pressure is confirmed as the cause of poor invertebrate quality when an index sensitive to that pressure shows a response.

RPDS includes several other tools, maps and reports to help users interpret their results. Comprehensive online help is available on the current system, as well as detailed descriptions of each of the taxonomic, ecological and environmental parameters included in the system.

The data sets compiled for RPDS with data from matched biological, chemical and RHS monitoring sites are useful in their own right – for example, they have been used to help set more realistic environmental standards for chemicals.

Two useful references are Paisley *et al.* (2011),[127] and Trigg (2020).[128]

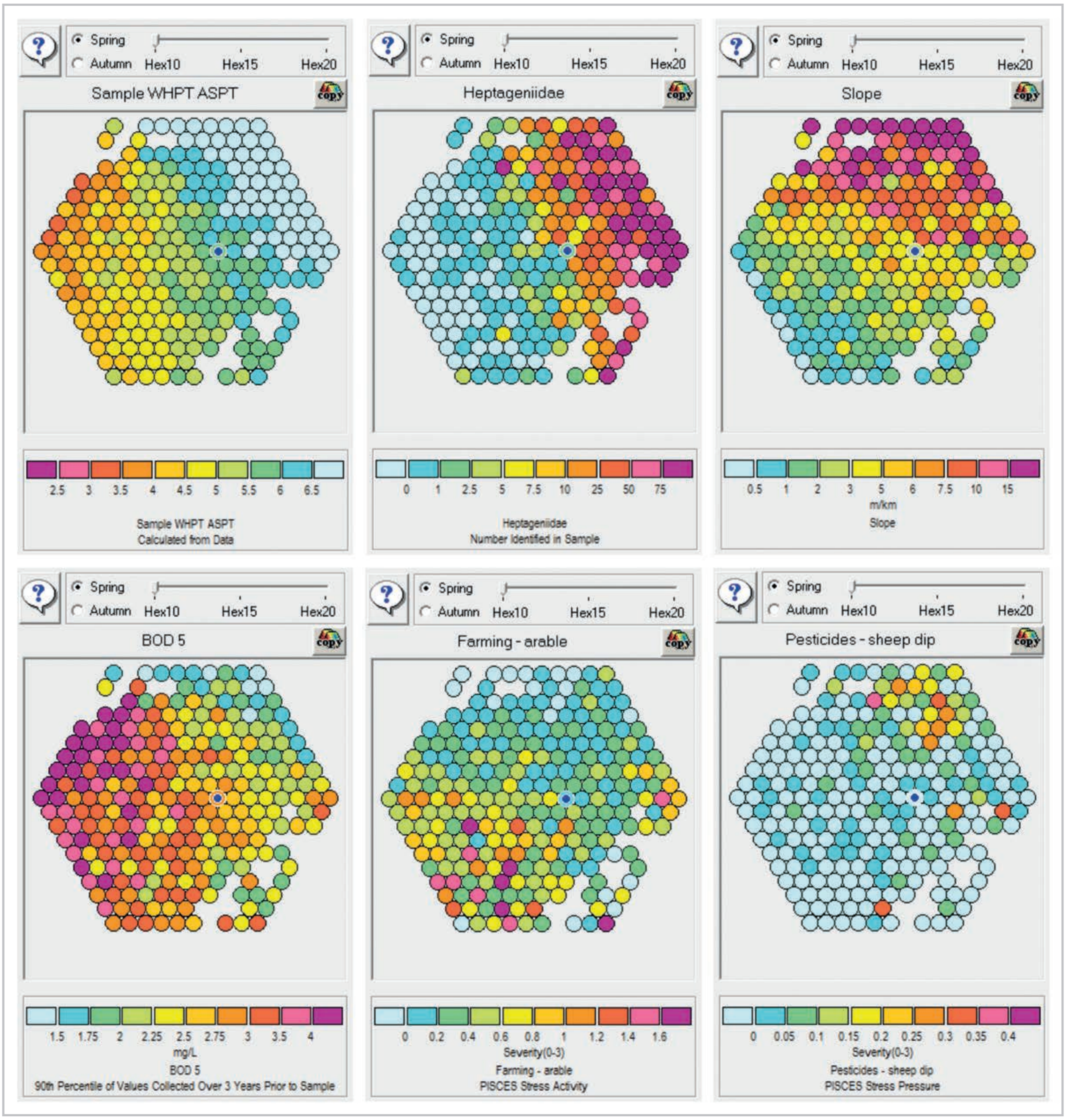


Figure 5.16

Example of the average values of various parameters mapped onto invertebrate community types, showing how the biota and many pressures are associated

Conclusion

This chapter reviews some of the key sample and data analysis methods used for investigations and to better understand the complex interactions between environmental factors and invertebrate communities. Some may be impacted by natural ecosystem change and others due to anthropogenic activity, including pollution, water abstraction, flood alleviation activity and climate change.

As a reminder, investigative monitoring is usefully defined as follows by the EU WFD:

To identify the causes of poor environmental quality (diagnosis) and their timing and source so that an appropriate programme of measures can be implemented to restore quality.

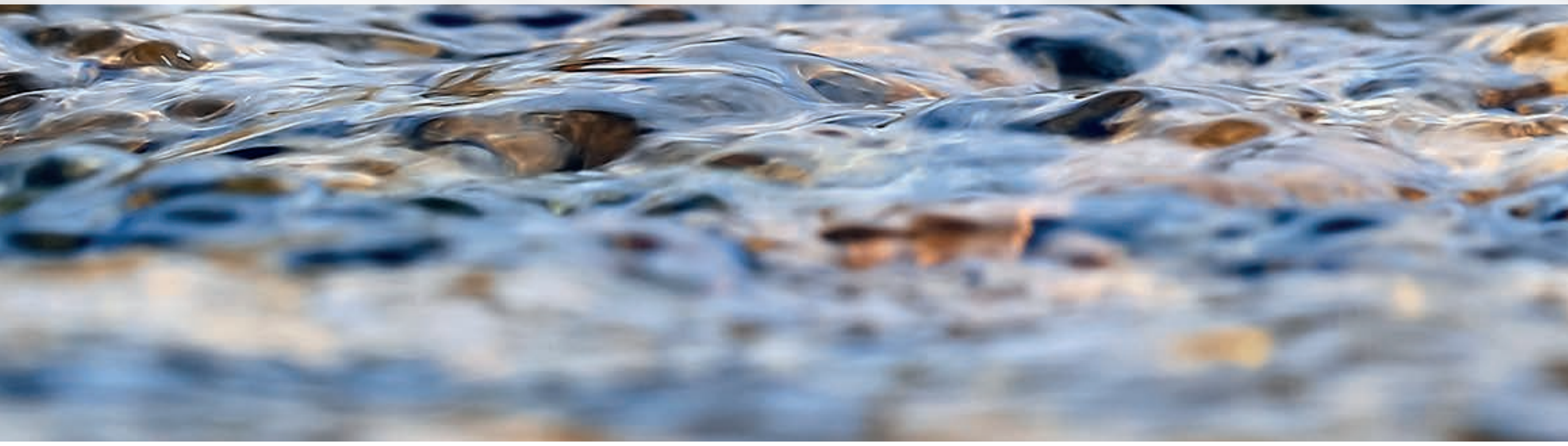
Investigative monitoring is undertaken in close association with **surveillance monitoring** which assesses long-term changes in the environment due to natural and widespread anthropogenic activity. It is the basis for formal classification and reporting of water quality and drives the infrastructure investment programmes. **Operational monitoring** is used to confirm the status of water bodies at risk from known pressures and to assess the efficacy of improvement programmes.

As previously stated, it is important to differentiate the data sets from these monitoring activities, as the data from investigative monitoring may bias classification toward a specific short-term event, such as a transitory pollution incident. Care must be taken in sample programme design to optimise these complementary activities and yet ensure high quality information for decision making and reporting.

Biotic indices are numerical values that relate the presence of taxa to environmental pressures. Their role is to simplify complex biological data so that ecologists can explain their results to environmental managers who may have little knowledge of ecology.

Although their format is intentionally very simple (usually a single number or letter), biotic indices are actually very complex, and most do not behave as parameters on a continuous scale of equal intervals. Biotic indices should not be used as a basis for statistical analysis.

Indexes are easily misused and can be misleading. Users should be wary of using biotic indices, particularly without understanding the extent of the data from which they were derived, or the specific uses for which the indexes are designed.



There are a considerable number of indices available, and this chapter provides an overview of the range and diversity of the issues targeted. Some are designed to solve a specific need and others have a broader application. Variations on these are being developed around the world and for specific river basins and applications. However, the core principles are common and access to this information may prevent duplication of effort and could speed up the adaptation of methods to suit new situations. **Be careful not to fall into the common traps and potential misuse of these useful tools.**

There are a number of powerful data analysis tools available, some very powerful with complex underlying statistics and mathematical analysis. Many are in the public domain and can be utilised by regulators, research groups, and individuals. However, they are only as good as the data input and must be used by informed and competent aquatic ecologists who understand the complexities of environmental interactions.

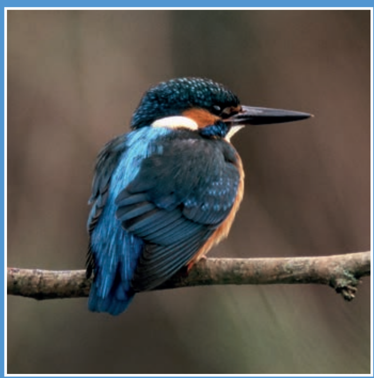
Finally, invertebrates respond to the integrated effect of all environmental pressures (both natural and anthropogenic) and it is impossible to apportion impact to individual pressures unless an environmental pressure is so severe that it is overwhelming. This will increasingly be the case as gross pressures are eliminated by environmental regulation. The corollary, that the impact of one environmental pressure can be mitigated by reducing other unrelated pressures, is not widely recognised.

Chapter 6 reviews the use of this information to inform water resources strategies, comply with national and international legal obligations, for statutory reporting, and to drive improvement programmes. Reporting and public access to high quality information is critical to effective water resource regulation, effective protection, and improvement programmes.

Interventions need to be evidence led and targeted to ensure maximum environmental outcomes, through the most effective improvement programmes. Monitoring programmes and data analysis, to provide high quality information, are a critical component of water resource protection and improvement.

Chapter 6

COMPLIANCE ASSESSMENT AND REPORTING




Including the post-Brexit UK
25-Year Environment Plan.

1
INTRODUCTION

Once the data analysis and classification has been undertaken it is important to complete the monitoring cycle and report the results in a clear and easily understandable format for the public, catchment partners, industry, regulators, and government. It must also inform decisions on water management and protection. In addition, it is used to review and improve monitoring in future programmes. We aim for constant improvement in monitoring and classification where possible.

The core elements addressed in this chapter are shown in Figure 6.1, delineated by the red circle.

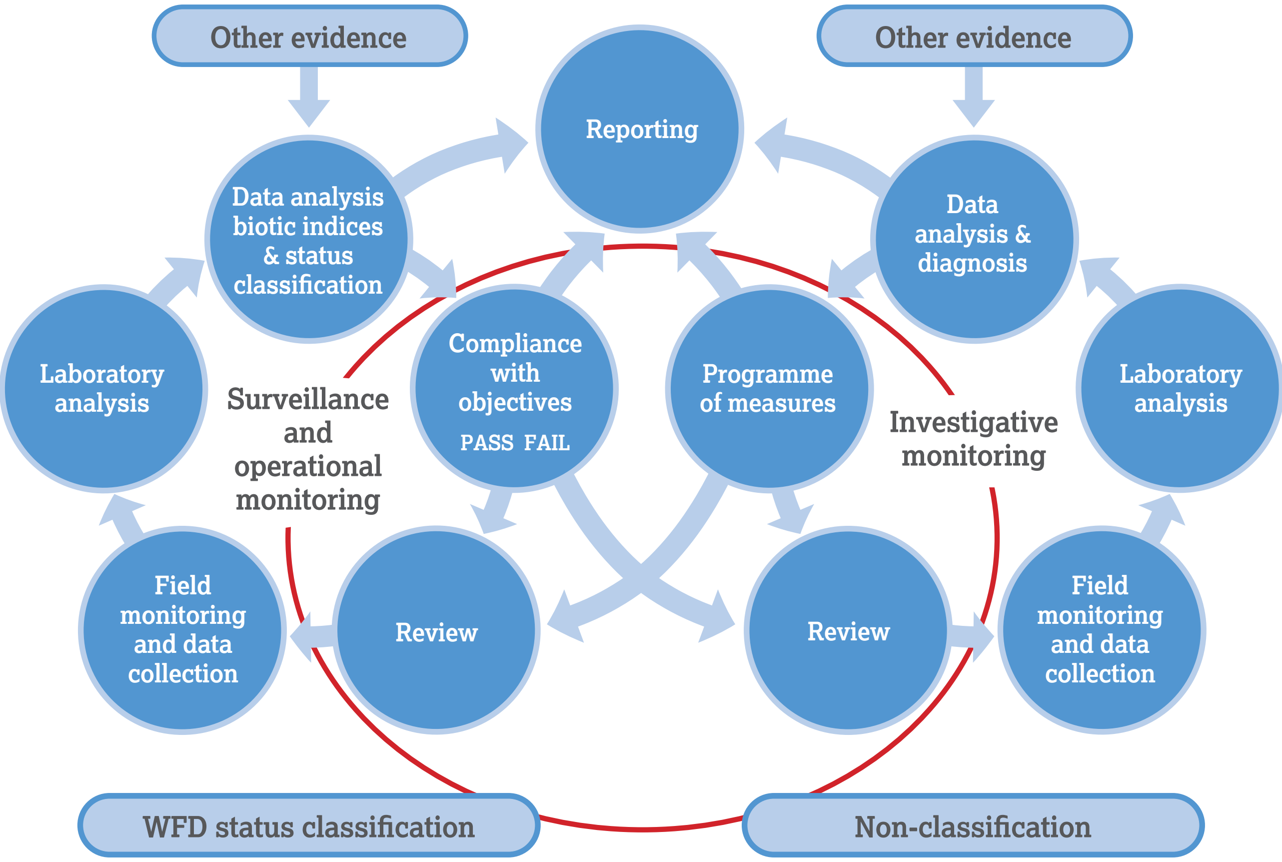


Figure 6.1
Monitoring and classification overview

These assessments inform the River Basin Management Plans and the development and implementation of the Programme of Measures.

At European level, the State of the Environment reports are compiled by the European Environment Agency (EEA) based in Copenhagen. These reports are totally dependent upon data and information being made available by EU Member States.

In the UK, the Environment Agency and the devolved agencies – Environment Agency Northern Ireland, Natural Resources Wales, and Scottish Environment Protection Agency – have statutory duties to produce a State of the Environment report. They also have a statutory duty to undertake environmental monitoring programmes and to advise ministers on environmental issues.

In addition, environmental data held by the environment agencies is available to the public and any other interested parties. This chapter gives examples of the data available.

Post-Brexit: for now the UK will continue with the Water Framework Directive (WFD) planning and implementation process and the WFD remains in place through the European Union (Withdrawal) Act 2018. [129]
https://www.legislation.gov.uk/ukpga/2018/16/contents/enacted

A UK 25-Year Environment Plan is now in place. [130]
https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/693158/25-year-environment-plan.pdf

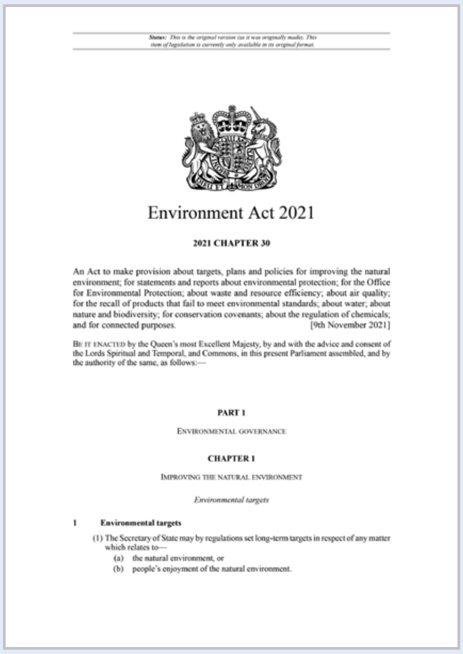
It will increasingly become the focus for environmental improvement policy, including the WFD principles. This will be consolidated and taken forward through the UK Environment Act 2021. [131]
https://www.legislation.gov.uk/ukpga/2021/30/contents/enacted

The UK Environment Act (2021) passed into UK law in November 2021. The Act comprises two thematic halves:

- The first provides a legal framework for (UK) environmental governance (post-Brexit).
- The second makes provision for specific improvement of the environment, including measures on waste and resource efficiency, air quality and environmental recall, **water, nature and biodiversity,** and conservation covenants.

It will enable the implementation of the 25-Year Environment Plan.

From: Environment Bill Explanatory Notes. [132]
https://bills.parliament.uk/publications/41685/documents/327



Status: This is the original version (as it was originally made). This item of legislation is currently only available in its original format.

Environment Act 2021

2021 CHAPTER 30

An Act to make provision about targets, plans and policies for improving the natural environment; for statements and reports about environmental protection; for the Office for Environmental Protection; about waste and resource efficiency; about air quality; for the recall of products that fail to meet environmental standards; about water; about nature and biodiversity; for conservation covenants; about the regulation of chemicals; and for connected purposes. [9th November 2021]

BE IT ENACTED by the Queen's most Excellent Majesty, by and with the advice and consent of the Lords Spiritual and Temporal, and Commons, in this present Parliament assembled, and by the authority of the same, as follows:—

PART 1

ENVIRONMENTAL GOVERNANCE

CHAPTER 1

IMPROVING THE NATURAL ENVIRONMENT

Environmental targets

1 **Environmental targets**

(1) The Secretary of State may by regulations set long-term targets in respect of any matter which relates to—
(a) the natural environment, or
(b) people's enjoyment of the natural environment.

Locally, Rivers Trusts and other Non-Governmental Organisation (NGO) projects are also engaged in collecting data and acting on the results. Additionally, the public, through citizen science projects, contribute to and utilise this information.

Information from State of the Environment reports informs future policy for the environment and industry. This information is a core element of the River Basin Planning initiatives, specifically the WFD.

2

UK ENVIRONMENT PLAN – POST-BREXIT ARRANGEMENTS – EXAMPLE

Post-Brexit, the Water Framework Directive remains in force via the European Union (Withdrawal) Act 2018. [129] **https://www.legislation.gov.uk/ukpga/2018/16/contents/enacted**

Until this is changed, the UK is committed to implementing the WFD. It will continue with the River Basin Management Plans, monitoring and reporting requirements, and the Catchment Based Approach (CaBA) to water environment management. A major difference is that, instead of reporting to the EU, it will report to the newly created, independent oversight body in the UK, the Office for Environmental Protection (OEP).

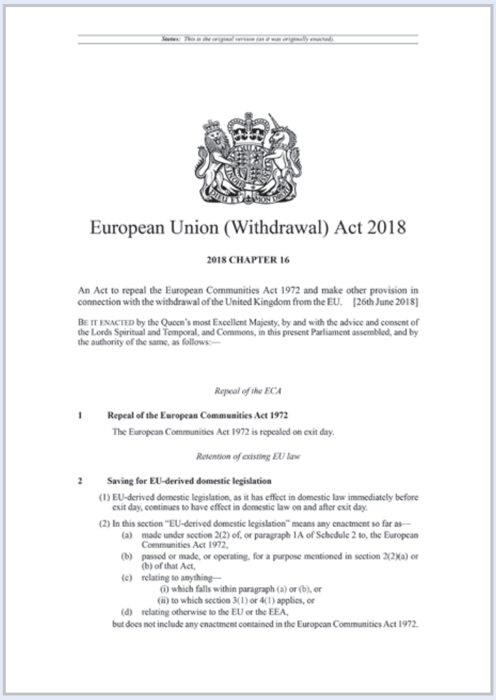
The OEP was formally established in UK law by the Environment Act in November 2021, following a short-lived interim OEP that was set up in July 2021. [133] **https://www.theoep.org.uk**

The OEP oversees the government's environmental activities, including progress with WFD. The Environment Agency in England and its devolved counterparts remain as competent authorities to implement the WFD. This includes monitoring and reporting to the OEP. In Scotland, many of the functions of the OEP are undertaken by Environmental Standards Scotland (ESS), which reports to the Scottish Parliament. **https://environmentalstandards.scot/**

In principle, it would be possible for the UK to make the monitoring and reporting information available to the EU. Comparative assessments of the EU and UK could be made on an informal basis, if agreed.

The best current indictor of the future of water management and the role of the WFD is set out in the **UK 25-Year Environment Plan**. Incidentally, the European (Withdrawal) Act also made the 25-Year Environment Plan statutory, so this has legal force.

European Union (Withdrawal) Act 2018



Status: This is the original version (as it was originally enacted).

European Union (Withdrawal) Act 2018

2018 CHAPTER 16

An Act to repeal the European Communities Act 1972 and make other provision in connection with the withdrawal of the United Kingdom from the EU. [26th June 2018]

BE IT ENACTED by the Queen's most Excellent Majesty, by and with the advice and consent of the Lords Spiritual and Temporal, and Commons, in this present Parliament assembled, and by the authority of the same, as follows:—

Repeal of the ECA

1 Repeal of the European Communities Act 1972

The European Communities Act 1972 is repealed on exit day.

Retention of existing EU law

2 Saving for EU-derived domestic legislation

(1) EU-derived domestic legislation, as it has effect in domestic law immediately before exit day, continues to have effect in domestic law on and after exit day.

(2) In this section "EU-derived domestic legislation" means any enactment so far as—
(a) made under section 2(2) of, or paragraph 1A of Schedule 2 to, the European Communities Act 1972,
(b) passed or made, or operating, for a purpose mentioned in section 2(2)(a) or (b) of that Act,
(c) relating to anything—
(i) which falls within paragraph (a) or (b), or
(ii) to which section 3(1) or 4(1) applies, or
(d) relating otherwise to the EU or the EEA,
but does not include any enactment contained in the European Communities Act 1972.

3

UK 25-YEAR ENVIRONMENT PLAN

The UK-25 Year Environment Plan *A Green Future: Our 25-Year Plan to Improve the Environment* (2018) is an important document regarding biological and environmental monitoring, covering aquatic and terrestrial environments.[130]

The environmental information collected will be used to inform improvement programmes and to report progress against the objectives determined by the plan. Indeed, the biological monitoring information would have been a key element of developing the 25-Year Plan and in setting its strategic goals and objectives.

https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/693158/25-year-environment-plan.pdf

This 25-Year Plan covers the UK, but it may be implemented differently by the devolved administrations. The government is committed to this and states 'we will continue to work with the devolved administrations on our shared goal of protecting our natural heritage'.

The most relevant high-level aim for the aquatic environment is provided by the Plan's goal – **Clean and Plentiful Water** (Box 6.1). Several other related policies in the plan also contribute to this aim, including *Using and managing land sustainably*, and *Recovering nature and enhancing the beauty of landscapes.* This also links to industrial policy, especially for the water sector and agriculture.

25-Year environment PLAN

Figure 6.2
The UK 25-Year Environment Plan [130]

Box 6.1

UK 25-Year Plan: Goal – Clean and Plentiful Water

We will achieve clean and plentiful water by:

Improving at least three-quarters of our waters to be close to their natural state as soon as is practicable by:

- reducing the damaging abstraction of water from rivers and groundwater, ensuring that by 2021 the proportion of water bodies with enough water to support environmental standards increases from 82% to 90% for surface water bodies and from 72% to 77% for groundwater bodies
- **reaching or exceeding objectives for rivers, lakes, coastal and groundwaters that are specially protected, whether for biodiversity or drinking water as per our River Basin Management Plans**
- supporting Ofwat's ambitions on leakage, minimizing the amount of water lost through leakage year-on-year, with water companies expected to reduce leakage by at least an average of 15% by 2025
- minimizing by 2030 the harmful bacteria in our designated bathing waters and continuing to improve the cleanliness of our waters. We will make sure that potential bathers are warned of any short-term pollution risks.

In the context of this handbook, note the second bullet point above.

'Monitoring and Metrics' from the 25-Year Plan is particularly relevant to this book and the monitoring and assessment methods described herein. We would expect these methods to evolve and assist in meeting and evaluating progress with the 25-Year Plan and the WFD (see Box 6.2).

Box 6.2

Extract from 25-Year Plan – Monitoring and Metrics

Measuring the impact of the 25-Year Environment Plan

Metrics are a critical part of the 25-Year Environment Plan. They enable us to comprehend the complexity of the environment and allow us to:

- understand how the environment as a whole is changing – the pressures, the state of assets, and the flow of benefits
- assess the effectiveness of our policies and show how we are delivering our domestic and international commitments
- inform decisions and promote action within and outside government, locally and nationally.

We have a large number of existing indicators and associated statistics, data and monitoring systems. A Natural Capital approach will require careful selection of these and development of further indicators.

> **The Plan makes clear that good evidence is the cornerstone of effective policy making, which is one of the core objectives of this book.**

To promote the use of evidence-based policy, the government has published a supporting evidence pack for water: *State of the Water Environment Indicator B3: Supporting evidence* (May 2021). [134]

https://www.gov.uk/government/publications/state-of-the-water-environment-indicator-b3-supporting-evidence/state-of-the-water-environment-indicator-b3-supporting-evidence

This utilises the biological and ecological monitoring evidence from England and combines it into a wider dashboard of information relating to the state of the water environment (Figure 6.3).

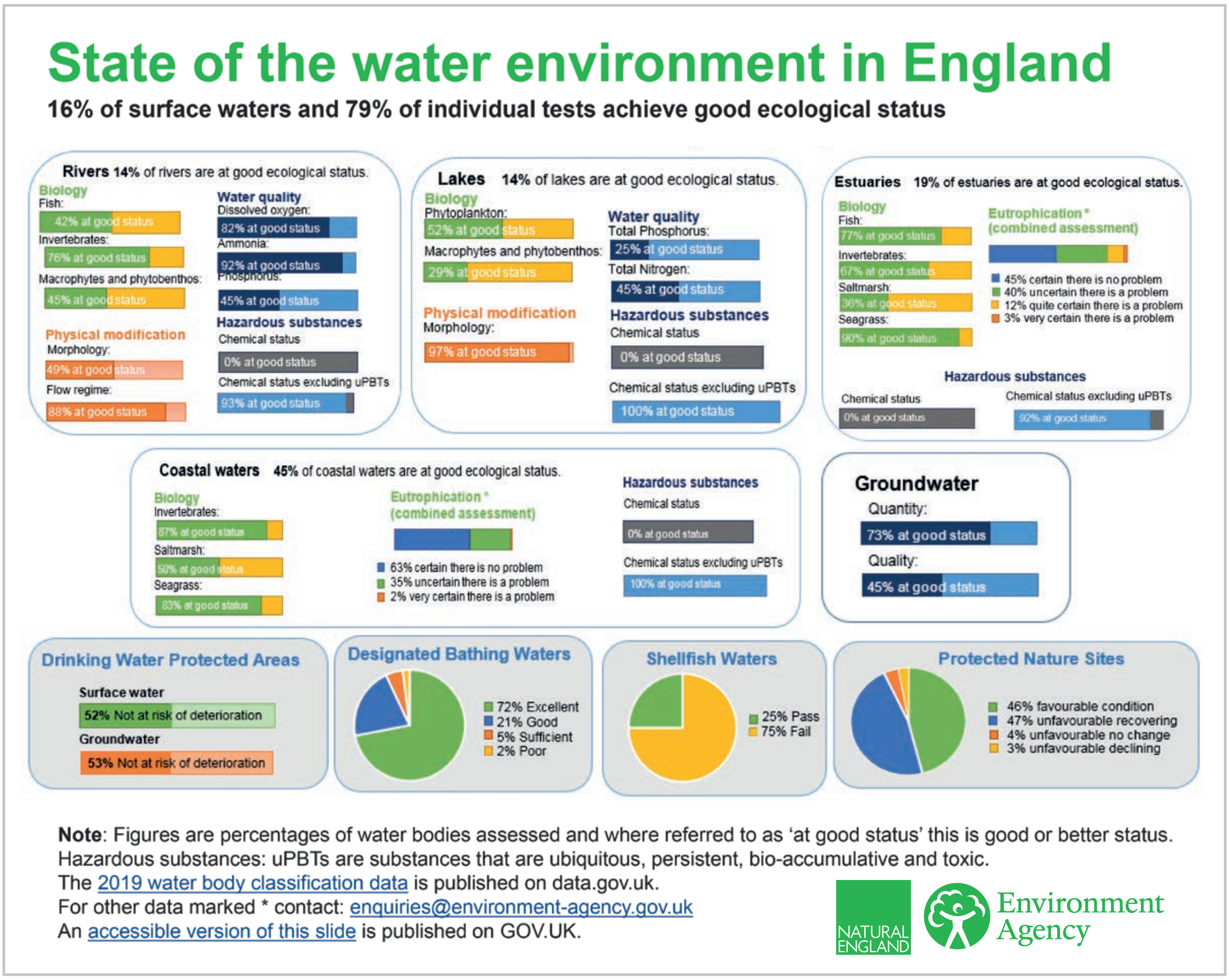


Figure 6.3

State of the water environment dashboard for England, 2021 (from *State of the Water Environment Indicator B3: Supporting evidence*).[134] **https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/988105/B3-indicator-supporting-evidence-May-2021.pdf**

4

ENVIRONMENTAL PRESSURES AND IMPACTS

A good understanding of the pressures and impacts that cause water bodies to fail their environmental objectives is important in targeting improvement programmes. The *State of the water environment indicator B3: supporting evidence* report summarises the key reasons for failure, in terms of issues such as physical modifications, water pollution, and changes to flow. It also gives an overview of the key industrial sectors impacting on the water environment.

From Figure 6.4 we can see that the two key sectors that cause failures in the UK are:

agriculture (45%)

water sector (44%)

Similar pressures and sectors are identified across Europe.

Key issues and sectors affecting water bodies in England

Issue	Agriculture and rural land management	Industry	Mining and quarrying	Navigation	Urban and transport	Water Industry	Local and Central Government	Domestic General Public	Recreation	Waste treatment and disposal	No sector responsible	% of water bodies impacted by each issue
Physical modifications	12.9%	1.9%	0.1%	1.9%	10.9%	7.9%	14.3%	0.3%	2.9%		0.1%	41%
Pollution from waste water	0.1%	0.5%			0.6%	35%	0.2%	1.1%		0.1%		36%
Pollution from towns, cities and transport	0.1%	3.4%	0.1%		10.1%	0.8%		6.4%	0.2%	0.3%	0.1%	18%
Changes to natural flow and levels of water	1.3%	0.4%		0.1%		9.8%	0.2%		0.1%			15%
Non-native invasive species											23%	23%
Pollution from rural areas	40.0%											40%
Pollution from abandoned mines			3.2%									3%
% of water bodies impacted by each sector	45%	6%	3%	2%	18%	44%	14%	8%	3%	0.3%	23%	

Note: the bottom row and right hand column are not summations of the rows and columns in the main table – see Introduction

Where:

	High (>30%)
	Medium (<30% and >10%)
	Low (<10% and >1%)
	Very Low (<1% and >0.1%)
	Insignificant (<0.1%)

The figures in the separate row at the bottom of the table '% of water bodies impacted by the activity of each sector', and those in the separate column on the right of the table '% of water bodies impacted by each issue' are not summations of the figures displayed in the main table. These percentages have been calculated by only counting any particular water body once per sector or per issue and so avoid including multiple entries as outlined above.

Reference: Environment Agency Challenges and Choices 2019.

An accessible version of this table is published on GOV.UK.

Environment Agency

Figure 6.4

Key issues and sectors affecting water bodies in England.[135] **https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/988105/B3-indicator-supporting-evidence-May-2021.pdf**

4.1 Agriculture

Changes to agricultural policy, and retargeting of subsidies can be implemented to optimise improvement to the water environment. Elements of the EU Common Agricultural Policy, specifically Pillar 2, are designed to support rural areas of the EU and to meet the wide range of economic, environmental and societal challenges of the 21st century. (136)
https://www.europarl.europa.eu/factsheets/en/sheet/110/second-pillar-of-the-cap-rural-development-policy

The UK is in the process of changing these agricultural environment arrangements, post-Brexit. It aims to align agricultural land management schemes, to reward environmental benefits, and to optimise progress with the 25-Year Environment Plan.

Current thinking is outlined in *The Path to Sustainable Farming: An Agricultural Transition Plan 2021 to 2024.* (137)
https://www.gov.uk/government/publications/agricultural-transition-plan-2021-to-2024

4.2 Water Industry

In England and Wales the privatised water industry has a clear five-year investment programme, funded by water charge payers.

In 2019, the regulator, the Water Services Regulation Authority (Ofwat), revealed a spending package of £51 billion for the five years, 2020 to 2025. (138) A quarter of this, around £13 billion, will be investment dedicated to providing resilient services and a better environment in the face of a growing population and climate change. **https://www.ofwat.gov.uk/pn-23-19-ofwat-gives-green-light-to-massive-investment-programme-to-transform-water-sector/**

Of the £13 billion, the Water Industry National Environment Programme (WINEP) is the programme of work water companies in England are required to do to meet their obligations to environmental legislation and UK government policy.(139) WINEP is the most important and substantial programme of environmental investment in England. From 2020 to 2025 it consists of £5.2 billion of asset improvements, investigations, monitoring, and catchment interventions. It sets out how the water industry will contribute to improving the natural environment. **https://www.gov.uk/government/consultations/review-of-the-water-industry-national-environment-programme-winep**
A consultation exercise will refine the options to meet the environmental outcomes required.

Detailed information, including the environmental drivers for investment, can be found in the Water Industry National Environment Programme publication. (140) **https://data.gov.uk/dataset/a1b25bcb-9d42-4227-9b3a-34782763f0c0/water-industry-national-environment-programme**

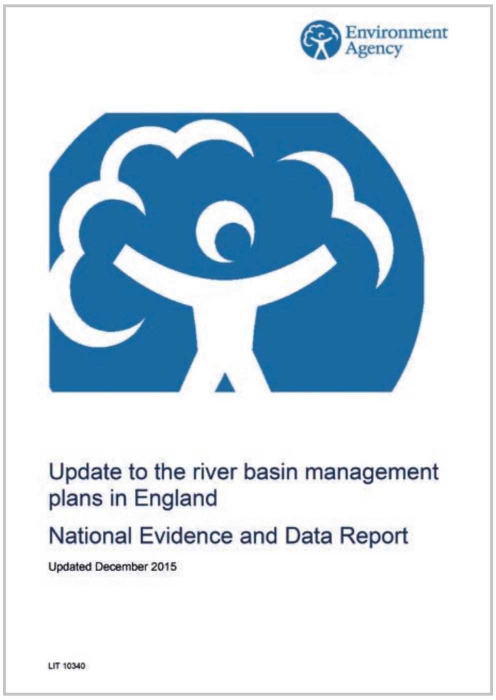
5

WATER FRAMEWORK DIRECTIVE – RIVER BASIN MANAGEMENT PLANS – ENGLAND NATIONAL LEVEL OVERVIEW – EXAMPLE

The WFD River Basin Management Plans remain as the core tool used to protect and improve the UK's water environment. This has been absorbed into the UK 25-Year Environment Plan, but the planning and implementation according to WFD principles remains in place.

River Basin Management Plans are updated every 6 years. [141] The Plans for England are available at **https://www.gov.uk/government/collections/river-basin-management-plans-2015#history**

The current Plans are those produced in 2015; their review and revision is underway, and draft Plans were published for consultation in October 2021. At a national level, an overview is provided in the National Evidence and Data Report 2015. [142] This can be found at **https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/514944/National_evidence_and_data_report.pdf**


Environment Agency

Update to the river basin management plans in England

National Evidence and Data Report

Updated December 2015

LIT 10340

Figure 6.5

Cover of *Update to the river basin management plans in England* (2015) [142]

Rhyacophila dorsalis larva.

Table 6.1 indicates the current ecological status of English waters in 2015. Comparable information is also available for the devolved administrations.

Table 6. 1

Overview of the ecological and chemical classification for surface waters in England for 2015

	Ecological status or potential					Chemical status	
No. of water bodies	Bad	Poor	Mod	Good	High	Fail	Good
4,679	136	765	2,966	805	7	137	4,542

A summary of objectives for ecological status or potential and chemical status by 2015 and beyond is shown in the next table (6.2).

Table 6. 2

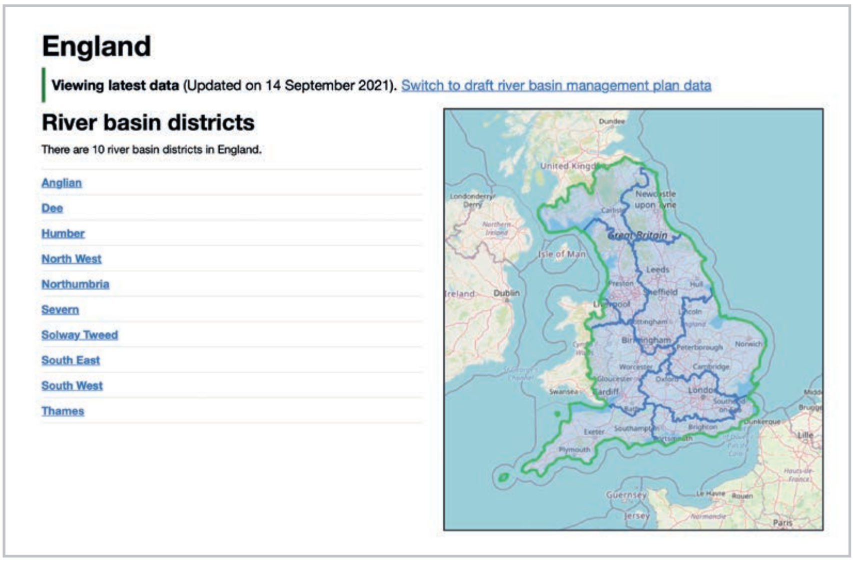
Summary of ecological status or potential and chemical status objectives for surface water bodies (by number of water bodies) including those with less stringent objectives and extended deadlines (blue-shaded cells) – England overview.

	Ecological status or potential						Chemical status			
	Bad	Poor	Mod	Good	High	Total	Fail	Good	Total	
By 2015	19	107	841	806	7	1,780	14	4,542	4,556	
By 2021	0	2	30	166	0	198	0	1	1	
By 2027	0	12	162	2,525	0	2,699	0	122	122	Extended deadline
Beyond 2027	0	0	0	2	0	2	0	0	0	
Total	19	121	1,033	3,499	7	4,679	14	4,665	4,679	
	Less Stringent						Less Stringent			

6
RIVER INVERTEBRATE CLASSIFICATION REPORTING UK

Results of the most recent river invertebrate classifications in England are available from WFD Cycle 2 River Invertebrate Classification. (143) They can be downloaded from **https://data.gov.uk/dataset/77368497-f25b-4fdc-a3d8-0468724ff752/wfd-cycle-2-river-invertebrate-classification**. These are used in combination with the results of classifications based on other quality elements to determine the ecological status and the overall status. Classification results for all elements for all water bodies are available from the Catchment Data Explorer. (144) **https://environment.data.gov.uk/catchment-planning/**

Data from the Environment Agency's river invertebrate monitoring in England are available from Ecology and Fish Data Explorer. (145) They can be downloaded from **https://environment.data.gov.uk/ecology/explorer/**



Nepa cinerea nymph

Crangonyx pseudogracilis

Helophorus grandis

7

CATCHMENT DATA EXPLORER – ENGLAND

The 2019 classification results for England were published in 2020, but no specific national report was produced to summarize or interpret them. **https://deframedia.blog.gov.uk/2020/09/18/latest-water-classifications-results-published/**

Instead, the results are mapped on the Catchment Data Explorer tool.[144] **https://environment.data.gov.uk/catchment-planning/** – from here you can download the results by River Basin District as a CSV file that can be opened in Excel, or you can drill down to individual water bodies. Results can also be obtained from the Catchment Based Approach website **https://data.catchmentbasedapproach.org/** [146]

An example output map is shown in Figure 6.6. This is for the River Kennet in southern England, a tributary of the River Thames.

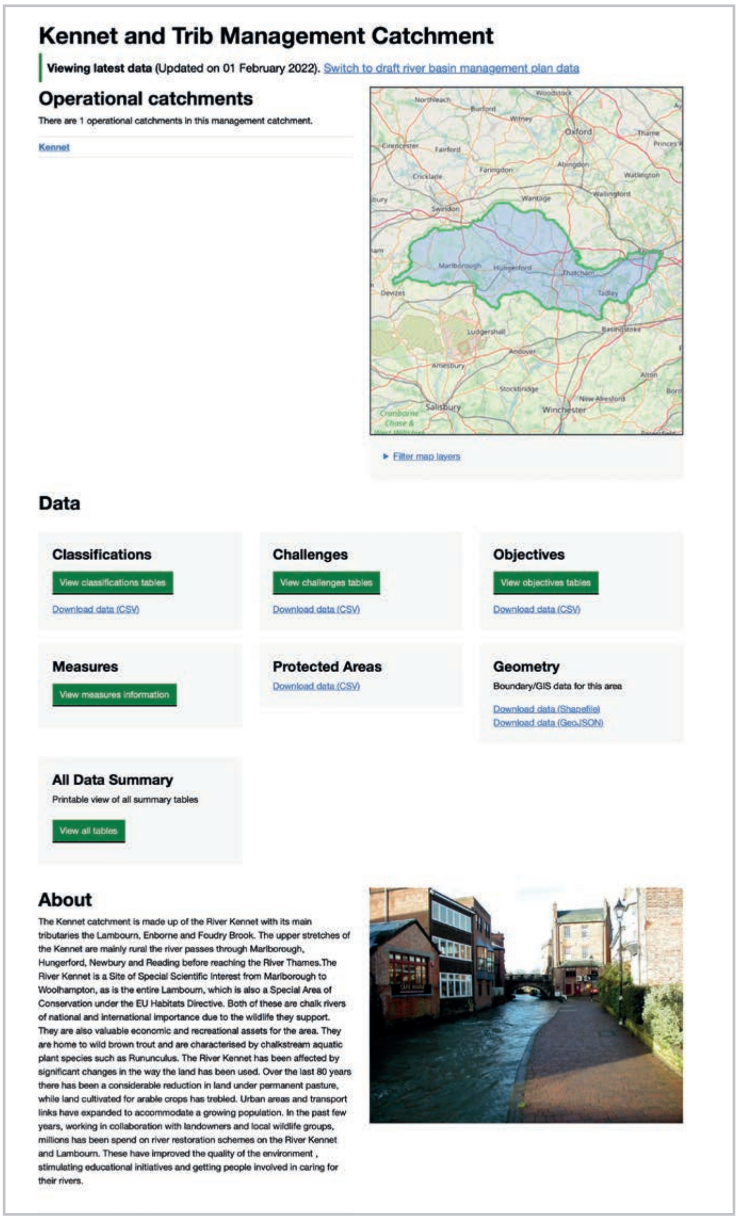


Figure 6.6

Catchment Data Explorer – top-level page for a river management catchment, the Kennet in the Thames River Basin District

The summary page (Figure 6.6) provides overall summaries including tables for the status classification (Figure 6.7), the reasons for not achieving Good status, ie the reasons for failure to achieve environmental objectives (Figure 6.8), the environmental objectives (Figure 6.9), summary statistics (Figure 6.10) and information about the programme of measures. The information is updated regularly and can be switched from the latest data to the data in the River Basin Management Plan.

Far more detail is provided in the data sets that can be downloaded from links on the river management catchment summary page, including details for the classification of all elements in all monitored water bodies, reasons for not achieving Good status or deterioration for every failing element in every water body, measures to restore good quality, environmental quality objectives for every water body, and the protected areas in each water body.

Ecological status for surface water bodies

Ecological status or potential	Bad	Poor	Moderate	Good	High	Total
Number of water bodies	0	3	23	7	0	33
Number of water body elements	0	9	40	42	206	297

Figure 6.7

Catchment Data Explorer – summary of classification data (Kennet)

Reasons for not achieving Good status and reasons for deterioration in this management catchment

The table below shows the number of reasons for not achieving Good status (RNAGS) and reasons for deterioration (RFD), split by sector.

Significant Water Management Issue	Changes to the natural flow and level of water	Invasive non-native species	Physical modifications	Pollution from abandoned mines	Pollution from rural areas	Pollution from towns, cities and transport	Pollution from waste water
Agriculture and rural land management	0	0	3	0	2	0	0
Domestic general public	0	0	0	0	0	0	0
Industry	0	0	0	0	0	0	0
Local & central government	0	0	0	0	0	0	0
Mining and quarrying	0	0	0	0	0	0	0
Navigation	0	0	0	0	0	0	0
No sector responsible	0	0	1	0	0	0	0
Other	0	0	6	0	0	0	0
Recreation	0	0	3	0	0	0	0
Sector under investigation	0	0	0	0	0	0	0
Urban and transport	0	0	0	0	0	0	0
Waste treatment and disposal	0	0	0	0	0	0	0
Water Industry	2	0	1	0	0	0	10
Total	2	0	14	0	2	0	10

Figure 6.8

Catchment Data Explorer – summary of reasons for not achieving Good status (Kennet)

Ecological status or potential objectives for surface water bodies

Including those with less stringent objectives and extended deadlines.

Status	Bad	Poor	Moderate	Good	High	Total
By 2015	0	0	5	5	0	10
By 2021	0	0	0	3	0	3
By 2027	0	0	0	12	0	12
By 2033	0	0	0	1	0	1
By 2039	0	0	0	7	0	7
Total	0	0	5	28	0	33

Figure 6.9

Ecological status or potential objectives

Summary statistics data for Kennet and Trib Management Catchment

Viewing draft River Basin Management Plan data.

Ecological status and potential

Summary statistic	Rivers, Canals and SWTs	Lakes	Estuaries	Coastal	Surface Waters Combined
% of water bodies at good or better ecological status/ potential	21%	25%			21%
% of biological elements, phys-chem elements and specific pollutants at good or better status	83%	57%			82%
% of water bodies with an objective of good ecological status/potential or better	83%	100%			85%
% of biological elements, phys-chem elements and specific pollutants with an objective of good status or better	97%	100%			97%

Figure 6.10

Summary statistics for ecological status and potential, this page includes similar information for chemical status

8
ENGLAND – BIOLOGICAL QUALITY MAPPING

For many years, water quality information, in terms of both biological and chemical classification, was published in the form of an overview map, with colours on rivers indicating status – Figure 6.11. This was usually published as part of 5-yearly State of the Environment reports, issued as hard copy and PDF. However, this has been replaced by the interactive catchment maps and electronic reports described earlier.

Accessing national-scale information in graphical format is now difficult in the UK. Figure 6.11 is the most recent biological quality information that we could find in this format. This map is based on pre-WFD General Quality Assessment Classification and will not be directly comparable.

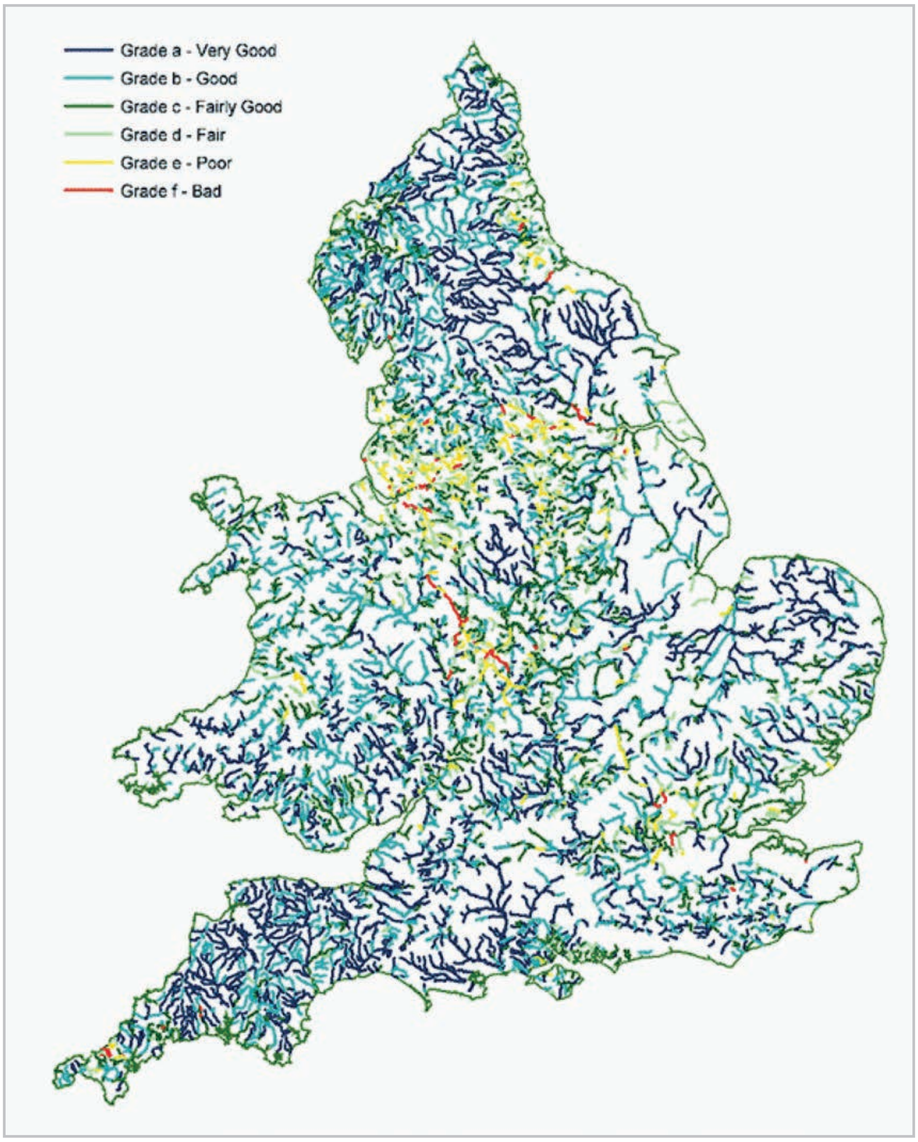


Figure 6.11
Status of rivers (England and Wales) – biological quality assessed according to the GQA classification

9

INVESTIGATIVE MONITORING – UK EXAMPLE

The impacts of pollution or other incidents may need to be investigated by investigative monitoring. Such information should not be used for classification because the intensity of sampling may distort the long-term picture. However, the surveillance monitoring data provides the baseline information, so changes from this may indicate that a pollution event has damaged the biota.

An example of the impact of a series of organic pollution events, together with the scale of impact on the biota, is shown in Figure 6.12. It also demonstrates that the combination of chemical and biological monitoring information provides evidence of the cause and effect of the pollution incidents.

This figure shows results from a monitoring site on a lowland river in southern England, the River Thame, a tributary of the River Thames. This monitoring site was downstream from a sewage treatment works that was allowed to discharge a very poorly treated effluent. When the major incident occurred is clear from the monthly chemical monitoring results. The oxygen concentration fell to less than 10% saturation – the Bad status boundary for this type of river is 45% (Defra 2016).[147] Although low oxygen saturation had been recorded in the past, it was accompanied by a very high concentration of ammoniacal nitrogen of more than 7.5 mg/l – the Bad status boundary in this type of river is 2.5mg/l N – which is indicative of untreated organic waste.[145] Although an invertebrate survey was not undertaken until about 4 months after the event, its impact on the invertebrate fauna was still clear.

The delayed recovery of biota enables biological assessments to detect pollution long after the pollution has ceased. Although WHPT ASPT and WHPT NTaxa were not particularly high before the incident, both dropped substantially after it: WHPT ASPT fell from 5.1 to 3.3 and WHPT Ntaxa from 23 to 11. Similar changes were seen in other invertebrate indices.

This information was used in enforcement action by the Environment Agency, in the courts, against a privatised water company. The water utility company received a £20M fine because of its poor management at both this and other wastewater treatment plants.

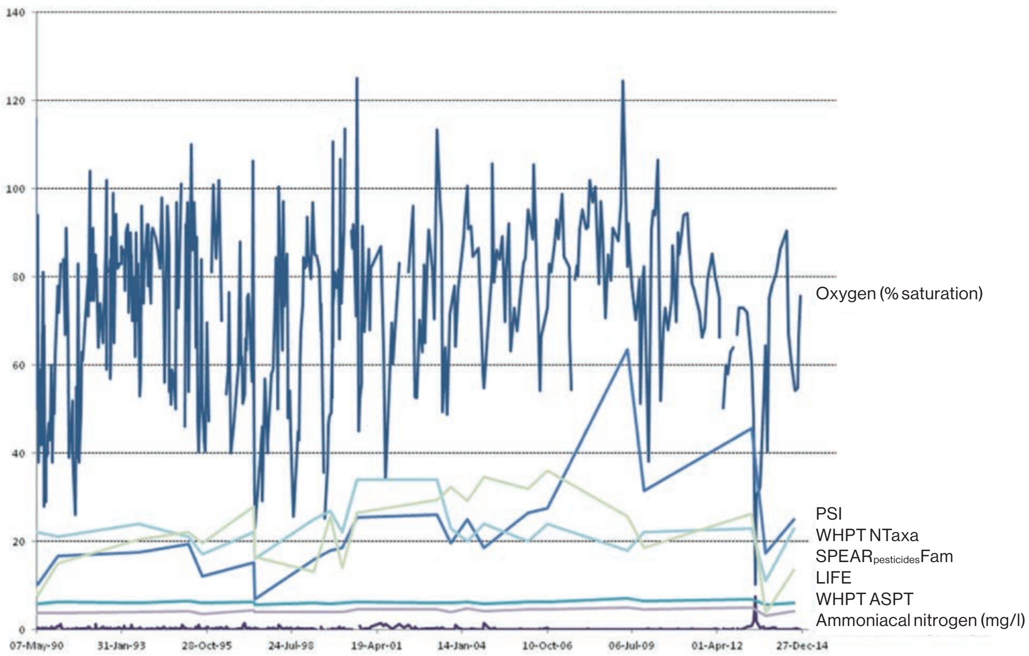


Figure 6.12
Chemical and invertebrate impacts from untreated wastewater discharges to a lowland river

10

UK – NATIONAL MACROINVERTEBRATE SURVEYS AND PUBLICLY AVAILABLE DATA

Environment Agency

10.1 Environment Agency biological survey data

The Environment Agency holds its biological data in an Oracle database known as Biosys, and holds its fish data on the National Fish Population Database.

The data can be accessed from the Ecology and Fish Data Explorer **https://environment.data.gov.uk/ecology/explorer** [145] This holds data from fish, diatom, macrophyte and macroinvertebrate samples. A user guide is embedded in the system.

All the data can be downloaded in one request as a bulk download, but a system with sufficient capacity will be needed. Alternatively, a sub-set of data can be selected by drawing a polygon on the map interface to define the area of interest. The map will show the survey sites within it for the different types of biota (Figure 6.13). A description of the data that is available for a specific site can be seen by clicking on its dot on the map.

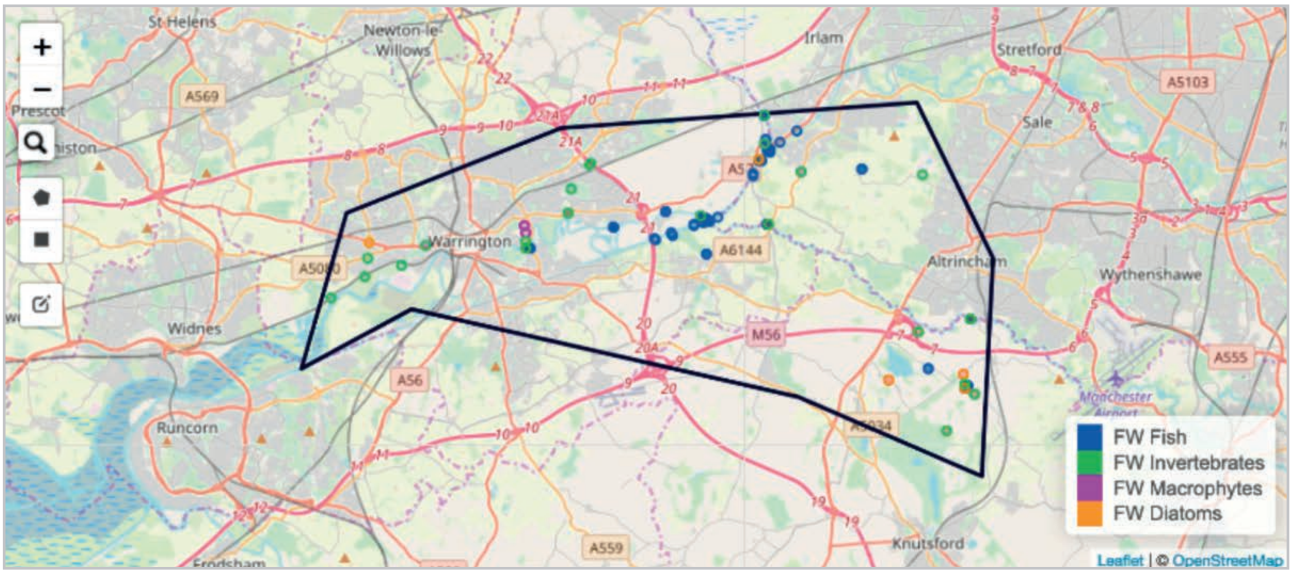


Figure 6.13

Map interface of the Ecology and Fish Data Explorer showing monitoring sites for different types of biota within an area selected by the user

Data download can be specified for specific sites and for specified time periods. Most of the data is from 1990 or later.

An Excel tool to format the invertebrate monitoring data, including the associated environmental data from the Data Explorer so that it can be uploaded to the River Invertebrate Classification Tool (RICT2), is available from the RIVPACS/RICT website RICT2 Data Extraction Template: **https://www.fba.org.uk/rivpacs-and-rict/rict-rivpacs-user-guides**

This template converts data extracted from the interactive map (but not data from the bulk download) into a form that can be pasted into the RICT data input template.

A link to the Data Explorer can be found both in the tool and in the Supplementary Data section of the RIVPACS/RICT User Guides web page.

Take care when analysing historical data. Check the method of sampling and analysis. Samples collected before 1990 are likely to have been collected and analysed by methods other than the standard methods described in **Chapter 2**.

Bankside assessments are less precise than laboratory analyses and are unsuitable for status classification because bias and probabilities of class assume more precise laboratory analysis. Before about the year 2000, only families included in BMWP indices (ie RIVPACS taxonomic level TL1) were recorded at most sites, but from around 2000, additional families included in WHPT were included in family-level analyses (RIVPACS taxonomic level TL2). From about 2013, the Environment Agency standardized on mixed-taxon (species) analysis (RIVPACS taxonomic level TL5), although some areas have analysed to this level for a long time, particularly in East Anglia.

10.1.1 Resource pressure on monitoring programmes

There is a reality that financing for public regulatory organisations goes through cycles, dependent on the overall economic climate, and on public and political priorities.

At all times it is crucial that priorities are set, and the monitoring programmes are as efficient and effective as possible.

However, long-term monitoring is often seen as an opportunity to make savings. But, compared to the cost of expensive infrastructure improvements, monitoring is cheap, and there is a risk that infrastructure improvements are wrongly specified because of a lack of data to determine the design and operational requirements.

Figure 6.14 is an example of the impact on biological monitoring programmes, undertaken by the Environment Agency, caused largely by budgetary constraints. This has led to prioritised changes in the nature of the monitoring programmes. Operational and investigative monitoring, which occurs at sites where problems are either suspected or being worked on, has been retained at the expense of surveillance monitoring at sites where quality has not changed, or where the risk of poor quality is low. However, the focus on operational and investigative monitoring means that there is a risk that the overall information base cannot provide an unbiased measure of the overall state of the environment. Because of these changes in focus, analyses of long-term changes in quality should be undertaken with caution.

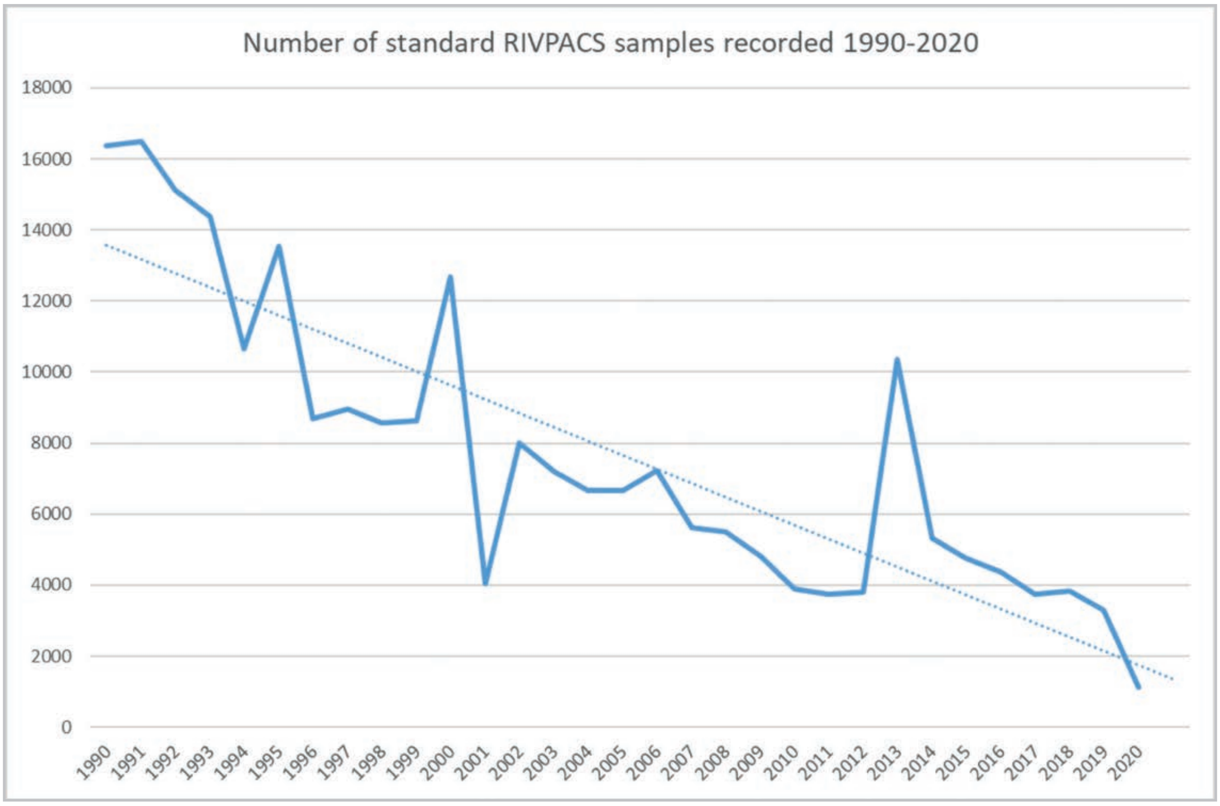


Figure 6.14

The number of standard river invertebrate samples recorded 1990–2020. This record excludes samples from investigations; additional samples were collected in 2020 but had not been analysed when this data was collated in October 2021. This may give a low estimate for the 2020 value. *Source: Environment Agency*

In response to these concerns, and to optimise this reduced monitoring resource, the Environment Agency initiated a new River Surveillance Monitoring Network in 2021. Its aim is to provide a measure of the overall quality of rivers in England and to enable changes in quality, at large scale, to be detected and measured.

This is a minimal programme in which 500 sites are monitored each year using a generalised random tessellation stratified (GRTS) design that balances the need to measure change with the need for extensive and unbiased geographical coverage. Some sites are surveyed every year, but to increase the geographical coverage, other sites are surveyed for two consecutive years once every 5 years, and others only once every 5 years. The whole programme will be repeated every 5 years and comprises 1600 sites (Table 6.3). Sites are selected randomly from a 1:50,000 scale river network to ensure that the surveillance network is unbiased. The network includes small and ephemeral streams that are known to be important (Riley *et al.* 2018) [148] but were not covered adequately by previous monitoring networks. It excludes artificial drainage ditches. Unlike the operational programme, sites are not designed to assess the quality of individual water bodies, but to be analysed together to provide wider-scale information. The network therefore complements the operational network but does not contribute to it.

Table 6.3

The River Surveillance Network GRTS design with a fixed panel of sites monitored every year, 5 panels of sites monitored in 2 consecutive years every 5 years, and 5 panels of sites monitored once every 5 years

	Year																			
	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20
Fixed panel	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100
Rotating panel 1	100	100				100	100				100	100				100	100			
Rotating panel 2		100	100				100	100				100	100				100	100		
Rotating panel 3			100	100				100	100				100	100				100	100	
Rotating panel 4				100	100				100	100				100	100				100	100
Rotating panel 5	100				100	100				100	100				100	100				100
Rotating panel 6	200					200					200					200				
Rotating panel 7		200					200					200					200			
Rotating panel 8			200					200					200					200		
Rotating panel 9				200					200					200					200	
Rotating panel 10					200					200					200					200
No. sites per year	500	500	500	500	500	500	500	500	500	500	500	500	500	500	500	500	500	500	500	500
No. unique sites	500	800	1100	1400	1600	1600	1600	1600	1600	1600	1600	1600	1600	1600	1600	1600	1600	1600	1600	1600

10.1.2 Trends and patterns revealed by Environment Agency monitoring data

There have been surprisingly few analyses of the long-term trends in the ecological quality of English rivers. Globally, there is evidence that freshwater insects are not suffering the declines observed in terrestrial insect communities (Outhwaite *et al.* 2020). [149] Analysis of Environment Agency monitoring data by Vaughan and Ormerod (2012, 2014) at Cardiff University suggest that this is also true of the invertebrate faunas in English rivers. [150] [151]

The original aim of their evaluation was to show the impact of climate change, but instead of a degradation, they found that invertebrate communities were improving. The improvements were greatest in urban areas, but they were also seen in rural areas. They ascribed this to improvements in water quality. This demonstrates how biological communities respond to the integrated effect of all pressures and that the impacts of a pressure (in this case, global warming) can be mitigated by reductions in a completely unrelated pressure (water quality).

There was concern that this improving trend had ceased or even reversed, but a recent extension of Vaughan and Ormerod's studies indicate that although increases in taxonomic richness have slowed or halted, the proportion of invertebrate taxa that are sensitive to oxygen and organic pollutants continues to improve (Pharaoh *et al.* 2021). [152] **https://www.gov.uk/government/publications/an-analysis-of-national-macroinvertebrate-trends-for-england-1991-2019**

Another study, by Craig Macadam (2021) [153] using Environment Agency monitoring data, demonstrates the co-variance of many biotic indices (Figure 6.15, see also **Chapter 5 Section 3.1**). This could be because the indices respond to similar physiological attributes or because pressures occur together.

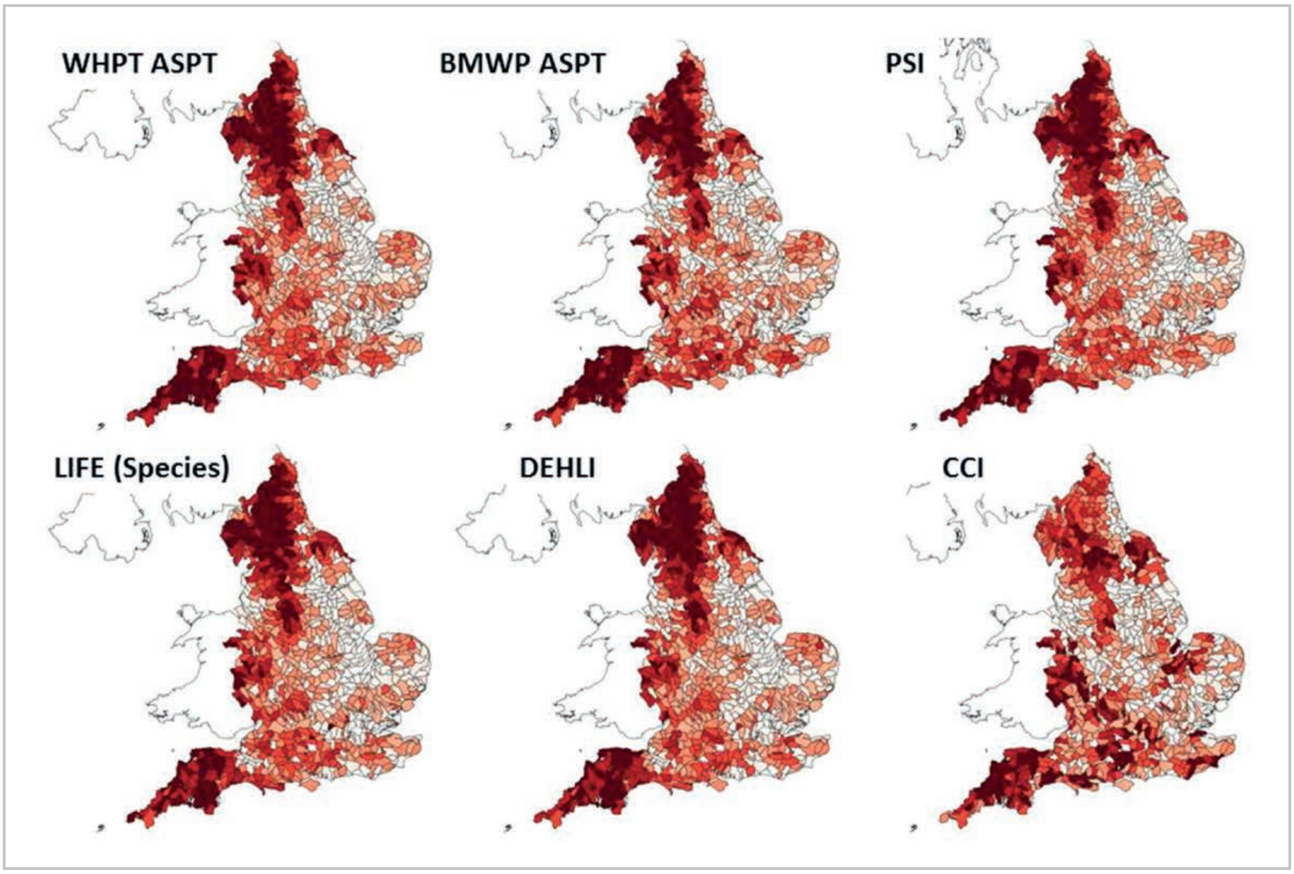


Figure 6.15

Co-variance of invertebrate biotic indices indicated by an analysis of Environment Agency monitoring data *(from Craig Macadam)* [153]


There have been surprisingly few analyses of the long-term trends in the ecological quality of English rivers.

10.2 Countryside Survey

The Countryside Survey (**http://www.countrysidesurvey.org.uk/**) monitors the natural resources of the UK's countryside and has been undertaken periodically since 1978. It is funded by the Natural Environment Research Council (NERC) and the Department for Environment, Food and Rural Affairs (Defra) and it is co-ordinated by the UK Centre for Ecology & Hydrology (CEH), who also undertook the most recent survey in 2007 (**https://www.ceh.ac.uk/our-science/projects/countryside-survey**).

The field survey is a very detailed study of more than 591 x 1 km squares located over England, Scotland and Wales. The squares are chosen so that they represent all major habitat types in the UK. Enough squares are selected for each type to make sure that the statistical analysis for that habitat is robust and reliable. The location of the study squares is kept confidential to avoid any deliberate influences that could affect them or the features within them. In this way the sample squares will remain a true reflection of changes in the wider countryside; they will continue to provide a reliable comparison for future surveys. However, lack of location data usually makes it unsuitable for other investigations or for combining with data for other surveys.

The countryside survey covers both terrestrial and freshwater environments. The freshwater surveys encompass both standing and running waters within each surveyed grid square. Because they are the most common type of water bodies, headwater streams and ponds predominated the freshwater surveys and separate reports were produced after the 2007 survey for these habitats (Dunbar *et al.* 2010 [154] and Williams *et al.* 2010). [155]

Standard methods are used to allow the results to be compared with those from previous surveys and therefore to enable changes in the quantity and quality of the UK's countryside to be detected. Freshwater methods are described in Murphy & Weatherby (2008). [156] Macroinvertebrate, aquatic macrophyte and river habitat surveys are undertaken.

Data from the Countryside Survey is available from **https://countrysidesurvey.org.uk/data**

10.3 UK Acid Waters Monitoring Network/ Upland Waters Monitoring Network

The UK Acid Waters Monitoring Network (UKAWMN) covered 11 lakes and 11 streams across the UK that were monitored chemically and biologically from 1988 to assess the ecological impact of acid deposition in areas believed to be sensitive to acidification. It is managed by Defra (**https://uk-air.defra.gov.uk/networks/network-info?view=aw**) and coordinated by the Environmental Change Research Centre (ECRC) at University College London. Results are stored in a database managed by CEH Wallingford and are available via links at Defra's web page. [157]

From 2013, the UK Acid Waters Monitoring Network became the Upland Waters Monitoring Network (UWMN) **https://uwmn.uk** designed to track changes in surface water quality and freshwater biodiversity across all upland regions of the UK, not only those sensitive to acid deposition. The network covers 12 lakes and 13 streams across the UK, which are monitored chemically and biologically.

The network is surveyed for water chemistry, fish, macroinvertebrates, aquatic macrophytes, diatoms, chironomids and zooplankton. Where appropriate, sediment traps, thermistors (from 2013), and sediment coring are used. Standard methods are used to collect data from the network, described at **https://uwmn.uk/methods**. These methods are generally those used for standard WFD status assessment.

UK Environmental Change Network

10.4 Environmental Change Network

The Environmental Change Network (ECN) was established in 1992 by NERC to monitor long-term environmental change and its effects on ecosystems (**http://www.ecn.ac.uk/**). It is co-ordinated by CEH. Since 1992, the Environmental Change Network has operated sites across the UK at which its partner organisations make a wide range of environmental measurements. Data are sent to the ECN Data Centre (**http://data.ecn.ac.uk/**), where they are checked and added to the ECN database and made freely available for research, education, and other non-commercial purposes.

ECN includes both terrestrial and aquatic environments. The freshwater component of ECN covers both lakes and rivers/streams and includes surface water chemistry and quality, surface water discharge, phytoplankton, aquatic macrophytes, epilithic diatoms, zooplankton, and macroinvertebrates. Methods are described at **http://www.ecn.ac.uk/measurements/freshwater**

Data from most sites are collected by the environment agencies in Scotland, Northern Ireland, England and Wales as part of their monitoring networks for status classification, so standard methods are used. The remaining sites are operated by a range of ECN partners. Several of the sites are Upland Waters Monitoring Network sites (**Section 10.3**).

10.5 Riverfly Census

The Riverfly Census is undertaken by Salmon & Trout Conservation UK to ensure that action is taken to protect the quality of rivers used for game fishing **https://salmon-trout.org/projects/riverfly-census/**

The first Riverfly Census was undertaken in 2015 and involved surveying sites on 12 English river systems (S&TC UK, 2015).[158]

Five sites were sampled on each river, in spring and autumn, using RIVPACS sampling methods. These were analysed to species in order to derive a diverse group of measures to 'fingerprint' their quality.

These measures comprised:

- Biological Monitoring Working Party (BMWP) score
- Average [BMWP] Score Per Taxon (ASPT)
- Species Richness (number of species, R)
- Ephemeroptera Plecoptera Trichoptera (EPT) richness
- Community conservation index (CCI)
- Total invertebrate abundance
- Total Reactive Phosphorus Index (TRPI)
- Saprobic Index (SI)
- Proportion of Sediment-Sensitive Invertebrates (PSI)
- Lotic Invertebrate index for Flow Evaluation (LIFE)
- the abundance of *Gammarus*
- the abundance of *Seratella ignita.*

The report, and the analyses that it described, was designed to be read by a general audience, not scientists, so particular care was given to clear presentation.

A colour-coded 'traffic-light system' was used to classify the face values of LIFE, PSI, SI and TRPI on a 5-class scale. The census was expanded in 2017 to cover more rivers, and an updated survey was undertaken in 2019 (S&TC UK, 2019).[159]

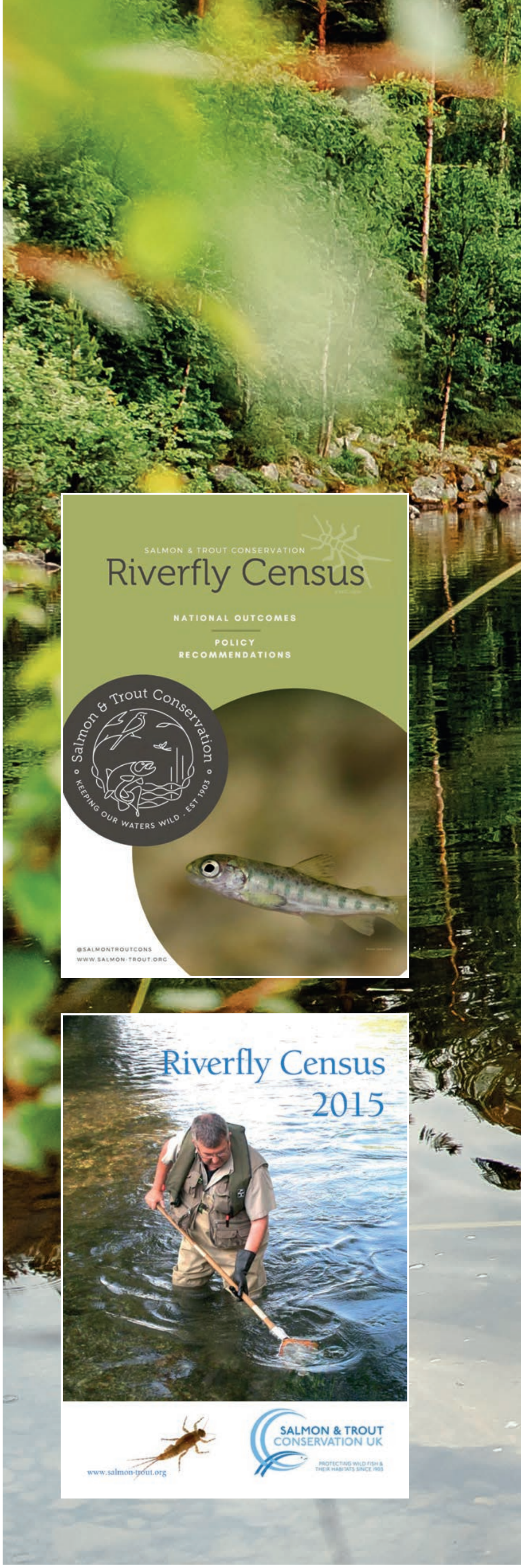


Figure 6.16
Front covers of the 2015 and 2019 Riverfly Census reports [158] [159]

11

EU STATE OF THE ENVIRONMENT REPORT

The latest publication of the European State of the Environment report, by the European Environment Agency (EEA), was produced in 2018, entitled *European Waters, Assessment of Status and Pressures 2018.* [160] **https://www.eea.europa.eu/publications/state-of-water** (Figure 6.17).

The EEA report aims to present results on:

the status of EU waters based on the second River Basin Management Plans

the pressures that are causing less than Good status

the progress that was achieved during the first RBMP cycle (2010–2015).

The report presents results on the status of surface waters and groundwater in Europe, providing overviews at EU, Member State, and River Basin District (RBD) levels.

The key messages on Ecological Quality are given in Box 6.3.

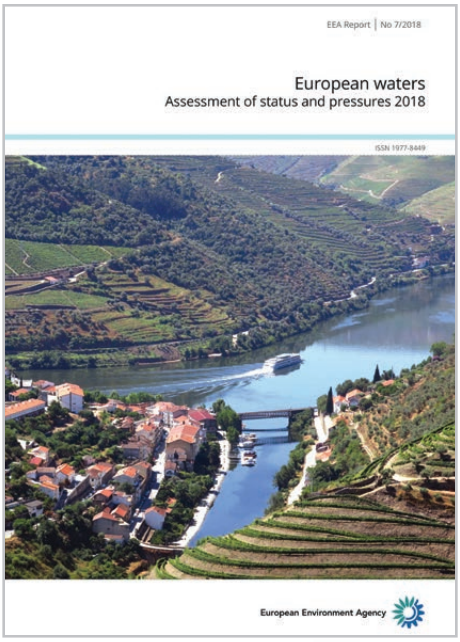


Figure 6.17

Cover of *European waters, assessment of status and pressures 2018* [160]

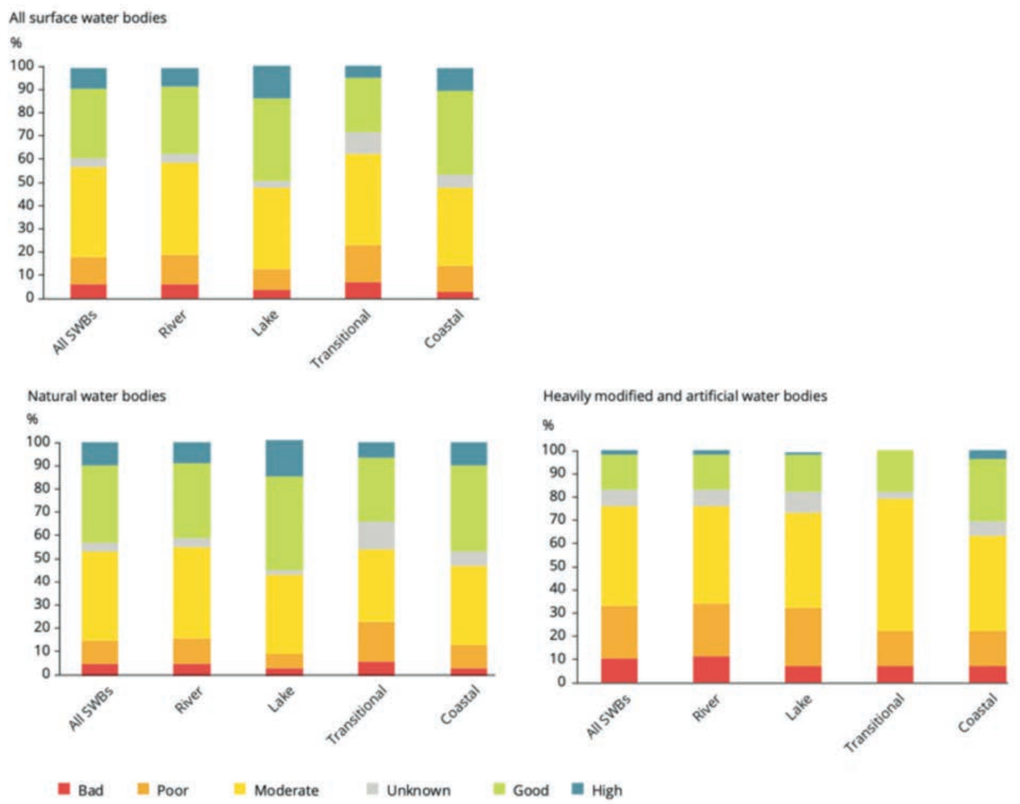
Box 6.3
EEA Report, 2018

Key messages

- On a European scale, around 40% of the surface water bodies are in Good or High ecological status or potential, with lakes and coastal waters having better status than rivers and transitional waters.
- The status of many individual elements (biological quality elements and supporting physico-chemical and hydromorphological quality elements) that make up the ecological status is generally better than the overall ecological status.
- The overall ecological status has not improved since the first RBMPs, but has improved for some biological quality elements from the first to the second RBMPs.
- The main pressures are point and diffuse source pollution, and various hydromorphological pressures. Diffuse source pollution affects 38% of surface water bodies and point source pollution affects 18%, while hydromorphological pressures affect 40%.
- The main impacts of the pressures on surface water bodies are nutrient enrichment, chemical pollution and altered habitats due to morphological changes.
- Member States have made marked efforts to improve water quality and hydromorphology. Some of the measures have immediate effect; others will result in improvement in the longer run. Effects are usually visible at the level of individual quality elements but often do not translate into an overall improved ecological status.

The EEA report provides the following overview of Ecological Status across Europe:

> ***Overall, around 40% of the surface water bodies are in Good or better ecological status, while 60% did not achieve Good status (Figure 6.18). Lakes and coastal waters are in better status than rivers and transitional waters. The ecological status of natural water bodies is generally better than the ecological potential of heavily modified and artificial water bodies.***



Notes: All SWBs means all surface water bodies, comprising rivers, lakes, and transitional and coastal waters. The classification of rivers by length and of the other water categories by surface area shows a similar distribution of status classes to the classification by number of water bodies (see below links), except that a lower proportion of the area of transitional waters is classified as being in high or good quality status.

Source: Results are based on WISE-SoW database including data from 25 Member States (EU-28 except Greece, Ireland and Lithuania). Surface water bodies: Ecological status or potential and chemical status, by category.

Figure 6.18

EEA – Water ecological status or potential, second RBMP, 2018

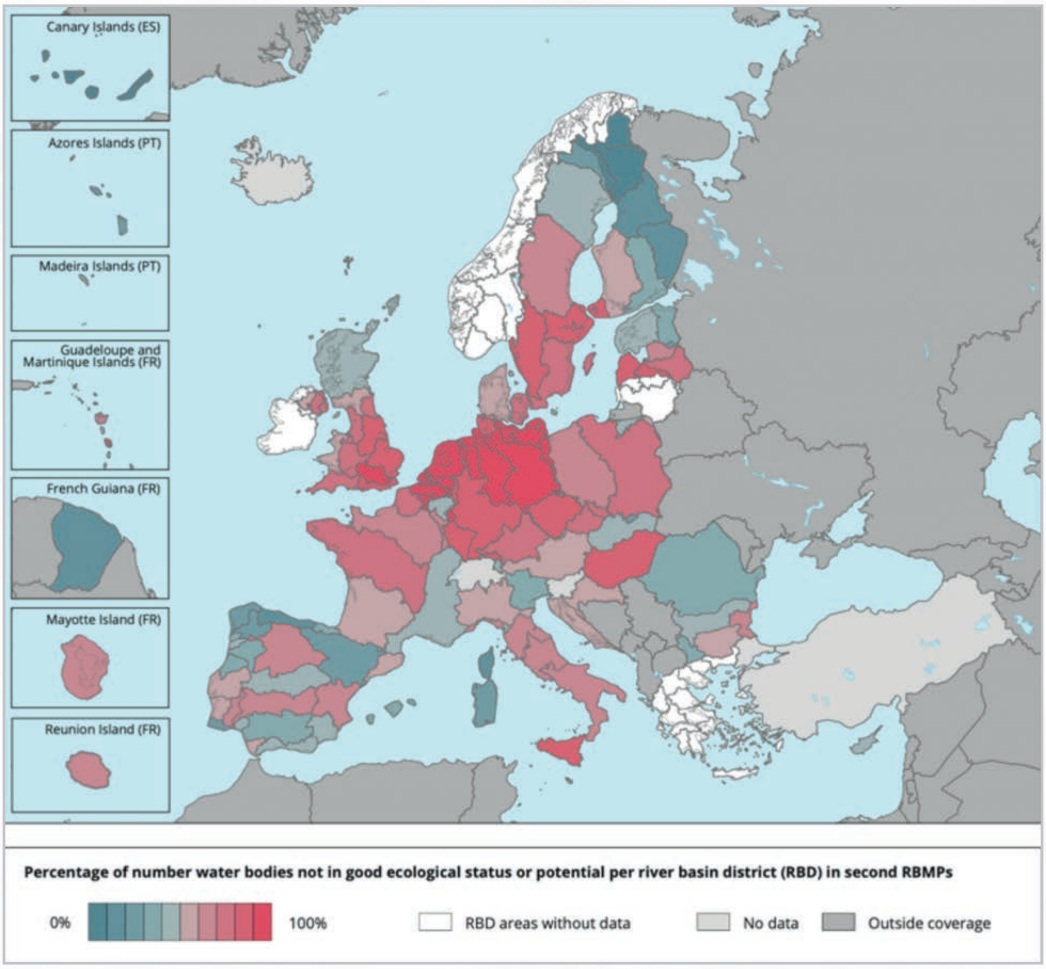


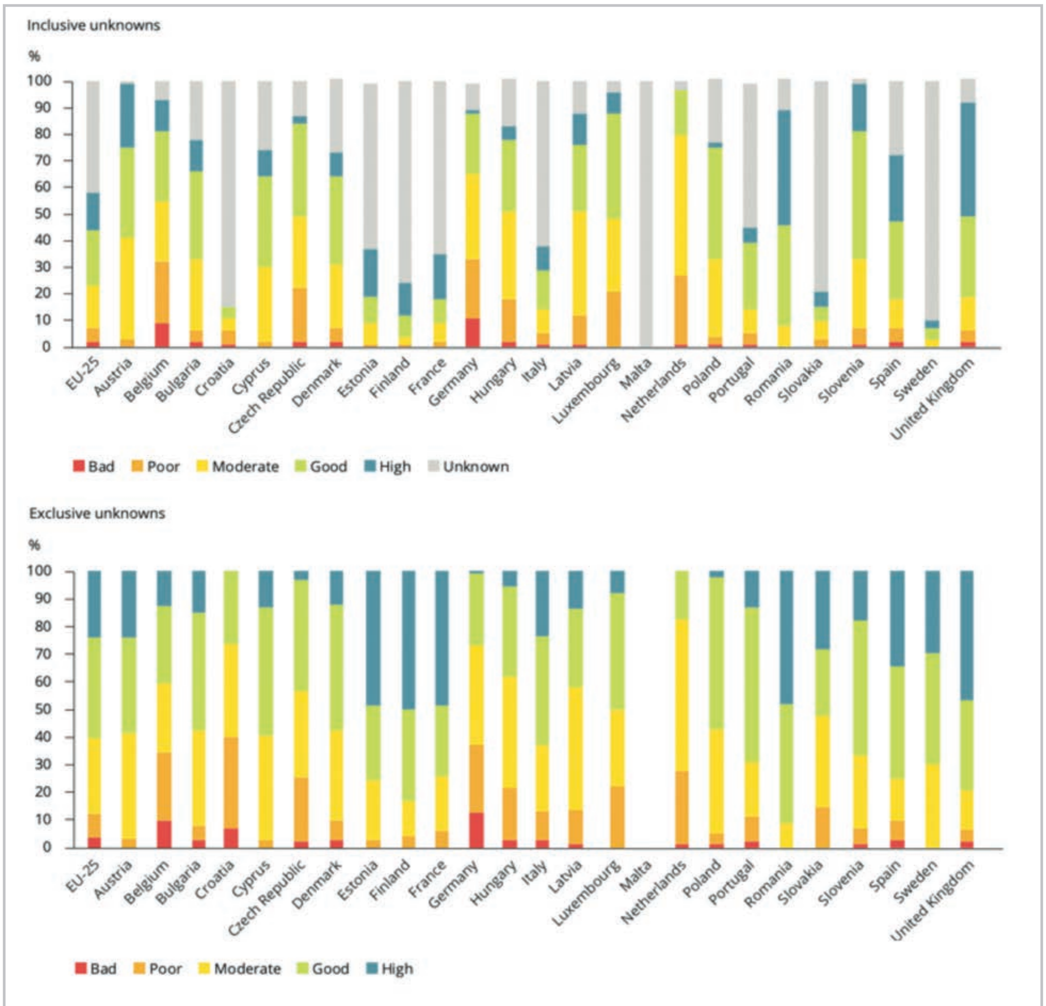
Figure 6.19

Water bodies that are not in Good ecological status or potential according to the second RBMPs

At a European river basin level, this map (Figure 6.19) provides an overview of Good ecological status according to Member State monitoring programmes.

For one of the biological quality elements (benthic invertebrates in rivers), Figure 6.20 illustrates the differences in ecological status according to Member States. In several Member States more than half of the river water bodies have not been assessed for benthic invertebrates (see top panel). The rivers with the best ecological status for benthic invertebrates are found in Romania, Finland, and the United Kingdom, while those with the worst are found in the Netherlands, Germany, and Croatia (see bottom panel).

The WISE-Freshwater tool makes it possible to explore similar results for other quality elements and categories. This EEA database can be accessed at **https://www.eea.europa.eu/themes/water/european-waters/water-quality-and-water-assessment/water-assessments/quality-elements-of-water-bodies**



Notes: Classification of ecological status for benthic invertebrates in rivers including the water bodies with unknown status for this biological quality element (grey bars) (top panel) and excluding water bodies without unknown status (bottom panel).

Source: Results are based on WISE-SoW database including data from 25 Member States (EU-28 except Greece, Ireland and Lithuania). Surface water bodies: QE status, by quality element and country.

Figure 6.20

Benthic invertebrate status across EU Member States (from EEA report)

Across Europe:
overall, around 40% of the surface water bodies are in Good or better ecological status, while 60% did not achieve Good status.

Source European Environment Agency (2018)

12

COMPLIANCE ASSESSMENT AND REPORTING – SUMMARY AND WAY FORWARD

The WFD and the UK 25-Year Plan focus on **outcome-based** water policy and implementation, in comparison to many of the previous **process-based** Directives such as the Urban Wastewater Treatment Directive. The outcomes include Good Ecological Status, which is reliant on assessment of biological and ecological indicators of water health.

The biological monitoring and assessment methods are therefore critical in optimising water management and improvement programmes. These potentially drive very expensive investment programmes (eg the UK Water Industry National Environment Programme). This emphasises the need for high quality information from well-constructed monitoring and assessment programmes, with appropriate quality assurance methodology. We have tried to give wide access to best practice methods via this handbook.

We are increasingly reliant on data and reporting to make effective policy and water management decisions. Integration of biology, ecology, chemical and hydrology information is critical. All UK improvement programmes are modelled to ensure the most cost-effective options, or combinations of options, are selected. The quality of the model outputs is directly related to the data available. Unfortunately, monitoring is often one of the first cuts to be made in cost-saving initiatives, even though these may influence significant and expensive investment. Without this information, the effectiveness of the investment may be difficult to assess and the feedback loops to the next investment cycle may be broken.

We are increasingly seeing ‘citizen science’ programmes (see **Chapter 5**) being used at local and river basin levels. Integrating this information into the formal monitoring programmes, usually undertaken by the environment agencies, is difficult, but work is ongoing to optimise this valuable additional information. It also has the benefit of engaging a wider societal interest and understanding of the issues.

Wider engagement drives a more open approach to data availability and use, together with improved communication via reports and graphical representation of information. New Internet and mapping-based information systems, with access to the primary information, are being made available. However, high-level summary information, in terms of published maps and reports, is generally less available – possibly a retrograde step.

High quality river basin planning requires constant improvement of monitoring, assessment, and reporting, without reducing the precision needed to facilitate complex decision making.

We hope this handbook will allow this to be developed further to continue the protection and improvement of the water environment – in the UK, the EU and globally.

Concluding Statement

This handbook provides an overview of the biological and ecological methods used to assess the status of the freshwater environment. Good river health is the key outcome and aim of this work.

Chapters 1 to 6 of the current handbook provide the context, development and core approaches used and developed by the UK and the EU to improve and protect freshwater quality, much of this aligned to the delivery of the Water Framework Directive. The principles remain constant and feed forward into possible reshaped UK approaches in the future.

Chapter 1	provides an overview, including the legal framework for freshwater biological monitoring.
Chapter 2	is a practitioner’s guide to the standard methods for invertebrate sampling and data collection.
Chapter 3	provides an understanding of current river invertebrate classification methodologies, focusing on RIVPACS and surveillance monitoring.
Chapter 4	looks at other sampling methods for investigative monitoring.
Chapter 5	looks at indices and data analyses for investigations, including the increasing contribution from citizen science programmes.
Chapter 6	considers the reporting methods used in the UK and the EU, specifically with links to investment programmes, driven by the monitoring and assessment information. It also provides links to publicly available data sets.

The focus, so far, has been on river invertebrate methodologies and on status classification using UK RIVPACS to provide a working example of what is needed to set up a biological monitoring programme for a national initiative, a river catchment or a specific tributary. Most invertebrate methods utilise these key principles and we expect users to modify and adapt methods to their specific situations as needed.

Several key biological and ecological methods are not covered in this handbook, including fish, macrophytes, diatoms, river restoration methodologies, still-water methods, and statistics and computing methods. We invite other specialists to contribute to add additional chapters or sections to expand its coverage.

Future developments in biological monitoring will be important and should also be considered for update and later inclusion in this handbook. These may include DNA analysis, remote sensing, and use of social networks for communication of river health and environmental issues.

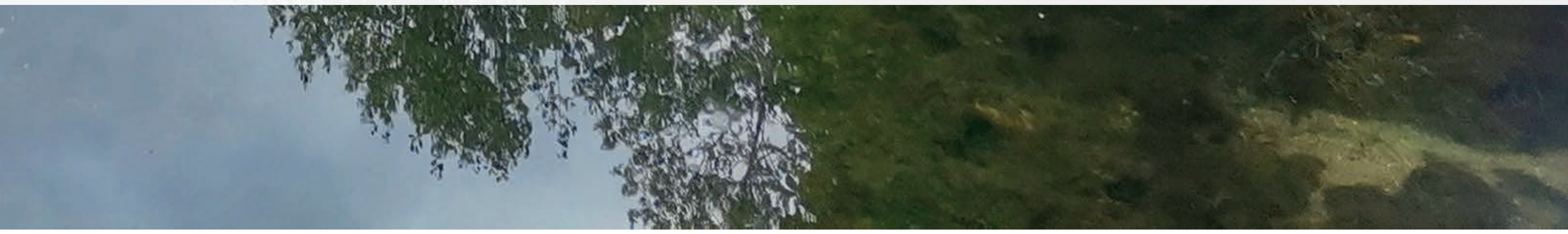
Increased community science initiatives and public participation will also help in data gathering and presentation. However, it is important that these are robust and fully integrated with core data sets where possible. These potentially drive significant investment in infrastructure and land use management, so need to be reproducible and consistent.

We hope that making this publication free of charge, and for public good, will accelerate the understanding of freshwater systems around the world. The core elements described here are the basis for training programmes and university teaching, to provide the expertise to consolidate the improvement of river health into the future. Access to key texts and references to the original documents will also be invaluable to practitioners.

We regard this handbook as being an open and living publication. Improvements, new sections, and examples of good practice are welcomed. We will seek to ensure that appropriate mechanisms for additions and improvements to its content are put in place.

Finally, we hope that this provides a useful insight for civil servants, water managers, specialists, and river conservation groups working to improve and protect our invaluable freshwater environment.

We invite you to contribute.



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Paul Logan has assisted as technical advisor and reviewer of this book. Paul was one of the key water industry biologists in the UK and worked in a number of policy and strategy roles with the Environment Agency for England, and its predecessors. He was one of the specialists who contributed to the development of the biological monitoring and assessment elements of the EU Water Framework Directive. He recently retired from the Environment Agency and chairs British (BSi) and European (CEN) standards' methods groups.

Dr Judy England was instrumental in promoting the rewriting and production of this handbook. She contributed the section on ephemeral rivers, commented on other sections, and provided photographs. Judy is an aquatic ecologist. Her areas of interest include the environmental effects of flow regulation on river biota and biota-habitat relationships, including resilience to climate change. Her interest in river restoration appraisal was the subject of her PhD at the University of Hertfordshire.

Nick Mott is a freshwater ecologist who has worked with The Wildlife Trusts for over 25 years; his chief role has been to undertake over 70 stream, river and floodplain restoration schemes in partnership with an eclectic cast of scientists, sponsors, landowners and digger drivers. Invertebrates are Nick's main interest especially crayfish, mussels, dead wood hoverflies, river shingle beetles and montane mayflies.

Dr Alistair Ferguson was the lead scientist for monitoring of the aquatic environment with the Environment Agency and its predecessors. He was a key advocate and architect for the development of biological methods to assess aquatic status and their integration within the UK approach. He was highly influential in their incorporation as a component of the WFD. A key advisor, now retired.

Dr John Davy-Bowker has contributed significantly to the development of this handbook and to the advancement of river biology and assessment. He has a keen interest in macroinvertebrates including conservation, species identification and sample processing methods, biomonitoring, citizen science, DNA collections, and long-term monitoring. Being passionate about protecting freshwater biodiversity, he has had a long involvement with the RIVPACS predictive model, and has led most of the research and development projects on this in recent years.

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Lee Knight has been working as a freshwater ecologist for almost 30 years, first for the NRA/EA, and freelance since 1997. He developed an interest in subterranean and groundwater ecology, and began by investigating the occurrence of the British endemic *Niphargus glenniei* in Devon and Cornwall, which lead to it being included in the UK's Biodiversity Action Plan in 2007 as a priority species. He has investigated other subterranean Malacostraca across Britain and Ireland, and set-up the Hypogean Crustacea Recording Scheme in 1999. He has authored or co-authored almost 30 papers and articles on various aspects of subterranean ecology in the British Isles.

Jonathan (Jon) Webb has worked for Natural England for 20 years, serving most of his time as an entomologist. Throughout this period he focused on wetlands, riparian habitats and dead wood habitats. He has spent considerable time undertaking field work around England and beyond, often with Nick Mott, and is the lead author of Pantheon (pantheon.brc.ac.uk).

Nicola Edgar has worked as a freshwater ecologist for the Environment Agency for 15 years, working in biological monitoring teams specialising in invertebrate identification. She later moved into the partnership delivery of environmental schemes, and leading citizen science initiatives. This led to her involvement in the development of the Urban Riverfly initiative.

Prof Simon Spooner commissioned the initial Chinese publication of the Ecological and Biological Monitoring Handbook in 2012 as part of the EU knowledge exchange programme with China. Simon is an expert in water quality modelling and national and international water policy. From 2007 to 2012 he was team leader of the Yellow River component of the EU-China River Basin Management Programme, then Technical Director for Water & Environment in Atkins China. Returning to the UK in 2020 he now leads Atkins modelling of the river impacts of sewer overflows using his SimBasinQ model framework.

Naomi Lumsden assisted the authors with the technical develpment of Chapter 1. She joined the Freshwater Biological Association to undertake this role and to provide technical support across FBA. She was Publications & Content Officer for FBA and is now a Coordinator at the Riverfly Partnership. She has recently completed an MSc in Conservation and Biodiversity.

Reindeer Torenbeek was the co-author, with Martin Griffiths, of the original Ecology and Biological Monitoring Handbook. He is an aquatic field biologist from the Netherlands. He worked as a consultant in this field on a number of EU projects including the STAR-AQM project, and on a number of knowledge exchange programmes in China and elsewhere. He currently teaches in the Netherlands.

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