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Ecological indicators of the effects of abstraction and flow regulation; and optimisation of flow releases from water storage reservoirs

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EXECUTIVE SUMMARY

WFD21D: Ecological indicators of the effects of abstraction and flow regulation; and optimisation of flow releases from water storage reservoirs

Project funders/partners: SNIFFER, Scottish Environment Protection Agency, Environment Agency, Northern Ireland Environmental Agency.

Key words: Water abstraction, impounding reservoirs, hydroelectric power, conceptual models, ecological indicators, river regulation.

Background to research

The UK Technical Advisory Group for the Water Framework Directive is currently undertaking a review of its guidance on classification of water bodies and the environmental standards it established to regulate the abstraction and impoundment of water in rivers. At present there is not a good relationship between the environmental standards for river flows and the biological classification of many water bodies. This is seen in water bodies across all of the classification bands, including those at Poor and Bad status. Additional ecological supporting information is required to improve the certainty of the classification of water bodies that are subject to major and severe hydrological impacts, and to increase the weight of evidence needed to identify where mitigation measures are needed be put in place to improve river flows.

Objectives of research

Two key work requirements have emerged:

1. The identification of simple, field measurable ecological indicators of major and severe hydrological impacts, consistent with Poor and Bad status in rivers and adjacent wetlands, as a result of a) water abstraction and b) flow regulation from water storage reservoirs.
2. A decision support framework to help environmental protection agencies decide how water is best released from water storage reservoirs to optimise ecological benefit (flow optimisation framework).

These are underpinned by working descriptions (conceptual models), which distil existing knowledge and describe the adverse ecological effects on rivers and dependent wetland habitats that result from changes to river flow regimes.

Key findings and recommendations

Conceptual Models

The report illustrates the changes to flow regimes resulting from the abstraction and impoundment of water, and elucidates the important connections between flow modifications and ecological impacts in rivers and flow-dependent wetlands.

The conceptual models describe the ecologically important components of the river flow regime that should be the focus for management effort and that form the basis of a framework for optimising water releases from impoundments in rivers:

- extreme or extended low flows;

- enhanced and stabilised low flows;
- loss of high flow pulses (return period <1 year) or small floods (2-10 year events);
- loss of large floods (>10 year events);
- extreme high or untimely discharge; and
- rapidly changing flows.

Alterations to these ecological flow components changes hydrological, hydraulic and geomorphological parameters in rivers and riparian wetlands. These combine to create the habitat state – the conceptualisation of the physical environment that supports aquatic organisms. Emergent properties of the habitat state have been identified that are important to allow aquatic organisms to reproduce and progress through their life-cycles, and form the basis of identifying abiotic ecological indicators of the severe effects of river flow alteration:

- **size** of the habitat (area/volume of aquatic habitat space);
- **connectivity and juxtaposition** of habitat; and
- **character** and diversity of the habitat (ecological 'quality' of the habitat).

This conceptualisation provides some understanding of the reasons why current biological classification methods, which are designed to detect water quality impairment, are often insensitive to hydromorphological pressures. The conceptual model illustrates why this is the case, and identifies ecological indicators that might support current classification tools in identifying major and severe impacts of hydrological alteration.

By incorporating existing research on the on the flow requirements of aquatic organisms, the conceptual model provides some useful information for the management and regulation of low flows in rivers. Minimum flow requirements are wide-ranging among different organisms and rivers, but there is still insufficient quantitative information to define the flow requirements of aquatic organisms more precisely than achieved by previous SNIFFER research reports (WFD 48; SNIFFER, 2006a) and UKTAG guidance (UKTAG, 2008a, b). There remains, therefore, considerable uncertainty in the existing environmental standards and condition limits for managed river flows, and the conceptual model supports the use of any prescribed minimum flow standards only in a risk-based, adaptive management context, and not as fixed values without latitude.

This report includes interpretations of UKTAG recommendations on river flow standards, flow condition limits and flow mitigation measures for heavily modified water bodies. These interpretations may not necessarily reflect the intent of UKTAG's recommendations or how the standards, condition limits and mitigation measures are used in practice by the UK environment agencies.

Ecological indicators of major and severe effects of abstraction and impoundment in rivers

The conceptual models have described a suite of biotic and abiotic ecological elements from which 54 candidate ecological indicators have been identified. The ecological indicators are mostly easily measurable in the field or can be derived from existing biological sample data, and do not require extensive specialist expertise.

Ecological indicators will be subject to local influences and their behaviour is likely to be river type-specific. Specific combinations of indicators are likely to apply to different river types and situations. However, when taken together it is expected that the ecological indicators will be able to provide a weight of evidence approach to identify river sites that are most severely affected by river flow alterations. This in turn will improve the certainty of classification of Poor and Bad status and improve the weight of evidence for prioritising mitigation measures in the most severely impacted water bodies.

Consultation with a wide range of experts and practitioners in hydro-ecology and water management through an expert workshop has been an important feature of the project. This agreed that the strength of the ecological indicators is in the combination of biotic, abiotic, multi-taxa and multi-trophic level indicators. Inevitably, however, some groups of ecological indicators offer greater certainty and potential for further development. These included: freshwater macroinvertebrate indices (Lotic invertebrate Index for Flow Evaluation [LIFE] and Proportion of Sediment-sensitive Invertebrates [PSI]), combinations of hydraulic measures and fine sediment deposition, bryophytes and terrestrial plants on exposed mid channel substratum and depositional features, and diatom indicators.

Key recommendations:

- Develop specific survey methodologies and undertake field trials in a range of water bodies that are subject to major and severe hydrological alterations, and comparable control water bodies.
- Refine the diagnostic capabilities of different combinations of ecological indicators and generalities within river types.
- Develop the LIFE methodology for use in Scotland and Northern Ireland and for diagnosing the severe ecological effects of river flow regulation downstream of impoundments across the UK. Using local reference sites might reduce the uncertainty around modelled reference values in specific water bodies. PSI might improve the diagnostic power of LIFE, especially at locations that are most severely affected by altered river flows.
- Remote sensing techniques have advanced rapidly over the past few years. Remote sensing could provide a solution to mis-matches between spatial scale of observation relative to the scale of environmental impact described in the conceptual model and enable combinations of ecological indicators to be assembled cost-effectively at larger spatial scales. Remote sensing also offers the possibility of surveying previously inaccessible locations.

Optimisation Framework

The optimisation framework sets out a generic decision support framework for determining how available water should be released from impoundments to reduce adverse ecological impacts and to enhance the ecological potential in downstream water bodies. The work leads directly from the recommendations in SNIFFER research project WFD 82 (SNIFFER, 2007; Acreman et al. 2009) and considers further work undertaken since. The optimisation framework is based upon the Building Block Methodology and is expressly designed to use the conceptual models presented in this report.

Given the extreme uncertainty in quantifying river flow-ecology relationships at the scales appropriate for river management, we advocate a risk-based approach. This identifies and prioritises risks and flow needs for the chosen habitat or ecological element, and rather than focussing upon formal objectives (e.g. WFD standards), identifies risk areas resulting from potential flow modifications.

Consistent with the recommendation in modern river regulation studies, the optimisation framwork is designed for local solutions to be based upon local information coupled with effective monitoring and adaptive management; the implementation of the optimisation framework should be treated as planned experiments.

Key recommendations:

- The water release optimisation framework should be used in a true adaptive management context in that its implementation should be treated as deliberate, large-scale experiments. In this way, uncertainty can be embraced by decision makers in making policy choices
- The optimisation framework should be trialled at a number of key sites and monitoring data collected. It is apparent in the literature that few studies have implemented this kind of framework and have collected data suitable for informing scientific-based decision making.

TABLE OF CONTENTS

1 INTRODUCTION

1.1 Background

The UK Technical Advisory Group (UKTAG) on the Water Framework Directive (WFD) was set up by the UK environment and conservation agencies to provide them with technical guidance on the development of tools and environmental standards to help implement the WFD in the UK.

In order to deliver the objectives of the WFD, the ecological status of surface water bodies needs to be classified. The current water body classification system across the UK relies on an assessment of a number of biological quality elements, chemical supporting elements and supporting hydromorphological conditions. Hydrological pressure is a component of the hydromorphological supporting condition and is measured by a specific assessment against current river flow standards that are considered necessary to support good ecological status. SNIFFER research project WFD 48 (SNIFFER, 2006a; Acreman et al. 2008) defined environmental standards used for regulating water resource use in the first cycle of River Basin Management Plans (RBMPs). Another SNIFFER research project, WFD 82 (SNIFFER, 2007; Acreman et al. 2009) set out the principles for and produced guidance on environmental flow releases from impoundments to meet WFD objectives.

The UKTAG Water Resources Task Team is currently undertaking a review of the environmental standards and its guidance on classification of ecological status and potential it established to protect the water environment from the impacts of abstraction and impoundment. At present there is not a good relationship between the assessment of river flow standards and the biological classification. This is seen in water bodies across all the classification bands, including those at Poor and Bad status.

Whilst research is ongoing to improve the certainty of ecology-hydromorphology pressure relationships, the UK environment and conservation agencies have an immediate need to be able to identify water bodies that are severely affected by water abstraction and river impoundments and to put in place appropriate environmental improvement measures. To achieve this in a cost-effective and consistent manner, the agencies need a tool box of ecological indicators that can be easily measured in the field and can be used together with other environmental standards and monitoring data to form the weight of evidence needed to improve certainty in the classification of water bodies that are affected by hydromorphological pressures.

Hydroelectric power (HEP) is a well-established renewable technology that is likely to be very important for the UK in achieving its renewable energy targets under the Renewable Energy Directive (2009/28/EC). Both large and small scale HEP installations however have altered the quantity and dynamics of river flows which can affect the ecology in both upstream and downstream reaches. Similarly, reservoirs for public water supply can adversely affect the ecology in upstream and downstream river reaches. The environment agencies need a framework which they can use to allocate the water that is available from water storage reservoirs for optimising the benefit for river ecology downstream river reaches.

1.2 Project aims

The project has two major aims:

1. To identify simple, field measurable indicators that can be used together with other environmental data to form the weight of evidence needed to identify where flow changes are causing major and severe impacts consistent with Poor and Bad status.
2. To propose a framework to design releases from impoundments in order to optimise the ecological benefits, to minimise adverse ecological impacts, and to enhance the ecological potential on heavily modified water bodies (HMWBs).

1.3 Project scope

The focus of this project is on distilling existing knowledge. The development of new and complex hydroecological methodologies is expressly not within the scope of this project.

The project outputs consider only directly discharge mediated effects arising from the pressures, and therefore exclude associated or confounding influences that may operate alongside discharge mediated effects.

The project outputs consider only the main effects on river reaches downstream of impoundments; they are not intended as a comprehensive treatment of all potential effects.

The project outputs exclude effects on lake (lentic) ecosystems (including that of the impounded reservoir) and transitional waters.

Whilst it is widely recognised that riverine environments encompass temporarily terrestrial and transitional habitats in river flood plains, as well as the base flow river channel, this report considers only the dominantly 'wet' habitats supporting aquatic and wetland ecosystems. The exception is invertebrates of exposed riverine sediments, which are considered here because of their high conservation value and sensitivity to water level changes.

The spectrum of these predominantly 'wet' habitats can be broadly categorised as the river channel (comprising the water column and benthos), the riparian zone (marginal floodplains), and the hyporheos (the subsurface wetted environment beneath the channel and marginal floodplain). However, given the purpose of this conceptual model, changes to the hyporheos are considered chiefly in terms of any subsequent effects they may have on the channel and floodplain.

1.4 Project approach

Both ecological indicators and the optimisation framework are underpinned by an evidence base. Contemporary thinking in environmental flow management recognises the incompleteness of the knowledge base, and seeks to achieve decision making through expert groups and stakeholder engagement. The project has therefore been undertaken in wide consultation with experts in environmental flows from different sectors and European countries.

Recognising that there are many recent reviews on environmental flows, the evidence base has therefore been developed beyond a literature review to a conceptual model of abstraction and flow regulation effects in rivers. It is intended that this will facilitate wider uptake and understanding during this current project and in subsequent projects.

Recognising the diversity of the rich review literature on hydroecology and conceptual models we have as far as possible tried to adopt and adapt previous accounts and to standardise the terminology. The main advance lies in combining well-established conceptual models to offer an integrated model that supports the identification of ecological indicators and the framework for optimising water releases from impoundments.

2 CONCEPTUAL MODEL

Water flow is important because it is the primary control of the physical character of river channels, which in turn is a major influence on the organisms living there (Bunn and Arthington, 2002; Petts, 2009). The connections between hydrological alteration and ecological impacts in rivers are complex; involving multiple interacting parameters; operating at different spatial and temporal scales. To be able to manage river flows effectively and consistently for ecological objectives, a working conceptual understanding of these connections must be established.

In the simplest of terms, abstractions and impoundments affect water flow by modifying the natural flow regime in different ways, according to their specific function, structure and location (Petts, 1984; Acreman et al. 2008). This in turn alters the hydraulics and physical structure of river channels, and adjacent riparian zones: the combination of which forms the river channel landscape occupied by organisms and their ecosystems.

Conceptual models of dynamic systems like rivers can be made almost arbitrarily complex (Feld et al. 2010) and the functional connections and feedbacks amongst discharge, hydraulic and geomorphological parameters are well-described in the literature (e.g. Hynes, 1979; Lewin, 1981; Allan, 1996). The conceptual models and supporting evidence base therefore simplify these interconnections, providing relevant detail only and elucidating the important pathways. As such, the conceptual model represents a highly simplified picture of the most important linkages between pressures and impacts.

The conceptual model presented in this report provides a route map through the important connections from the drivers for human water use to the resulting ecological impacts in rivers. As far as possible a process-based approach (complementing empirical patterns) is taken, because this is more robust in challenging understanding and recognising knowledge gaps.

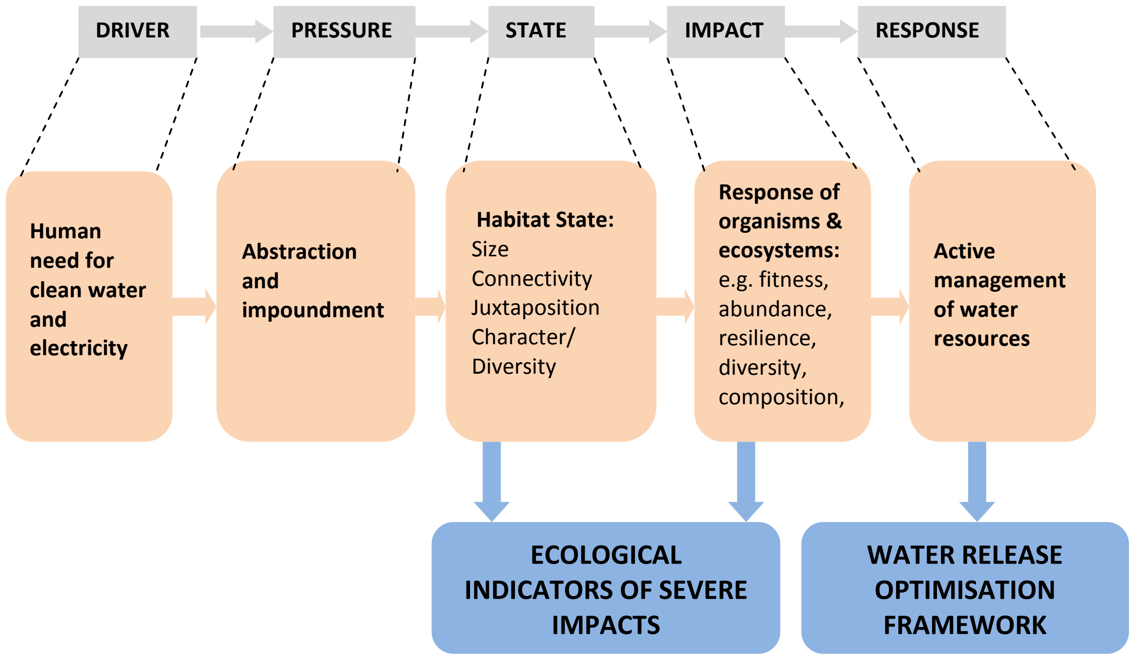
The aims of the conceptual model in this report are:

- to conceptualise the complex interconnections between the drivers, pressures, states and impacts of river flow modifications to provide transparency and understanding for non-technical users and clarity for scientific workers;
- to provide a clear display of the linkage pathways and scientific rationale behind the selection of ecological indicators and the development of the framework for optimising water releases from impoundments, that is open to constructive criticism, challenge and future development;
- to provide a demonstration of the validity, strengths and weaknesses of ecological indicators and the framework for optimising water releases from impoundments, and their application;
- to provide a framework to modify ecological indicator choice, metrics and use as knowledge and understanding develop through future research and adaptive management;
- to provide a traceable, rational connection between the report outputs and policy aims; and
- to generate hypotheses for further testing.

2.1 The framework for the Conceptual Models

To structure the connections between the hydrological effects of human water use and impacts on aquatic biota, the conceptual model adopts the DPSIR (Drivers, Pressures, State, Impact and Response) framework. The DPSIR framework has been used in many environmental impact contexts and is the approach used across Europe by the European Environment Agency (EEA) to link socio-economy with ecology (EEA, 2007), and in ecological research to support the implementation of the WFD (Feld et al. 2010; 2011). This is illustrated in Figure 2.1 with the equivalent steps relevant to river flow management and the derivation of the two report outputs: ecological indicators and the framework for optimising water releases from impoundments.

Figure 2.1 - Application of the DPSIR framework (grey) to defining river flow impacts and management (pink). The origins of this project's products of ecological indicators and the optimisation framework are shown in blue



In this framework, *Drivers* are the societal driving forces behind river flow modification, including, for example, agriculture, industrialisation and population growth. Drivers create the need for water consumption and result in developments that cause *Pressures* on river flows, including the different ways to abstract and impound water.

Pressures directly affect physical parameters, in this case the hydrology, of river channels, which modify the *state* of the environment (habitat) in which aquatic organisms live. Each type of pressure affects water flow by modifying the natural flow regime in different ways, according to their specific function, structure and location (Petts, 1984; Acreman et al. 2008).

State describes the effects that these pressures cause on the physical environment of rivers. In ecological terms, the physical environment of rivers represents the wet habitat that supports aquatic and wetland organisms.

Habitat comprises the hydraulics and physical structure of river channels and adjacent riparian zones; the combination of which forms the habitat state, supporting organisms and their ecosystems.

The state of the environment can result in *impacts* on aquatic organisms, from the responses of individual organisms, through species populations and communities to ecosystem functions.

Response is the human management action to resolve impacts. In this report, responses are restricted to the specific application of the framework for optimising water releases from impoundments. Moreover, the conceptual model is designed to identify the ecological and abiotic impacts of river flow modifications, rather than to substantiate the effect of management intervention. The full range of management responses are therefore not outlined in the conceptual model.

2.2 Using the Conceptual Model

The conceptual model comprises:

1. A textual narrative describing the important links and highlighting the spatial and temporal processes involved that underpin the identification of ecological indicators (Section 3) and the framework for optimising water releases from impoundments (Section 4).
2. An evidence base providing more specific details and literature references of the different ecological elements leading to the identification of ecological indicators (Appendix I).
3. Abbreviated conceptual models designed to guide the application of the optimisation framework. This comprises of process diagrams that link flow changes to habitat state and biotic impacts, and impact tables that summarise the nature and timing of risks to different ecological elements (Appendix II).

The evidence base includes both habitat (hydraulic and geomorphological) and biotic elements, detailed separately in Table 2.1 and Table 2.2.

Table 2.1 - Habitat elements used in the conceptual model and the evidence base reference

Habitat element	Conceptual Model Reference
Wetted perimeter	I.1
Surface flow types	I.2
Volume of fine sediment in channel bed	I.3
Channel bed armouring	I.4
Stability of channel bed	I.5
Stability of channel banks	I.6

Adjustments at tributary confluences	I.7
Spacing of riffles	I.8

Table 2.2 - Ecological elements used in the conceptual model and the evidence reference

Ecological element	Conceptual Model Reference	Ecological element	Conceptual Model Reference
Atlantic salmon (*Salmo salar*)	I.9	Amphibians	I.26
Brown trout (*Salmo trutta*)	I.10	Freshwater pearl mussel (*Margaritifera margaritifera*)	I.19
Grayling (*Thymallus thymallus*)	I.12	White-clawed crayfish (*Austropotamobius pallipes*)	I.20
River lamprey (*Lampetra fluviatilis*)	I.13	Aquatic macroinvertebrates	I.23
Sea lamprey (*Petromyzon marinus*)	I.14	Invertebrates of exposed riverine sediments	I.24
Brook lamprey (*Lampetra planeri*)	I.15	Aquatic macrophytes	I.21
European Eel (*Anguilla anguilla*)	I.16	Riparian vegetation	I.22
Bullhead (*Cottus gobio*)	I.17	Diatoms	I.25
Coarse fish	I.18	Bryophytes	I.27

2.3 Drivers

Drivers for the diversion of water away from rivers by humans include the need to supply freshwater for domestic, agricultural and industrial purposes, and the need to generate HEP. These result in developments that capture water by abstraction from rivers and catchments, and by impounding rivers. These are expected to increase in the future with expanding global populations and as the demand for clean, renewable energy increases. This conflicts with the needs of river ecosystems.

Detailed consideration of drivers is not part of the scope of the conceptual model; what is important is that the pattern of demand created by different drivers influences the behaviour of pressures, thereby influencing how the flow regime is altered, and thus giving rise to different types of ecological effect.

2.4 Pressures

Flow changes are typed into six pressures with the effect of individual abstractions or impoundments considered to be the same within each type:

- steady abstraction (typically unvarying abstraction rates or variable abstractions highly attenuated by groundwater storage, e.g. groundwater and surface water abstraction, and run-of-river hydropower);
- seasonally varying abstractions (e.g. spray irrigation);
- direct supply reservoirs for water supply;
- regulating reservoirs for water supply;
- regulating reservoirs for HEP generation; and
- pumped storage reservoirs

Despite important within-type variation, each of these pressures have commonalities of behaviour that are described for impoundments in Richter and Thomas (2007).

Steady abstraction reduces the magnitude of flows, but in absolute terms, steady abstraction does not change their pattern. Therefore, natural flow variability is maintained. Relative to natural flows, steady abstraction has a greater effect during natural low flow periods. Their effect on low flows has therefore received the most attention in the literature, and any effects on larger flows such as spates and floods can be considered minor. For the purposes of the conceptual model, these can be disregarded.

Variable abstraction reduces the magnitude of flows at particular times only, as, for example, with spray irrigation, which is typically confined to a defined season. Because of their variability, these abstractions superimpose an operational abstraction regime onto natural flow variability. However, natural flow variability is usually of greater magnitude than changes to the abstraction, and the effect of variable abstractions is again greatest relative to natural low flows. Therefore, the main effect of a variable abstraction is to affect flows during defined periods only. Changes to the natural flow variability can be considered of lesser importance and can be disregarded in the conceptual model.

Direct supply reservoirs store water and pipe it from the reservoir away from the downstream river reaches, thereby denying these reaches part of their natural flow. The regime immediately downstream is governed to a great extent by artificial releases rather than natural processes: During low and average flows leakage may maintain a minimal supply of water downstream, and a ‘compensation flow’ (not always made) may augment this further. These compensation flows may be unnaturally high or low, and are in only a relatively few cases varied. Thus, flow variability is maintained only by spills, artificially released spate flows (artificial freshets), and occasional maintenance activity such as scour valve operations. Direct supply reservoirs therefore have an abstraction effect, removing water from the system, and also profound effects on the pattern of flow variability. Attenuation effects on the floodwave characteristics of direct supply and regulating reservoirs are by contrast considered minor and are not considered further in the conceptual model.

Regulating reservoirs share many features with direct supply reservoirs; leakage, compensation flows, artificial freshets, spills and occasional operational releases can all be important aspects of the flow regime immediately downstream. Regulating reservoirs, however, also release water for supply into the downstream watercourse. In the UK, losses in flow (due to evaporation) are therefore comparatively minor. The main effect of regulating reservoirs is instead to change the pattern of flows, which are strongly influenced by supply needs: Downstream of water supply reservoirs, flows are typically higher than natural during the drier seasons of late spring and summer, and lower than natural during autumn and winter, when water is stored for the drier seasons. Pulses are also not synchronised with catchment inputs of sediment and nutrients. For HEP generation, numerous (many per day) abrupt releases are often made to meet peak electricity demands, followed by an equally rapid decline to the compensation level (hydropeaking). These sudden variations are not synchronised with catchment inputs and have no analogue in nature.

Pumped storage reservoirs are a special case of direct supply reservoirs which have a very large storage capacity relative to their catchment; for example the Dinorwic HEP scheme has significant storage on a very small catchment, and the very large Rutland Water water supply reservoir is located on the modest catchment of the River Gwash. Where direct supply or regulating reservoirs may remove spate flows and even small floods, pumped storage schemes can remove even larger floods from the downstream regime.

2.5 States

There are four components of state:

- the direct hydrological effects that result from the pressures;
- and hydraulic effects that result from hydrological changes;
- the direct or indirect geomorphological effects; and
- the combination of these (alongside other physico-chemical properties) create the habitat state in which aquatic organisms live which is the principal link between the pressures exerted by human water use and aquatic organisms.

The different parameters that comprise the habitat state are described in the following sections. This part of the conceptual model identifies emergent properties of the habitat state that provide potential ecological indicators of the severe effects of river flow alteration.

2.5.1 Hydrological parameters

2.5.1.1 Hydrological variability

The hydrological effects (i.e. the loss of or change in the pattern of discharge) of abstraction or reservoir operation have both spatial and temporal dimensions.

Spatially, the hydrological alteration decays irregularly as it propagates downstream, as catchment inflows, tributaries and other artificial influences progressively reduce the proportionate impact of abstraction and introduce variability downstream. It follows from this that there is not only some scheme-specificity of the patterns of hydrological change at an abstraction or impoundment, but that the location of the pressure within the catchment is also important.

Temporally, changes due to abstraction or impoundment may have a very wide range of effects, but take place against a background of natural variability, which can be far in excess of the artificial effect. It follows from this that any artificially introduced hydrological alterations must be set in the context of natural variability, and that this variability must be defined in some way.

Discharge is spatially quite conservative, and provided the time varying inputs (chiefly rainfall) can be characterised, the pattern of flow changes in time and space can be predicted to a degree of accuracy that may be sufficient for use in ecological relationships even with generalised (i.e. inexpensive) models.

Summarising the hydrological variability, however, is problematic – and important. Petts (2009), quoting Naiman et al. (2002), states that 'the fundamental ecological principle for the sustainable management of riverine ecosystems is the need to sustain flow variability that mimics the natural, climatically driven variability at least from year to year and season to season, if not from day to day'.

Historically, abstraction effects have been summarised using the flow duration curve, which is akin to (but not the same as) a cumulative frequency distribution of flows. However, these are of very limited use in describing the changes in pattern experienced downstream of abstractions. A more comprehensive scheme is given by Richter et al. (1996), which categorises hydrological change in terms of:

- magnitude (e.g. mean or other measure of central tendency, over a period: hour, day, month, year);
- duration (how long a flow component persists for);
- timing (when a flow component occurs);
- frequency (how often an ecological flow component occurs over a time period);and
- rate of change ("flashiness", how quickly flows change between components).

Ziegler and Schofield (2007) have additionally differentiated sequencing (timing of events relative to one another) from timing (timing of events relative to the seasonal cycle or calendar year).

Richter's scheme has been shown to characterise the important aspects of the flow regime (Olden and Poff, 2003). Poff and Zimmerman (2010) further report relationships to ecology using all the Richter attributes, in the order shown above (i.e. with more papers establishing links to measures of magnitude and least to rate of change). All of these properties of hydrological alteration must therefore be borne in mind when addressing flow changes downstream of impoundments.

2.5.1.2 Ecological flow components

Although the general scheme of Richter has achieved wide acceptance, there are no universally accepted measures with which to define the different aspects of flow variability. Richter et al. (1996) identified 32 'Indicators of Hydrological Alteration' (IHAs), for which the central tendency (mean) and variability both within and between years were used. These were adopted by SNIFFER research project WFD 82 (SNIFFER, 2007) and Acreman et al. (2009), which for most measures also offered proxies which could be estimated using LowFlows2000. The indicators of hydrological alteration, however, have numerous shortcomings; Olden and Poff (2003) demonstrate significant multicollinearity and redundancy in the indicators and they are not claimed to have particular ecological relevance (e.g. Monk et al. 2007).

Hundreds of alternative indicators have also been offered (e.g. Matthews and Richter, 2007, Olden and Poff, 2003), and since WFD 82 (SNIFFER, 2007), work reported by Petts (2009) offers some scope to simplify to indices of overall alteration. The Hydroecological Integrity Assessment (HIA) component of the Ecological Limits Of Hydrological Alteration (ELOHA) framework (Poff et al., 2010a), which is offered as a consensus approach among numerous international, inter-disciplinary river scientists, also adopted an alternative approach based upon a statistical redundancy procedure to select from descriptors of flow variability.

In view of the complexity of the above measures, and given that given that impacts cited in the ecological literature cannot be related precisely to specific hydrological indices, hydrological variability is summarised using an extension of the Ecological Flow Components (EFCs) used by Richter and Thomas (2007), comprising:

- artificially extreme or extended low flows;
- artificially enhanced and stabilised low flows (compensation or augmentation flows);
- loss of high flow pulses (return period<1yr) or small floods (2-10 year events) (reservoirs);
- loss of large floods (>10yr events) (pumped storage reservoirs);
- extreme high or untimely discharge (regulation reservoirs or HEP); and
- rapidly changing flows (HEP).

2.5.2 Hydraulic parameters

2.5.2.1 Veclocity and depth pathways

The hydraulic state is the physical behaviour of water, and results from the interaction between hydrological changes (the quantity of water, or discharge) and the physical structure (morphology) of the river channel. Hydraulic interactions are well described in established texts (e.g. Chow, 1959), and can be modelled, and summarised, in a similar way to discharge.

Hydraulic interactions change the dimensions and connectivity of aquatic space, and the character and diversity of flow types within it. Thus, they have wide-ranging effects on conditions within the channel, riparian zone and the hyporheos. These complex responses are broadly categorised here into two hydraulic pathways: velocity and water depth.

Changes in velocity result in changes in shear stresses within the water column and at the bed, differentiating, for example, between slack or slow moving, low shear stress waters from higher energy, higher stress environments. This distinction is of demonstrated importance to both invertebrates and fish (eg Brooks et al. 2005, Statzner and Higler, 1986).

Changes in water depth result in changes in wetted width and wetted perimeter, and therefore the dimensions of water and bed space in the river channel. It also controls the hydraulic head (pressure) in the hyporheos.

As well as these primary or direct controls, velocity and depth have indirect or interacting hydraulic effects:

Firstly, the interaction of velocity and depth is an important control on the *type* of flow, both in the water column and at the bed. This is expressed hydraulically by turbulence (denoted by the Reynolds Number[1]) and flow intensity (denoted by the Froude number[2]). Both of these properties are important descriptors of meso-scale habitat type (e.g. riffle, run, glide and pool), and upon microhabitats within these: Jowett (1993), for example, identified consistent and scalable ranges of Froude number that correlated to habitat types; Brooks et al. (2005) demonstrated the value of shear stress, Froude number and roughness Reynolds number in defining benthic microhabitats. Heritage at al. (2009), further cite robust associations between the variety of physical conditions and biotic diversity. Thus, these hydraulic properties distinguish between slackwater and flowing waters (defining the lentic and lotic environments), and also differentiate between flowing water types (such as riffles and pools) and distinct 'patches' within them (such as riffle crests and riffle margins).

Secondly, water depth in particular is a control on the connectivity of aquatic habitat, both within the channel and between the channel, riparian and hyporheic zones. In this conceptual model, hydrological connectivity refers to the presence or absence of flow paths between persistent or temporary patches of aquatic habitat, and is an important property of the habitat state that affects the movement and dispersal of aquatic organisms in rivers (Larned et al. 2010), and exchanges of nutrients and energy between aquatic environments. Within the river channel, depth and its interaction with channel form controls the fragmentation and likages of aquatic space, and through this

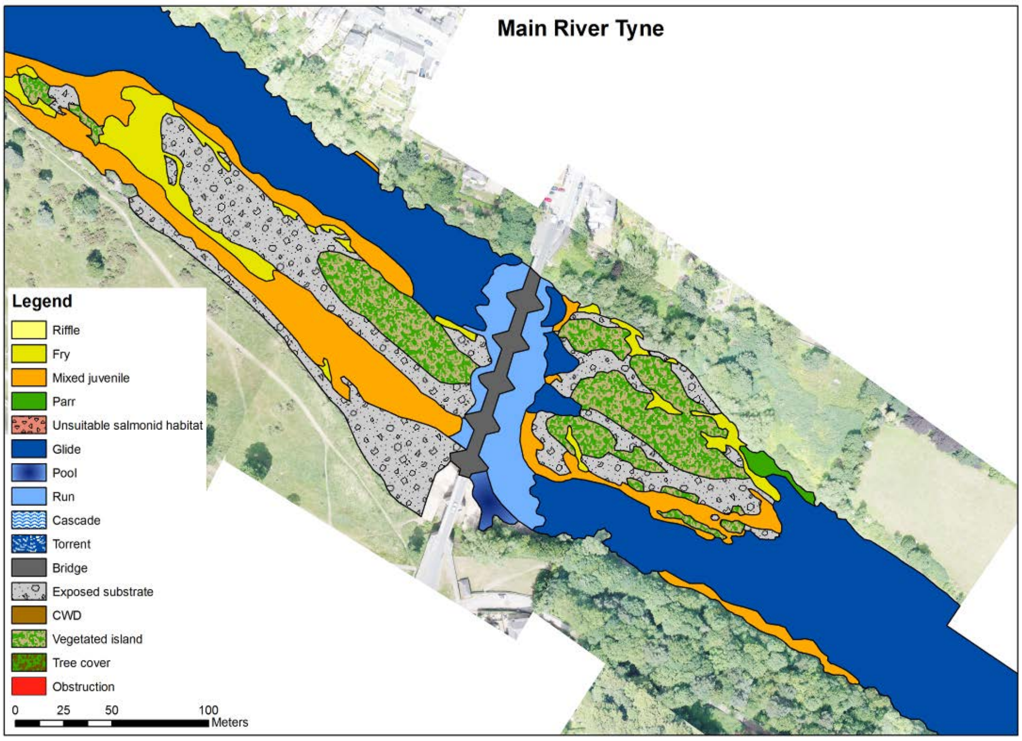
[1] The relative importance of inertial and viscous forces in the water.

[2] The relative importance of gravitational and inertial forces in the water, which can be envisaged as the forces pulling water down the channel and those resisting this movement.

the inundation and drying regimes of channel margins, channel features and of ephemeral channels. In the riparian zone, water depth controls the inundation regimes of marginal habitats and the riparian zone, establishing connectivity with floodplain and backwater habitats. Through its effect on the hydraulic head within the hyporheos, water depth also influences the rate and direction of water movement in the hyporheos, the moisture content of the riparian root zone and upwelling in (and through this, oxygenation of) bed substrate.

Importantly, because channel geometry is highly variable in space, hydraulic behaviour varies spatially much more than discharge does. Even if discharge is steady (i.e. does not vary with time), hydraulic behaviour varies laterally (across the channel), longitudinally (along a reach) and vertically (with depth), and it does so at a number of spatial scales. The effect is to create a mosaic of hydraulic conditions along the river reach (Figure 2.2). With changing (unsteady) discharge, this mosaic is also variable in time, and the static image shown in Figure 2.2 would become an animation.

Figure 2.2 - Example habitat mosaic. Note that in this example, combinations of depth and flow type have been used to define functional habitat for salmonid fish



2.5.2.2 Relative importance of depth and velocity pathways

The relative importance of the depth and velocity pathways varies between sites; for example, below bankfull, straight channels tend to speed up with increased discharge, whereas braided or meandering channels tend to spread out (i.e. increase their wetted perimeter) (Gordon et al. 2004).

The relative importance of these two hydraulic pathways also varies at different discharges at the same site, depending upon the discharge, the wetted cross sectional area, bed friction, water surface gradient.

In an even-sided, regular channel without downstream hydraulic controls, and with flow between the bed and bankfull, both depth and velocity pathways can be important. Reductions in discharge tend to cause quite regular (though not linear) rates of decrease in both velocity and depth. With these changes come progressive reductions in wetted width and wetted perimeter (although these tend to be relatively small per unit reduction in depth), loss of head within the hyporheos, and, often (but not universally), decreases in shear stress, flow intensity and turbulence.

Over average flows between bed level and bankfull, changes in hydraulic behaviour may, on average, be quite gradual. To conceptualise this as flow increases or decreases, an animation of Figure 2.2 might show only progressive expansions and contractions in wetted perimeter or marginal habitat, or changes in the proportions of riffles or pools. In many channels, it may only be with higher and lower flows that a tendency to more abrupt change might become evident, as important thresholds are crossed.

Importantly, however, even regular rectangular or trapezoidal channels have two important breakpoints in hydraulic behaviour; once flow exceeds bankfull, or is insufficient to achieve bed coverage, wetted perimeter and width change rapidly (Gippel and Stewardson, 1998). In some steep channels, velocity changes can be relatively minor as the bed becomes exposed, such that the river is effectively miniaturised, becoming smaller whilst the type of flow is maintained. Mainstone (2010) describes that the effects of low flows are often to miniaturise habitats before their character is ultimately changed as flow over riffles and runs is lost.

2.5.2.3 Hydraulic thresholds as indicators

The existence of breakpoints in hydraulic behaviour offers the prospect of defining points with which river managers can set flow standards to protect the environment (Acreman et al. 2009). This is important, because the need for consistent regulation across rivers has prompted an effort to define generalised hydraulic behaviour, and in particular a search for such threshold mechanisms.

The two most obvious breakpoints are at bankfull and the point at which the channel bed is wetted. Bankfull discharge governs the regime of riparian inundation. In humid climes, lowland alluvial rivers with simple channels and without morphological alterations (i.e. where the channel may be in an equilibrium condition), bank full discharge tends to be reached during small floods of return period c.1.5 years (Wolman and Miller, 1960). Bankfull discharge is not considered further as a seperate ecological indicator, but contributes to effects on riparian vegetation (Appendix I.22), and is a potentially useful guideline for use in the optimisation framework.

A breakpoint in the discharge – wetted perimeter relationship may delineate a threshold at which, *on average*, wetted width or perimeter decreases more rapidly per unit decrease in discharge. The diversity of flow types may also tend to homogenise with increases above it. Studies on regulated rivers in the UK for setting compensation releases from reservoirs (Environment Agency, 2009), demonstrated that increasing discharges between bed level and bankfull not only offered a lower rate of change in important hydraulic parameters, but also tended to reduce its diversity: once the full channel width was wetted, further increases in water level tended to homogenise hydraulic behaviour.

The breakpoint in the discharge – wetted perimeter relationship has been investigated in an attempt to derive general relationships on the basis of river typology to inform the WFD environmental standards for river discharge in the UK (SNIFFER, 2007; Acreman et al. 2009). It provides useful information about the spatial distribution of available habitat and is also a useful visual aid for demonstrating the impacts of various flow modification scenarios on the habitat state (Gordon et al. 2004); it is explored further in the Evidence Base (I.1) as of potential use both as an ecological indicator, and as a general guide for use in the optimisation framework. The conceptual model of Boulton (2003), which is described in Section 2.6.2, shows the ecological significance of water depth thresholds in rivers.

Flow type also offers potential as an ecological indicator, and is widely used in qualitative classification schemes, such as the River Habitat Survey (Environment Agency, 2003). Differences in hydraulic conditions in the water column and at the bed can manifest as differences in appearance at the water surface (e.g. Heritage at al. 2009). The potential use of surface flow type as an indicator of abstraction or impoundment impact is expanded upon in the Evidence Base (I.2).

2.5.2.4 Hydraulic complexity

Despite the potential for identifiable hydraulic thresholds, important properties of hydraulic behaviour complicate generic application and are therefore worth further emphasis here.

Firstly, the underlying hydraulic responses to flow changes are not linear with discharge. Many are only gradual or do not necessarily increase with discharge at all. As noted above, the diversity of flow types does not necessarily increase with discharge, and despite general tendencies, it cannot always be assumed that changes in discharge cause significant changes in hydraulic type, or that higher flows will result in an expansion of particular 'higher stress' flow types. With increasing discharge, riffles can become runs, and deep runs can become pools.

Secondly, in irregular channels the pattern of change in velocity, depth and associated hydraulic properties is more complex than the simple behaviour described above. Local irregularities at various scales - in channel cross or longitudinal section, or in substrate composition of form, create distinct and localised hydraulic behaviour. Abrupt changes in wetted space or flow type may therefore occur as irregularities in channels are inundated, banks overtopped into riparian land, or hydraulic jumps develop or are drowned out.

Thus, although the broad nature of hydraulic relationships is generally well understood, their complexity, and the control exerted by local morphology, means that characterising hydraulic behaviour requires comprehensive site-specific information. Generalised hydraulic geometry (e.g. Leopold and Maddock, 1953, Padmore, 1997, Booker and Dunbar, 2008) indicates that without such site-specific information, characterisation of hydraulic response to flow change is likely to be both uncertain, limited by typological distinctions (the catchment and geological setting of the channel Gilvear et al. 2002) and affected by the history (both short- and long-term) of hydrology, sediment input and morphological alteration. The problem of summarising hydrological variability in time therefore develops into one of summarising complex hydraulic variability in time and in space, and over multiple scales.

2.5.3 Geomorphological parameters

2.5.3.1 Geomorphological responses to hydraulic behaviour

Geomorphological state defines the response of channel morphology to hydraulic behaviour. The interaction of hydraulics with the channel boundary operates through the erosion of bed and banks, and through the entrainment, transport and deposition of sediment. These processes thereby direct the evolution of channel form, which is constantly adjusting to the prevailing flow and sediment conditions. As general rules:

- Periods of low and average flows tend to cause immobility or, if sediment is available, gradual accretion of finer material.

- Pulses of higher flows, during which shear stress (a function of water velocity) and stream power (a function of discharge) are competent to flush the fine material downstream (Acornley and Sear, 1999) occur seasonally, perhaps two or three times a year (King et al. 2008). These prevent clogging of the coarser matrix.

- Small floods, with a return period between one and five years (King et al. 2008), have the capacity to mobilise coarse sediment on the channel bed, breaking up any coarse surface (armour) layer, releasing finer subsurface sediment and replenishing sediment in bars and on riffles. Through their relative frequency and significant effect on sediment erosion, transport and deposition, small floods of between one and two year return period are also cited by King et al. (2008), as being the most important in maintaining channel form.

- Large floods have greatly increased erosive power and can cause catastrophic and longstanding effects on substrate and channel form, thereby having an important effect on channel morphology, despite their low frequency (e.g. Harvey, 2001).

Thus, low discharges are affected by channel form, bedform and substrate composition, and at high discharges the reverse is often true. Inevitably, there results a close correspondence between prevailing hydraulic conditions in the watercolumn and substrate/ channel form of the benthos. For example, in the riffle and pool sequences that are a common bedform in gravel and cobble bedded rivers with active sediment transport regimes, the high velocities and shear stresses that prevail in riffles and runs are associated with coarser substrate (and because of this, better oxygenation of pore spaces (Greig et al. 2007)). In intervening slower flowing glides and particularly in pool habitats, finer sediment is more prevalent (Hirsch and Abrahams, 1981). The distinct physical structures and associated hydraulic properties (velocity, depth, shear stress, flow intensity, turbulence, etc.), and morphological properties (slope, substrate, etc.) create 'physical biotopes'[3], the physical framework for characteristic ecological structures that utilise them. ('functional habitats').

2.5.3.2 Geomorphological integration and feedback

Geomorphological processes are highly non-linear to the hydraulic forcing variables (themselves non-linear to discharge). They are also disproportionately influenced by infrequent disturbance events, and are further complicated by interactions with biological responses, particularly macrophyte growth (e.g. Hatton-Ellis, Greive and Newman, 2003). Threshold mechanisms are important (e.g. Hjulstrom, 1939), uncertain in calculation, difficult to translate to reach scale behaviour and complicated by mixed or structured substrate. King et al. (2008), for example, demonstrate the large uncertainties in using three alternative and widely used equations (Mannings, Darcy Weisbach and Lacey) for calculating the critical particle entrainment velocity in different envirnments, and these difficulaties at a particle scale can translate to difficulties in determining the aggregate response.

Geomorphological interactions also occur over a range of timescales, from single floods to decades or centuries, such that natural channels are continually evolving to achieve equilibrium with the changing hydraulic regime. Bed composition and structure therefore integrate over a range of preceding hydraulic conditions. Importantly, channel form, bedform and substrate character also have a feedback to hydraulic behaviour, strongly influencing hydraulic conditions at the bed.

Thus, although the nature of geomorphological interactions is well established and described in geomorphological texts (e.g. Thorne, 1997), the linking of hydraulic conditions to geomorphological response is not easily quantified. Moreover, hydraulic changes observed in the short-term may not be representative of those in the longer-term, and the effect of changes in discharge cannot necessarily be rectified simply by reversing the alterations to the discharge regime.

2.5.3.3 Substrate and channel form as indicators

The integrative nature of geomorphological changes, and the importance of low frequency events, adds the need for a longer-term perspective of landscape evolution to the complexities of temporal and spatial variability of hydrology and hydraulics. However, they also stand in contrast to the more rapidly varying hydrological and hydraulic conditions. Accepting that the physical presence of dams and weirs also affect substrate and channel form because of the reduced rates of sediment input, this intergrative property makes them of potential use as indicators of flow effects.

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[3] In river management terms, 'biotopes' define physical habitat structures (riffle, pool etc.), which have associated hydraulic properties (velocity, depth, shear stress, flow intensity, turbulence, etc.), morphological properties (slope, substrate, etc.), and characteristic ecological structures.

Prevailing low flows or lack of flushing, if combined with a supply of fine sediment (sand, silt and clay), may result in accumulation of a fine surface sediment layer and/or a higher proportion of fine sediment within the interstices of river bed gravels (Sear, 1995; Wood and Armitage, 1997; Acornley and Sear, 1999). While accumulation of fine sediment in the channel bed is a spatially and temporally heterogeneous phenomenon, it is interrupted in a natural system by disruptive events; prolonged accumulation of fine sediment within the gravels of typically high energy bedforms (such as riffles) would indicate excessive fine sediment deposition.

Where flows are sufficient to transport only finer gravels (as where, for example, spate flows are common but high flows are truncated), fines may be absent from a surface layer composed entirely of interlocking coarser material (bed armouring or paving). Bed armour periodically breaks up during high magnitude flows (Vericat et al. 2006, 2008), providing a supply of sediment to downstream reaches and maintaining a loose bed structure, but where high magnitude flows do not occur, armouring and stabilisation of the bed will become extreme (e.g. Sear, 1995).

Over the longer term (decades), an imbalance between the flow regime and the sediment supply may result in changes to channel form at larger scales. Should sediment transport become inactive, riffles may become degraded, while pools may become aggraded (Sear, 1995), altering the spacing of these features, or stagnating the riffle-pool sequence, which may maintain flow diversity but reduces its usefulness as spawning habitat.

Channel widening or narrowing may occur, depending on the nature of the changes to flow and sediment supply. These will result in changes in the width to depth ratio in the channel, with possible formation of terraces and lowering of base level (Brandt, 2000). Where there are reduced high flows and a corresponding lack of sediment transport, channel stabilisation and narrowing typically occur, with bars becoming vegetated and developing into benches (e.g. Sear, 1995; Gilvear, 2004). Reduced sediment transport capacity in a regulated channel may also be manifest by the deposition of bars downstream of unregulated tributary confluences (Curtis et al. 2010), and where flows in the main channel do not have the competence to mobilise these deposits, aggradation and narrowing will occur (e.g. Gilvear, 2004). Other long-term changes may include alteration of the flow type within the channel, such as through degradation of riffles and aggradation of pools (Sear, 1995).

These potential indicators are expanded upon in the Evidence Base (I.3to I.8).

2.5.4 Habitat state

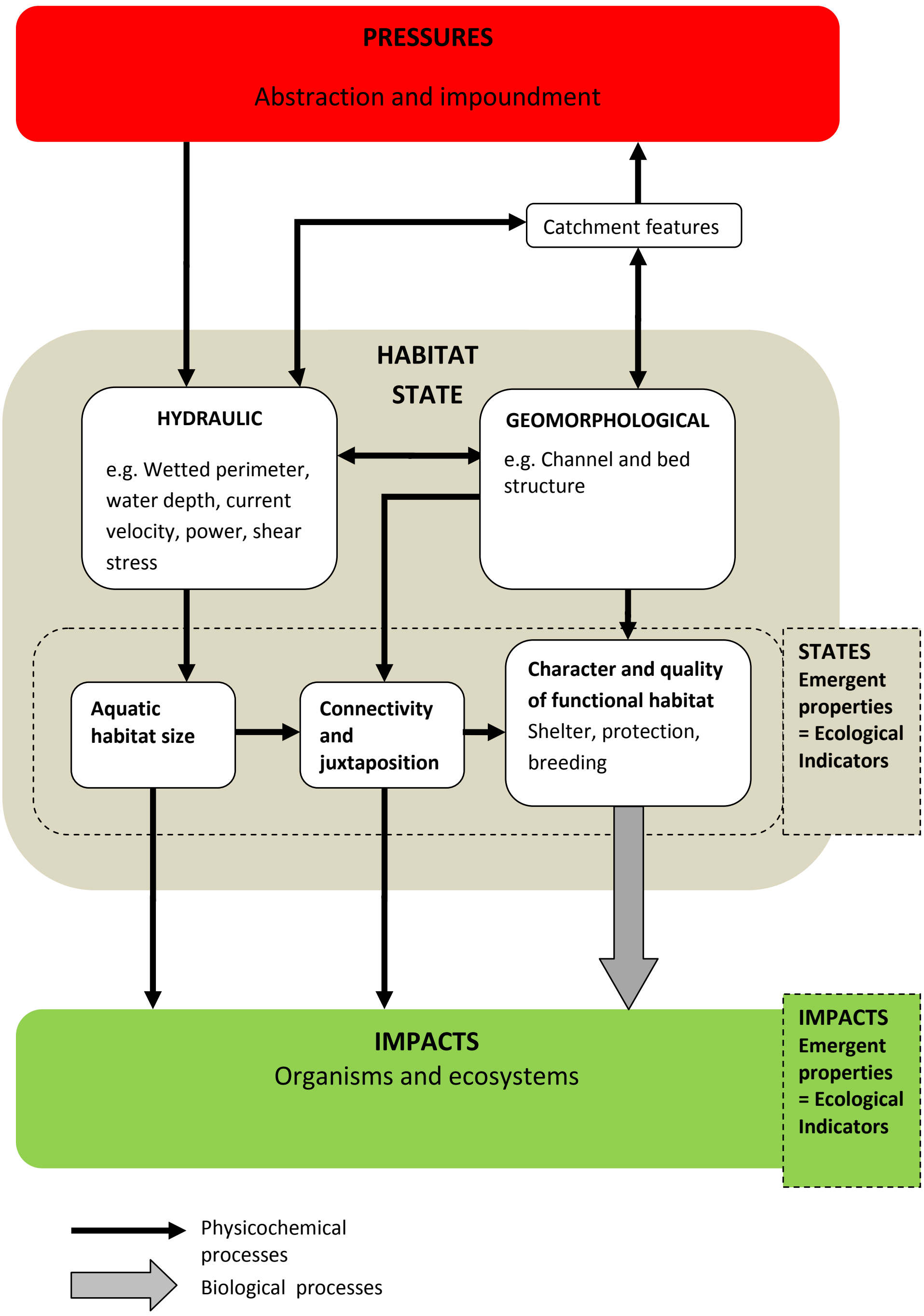
2.5.4.1 Habitat: an emergent property of physico-chemical condition

The habitat state integrates the hydrological, hydraulic and geomorphological effects described above, with other physico-chemical parameters (thermal and water quality parameters) not included in this conceptual model (Figure 2.1).

The interacting physical parameters that create the habitat state in rivers have emergent properties, which have major proximate influences on the biota such as: shelter and protection, productivity, resilience, stability, juxtaposition, and connectivity between habitats. These variables together with features of adjacent riparian zones and land, forms the habitat landscape occupied by the organisms and their ecosystems. Thus the habitat state supports the aquatic biota, providing the physical framework to support aquatic ecosystems, and from this derives the importance of timing flows with catchment inputs.

In current WFD terms, the habitat state is referred to as the hydromorphology supporting condition of the biological quality elements that determine the target of good ecological status in water bodies.

Figure 2.3 – The role of habitat state linking abstraction and impoundment pressures to ecological impacts



2.5.4.2 Properties of the habitat state

Hydraulic and morphological changes equate to alterations in habitat distribution and character, providing a link between flow alteration and ecological response. Because river channel landscape has both local character (quality) and dimensions (space, occupied by the organisms and plants), and because organisms are adapted to their habitat, these properties have significant roles in determining biological impacts. Relevant hydraulic and morphological changes and features can therefore be used as indications that flow alteration has had an impact on the geomorphology of the channel, with the expectation that this will have subsequent effects on aquatic ecology.

During the course of their life cycles organisms may depend upon many types of habitat either adjacent or far apart. Therefore, properties such as juxtaposition and connectivity and diversity of habitat patches or mesohabitats must also be important.

Thus, the properties of the habitat state described in this conceptual model are:

- size of the habitat (area/volume of aquatic habitat space);
- connectivity and juxtaposition of habitat; and
- character and diversity of the habitat (ecological 'quality' of the habitat).

Important considerations for ecological indicators

Understanding these properties of the habitat state is fundamental to the identification of ecological indicators and to the development of ecological tools for river flow management. For example, the standard macroinvertebrate biological assessment and classification methods for rivers in the UK are designed to standardise the effects of habitat size (space) to enable comparisons of the character (quality) of the aquatic habitat across sites (e.g. Wright, Sutcliffe and Furse, 2000). This and other standard biological assessment methods are therefore insensitive to the effects of changing aquatic habitat size (Armitage and Pardo, 1995; Armitage and Cannan, 1998), resulting in incomplete or uncertain estimates of the impacts of hydromorphological pressures (Mainstone, 2010). Mainstone (2010) describes that the effects of low flows are often to miniaturise habitats before their character is changed and suggests that standard sampling methods that are designed for water quality assessments provide incomplete information about the biological effects of reduced discharge in rivers. An important consideration of the ecological indicators is to describe major and severe impacts of flow alteration through effects on habitat size, character, connectivity and juxtaposition.

The importance of scale

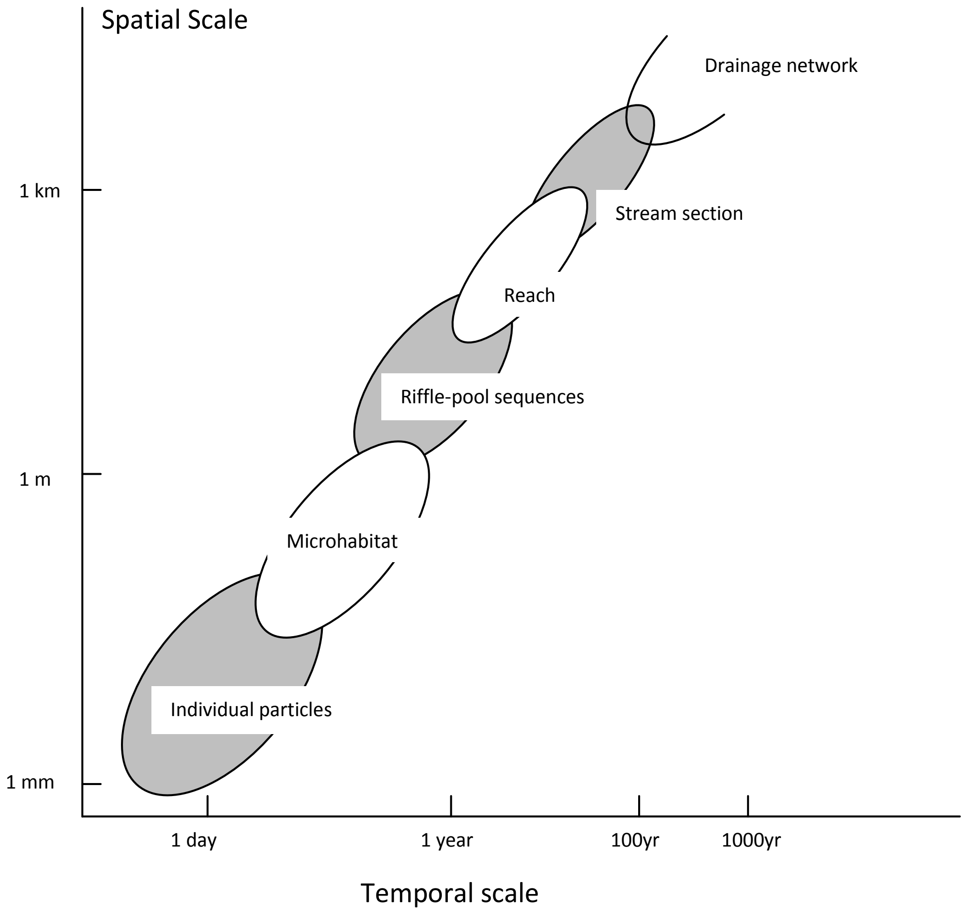
The above properties can be defined at a range of scales in both space and time. River channel landscape is dynamic in both time and space (Frissell et al. 1986, Allan, 1996; Poff et al. 1997; Folt et al. 1999; Maddock, 1999; Woodward and Hildrew, 2002; Durance et al. 2006) (Figure 2.4), and the result is a changing mosaic of habitat structure and variation in impacts at all trophic levels (Woodward and Hildrew, 2002; Durance et al. 2006), and over multiple scales (Table 2.3). These scale effects arise because of the interaction of discharge, which is highly time-variant, with channel geometry and geomorphic variables, which vary greatly laterally across channels and longitudinally from source to sea.

Each scale (Table 2.3) displays a different relationship with flow variation, because the processes governing states and organism life cycle responses are different at different scales (e.g. Folt et al. 1999; Maddock, 1999; Woodward and Hildrew, 2002; Durance et al. 2006). Thus, matching indicators and measures for river flow management with scales becomes vitally important. The spatial and temporal dynamism of rivers, described above, must be taken into consideration before any relationships among state parameters have any ecological meaning or relevance in deriving environmental standards or river management tools. This is an area of concern among many river scientists regarding the applicability of hydraulic-habitat models (such as the PHABSIM approach, Bovee, 1982; Bovee et al. 1998) to ecologically-based river management (e.g. Orth, 1987; Gore and Nestler, 1988; Stewardson and Gippel, 2003).

Table 2.3 - River unit terminologies and contrasting response times

Spatial scale/system level	Definitions adapted from Maddock (1999) and others	Response rate / time scale (of change)
Microhabitat	zones of varying depth, velocity, substrate, cover immediately influencing behaviour, protection and feeding	high (hrs-days)
Mesohabitat	=hydromorphological units = biotopes =ecological unit, eg riffle, run glides, pools, gravel bars	High (days-10^1 yrs)
Macrohabitat	e.g. riffle-pool sequence. Functional combinations of hydromorphological unit, "offering full set of habitat to complete life cycle for resident species" (Maddock,1999)	High/Mod
Reach	River section comprising broadly similar land form and channel gradients	Mod (10^2 yrs)
Segment	=sector (Maddock 1999)? Characterised by same "river type", or channel type and = "landscape"	Low (10^3-10^4 yrs)
Stream system = sub-catchment = tributary	part of the catchment area characterised by different river types	Low
catchment = drainage basin	Discrete drainage area draining to the sea	Low (10^5 – 10^6 yrs)

Figure 2.4 - Approximate spatial and temporal scales over which physical change takes place in rivers (after Allan 1996 and Frissell et al. 1986)



2.6 Impacts

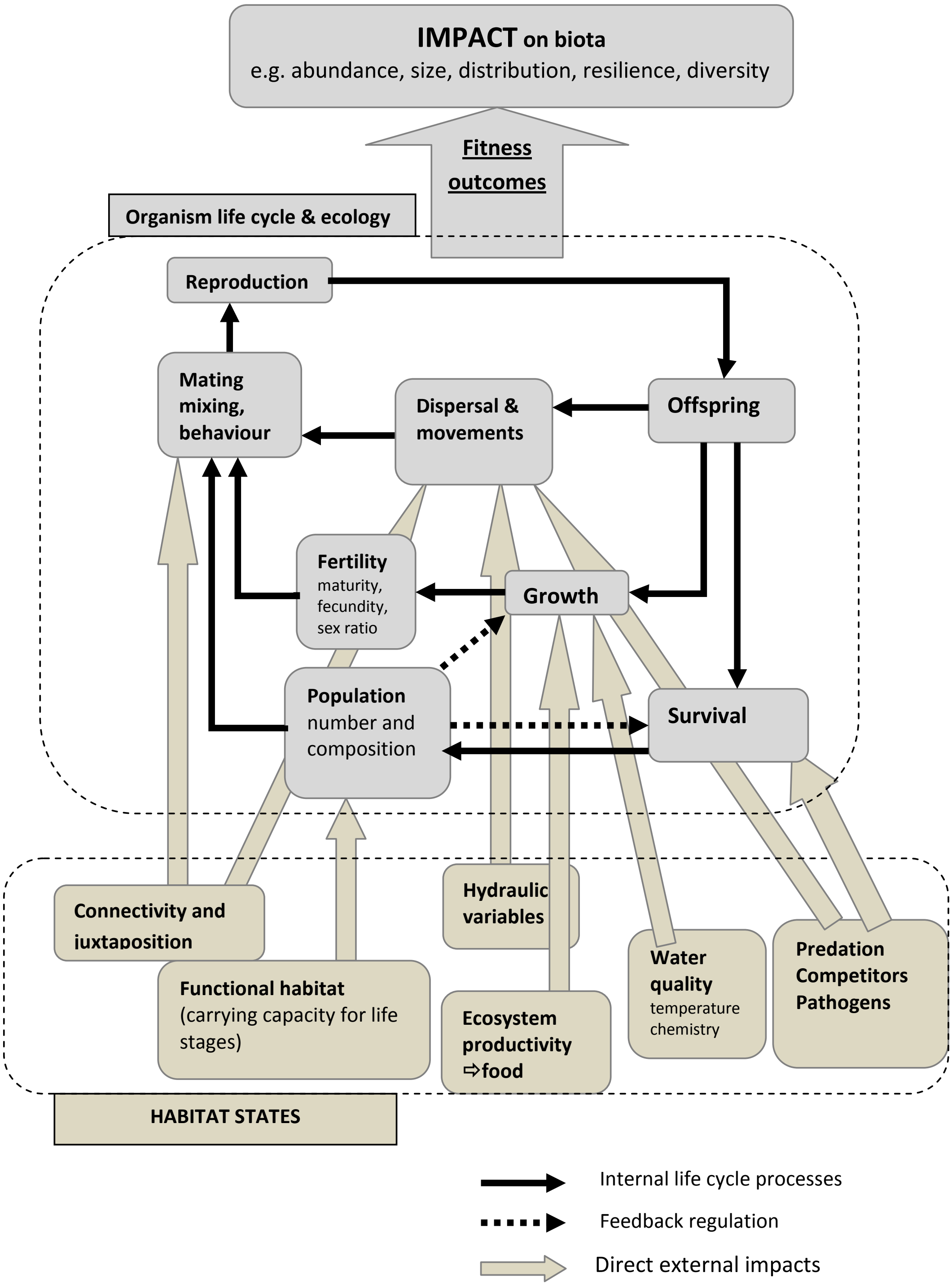
2.6.1 Biological and ecological responses

Organisms and ecosystems do not generally respond to discharge directly, but to the resulting hydraulic and geomorphological state variables that combine to create the habitat state. That habitat state and its emergent properties influence aquatic biota and their reticulate connections with ecosystem components is a cornerstone of freshwater ecology and there is a huge literature outlining models relating biota to flow-related habitat states (e.g. Fausch et al. 1995; Orth, 1995; Baglinière and Maisse, 1999).

Impacts arise through the responses of organisms and these can be defined at several level of organisation, from the responses of individual organisms, through species populations and communities to ecosystem functions. The responses of these are briefly outlined through a process conceptual model of fitness-based life cycles (Figure 2.5).

The details of fitness estimation are not relevant in this conceptual model; however, the life cycle processes are, because it is through them that environmental states operate via direct or reticulate processes and some of the basic impact routes are shown in Figure 2.5. All organisms have life cycles that achieve three main outcomes: survival, reproduction and dispersal. The latter is identified separately here to emphasise its functional significance in relation to flow, but can be subsumed within survival and reproduction. These attributes combine to determine the life time fitness of organisms, which is their ability to transmit genes to subsequent generations and thus contribute to survival of populations and the species.

Figure 2.5 - Schematic of how states exert impacts on aquatic biota through life cycle responses and evolutionary fitness



States exert their impacts through the links shown by arrows in Figure 2.5, which shows one indicative life cycle, but this is repeated in multiple forms for other populations and taxa combinations, each having their own feedback and control mechanisms, coupled through community and ecosystem webs. There are numerous feedbacks and regulatory processes many of which are poorly understood. Needless to say the complexity of all the interactions is vast and the aim in this report is to focus down to main links and processes for which water flow has particular importance.

The combined effects of habitat states, fitness across taxa and their many interactions give rise to community and ecosystem responses. In contrast to organisms (species and populations), ecosystem (including communities) responses, while intimately involved with other levels of biological organisation, do not display fitness characteristics in the same way. They have no genome and are not, as far as we know, evolving entities, so do not show the same fitness adaptations or proximate functional responses to flow (or other environmental factors), although they are influenced by flow acting on their component processes. However, they are simultaneously the matrix within which fitness of species operates and the support system for the component organisms. Therefore, the biota is adapted at individual, community and ecosystem levels to physical states over different spatial and temporal scales.

The habitat state in rivers is dynamic, mediated by temporal changes in river flows. The processes described above that give rise to the observed biological impacts do not occur in a steady state. Most aquatic species possess life history traits that enable individuals to survive and reproduce within a certain range of environment variability (Townsend and Hildrew, 1994). This forms the basis of well-documented concepts of disturbance in river ecology which emphasise that disturbance mediated by flow variability is a critical driver for ecosystem function (e.g. Death and Winterbourn, 1995; Clausen and Biggs, 1997; Townsend and Scarsbrook 1997). The organisation and function of river communities is partly the product of the interaction between abiotic disturbance (the magnitude, duration, frequency, timing and rate of change of high and low flows) and the rate of competitive exclusion. At high levels of flow disturbance, organisms are controlled by their tolerances to abiotic conditions; at low levels of flow disturbance, biotic controls (competition and predation) become more important. The highest total densities and diversity of aquatic macroinvertebrates, for example, often occurs at intermediate levels of flow disturbance (e.g. Death and Winterbourn, 1995; Clausen and Biggs, 1997; Townsend and Scarsbrook, 1997).

These concepts are fundamental to river managers charged with controlling water releases from impoundments to optimise ecological benefit in downstream reaches and will be revisited in Section 4 (Flow optimisation framework). The concepts of abiotic and biotic controls on aquatic organisms under different habitat states are important in identifying ecological indicators of severe impacts of flow alteration and these are described in more detail in the following section.

2.6.2 Impacts at thresholds

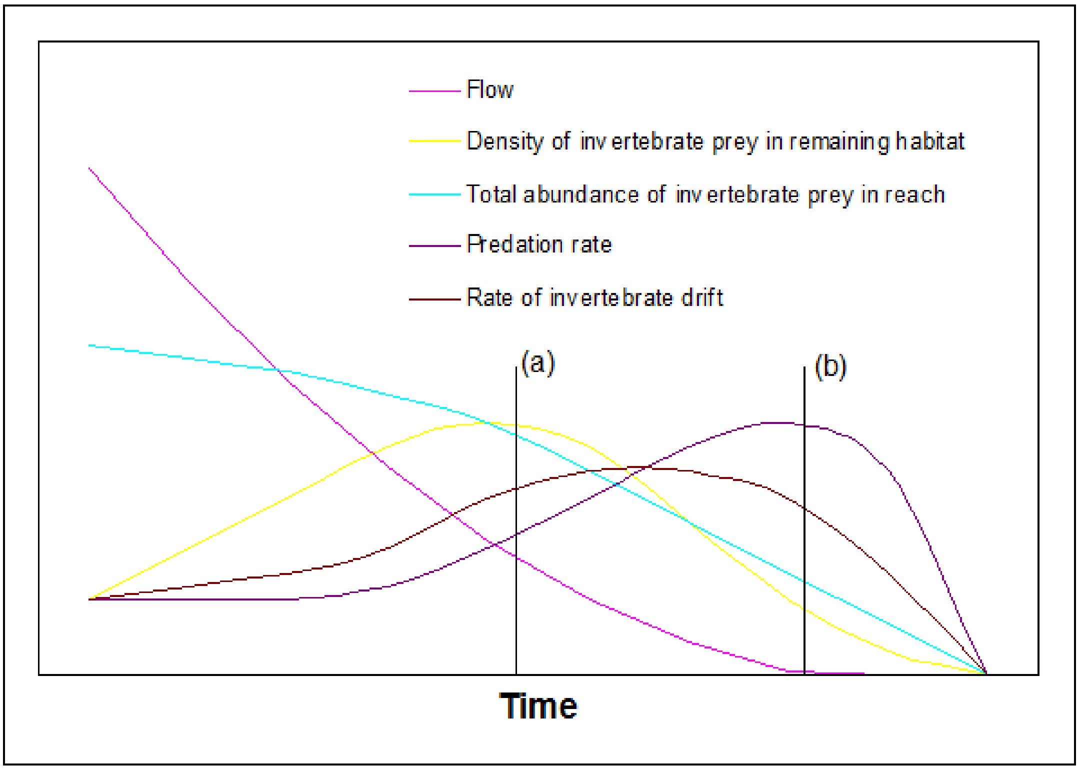
Links between habitat state and biotic response have been described in Section 2.5.4**Error! Reference source not found.**, and relevant individual state-impact links are further described in the evidence base. However, selected biotic responses at potentially important habitat thresholds are offered below to complement the more generic process-description above. The examples illustrate the effect of drought on macroinvertebrates but might easily be translated to effects due to abstraction or impoundment, and to other ecological elements. These examples also highlight the importance of biotic interations in determining the observed ecological impacts to severe reductions of water level in rivers – a point that existing biological assessment and classification tools do not consider. In this way these conceptualisations are useful in working towards the indentification of ecological indicators of severe river flow alterations.

Mainstone (2010) presents a simple conceptual model of the biotic interactions that occur among riverine macroinvertebrates in response to decreasing habitat space due to declining discharge in rivers:

- As river flow decreases, individuals of all aquatic organisms are initially concentrated into smaller habitats, causing an initial increase in density (apparent abundance when using standard sampling methods) (Wright and Berrie, 1987; Suren and Jowett, 2006).
- Predation rate, density dependent mortality and drift rate increases (McIntosh et al. 2002; Peckarsky et al. 1990), causing a decrease in the density of prey species relative to predator species.

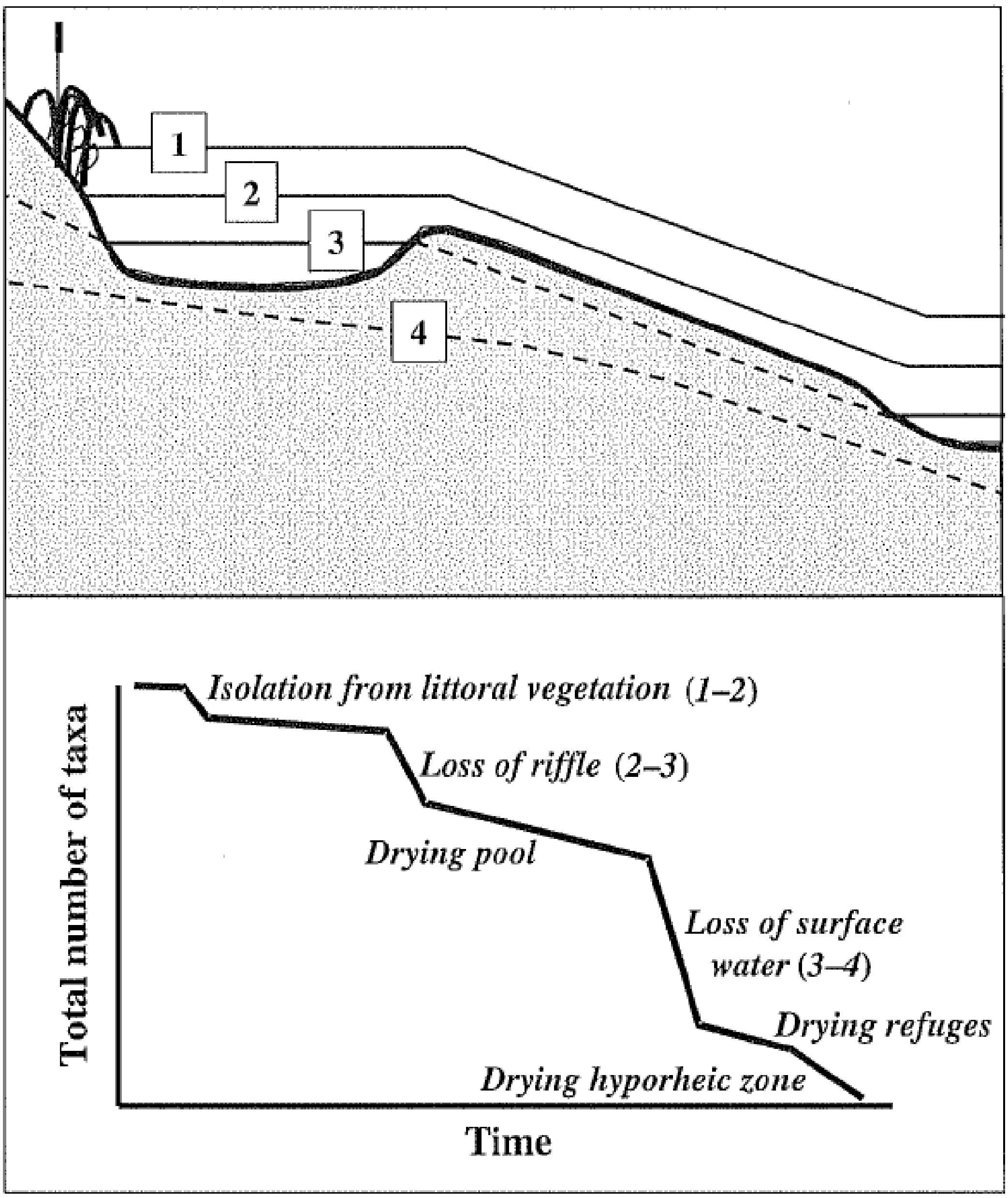
Boulton and Lake (1992) report that under conditions of extremely low flows in Australian rivers, typical lentic invertebrate taxa that have the ability to respire aerially colonise river reaches. Most of these taxa are large-bodied predators, such as dytiscids (diving beetles), notonectids (backswimmers), corixids (water boatmen) and amphipods (shrimps). Observations from rivers in the UK that have severe and chronic low flows downstream of abstractions and impoundments indicate that gerrids (water skaters) and odonate nymphs (dragonflies) are often abundant where flow has stopped or is barely perceptible (D. Bradley, Pers. Obs). Chironomids that use haemoglobin to enhance oxygen uptake are also characteristic of hypoxic conditions that often occurs in rivers just before flow ceases (Boulton and Lake, 1992).

Figure 2.6 - Conceptualised macroinvertebrate responses to summer flow recession during drought. Routine observation at time (a) – high invertebrate prey density, low predation rate; routine observation at time (b) - low prey density, high predation rate (reproduced from Mainstone, 2010)



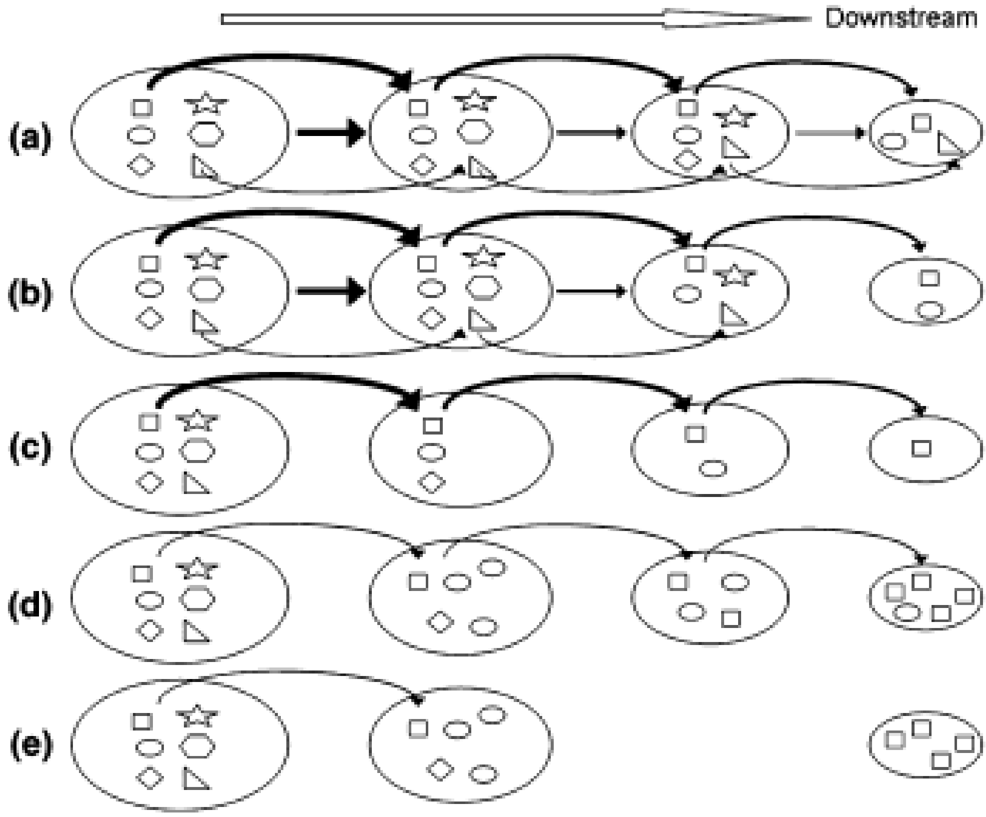
Boulton (2003) presents a further conceptualisation of how aquatic macroinvertebrates respond to the abiotic controls and character of habitat state as river water levels fall during drought (Figure 2.7). This model suggests that macroinvertebrate assemblage composition changes in a 'stepped' fashion as water levels cross thresholds (indicated by numbers in Figure 2.7). As water levels decline, total numbers of taxa are expected to decline sharply when submerged or trailing vegetation is isolated from the open water (step 1-2), then as water levels fall below the riffle and wetted reaches become fragmented between isolated pools (step 2-3), then when surface water disappears (step 3-4). This model is described at the reach scale, but can be translated across larger spatial scales

Figure 2.7 – A conceptual model of 'stepped' changes in macroinvertebrate assemblage composition in response to declining water levels in a river (Source: Boulton, 2003)



The concept of metacommunities has also been used to describe the ecological impacts of hydrological discontinuity at multiple scales in rivers (Larned et al. 2010). This model suggests that fish and aquatic invertebrate communities are generally depauperate in disconnected aquatic habitats compared to connected aquatic habitats in rivers, and that disconnected aquatic communities are often sub-sets of connected aquatic communities (Meyerhoff and Lind, 1987; Taylor and Warren, 2001; Bonada et al. 2006). The metacommunities concept acknowledges that rivers are frequently disturbed environments and the duration of connection and disconnection of aquatic habitats determines the structure of the ecological communities.

Figure 2.8 - A conceptual model of metacommunity structure in longitudinal arrays of aquatic habitat in rivers. a) connected habitats; b) partially connected habitats; c) recently isolated habitats; d) long-isolated habitats; e) long-isolated habitats with complete drying. Shapes within habitats represent different types of aquatic taxa: squares, ovals and stars represent taxa with an aerial phase; squares and ovals represent taxa that fly along wetted or dry channels, stars represent taxa that that only fly along wetted channels; squares represent the most desiccation-resistant taxa. Only downstream dispersal is shown. Source: Larned et al. (2010)



Scale is a central issue in freshwater connectivity because freshwaters are spatially structured in a scale-dependent fashion. Scale affects the way different processes operate and can be detected, and organism body size determines the distances that organisms need to move between aquatic habitat patches (Ormerod et al. 2011). For example, organisms with larger body sizes (e.g. Atlantic salmon) occupy larger home ranges than those with smaller body sizes (e.g. aquatic invertebrates), although dispersal can involve long distances with species that possess small, vagile propagules (Ormerod et al. 2011).

Related to connectivity is the concept that once a threshold of water level is reached in a river channel, further increases in water level cause the habitat state to homogenise. Larned et al. (2010) extended this conceptual model to the response of aquatic organisms and suggested that river-wide taxon richness initially increases with water level as wetting of a dry channel initially causes an increase in the abundance of aquatic habitat patches. At some point before bankfull water level is reached, however, taxon richness starts to decrease as the aquatic habitat is homogenised as patches coalesce (van der

Nat et al. 2002; Larned et al. 2010), so reducing the diversity of functional habitats for aquatic organisms.

This model is consistent with the conclusion of experimental studies (Section 2.5.2.3), which, in contrasting compensation flows in the Rivers Loxley and Rivelin (Environment Agency, 2009) demonstrated that compensation flows set too high had a negative effect on macroinvertebrate communities in downstream river reaches by homogenising habitats. The concept that high and stable compensation flows are not optimal for riverine organisms has important implications for a framework for optimising water releases from river impoundments.

Conceptually equivalent mechanisms can also apply at higher flows and at different levels of organisation; thus serving to define both species preferences and the conditions necessary for biotopes. Figure 2.9 and Figure 2.10 presents such a model, describing the dependence of riparian wetlands upon habitat state, with thresholds used both to define the riparian wetland and to classify the community structure within it.

Figure 2.9 - Schematic to show wetland characteristics

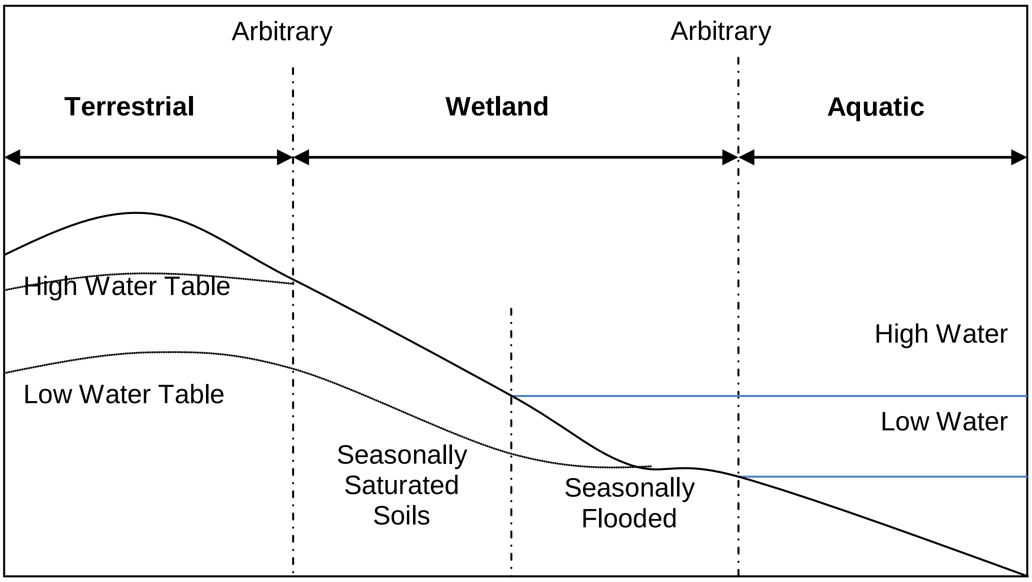


Figure 2.10 - Water table depth zones for MG13

Seasons and Variable	Tolerable Amber Range	Not Tolerable beyond Red limit
Spring (Mar-May)		
A Mean Water Table Depth (maximum) /m	0.3 - 0.55	0.55
B Mean Water Table Depth (minimum) /m	0.3 - ?	-
C Max duration of surface water flooding episode covering >10% of ar	-	-
D Cumulative duration of flooding during season/days	-	-
Summer (June-Aug)		
A Mean Water Table Depth (maximum) /m	0.8 - ?	0.55
B Mean Water Table Depth (minimum) /m	0.3 - 0.1	0.1
C Max duration of surface water flooding episode covering >10% of ar	8 - 20	20
D Cumulative duration of flooding during season/days	30 - 60	60

Figure 1.3 Variables for MG13 Grassland

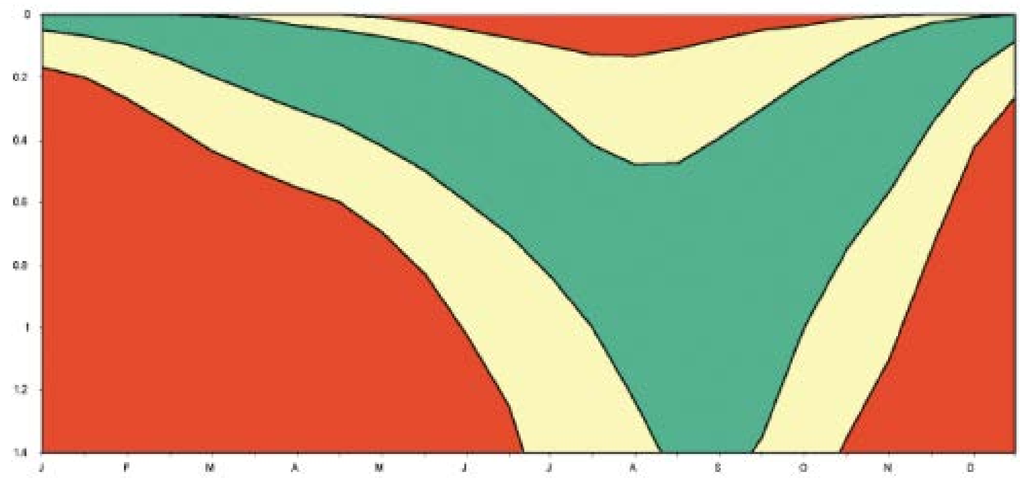


Figure 1.4 Water Table Depth Zones for MG13 Grassland
Water Table zones, illustrating the range of depths that are "desirable" (green) and "tolerable for limited periods" (amber). Values are based on the mean of at least three readings, taken at least 7 days apart, but all within a four-week period.

2.6.3 Properties of biological responses

The potential existence of thresholds of biotic response appears to offer the prospect of defining break points and standards to protect the environment. However, the existence of thresholds may in part be artefacts of categorisation and of scale; arbitariness is, for example, evident in the conceptaulisation of what is in fact a continuum of wetland character (199). Moreover, thresholds evident at one scale may not be so at another. For example, Gippel and Stewardson (1998) demonstrate that breakpoints in the discharge-wetted perimeter relationship evident at transects even out when many transects are considered across a reach (140). In practice, there is a continuum of ecological responses to flows, with substantial variation between taxa, within species according to life stages and season, between river and mesohabitat types and along rivers. This reflects varying combinations of genuine environmental differences, genotypic and phenotypic variation and the effects of confounding factors (e.g. Dudgeon et al. 2006; Vaughan et al. 2009).

Further features of biological responses also preclude a mechanistic application of habitat measures. In some cases it is the emergent state properties (e.g. food production, habitat connectivity and juxtaposition) that have dominant effects rather than just the individual state variables such as hydraulics. The limited or inconsistent role of hydraulic variables observed in most models may reflect the flexible use that many organisms can make of habitat resources. Clearly this varies with species and life stage, but this is one reason why the simple mechanistic cause-effect routes implied in the conceptual model cannot capture all the complexity of the biological systems.

Impacts through biotic responses can also only be properly assessed in relation to appropriate temporal and spatial scales. As well as states varying in that way, the wider habitat landscape is essential to maintain the life cycles and thus species, communities and ecosystems, and the spatial habitat requirements of co-existing organisms are often very different. Problematically, fitness of many larger organisms is often manifest at larger spatial scales than are conventionally studied for flow-impact purposes (Fausch et al. 2002; Durance et al. 2006). An extreme example is migratory Atlantic salmon which uses the whole river system and a good part of the North Atlantic. More modest spatial demands may be made by macrophytes and macroinvertebrates, but even these usually require upstream sources of propagules, delivered by for example upstream flight of adult insects.

The timescales relevant to different indicators also vary greatly. Some invertebrates complete their life cycle in days whilst fish may take several years. Lags in response to hydrological events, of which abstraction or impoundment pressures may cause or contribute to, arise from these lifecycle requirements that are highly variable between species, and may propagate up trophic levels via foodwebs etc in complex ways. Through these mechanisms, intermittent loss or severe reductions in available habitat during extreme conditions, such as drought or floods, may have disproportionate and longstanding effects on biota. Such pressures are natural phenomena and, at normal frequencies, may serve both to provide high selective pressure, which maintains fitness, and to eliminate invasions by species that are in marginally suitable habitats on the limits of their natural tolerance (e.g. Fausch et al. 2001). For example, loss of such peaks has been shown to enable colonisation by some non-native fish species (Marchetti and Moyle, 2001). Artificial floods have also been shown to lead to invertebrate impacts in species that are otherwise adapted to cope with the natural timing of rainfall and natural floods (Death, 2010).

2.6.4 Biological indicators

Biological responses are the terms in which impacts are defined, and are therefore the most appropriate ecological indicators determine and the most appropriate measures with which to optimise flow releases.

Candidate ecological indicators are therefore drawn predominatly from biotic measures. These are most easily defined in terms of species responses and are described in detail in the evidence base (Appendix I). Two common measures of fitness are life time egg deposition and intrinsic rate of population increase (e.g. Stearns, 1992; Gotelli, 2008). But these are difficult to measure and partial surrogates (variables and traits contributory to fitness, such as abundance, size etc) have to be used. Indicators that have ecological meaning and interpretive value should reflect the various outcomes of fitness (Figure 2.5) at individual level and these species level processes, acting within populations, combine to give responses of communities and ecosystems.

Inevitably, the longitudinal and lateral zonation of habitats results in biological zonation (e.g. Huet, 1959; Hynes, 1979) that must be accounted for in using biotic measures. Fortunately, these broad zonations are more or less predictable from habitat and catchment factors. For example, fish, such as roach and bream, living in lowland reaches are morphologically adapted to slow, deep water and to the prey items and their habitat. The zonations of biota also contribute to the longitudinal patterns down river networks in trophic web complexity, stability and dynamics that have consequences (not at this stage always understood or predictable) for responses to disturbances such as artificial flow modifications (Power and Dietrich, 2002; Woodward and Hildrew, 2002).

2.6.5 Ecosystem indicators

Ecosystem emergent properties also offer potential as indicators. Ecosystems have emergent properties, such as carbon sequestration, nutrient breakdown and cycling, biodiversity, stability and resilience; and these can be dramatically disturbed or eliminated by flow regime changes. Several of these are delivered through trophic webs and dynamics (Woodward and Hildrew, 2002) which may adjust in response to flow impacts and other disturbances (Orth, 1995; Death 2010). Moreover, ecosystems deliver the ecosystem services that riverine environmental quality supports (UK National Ecosystem Assessment, 2011) and are thus relevant to contemporary flow management.

To date, measures of ecosystem function of relevance to detecting the impacts of environmental stressors in rivers have focussed mostly on the decomposition of leaf-litter (see review by Friberg et al. 2011). Despite a large and rapidly growing body of literature showing a general increase in microbial activity relative to invertebrate-mediated decomposition of leaves at sites with altered riparian vegetation, higher temperatures and lower pH (see review by Friberg et al. 2011); usable bioassays are still in their infancy.

2.7 Recent scientific work quantifying ecological low flow requirements

The hydrological regime of rivers in the UK is regulated using environmental standards (termed Environmental Flow Indicators (EFIs) in England and Wales) that are considered appropriate to support GES (UKTAG, 2008a), and condition limits for managed flows (UKTAG, 2008b). Environmental standards for river flows prescribe the maximum abstraction allowable under different flow conditions in spring/summer and autumn/winter periods. Condition limits for managed flows prescribe the maximum deviation from natural flows allowable downstream of impoundments, under different flow conditions. UKTAG have recently reviewed the environmental standards for river flows at medium and high flows for Poor and Bad status (Gosling, 2012). This section of the report presents a summary of recent scientific work on the low flow requirements of riverine biota that has been undertaken since the existing environmental standards and condition limits were established in 2007 (Table 2.4). Section 2.8.2 describes the implications of this work to the management and regulation of river flows.

Table 2.4 - Recent scientific work quantifying ecological low flow requirements

Study	Author	Minimum flow values	UKTAG river type	UKTAG Q_n95 low flow standards or Q_n95 condition limits	Comments
Environment Agency North West Region Hands-off Flows for SAC rivers (2005)	Atkins Global (2005)	Cawdale Beck: Q_n89; Heltondale Beck: Q_n75; Swindale Beck: Q_n79	C2	7.5% (April-October) and 10% (November - March) deviation from Q_n95	Data supplied by Jane Atkins, Environment Agency at the Expert Workshop. Precautionary flow targets to protect the habitat requirements of Atlantic salmon. **Note that the driver for flow targets was the Habs Directive**
Environment Agency North West Region Low Flow Studies in SAC Rivers in Cumbria: 2008-2009 (2010)	Atkins Global (2010)	Minimum flows at 4 sites: Q_n96, Q_n98, Q_n96 and Q_n99. Precautionary flow targets at 4 sites: Q_n81, Q_n84, Q_n73 and Q_n81	C2	No deviation below the Qn95 condition limit	Precautionary flow targets to protect the habitat requirements of freshwater pearl mussels on an SAC. **Note that the driver for flow targets was the Habs Directive**
Severn Trent Water Ltd AMP 4 and AMP 5 Low Flow Investigation Sites (2010 - 2015)	APEM and ESI. Bradley et al. (in review)	60% deviation from Q_n75	B1	10% (April-October) and 15% (November - March) deviation from Q_n95	Indicative ecological status was determined using macroionvertebrate indices (LIFE O/E ratio) supported by detailed multivariate statistical analysis

Study	Author	Minimum flow values	UKTAG river type	UKTAG Q_n95 low flow standards or Q_n95 condition limits	Comments
Wessex Water Services River Avon SAC Low Flow Investigation (2009)	APEM (2009)	Macroinvertebrates; 50% deviation from Q_n95. Bullheads: >15% deviation from Q_n95	A2 (headwaters)	7.5% (April-October) and 10% (November - March) deviation from Q_n95	Indicative ecological status was determined using macroionvertebrate indices (LIFE O/E ratio) supported by detailed multivariate statistical analysis. Minimum flow for bullheads was determined by a significant reduction in densities. **Note that the driver for flow targets was the Habs Directive**
River Itchen Macroinvertebrate Community Relationship to River Flow Changes	Environment Agency. Exley (2006)	15.6 - 13.9% deviation from the long-term actual Q95	A2	10% (April-October) and 15% (November - March) deviation from Q_n95	Flow thresholds based on LIFE O/E ratio supported by detailed multivariate statistical analysis. Precautionary values due the SAC designation of the River Itchen. **Note that the driver for flow targets was the Habs Directive**
DRIED-UP 3	Dunbar et al. (2010 c)	Simulated reduction of 0.1 in autumn LIFE scores in response to to 10 - 30% reduction in mean summer Qn95	Multiple river types	7.5 to 20% deviation from Qn95 (April-October)	The significance of a reduction of 0.1 in LIFE scores was not described in terms of indicative ecological status

2.8 Summary

2.8.1 Process descriptions and impact tables

The conceptual model has demonstrated that there are ecologically important components of the river flow regime that should be the focus for management effort and the basis of a framework for optimising water releases from impoundments in rivers. In a full description, these include the mean and variability of the magnitude, duration, timing, sequencing and frequency of hydrological events, and the rate of change between them. To make flow management more tractable, the conceptual model summarises flow changes into six ecologically important components:

- (increased frequency of) extreme or extended low flows;
- enhanced and stabilised low flows;
- loss of high flow pulses (return period<1yr) or small floods (2-10 year events);
- loss of large floods (>10yr events);
- extreme high or untimely discharge; and
- rapidly changing flows.

The conceptual model has further demonstrated that alterations to these ecological flow components affect biota through the mediating influence of hydraulic and geomorphological parameters in rivers. These combine (with thermal, chemical and other characteristics) to create the habitat state – the conceptualisation of the physical environment that supports aquatic organisms. Emergent properties of the habitat state have also been identified that are important to allow aquatic organisms to reproduce and progress through their life-cycles. Habitat effects can be organised around these emergent properties that can form the basis of identifying abiotic ecological indicators and help structure application of the optimisation framework:

- size of the habitat (area/volume of aquatic habitat space);
- connectivity and juxtaposition of habitat; and
- character and diversity of the habitat (ecological 'quality' of the habitat).

The conceptual model further outlines general biotic and ecosystem processes that link these habitat properties to measures of species success (abundance, size, population structure, etc.), and community and ecosystem characteristics. These form the basis of identifying biotic ecological indicators of the severe effects of river flow alteration and also help structure application of the optimisation framework.

The mechanisms described in the conceptual model text are summarised in process diagrams and impact tables (Appendix II):

- process diagrams are presented for each ecological flow component and provide a visual route map from flow changes (causes) to biotic effect, via changes to the habitat state; and
- impact tables are presented for selected ecological elements (receptors), and summarise the nature and timing of risks.

The process diagrams and impact tables illustrate the derivation of the ecological indicators and facilitate the application of the water release optimisation framework, but are intended as adjuncts to, not replacements for, the conceptual model text; they lack the latter's treatment of scale and complexity.

The process diagrams adopt different approaches to describing the physical and biological aspects, reflecting the complexities in defining biological responses (and also perhaps the enduring differences in approach from the 'hydro' and 'ecology' traditions in hydro-ecological science):

- The physical processes are mapped out and can be prioritised. They are coloured to differentiate the different physical environments, and the usefulness of a change in habitat state as a physical indicator.
- The biotic responses represent different biotic processes, species and levels of biotic organisation. Colours denote the sign and degree of response, and therefore the sensitivity of biotic response to flow change, with further descriptive detail given for selected ecological elements in the evidence base.

The process diagrams do not present the effects of particular types of pressure, because the same type of pressure can have different effects; for example, a reservoir of any given type may have a low or a high compensation flow, or may or may not release freshets. Therefore, the hydrological effects of an abstraction or an impoundment can be summarised by a 'pick and mix' of ecological flow components, and therefore a conceptual model of any given abstraction or impoundment can be constructed from a series of modules, each describing separate ecological component modules. As a simple example, the main effect of abstraction – reducing flows during, and prolonging, natural low flow periods – is 'extreme or extended low flows'. By contrast, the more complex effect of a direct supply reservoir might be described by 'extreme or extended low flows' plus 'loss of freshets and small floods'.

2.8.2 Implications for river flow management

2.8.2.1 Overview

What does the conceptual model tell us?

In the first instance, the conceptual model demonstrates that river flows are a primary control upon river biota. Thus, they support the natural flow paradigm, that of a monotonic relationship between the degree of hydraulic alteration and the disturbance to the ecology (Richter et al. 1997), such that, with organisms adapted to their natural flow regime (Lytle and Poff, 2004) it can be hypothesised that a river's ecological integrity will tend to increase as flows more resemble naturalised regimes, and that deviations from this can be regarded as detrimental (Poff et al. 2010b). This is the starting point for the WFD water resources environmental standards and condition limits for managed flows in the UK (Acreman and Dunbar, 2004; Acreman et al. 2008; UKTAG, 2008a, b), and for water resource management internationally.

The conceptual models further demonstrate that, through the effect on disturbance events as well as on prevailing hydraulic behaviour, changes to *any* aspect of a natural flow regime must inevitably result in deviations from natural ecosystem composition and function, and therefore *all* aspects of the flow regime must be protected – not just the lowest flows.

Finally, the conceptual model also articulates why there is not a clearly definable relationship between the assessment of river flows and the biological classification of water bodies using existing biomonitoring tools. It has demonstrated that for ecological tools to be sensitive to hydromorphological pressures, they need to take account of a) the changing size of aquatic habitat space and b) biotic changes within individual meso-habitats and not just the overall character or quality of habitats as in traditional water quality assessments. The role of biotic interactions in determining the observed ecological impacts of severe low and stable flows has been highlighted, which are not captured by existing biotic indices and classification tools. Despite undoubted advances in the underlying science and in the tools available to river engineers (e.g. Petts, 2009), complexity and scale issues remain in both physical interactions and biological responses.

The conceptual model does not solve these problems, but it does is allow us to structure the solution better, and to identify inexpensive measures which might prove more amenable to widespread application than current tools. These form the basis of the ecological indicators and optimisation framework. The implications are discussed further below.

2.8.2.2 Dealing with uncertainty: adaptive vs prescriptive management

As the conceptual model illustrates, spatial and temporal dynamism, and local context are recurring features of both the physical and biological character of rivers and their connected wetlands. The complexity of the relationship between river discharge and biology, coupled with the lack of quantitative information is such that prescriptive management of river flows that will universally support good biology is not possible with high certainty. The conceptual model supports the widely held view that river flow management (whether by the application of hydrology standards and condition limits, or other methods), with its attendant uncertainties, should be undertaken in a risk-based framework, with adaptive mangement when applied at the local level (for example with DRIFT (King et al. 2010), (Souchon et al. 2008), and WFD 82 (SNIFFER, 2007)).

This conclusion is reinforced by international experience. That information about flow requirements of aquatic biota is limited; that ecological responses to artificially controlled river flows are highly variable between and within rivers (e.g. Poff and Zimmerman, 2010; Souchon et al. 2008); and that the tools to predict the effects of flow changes are of limited effectiveness (e.g. Bradford et al. (2011) are conclusions reached by many reviews (e.g. Acreman and Dunbar, 2004; Poff et al., 2010b Thorstad et al., 2008; Milner et al. 2011). Furthermore, robust scientific tests quantifying the outcome of prescribed flows for aquatic organisms are still very few (Armitage, 2006; Sabaton et al. 2008; Bradford et al. 2011), are mostly confined to fish and in many cases have been equivocal about the ecological benefits.

2.8.2.3 Hierarchy and structure in assessing impacts and designing flow regimes

By demonstrating the uncertainties in defining habitat changes and linking these to biotic effects, the conceptual model substantiates the hierarchy of tools presented in WFD 82 (SNIFFER, 2007) – that hydrological standards are of lower accuracy than hydraulic techniques, which are in turn of lower accuracy than estimates based on biological data.

Adopting this hierarchy, impact assessment and the design of remediating measures should, therefore, utilise relevant local biotic data wherever available, either to establish flow or habitat – biotic relationships, or to confirm the attainment of GEP in the waterbodies downstream of an impoundment. Using the structure above, changes in species abundance, size, population structure and community and ecosystem characteristics can be related to the nominative descriptions of status.

Habitat information is the next best alternative, but given the limitations of habitat approaches articulated in the conceptual model, it is emphasised that the use of hydraulically-derived habitat models is to derive an initial condition in an adaptive management approach and is not considered a long-term replacement for biotic data.

The conceptual model and supporting evidence base supports two broad approaches to the use of habitat information:

- Defined preference habitat modelling - where a hydraulic description is combined with species or community preferences (Appendix I.9).
- General habitat properties, where a hydraulic description is used to examine changes in more general habitat properties quantity, connectivity, character and diversity.

If general habitat properties are used, the conceptual model suggests that a holistic approach, and therefore a holistic suite of habitat-based indicators, must describe properties of:

- size of the habitat (area/volume of aquatic habitat space);
- connectivity and juxtaposition of habitat; and
- character and diversity of the habitat (ecological 'quality' of the habitat).

Potentially, a flow standard represents a fall-back in the absence of hydraulic, habitat or biotic information where highly simplifying assumptions must be made, for example in classifying large numbers of waterbodies, or addressing relatively minor water management problems. This is discussed in more detail in Section 2.8.3.

2.8.3 Implications for low flow standards and condition limits

The conceptual model and expert consultation through the project workshop has identified several points that will help inform future reviews of the existing environmental standards and condition limits for minimum ($<Q_n95$) flows. This report has highlighted limited new quantitative information on the minimum flow requirements of riverine biota since the environmental standards and condition limits were established (UKTAG, 2008a, b). The minimum flow requirements differ among organisms and river types and there remains a paucity of quantitative information on the flow requirements of aquatic organisms. There remains considerable uncertainty in the existing flow standards and condition limits and the conceptual model supports the use of minimum flow standards only in a risk-based, adaptive management context, and not as fixed values without latitude.

The conceptual model acknowledges limited evidence for a threshold at Q_n95 (Booker and Acreman, 2007), as a proxy for an average break point in discharge - wetted perimeter relationships, by which to support current enviuronmental standards. Furthermore, it notes significant associated uncertainties, the lack of representativeness of the sample sites used to derive this, and the potential for smoothing of breakpoints in the flow-wetted perimeter curves when aggregated across sites (Gippel and Stewardson, 1998). These uncertainties do not only concern the location of such a threshold, but also the existence of one that can be reliably applied at the reach scale. More importantly, the wetted perimeter chiefly addresses habitat quantity, albeit with subsidiary support to connectivity and limited evidence of associated changes in hydraulic diversity (e.g. Section 2.6.2). The character of habitat is not addressed by the use of an environmental standard, which assumes a natural physical habitat apart from flow, meaning that habitat quality is neglected by an approach based on flow standards alone.

The conceptual model further demonstrates the importance of naturally arising flow stress through ‘resetting’ the ecosystem for checking invasive or competitively dominant species, and maintaining an ecosystem that is characterised by abiotic disturbance and biotic resilience. Existing environmental standards prescribe the lowest allowable abstraction when flows are low (such as during natural drought conditions). This often coincides with peak demand for water (such as during hot dry summers). The conceptual model supports proposals that higher abstractions at minimum flows could be allowed during particular short-term periods, provided that measures are in place to allow the river ecosystem to recover afterwards. This would acknowledge the importance of natural variability of low flows and the resilience of riverine ecosystems to periodic low flow stress, and may provide significant benefits to operational yield and water use.

Likewise, enhanced compensation flows or ‘discharge rich’ reaches supplemented by effluents are undoubtedly deviations from natural condition. These can increase the local abundance of some biological elements, such as fish (John Armstrong, David Solomon pers. comm.), and yet have a negative impact on the riverine ecosystem as a whole and over the long-term. The same conflicts arise under natural flow regimes, but then each ecological element is adapted to the natural variation – the natural disturbance of wet and dry years is part of the selection pressure that acts on aquatic organisms (Lytle and Poff, 2004). Flow condition limits for managed flows prescribe unacceptable deviations above natural flows (+40%), but this is not considered by existing environmental standards for river flows, for example in situations where natural baseflows are augmented by effluents.

The conceptual model presents some evidence that ecological responses to changing water levels in rivers might occur in a stepped fashion as certain habitat thresholds are crossed rather than in a linear fashion as assumed by the existing environmental standards (i.e. higher abstraction allowed at Q_n60, lower abstraction allowed at Q_n95). Related to this point is the importance of the temporal sequencing of flows in relation to the timing of organism’s life-cycles that is not taken into account by the existing environmental standards or condition limits that are based on the flow duration curve. A refinement of the environmental standards might be to consider varying abstraction limits at minimum flows according to the duration of antecedent flows at different levels. For example, by allowing more abstraction at minimum flows after periods of higher flows, but allowing less abstraction at minimum flows after supra-seasonal or extended low flow periods.

2.8.4 Implications for hands off flows, compensation flows and river support

The conceptual model clearly demonstrates that hands off flows, compensation flows and river support should be set to mimic natural flows, accepting that some loss in the magnitude of flows is inevitable wherever there is direct abstraction. Clearly impoundments should therefore release water into the downstream channel throughout the year (assuming that flows do not cease naturally at some times of year). As well as maintaining some minimum flow throughout the year, compensation flows should maintain flow variability – ideally at all temporal scales (sub-daily to inter annual).

The support provided by the conceptual model for a dynamic component to regulating minimum flows (Section 2.8.3) would allow compensation flows (and other forms of river support) to be scaled by catchment inflows. This would make compensation releases from storage more consistent with abstraction problems (in which hands of flows already allow natural flow reductions below proscribed 'minimum flows'), and has been used internationally (e.g. Yin et al. 2011) to ensure coincidence of flow with catchment inputs, and maintain flow variability at a range of temporal scales.

The conceptual model also supports the reduction of enhanced compensation flows (Section 2.8.3), although this would need to be considered alongside other policy drivers and the needs of downstream water users. For example, water transfer from Kielder Water has been predicted to enhance salmon parr habitat to the River Wear by 10% and to reduce by 70% the reductions in habitat loss that would normally occur under natural flows (Gibbins and Heslop, 1998). Pragmatically, those are good outcomes, for fish; but the wider questions about the long-term fitness consequences of this, the effects on other ecological elements and wider ecosystem function, remain unanswered. The natural flow paradigm suggests, however, that over the longer-term, a variable flow regime may be of greater benefit to the ecosystem than enhanced or stable flows.

Variability at a range of timescales should be achieved, ideally by scaling to and coincident timing with catchment inflows. Failing this, limited inter-year variability can be provided for using wet, average and dry year release profiles (described, for example, in Gordon et al. 2004). The 'average year' compensation flow is set to protect diversity and abundance with a degree of uncertainty. A lower, dry year 'survival' flow, can then be set, scaled to reflect natural dry year conditions to allow reduced abundance (but not extinction) during periodic natural environmental stress, and a lower certainty of avoiding impacts. In wet years, higher flows (might also be used to correspond to natural conditions, or used after droughts to stimulate recovery.

In the absence of a detailed local hydraulic description, and assuming that defined species or community preferences are not targeted, the conceptual model favours separate quantitative assessment of the habitat properties of:

- size of the habitat (area/volume of aquatic habitat space);
- connectivity and juxtaposition of habitat; and
- character and diversity of the habitat (ecological 'quality' of the habitat).

Where this is not possible, the conceptual model lends limited support to a strategy of targeting bed coverage consistent with that found in a representative natural channel (which may not always achieve bed coverage). This maintains total in-channel habitat space and, through this, implicitly maintains a degree of connectivity within the channel. There is also some evidence to suggest that maintaining wetted bed may support in-channel habitat diversity and taxon richness, although the evidence is divided about whether these last habitat effects occur immediately bed space is wetted or at some point above this (Section 2.6.2) and (Appendix I.1).

Because of this uncertainty, a strategy of achieving close to natural bed coverage should be complemented by an assessment of other properties (particularly in-channel habitat quality or character), even if only a qualitative reality check is possible. This may be directed at specific questions, for example, to assess whether water is flowing or static, whether flow at barriers is sufficient to maintain longitudinal connectivity, whether sufficient head is maintained to sustain natural patterns of lateral and vertical connectivity, whether velocity is sufficient to prevent fines settling or maintain bed movement for channel maintenance flows, or the functional habitat requirements of ecological elements are likely to be achieved in at least part of the channel.

2.8.5 Implications for freshets and spills

The conceptual model demonstrates how spates in a natural system perform disturbance functions as diverse as flushing fine sediments, checking invasive and competitive species, maintaining flow variability in channel habitats, maintaining lateral connectivity between river and floodplain and enabling migration of salmonids. Higher pulse flows should also be competent to maintain channel structure and maintain active geomorphological development of the channel and riparian zone.

The most studied of these roles is for salmonid migration (Appendices I.9 and I.10), for which spate and flood flows:

- attract fish into rivers from the sea;
- stimulate movement of resting fish, follow periods of low flow or extended residence in holding areas;
- maintain movement once started;
- encourage fish to pass partial barriers (e.g. falls, shallow rapid areas), protecting them from predation, poaching or mortality through delays, crowding and disease transfer, high temperatures;
- distribute spawners throughout the catchment as widely as possible to maximise use of spawning and rearing habitat;
- enhance angling catches by making fish available to middle/upper reaches and enabling angling conditions;
- improve or support water quality in downstream areas of the main stem or estuary where holding adult salmonids may be vulnerable.

Invertebrates of exposed riverine sediments, which include many rare species of beetle and spider, depend on a regime of flooding and drying to maintain habitat character and to prevent the dominace of competitive terrestrial species. The life cycles of these invertebrates have evolved to enable them to be resistant to natural high flows that occur in winter and for vulnerable early stages to coincide with natural low flow periods in early summer. Artificial floods that do not coincide with the timing of natural events or that have a much quicker rate of change can impact these organisms that have high conservation value.

In the absence of natural flows, artificial freshet releases, spills and operational releases such as scour valve operation, must perform the various functions of natural pulse flows. Note also that aseasonal floods, for example from scour valve operation, can have greater effects on biota than those in normal flood seasons (Death, 2010), causing, for example, significant impacts such as washout of eggs and invertebrates on exposed riverine sediments.

The conceptual model and evidence base (Appendix I) provides guidelines for freshet releases in Appendix I.2 and Appendix I.4 (for channel maintenance) and Appendices I.9 and I.10 (for salmonid migration). The potential effects of aseasonal flood releases are summarised in the impact tables (Appendix II).

The conceptual model again supports freshet and spill characteristics that are as consistent as possible with natural flows in these properties, and further supports variability in the release pattern at a range of temporal scales. The conceptual model therefore supports an approach of scaling to and timing coincident with natural catchment inflows, and provides a scheme (Richter et al. 1996) by which to describe the properties of these inflows (magnitude, duration, timing, frequency and rates of change).

3 ECOLOGICAL INDICATORS OF THE SEVERE EFFECTS OF ABSTRACTION AND IMPOUNDMENT OF WATER IN RIVERS

3.1 Background

The current water body classification system across the UK relies on an assessment of a number of quality elements and supporting conditions. Hydrological pressure is a component of the supporting condition for the biological quality elements and is measured by a specific assessment against current river flow standards. Ideally, failure to meet a river flow standard would result in a measurable impact on biology, but at present there is not a good relationship between an assessment of hydrological pressure and the biological classification for numerous reasons. This is seen across all the classification bands from High to Bad. This uncertainty is currently being addressed through investigations looking at developing classification tools that are more sensitive to river flow alterations. This section of the report focuses on providing specific ecological supporting information for the worst affected waterbodies – those at Poor and Bad hydrological status. Ecological indicators are required to improve the weight of evidence needed to identify where measures should be put in place to improve river flows in water bodies and HMWBs.

Specifically, this section of the report uses the conceptual model to identify a suite of ecological indicators that can be used to:

- capture features of river environments that are not described by current biological classification tools, and can be used as signposts to help identify the most severe effects of water use, thereby improving certainty around the resulting classification of water bodies as Poor and Bad status;
- provide an additional ecological evidence base to support the current classification of surface water bodies considered to be at Poor and Bad status using current UKTAG biological classification tools, where there are severe pressures from water use;
- provide an additional ecological evidence base to support the current classification of surface water bodies considered to be at Poor and Bad status, where there are severe pressures from water use, but no local biological data; and
- improve the weight of evidence for prioritising mitigation measures on the most severely affected water bodies.

3.2 Requirements of the Ecological Indicators

3.2.1 General criteria

The conceptual model has identified indicators of direct measures of the impacts on animals and plants and indirect measures of the physical environment. Indirect, abiotic indicators have been identified where the conceptual model provides clear links to the biotic response. The ecological indicators include both quantitative and qualitative measures. Ecological indicators provide information on primary variables (e.g. size and abundance of populations) and emergent properties such as species richness, succession and resilience.

The general criteria for identifying ecological indicators are:

- relevance to ecological quality definitions, particularly those of severe and major effects on ecology;
- sensitivity of life stage and life history responses to flow modifications (noting that the project focuses on severe to major effects);
- seasonal applicability;
- temporal and spatial distribution and stability;
- ecosystem functionality (i.e. what particular links with other ecosystem components may lead to synergy or redundancy of information);
- sensitivity (and distribution) variation with river type;
- precision with respect to differentiating effects of abstraction vs flow regulation impacts;
- monitoring feasibility and cost; and
- accessible analytical and diagnostic routines (including required facilities and expertise).

3.2.2 Measurement

The ecological indicators include both quantitative and qualitative features of rivers and riparian habitats that can be easily measurable on site without the need to implement additional sampling programmes or which might be derived from data collected as part of existing sampling programmes. Field measurable indicators must be able to recorded quickly and easily, and will not require the use of expensive data gathering techniques. It is expected that the Ecological Indicators will be collected by a wide range of workers (e.g. ecologists, hydrologists, geomorphologists) whilst undertaking field monitoring for a wide range of purposes. The methods and spatial scale of measurement therefore need to be flexible to accommodate different situations, but sufficiently robust to provide reliable information.

Technological advances in high resolution aerial photography and image processing offer many advantages to the rapid collection of information on Ecological Indicators at larger spatial scales (whole water body or catchment scale) and at more remote locations than is possible or cost-effective from ground-based surveys (Bradley & Dugdale, 2009). The potential for Ecological Indicators to be measured using high resolution aerial images has therefore also been considered.

3.2.3 River type specificity

It is essential that the ecological indicators can be used reliably to provide consistent results across the UK. However, not all ecological indicators can be used in the same way in all water bodies. Therefore river type specificity has been defined for each ecological indicator. It was decided that a single river typology (such as that described in WFD 48) applied across all of the indicators, would be convenient, but would not be realistic or conducive to providing reliable results. River types have therefore been defined as appropriate to a given indicator.

3.2.4 Weight of evidence

The ecological indicators will be used together to provide a weight of evidence to identify the most severely affected water bodies due to water use, and to improve certainty in the classification of water bodies classed as Poor or Bad status. In some cases, the ecological indicators might provide weak information on their own, but when used in combination with other indicators and in certain combinations at certain locations, they will provide a useful weight of evidence.

3.3 Identification of the ecological indicators

3.3.1 Link to Conceptual Model

Ecological indicators have been identified in the conceptual model and the supporting evidence base. These provide the substantiating evidence base, including literature references, for the ecological indicators and the conceptual links from the water resource drivers, through the physical effects to the biological/ecological impacts. In this way the conceptual model provides the linkages between the abiotic ecological indicators and the responses of the biological quality elements that define Poor and Bad status.

Recall that the conceptual model presents two key concepts that determine the ecological impacts of flow alteration and are pertinent to the identification of ecological indicators of these effects:

1. The relative importance of abiotic and biotic controls on aquatic organisms under different modified habitat states.
2. The habitat state comprises not only the character (or 'quality') of habitat, but also the size of the habitat and the connectivity of habitats.

The conceptual model identifies the role of biotic interations in determining ecological impacts to flow alterations, particularly at extreme low and/or stable flows, where predation and density-dependent effects become intensified. Biotic indicators have been identified that capture the effects of both biotic interactions and abiotic effects that occur under conditions of severe hydrological alteration.

Abiotic indicators have been identified that provide information on the size, connectivity and character of instream habitats that indicate ecological impacts of severe hydrological alterations. Current biological classification and assessment methods provide information on the diversity and character of riverine habitats, but do not yield information regarding the total size (volume or area) of aquatic habitats (Mainstone, 2010). The effect of abstracting or impounding water in rivers is often to miniaturise aquatic habitats, whilst still maintaining the overall character. This is why, when sampled across available wetted habitat, current biological methods are often insensitive to measuring the effect of hydromorphological pressures (Mainstone, 2010). Ecological indicators have therefore been selected to provide information on the size of aquatic habitat space and connectivity of aquatic habitats in addition to indicators of habitat diversity and character.

3.3.2 Starting point – normative definitions of Poor and Bad status

Ecological indicators are needed to support decision making on waterbodies which are severely affected by water use, with Poor and Bad status under the WFD. The current normative definitions of Poor and Bad status in Annex V of the Directive (Directive 2000/60/EC) therefore provide the starting point for the identification of ecological indicators. The definition of Moderate status is also given to clarify the Moderate/ Poor boundary:

Moderate: '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'.

Poor: 'Waters showing evidence of major alterations to the values of 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'.

Bad: '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'.

3.3.3 Process of identifying Ecological Indicators

The ecological indicators have been identified by the project consultants and by expert consultees, who have contributed through a web forum, targeted discussion and an Expert Workshop held in Manchester on 6th October 2011. A record of the Workshop, comments received from the Expert Panel during and subsequent to the workshop, and a brief description of how this feedback has been used in the selection or refinement of the ecological indicators is contained in Appendix I. The final list of ecological indicators presented in this report represents the outputs of an Expert Workshop and subsequent review and consultation.

3.4 The Ecological Indicators

3.4.1 Physical Indicators

A total of 19 physical indicators have been identified (1a – 1s) as either having direct linkages to biological or ecological elements, or being symptomatic of hydraulic or geomorphological changes that have such linkages. Table 3. describes these indicators and provides details on links to the conceptual model river type specificity, potential confounding factors and additional comments from the expert group.

Table 3.1 Physical indicators

Indicator number	Indicator description	Driver application from Conceptual Model	River type specificity	Potential confounding factors	Link to Conceptual Model detailed evidence base
1a	Loss or absence of wetted channel. Absence of water in a river channel	Steady abstraction, spray irrigation, direct supply reservoir (water & HEP), regulating reservoir (water & HEP).	Not natural winterbournes	Natural drying, sinks (e.g. karstic streams) and winterbournes. Caution if used in extreme droughts	I.1
1b	Fragmentation of aquatic habitat in river channels	Steady abstraction, spray irrigation, direct supply reservoir (water & HEP), regulating reservoir (water & HEP).	Not natural winterbournes	Natural sinks (e.g. karstic streams) and winterbournes. Artificial structures (e.g weirs). Caution use in extreme droughts	I.1
1c	Loss of riffles/ runs, preponderance of pools	Steady abstraction, spray irrigation, direct supply reservoir (water & HEP), regulating reservoir (water & HEP).	Not natural winterbournes or large lowland rivers	Natural sinks (e.g. karstic streams) and winterbournes. Caution if used in extreme droughts	I.2, I.8
1d	Fine sediment covering sensitive habitats (riffles, runs, glides)	Steady abstraction; spray irrigation; direct supply reservoir(water & HEP),	Gravel and cobble bed rivers	Excessive inputs of fine sediment from the catchment	I.3

Indicator number	Indicator description	Driver application from Conceptual Model	River type specificity	Potential confounding factors	Link to Conceptual Model detailed evidence base
1e	Dense plume of fine sediment occluding water column when submerged substrate disturbed	Steady abstraction; spray irrigation; direct supply reservoir (water and HEP).	Gravel and cobble bed rivers	Excessive inputs of fine sediment from the catchment	I.3
1f	Absence of gravel from bed surface	Direct supply reservoir (water & HEP)	Gravel and cobble bed rivers, potentially also cascades and bedrock channels.		I.4
1g	Uniform cobble particle size on bed surface (armouring or paving), ‘static’ (ie not active) riffles.	Direct supply reservoir (water & HEP)	Gravel and cobble bed rivers		I.4
1h	All mid-channel substratum submerged during March-June for >1.5km downstream of impoundments	Regulating reservoir (HEP), Direct supply reservoir (water & HEP), Regulating Reservoir (water)	Gravel and cobble bed rivers	Naturally deep rivers.	I.1

Indicator number	Indicator description	Driver application from Conceptual Model	River type specificity	Potential confounding factors	Link to Conceptual Model detailed evidence base
1i	No active (unvegetated) channel bars	Regulating reservoir (HEP), Direct supply reservoir (water & HEP), Regulating Reservoir (water)	Gravel and cobble bed rivers		I.5
1j	Presence of stable (vegetated) channel bars without presence of active (unvegetated) bars	Regulating reservoir (HEP), Direct supply reservoir (water & HEP), Regulating Reservoir (water)	Gravel and cobble bed rivers		I.5
1k	Evidence of terrace formation	Regulating reservoir (HEP), Direct supply reservoir (water & HEP), Regulating Reservoir (water)	Gravel and cobble bed rivers		I.6
1l	No exposed substrate on channel banks	Regulating reservoir (HEP), Direct supply reservoir (water & HEP), Regulating Reservoir (water)	Gravel and cobble bed rivers		I.6
1m	Gradient of channel banks less than vertical	Regulating reservoir (HEP), Direct supply reservoir (water & HEP), Regulating Reservoir (water)	Gravel and cobble bed rivers		I.6

Indicator number	Indicator description	Driver application from Conceptual Model	River type specificity	Potential confounding factors	Link to Conceptual Model detailed evidence base
1n	Low width to depth ratio	Regulating reservoir (HEP), Direct supply reservoir (water & HEP), Regulating Reservoir (water)	Gravel and cobble bed rivers		I.7
1o	Steep, undercut or eroding tributary banks	Regulating reservoir (HEP), Direct supply reservoir (water & HEP), Regulating Reservoir (water)	Gravel and cobble bed rivers		I.7
1p	Tributary terraces	Regulating reservoir (HEP), Direct supply reservoir (water & HEP), Regulating Reservoir (water)	Gravel and cobble bed rivers		I.7
1q	Exposed tree roots in bottom of tributary channels	Regulating reservoir (HEP), Direct supply reservoir (water & HEP), Regulating Reservoir (water)	Gravel and cobble bed rivers		I.7
1r	Presence of active (unvegetated or semi-vegetated) bars downstream of tributary confluences	Regulating reservoir (HEP), Direct supply reservoir (water & HEP), Regulating Reservoir (water)	Gravel and cobble bed rivers		I.7

Indicator number	Indicator description	Driver application from Conceptual Model	River type specificity	Potential confounding factors	Link to Conceptual Model detailed evidence base
1s	Widespread gravitational bank collapse	Regulating reservoir (HEP), Direct supply reservoir (water & HEP), Regulating Reservoir (water)	Gravel and cobble bed rivers		I.6

3.4.2 Fish indicators

A total of eight fish indicators have been identified (2a – 1h). Table 3.2 describes these indicators and provides details on links to the conceptual model river type specificity, potential confounding factors and additional comments from the expert group.

Table 3.2 Fish indicators

Indicator number	Indicator description	Driver application from Conceptual Model	River type specificity	Potential confounding factors	Link to Conceptual Model detailed evidence base
2a	Trout and salmon (0+ to 2+) absent in otherwise suitable and accessible habitat as assessed by appropriate model.	Steady abstraction, water supply and HEP reservoir	All except lowland floodplain rivers	Trout are considered more reliable indicators than salmon given their ubiquity	I.9,I.10
2b	Increased growth rate of trout	Water supply reservoir	All except lowland floodplain rivers	Further development needed to establish reference growth rates at different sites	I.10
2c	Decreased growth rate of trout	Water supply and HEP reservoir	All except lowland floodplain rivers	Further development needed to establish reference growth rates at different sites	I.10
2d	Absence of adult salmon or migratory trout in autumn	Steady abstraction, water supply reservoir	Upland spate rivers		I.9, I.10
2e	Increased ratio of plant-spawning to gravel-spawning coarse fish	Steady abstraction; spray irrigation; direct supply reservoir	Chalk streams and lowland rivers. Excl. N.Ireland, much of Scotland.		I.18

2f	Poor first summer recruitment of phytophilic coarse fish	Steady abstraction; spray irrigation; direct supply reservoir	Chalk streams and lowland rivers. Excl. N.Ireland, much of Scotland.		I.18
2g	Poor winter survival of phytophilic and lithophilic coarse fish	Water supply (transfers), direct supply reservoir	Chalk streams and lowland rivers. Excl. N.Ireland, much of Scotland.		I.18
2h	Poor first summer survival of lithophilic and phytophilic coarse fish	Water supply (transfers) direct supply reservoir	Chalk streams and lowland rivers. Excl. N.Ireland, much of Scotland.		I.18

3.4.3 Macroinvertebrate indicators

A total of seven macroinvertebrate indicators have been identified (3a – 1g). Table 3.3 describes these indicators and provides details on links to the conceptual model, river type specificity, potential confounding factors and additional comments from the expert group.

Table 3.3 Macroinvertebrate indicators

Indicator number	Indicator description	Driver application	River type specificity	Potential confounding factors	Link to Conceptual Model detailed evidence base
3a	Major reduction in taxon richness	Steady abstraction, regulating reservoir (HEP), spray irrigation, direct supply reservoir (water & HEP), regulating reservoir (water)	All rivers	Water pollution. Artificial physical modification of the channel	I.23
3b	LIFE O/E >0.914 using RIVPACS III+ or RICT and family LIFE	Steady abstraction, regulating reservoir (HEP), spray irrigation, direct supply reservoir (water & HEP), regulating reservoir (water)	LIFE not tested in Scotland or Northern Ireland	Water pollution. Artificial physical modification of the channel	I.23
3c	Abundance of large bodied predatory invertebrates, such as Coleoptera larvae and adults (especially Dytiscidae), Hemiptera (Notonectidae, Corixidae and Gerridae) and Odonata nymphs in main river channel	Steady abstraction, regulating reservoir (HEP), Spray irrigation, direct supply reservoir (water & HEP), regulating reservoir (water)	All rivers	Washout from local still waters during floods. Do not include if present only in natural backwaters or vegetated margins of rivers. Can colonise river reaches rapidly in response to seasonal low flows and drought. Need to compare to local reference sites and use in combination with other ecological indicators	I.23

Indicator number	Indicator description	Driver application	River type specificity	Potential confounding factors	Link to Conceptual Model detailed evidence base
				of chronic impacts.	
3d	Presence or increased numbers of LIFE Flow Group V and VI species when not predicted by RIVPACS/RICT	Steady abstraction, regulating reservoir (HEP), spray irrigation, direct supply reservoir (water & HEP), regulating reservoir (water)	Fast flowing, stony bottomed rivers. LIFE not tested in Scotland or Northern Ireland	Water pollution. Artificial physical modification of the channel	I.23
3e	Absence of LIFE I-III species when predicted to occur by RIVPACS/RICT	Steady abstraction, regulating reservoir (HEP), spray irrigation, direct supply reservoir (water & HEP), regulating reservoir (water)	Fast flowing, stony bottomed rivers. LIFE not tested in Scotland or Northern Ireland	Water pollution. Artificial physical modification of the channel	I.23
3f	Presence of species described as winterbourne specialists (see Table I 13) in normally permanently flowing reaches near abstractions or downstream of impoundments	Steady abstraction, spray irrigation	Chalk streams	Natural sinks (e.g. karstic streams) and winterbournes	I.23

Indicator number	Indicator description	Driver application	River type specificity	Potential confounding factors	Link to Conceptual Model detailed evidence base
3g	Absence of baetid mayflies when predicted to occur by RIVPACS/RICT	Steady abstraction, regulating reservoir (HEP), spray irrigation, direct supply reservoir (water & HEP), regulating reservoir (water)	Unpolluted, stony/gravelly rivers. Not acidified streams with pH <5.5	Water pollution. Artificial physical modification of the channel	I.23
3h	Dominance or monopoly of *Gammarus* spp. downstream of impoundments	Regulating reservoir (HEP), Direct supply reservoir (water & HEP), regulating reservoir (water)	Not base poor catchments or newly wetted winterbourne channels	To be used as an indicator of Poor and Bad status only downstream of impoundments. Other factors can cause *Gammarus* spp. To dominate in other rivers (excessive allochthonous inputs, moderate organic enrichment, newly wetted winterbourne channels)	I.23

3.4.4 Macrophytes, bryophytes and diatom indicators

A total of 14 macrophyte, bryophyte and diatom indicators have been identified (4a – 1n).

Table 3.4 describes these indicators and provides details on links to the conceptual model, river type specificity, potential confounding factors and additional comments from the expert group.

Table 3.4 Macrophyte, bryophyte and diatom indicators

Indicator number	Indicator description	Driver application from Conceptual Model	River type specificity	Potential confounding factors	Link to Conceptual Model detailed evidence base
4a	Exposed cobbles, pebbles and small boulders in river channels covered by mosses and/or liverworts indicates chronically stable flows and greatly reduced frequency of erosive, inundation events	Regulating reservoir (HEP), direct supply reservoir (water & HEP), regulating reservoir (water)	Gravel and cobble bed rivers	A simple, reliable indicator of chronic low and stable flows in stony rivers. Might be developed in the future to include key species that are easily identifiable in the field and indicate degrees of wetting and drying.	I.27
4b	Dominance of emergent plants in relation to submerged plants across the river channel	Steady abstraction, regulating reservoir (HEP), spray irrigation, direct supply reservoir (water & HEP), regulating reservoir (water)	CB1, CB2, CB4 and CB6a (Hatton-Ellis & Grieve, 2003)	Do not include if present only in natural backwaters or vegetated margins of rivers	I.21
4c	Dominance of terrestrial plant species in relation to submerged and emergent aquatic species across the river channel	Steady abstraction, regulating reservoir (HEP), spray irrigation, direct supply reservoir (water & HEP), regulating reservoir (water)	All rivers	Do not include if present only in natural backwaters or vegetated margins of rivers	I.21

Indicator number	Indicator description	Driver application from Conceptual Model	River type specificity	Potential confounding factors	Link to Conceptual Model detailed evidence base
4d	Dominance of perennial terrestrial plant species in river margins in relation to aquatic species and annual species	Steady abstraction, regulating reservoir (HEP), spray irrigation, direct supply reservoir (water & HEP), regulating reservoir (water)	CB3, CB4, CB5 and CB6b (Hatton-Ellis & Grieve, 2003)	Do not include if present only in natural backwaters or vegetated margins of rivers	I.21
4e	>10% cover of perennial terrestrial vegetation colonising bars (e.g. perennial herbs, mosses, ferns, trees, bushes)	Steady abstraction, regulating reservoir (HEP), spray irrigation, direct supply reservoir (water & HEP), regulating reservoir (water)	Alluvial or semi-alluvial channels	Potentially useful and reliable indicator of chronic low and stable flows. >10% cover is a proposed starter value and not supported by literature.	I.21
4f	>10% cover of perennial terrestrial vegetation colonising channel banks (e.g. perennial herbs, mosses, ferns, trees, bushes)	Steady abstraction, regulating reservoir (HEP), spray irrigation, direct supply reservoir (water & HEP), regulating reservoir (water)	Alluvial or semi-alluvial channels	Potentially useful and reliable indicator of chronic low and stable flows. >10% cover is a proposed starter value and not supported by literature.	I.21
4g	Filamentous algae covering all submerged macrophytes or channel bed.	Steady abstraction, regulating reservoir (HEP), spray irrigation, direct supply reservoir (water & HEP), regulating reservoir (water)	All rivers	Do not include if present only in natural backwaters or vegetated margins of rivers	I.21

Indicator number	Indicator description	Driver application from Conceptual Model	River type specificity	Potential confounding factors	Link to Conceptual Model detailed evidence base
4h	Dominance of *R. peltatus* relative to *Ranunculus penicillatus* subsp. *psuedofluitans*	Steady abstraction, regulating reservoir (HEP), spray irrigation, direct supply reservoir (water & HEP), regulating reservoir (water)	CB1, CB2, CB4 and CB6a.(Hatton-Ellis & Grieve, 2003)	Natural sinks (e.g. karstic streams) and winterbournes	I.21
4i	Absence of submerged aquatic macrophytes in river types CB4, CB5 and CB6b (Hatton-Ellis & Grieve, 2003)	Steady abstraction, regulating reservoir (HEP), spray irrigation, direct supply reservoir (water & HEP), regulating reservoir (water)	All rivers	Has been cited as an important indicator of excessive abstraction and low flows in chalk streams such as the River Kennet. To be used with caution at most sites and only in combination with other key indicators (1d, 1e, 4g)	I.21
4j	Presence of non-rooted, free-floating species such as duckweed (*Lemna spp*) and floating filamentous algae in river channel	Steady abstraction, regulating reservoir (HEP), spray irrigation, direct supply reservoir (water & HEP), regulating reservoir (water)	Not in naturally very slow flowing lowland rivers	Washout from local still waters during floods. Do not include if present only in natural backwaters or vegetated margins of rivers	I.21

Indicator number	Indicator description	Driver application from Conceptual Model	River type specificity	Potential confounding factors	Link to Conceptual Model detailed evidence base
4k	Dominance of rooted species that are usually confined to still backwaters in main river channel (e.g. starwort *Callitriche*, milfoil *Myriophyllum*and crowfoot *Ranunculus*)	Steady abstraction, regulating reservoir (HEP), spray irrigation, direct supply reservoir (water & HEP), regulating reservoir (water)	Not in naturally very slow flowing lowland rivers	Do not include if present only in natural backwaters or vegetated margins of rivers	I.21
4l	Dominance of aerophilic diatom taxa	Steady abstraction, regulating reservoir (HEP), spray irrigation, direct supply reservoir (water & HEP), regulating reservoir (water)	All rivers	Do not include if present only in natural backwaters or vegetated margins of rivers. A potentially useful and previously underexploited indicator of severe low flows	I.25
4m	Occurrence of long filamentous diatomaceous biofilms	Steady abstraction, regulating reservoir (HEP), spray irrigation, direct supply reservoir (water & HEP), regulating reservoir (water)	All rivers	Do not include if present only in natural backwaters or vegetated margins of rivers. A potentially useful and previously underexploited indicator of severe low flows	I.25

Indicator number	Indicator description	Driver application from Conceptual Model	River type specificity	Potential confounding factors	Link to Conceptual Model detailed evidence base
4n	Increased relative abundance of motile diatom taxa	Steady abstraction, regulating reservoir (HEP), spray irrigation, direct supply reservoir (water & HEP), regulating reservoir (water)	All rivers	Do not include if present only in natural backwaters or vegetated margins of rivers. A potentially useful and previously underexploited indicator of severe low flows	I.25

3.4.5 Amphibian indicators

Two amphibian indicators have been identified (5a – 5b). Table 3.5 describes these indicators and provides details on links to the conceptual model, river type specificity, potential confounding factors and additional comments from the expert group.

Table 3.5 Amphibian indicators

Indicator number	Indicator description	Driver application from Conceptual Model	River type specificity	Potential confounding factors	Link to Conceptual Model detailed evidence base
5a	Presence of frog or toad tadpoles in river channel, especially in late spring – summer indicates long-term and severe low flows from abstraction and/or impoundment of water	Steady abstraction, direct supply reservoir (water & HEP)	Not natural winterbournes	Washout from local still waters during floods. Do not include if present only in natural backwaters or vegetated margins of rivers. Frogs and toads will breed in slow flowing lowland rivers with extensive vegetated and/or shallow margins. Tadpoles need to be present in abundance and all over the river channel for this indicator.	I.26
5b	Presence of newts in river channels indicates long-term still water conditions due to the severe effects of abstraction and/or impoundment of water	Steady abstraction, direct supply reservoir (water & HEP)	Not natural winterbournes	Washout from local still waters during floods. Do not include if present only in natural backwaters or vegetated margins of rivers	I.26

3.4.6 Riparian vegetation indicators

Three riparian vegetation indicators have been identified (6a – 6c). Table 3.6 describes these indicators and provides details on links to the conceptual model, river type specificity, potential confounding factors and additional comments from the expert group.

Table 3.6 Riparian vegetation indicators

Indicator number	Indicator description	Driver application from Conceptual Model	River type specificity	Potential confounding factors	Link to Conceptual Model detailed evidence base
6a	Loss of more aquatic Sphagna and perhaps transition to a different NVC community (e.g. M4 to M6).	Direct supply, regulating and pumped storage reservoirs for water supply and HEP.	Any	Morphological alteration, land management.	I.22
6b	Loss of wetland species and increased represenatation of more terrestrial species.	Direct supply, regulating and pumped storage reservoirs for water supply and HEP.	Any	Morphological alteration, land management.	I.22
6c	Depth and extent of water in the wetland during wet months	Direct supply, regulating and pumped storage reservoirs for water supply and HEP.	Any	Morphological alteration, land management.	I.22

3.5 Characteristics of the ecological indicators

The characteristics of the environment or community that each of the ecological idicators describe have been examined. This will guide the diagnostic features of different combinations of ecological indicators which form a weight of evidence.

Ecological indicators have therefore been identified that when summed up together within sites, provide a weight of evidence that describes the different characteristics of ecological impacts of severe river flow alterations.

Table 3.7 to Table 3.12 describe the environmental characteristics that each of the ecological indicators measures to guide the information that different combinations of indicators will produce.

Table 3.7 Environmental characteristics described by the 19 physical indicators

Indicator number	Diversity of habitat	Character of habitat	Size/volume of habitat	Connectivity of habitat
1a				
1b				
1c				
1d				
1e				
1f				
1g				
1h				
1i				
1j				
1k				
1l				
1m				
1n				
1o				
1p				
1q				
1r				
1s				

Table 3.8 Environmental characteristics described by the eight fish indicators

Indicator number	Diversity of habitat	Character of habitat	Size/volume of habitat	Connectivity of habitat
2a				
2b				
2c				
2d				
2e				
2f				
2g				
2h				

Table 3.9 Environmental characteristics described by the seven macroinvertebrate indicators

Indicator number	Diversity of habitat	Character of habitat	Size/volume of habitat	Connectivity of habitat
3a				
3b				
3c				
3d				
3e				
3f				
3g				

Table 3.10 Environmental characteristics described by the 14 macrophyte, bryophyte and diatom indicators

Indicator number	Diversity of habitat	Character of habitat	Size/volume of habitat	Connectivity of habitat
4a				
4b				
4c				
4d				
4e				
4f				
4g				
4h				
4i				
4j				
4k				
4l				
4m				
4n				

Table 3.11 Environmental characteristics described by the two amphibian indicators

Indicator number	Diversity of habitat	Character of habitat	Size/volume of habitat	Connectivity of habitat
5a				
5b				

Table 3.12 Environmental characteristics described by the three riparian vegetation indicators

Indicator number	Diversity of habitat	Character of habitat	Size/volume of habitat	Connectivity of habitat
6a				
6b				
6c				

3.6 Flow components described by the Ecological Indicators

Each ecological indicator was examined in relation to the ecological flow components introduced in the conceptual model. This analysis was based on the information provided by the conceptual model to elucidate the hydrological conditions that each ecological indictor might respond to (Table 3.13 to Table 3.18). This information, used in conjunction with the information provided for the environmental characteristics described by each ecological indicator can be used to interpret different combinations of ecological indicators which form a weight of evidence.

Table 3.13 Flow components described by the 19 physical indicators

Indicator number	Extreme and extended low Q	Enhanced and stabilised Q	Loss of small floods (<1yr)	Loss of large floods (>1yr)	Extreme or untimely large Q	Rapid Q change
1a	■					
1b	■					
1c	■					
1d	■		■	■		
1e	■		■	■		
1f		■				
1g		■				
1h		■				
1i	■	■	■	■		
1j	■	■	■	■		
1k	■	■	■	■		
1l		■				
1m	■	■	■	■		
1n	■					
1o						
1p						
1q						
1r						
1s					■	■

Table 3.14 Flow components described by the eight fish indicators

Indicator number	Extreme and extended low Q	Enhanced and stabilised Q	Loss of small floods (<1yr)	Loss of large floods (>1yr)	Extreme or untimely large Q	Rapid Q change
2a	■		■			
2b	■	■		■		
2c					■	■
2d	■		■	■		
2e	■		■	■		
2f					■	■
2g					■	■
2h					■	■

Table 3.15 Flow components described by the seven macroinvertebrate indicators

Indicator number	Extreme and extended low Q	Enhanced and stabilised Q	Loss of small floods (<1yr)	Loss of large floods (>1yr)	Extreme or untimely large Q	Rapid Q change
3a						
3b						
3c						
3d						
3e						
3f						
3g						

Table 3.16 Flow components described by the 14 macrophyte, bryophyte and diatom indicators

Indicator number	Extreme and extended low Q	Enhanced and stabilised Q	Loss of small floods (<1yr)	Loss of large floods (>1yr)	Extreme or untimely large Q	Rapid Q change
4a						
4b						
4c						
4d						
4e						
4f						
4g						
4h						
4i						
4j						
4k						
4l						
4m						
4n						

Table 3.17 Flow components described by the two amphibian indicators

Indicator number	Extreme and extended low Q	Enhanced and stabilised Q	Loss of small floods (<1yr)	Loss of large floods (>1yr)	Extreme or untimely large Q	Rapid Q change
5a						
5b						

Table 3.18 Flow components described by the three riparian vegetation indicators

Indicator number	Extreme and extended low Q	Enhanced and stabilised Q	Loss of small floods (<1yr)	Loss of large floods (>1yr)	Extreme or untimely large Q	Rapid Q change
6a						
6b						
6c						

3.7 Practical application

Survey methods of each ecological indicator were considered to determine the practicality of their use (Table 3.19 to Table 3.24). The following tables are to inform follow on work that is recommeded to develop specific survey methodologies and to trial the use of ecological indicators in the field.

Table 3.19 Survey methods for the 19 physical indicators

Indicator number	Field method	Aerial survey
1a	Walkover survey	Y
1b	Walkover survey	Y
1c	Walkover survey	Y
1d	Walkover survey	Y
1e	Walkover survey, biological sampling	N
1f	Walkover survey	Y
1g	Walkover survey	Y
1h	Walkover survey	Y
1i	Walkover survey	Y
1j	Walkover survey	Y
1k	Walkover survey	Y
1l	Walkover survey	Y
1m	Walkover survey	N
1n	Walkover survey	Y
1o	Walkover survey	N
1p	Walkover survey	Y
1q	Walkover survey	Y
1r	Walkover survey	Y
1s	Walkover survey	Y

Table 3.20 Survey methods for the eight fish indicators

Indicator number	Field method	Aerial survey
2a	Electric fishing	N
2b	Electric fishing	N
2c	Electric fishing	N
2d	Electric fishing	N
2e	Electric fishing/netting	N
2f	Electric fishing/netting	N
2g	Electric fishing/netting	N
2h	Electric fishing/netting	N

Table 3.21 Survey methods for the seven macroinvertebrate indicators

Indicator number	Field method	Aerial survey
3a	Kick/sweep sampling	N
3b	Bankside sort. Kick/sweep sampling	N
3c	Bankside sort. Kick/sweep sampling	N
3d	Bankside sort. Kick/sweep sampling	N
3e	Kick/sweep sampling	N
3f	Bankside sort. Kick/sweep sampling	N
3g	Bankside sort. Kick/sweep sampling	N

Table 3.22 Survey methods for the 14 macrophyte, bryophyte and diatom indicators

Indicator number	Field method	Aerial survey
4a	Walkover survey	Y
4b	Walkover survey	Y
4c	Walkover survey	Y
4d	Walkover survey	Y
4e	Walkover survey	Y
4f	Walkover survey	Y
4g	Walkover survey	Y
4h	Walkover survey	N
4i	Walkover survey	Y
4j	Walkover survey	Y
4k	Walkover survey	Y
4l	Walkover survey	N
4m	Walkover survey	N
4n	Walkover survey	N

Table 3.23 Survey methods for the two amphibian indicators

Indicator number	Field method	Aerial survey
5a	Walkover survey	N
5b	Walkover survey	N

Table 3.24 Survey methods for the three riparian vegetation indicators

Indicator number	Field method	Aerial survey
6a	Walkover Survey	Y
6b	Walkover Survey	Y
6c	Walkover Survey	Y

3.8 Certainty of Ecological Indicators

The evidence base provided by the ecological indicators was assessed using an adaptation of the 'uncertainty approach' used in the UK National Ecosystem Assessment (UK NEA, 2011).

The 'uncertainty approach' consists of a set of uncertainty terms derived from a four-box model (Figure 3.1) This is a semi-quantitative analysis combining qualitative consensus among experts for the evidence supporting each indicator (on a scale from speculative to well established) with an estimate of the level of certainty of each indicator identifying the severe effects of water use in rivers.

The scale for describing the 'certainty' of each Ecological Indicator was:

Virtually certain: >99% probability of occurrence
Very likely: >90% probability
Likely: >66% probability
About as likely as not: >33-66% probability
Unlikely: <33% probability
Very unlikely: <10% probability
Exceptionally unlikely: <1% probability

This model placed each of the indicators into one of four boxes:

Well Established: There is high agreement based on significant evidence. Ecological Indicators that can be used individually or in combination with an acceptable level of certainty.

Established but Incomplete: There is high agreement based on limited evidence. Ecological Indicators that have good potential, but are novel or not supported by much evidence. These indicators are recommended for further development, provided that they offer the potential to complement, improve upon, or provide a low cost alternative to, existing established indicators.

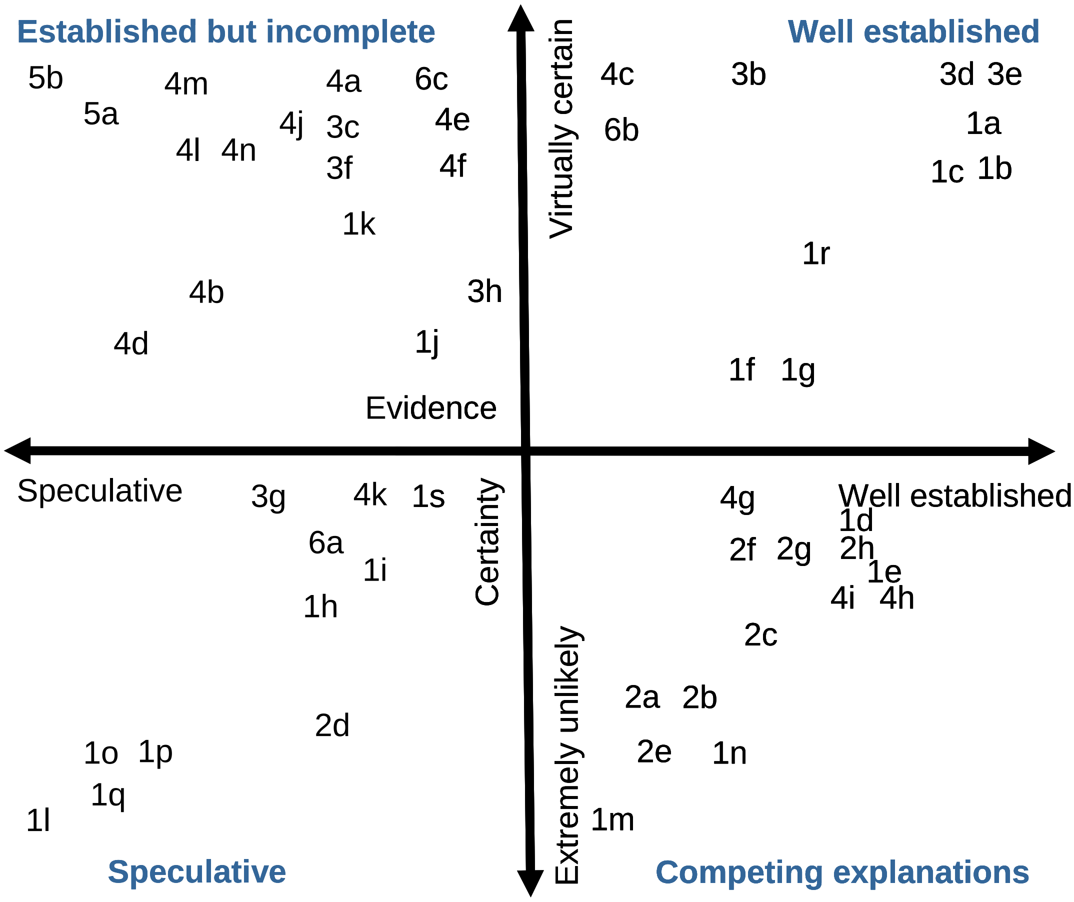
Competing Explanations: low agreement, albeit with significant evidence. Ecological Indicators that have reasonable supporting evidence but must be used with caution, for example due to their being responsive to other confounding factors, or being highly site-specific in their application. Used with caution, these may be useful in combination with other indicators to form a weight of evidence, particularly if the indicators are can be deployed at low cost.

Speculative: low agreement based on limited evidence. Ecological Indicators that are not well supported by evidence and are not considered to provide an acceptable level of certainty on their own, but may be useful in combination with other indicators to form a weight of evidence (particularly if the indicators are can be deployed at low cost).

This analysis identifies some useful tendencies:

- A grouping of potentially complementary, well-established indicators of habitat character and abundance, based around macroinvertebrate indices of character (LIFE and associated flow groups), and abiotic indices of habitat scale.
- A grouping of promising abiotic indicators specifically applicable to impoundment problems on gravel and cobble bed (typically upland) rivers.
- A grouping of promising potential indicators, which include further abiotic and macroinvertebrate indicators, but are largely comprised of macrophyte indicators of terrestrial succession, and bryophyte and diatom-based indicators. Bryophyte and diatom indicators in particular were well received by consultees at the Expert Worksop.
- A grouping of largely fish and aquatic macrophyte-based indicators that are well established, but for which there are difficulties in interpretation, and often relatively high survey costs.
- A grouping of (mostly) abiotic indicators, which although speculative individually, may be surveyed together at relatively low cost, and potentially across large spatial scales. Collectively, these indicators may add to the weight of evidence to support existing classification tools, or to help prioritise further investigation using more certain indicators.

Figure 3.1 Ecological Indicators uncertainty analysis



Key

Physical indicators
1a Loss of wetted channel
1b Fragmentation of aquatic habitat
1c Loss of riffles/ preponderance of pools
1d Fine sediment
1e Sediment plume
1f Absence of gravels
1g Bed armouring
1h Submerged substratum
1i No active bars
1j Stable bars
1h Terrace formation
1l No exposed marginal substrate
1m Bank gradient< vertical
1n Low width: depth ratio
1o Tributary terraces
1q Exposed tree roots in tributary channels
1r Active bars d/s tributary confluences
1s Gravitational bank collapse

Fish indicators
2a 0+ to 2+ trout and salmon absent
2b Increased growth rate of trout
2d Absence of adult salmon/ migratory trout in autumn
2e Plant: gravel spawning coarse fish
2f 1st summer recruitment of phytophilic coarse fish
2g Winter survivial of phytophilic and lithophilic coarse fish
2h 1st summer survival of phytophilic and lithophilic coarse fish

Macroinvertebrate indicators
3a Reduction in taxon richness
3b LIFE O/E ratio
3c Large bodied predatory invertebrates
3d Abundance of LIFE flow group V and VI species
3e Absence of LIFE flow group I and II species
3f Presence of winterbourne specialists
3g Absence of baetid mayflies
3h Dominance of *Gammarus spp*.

Macrophyte, bryophyte & diatoms
4a Mosses and liverworts on exposed cobbles
4b Domiance of emergent plants
4c Dominance of terrestrial plants
4d Dominance of perennial terrestrial plant species
4e >10% terrestrial plant cover on bars
4f >10% perennial terrestrial plant cover on banks
4g Filamentous algae on channel bed
4h Dominance of *R. peltatus*
4i Absence of submerged aquatic macrophytes
4j Presence of free floating macrophyte species
4k Dominance of rooted species
4l Dominance of aerophilic diatom taxa
4m Presence of diatomaceous biofilms
4n Relative abundance of motile diatom taxa

Amphibian indicators
5a Presence of tadpoles
5b Presence of newts

Riparian vegetation indicators
6a Loss of aquatic Sphagna
6b Loss of wetland species
6c Depth and extent of water during wet months

4 THE OPTIMISATION FRAMEWORK

4.1 Introduction

Flow releases from reservoirs for environmental benefit are common and are usually some combination of steady compensation flows and freshets, designed for fish passage, angling purposes or enhancement of water quality in critical downstream areas. For example on the River Tyne, Northumberland, releases from Kielder Reservoir are designed to accommodate variously fisheries, canoeing, channel flushing and maintaining estuarine water quality (Gibbins and Acornley, 2000; Archer et al. 2008).

In most cases the release regimes are not based on any closely formulated protective environmental principles and Gustard et al. 1987 (cited in SNIFFER, 2007) reported that 70% of reservoirs in the UK released a constant compensation discharge during the year. Releases to protect multiple ecological elements are even less explicitly addressed than different uses; although Gibbins and Heslop (1998) give an example where a water transfer was predicted to benefit two fish species.

Moreover, there is very little monitoring of artificial release regimes (Souchon et al. 2008), at least in the British Isles, so the benefits are mostly conjecture. Thus, in most cases there is no optimisation of the procedures. Optimisation implies some feedback into operational practice from monitoring; in other words adaptive management, and clear identification of what ecological elements are being optimised.

4.1.1 Starting point

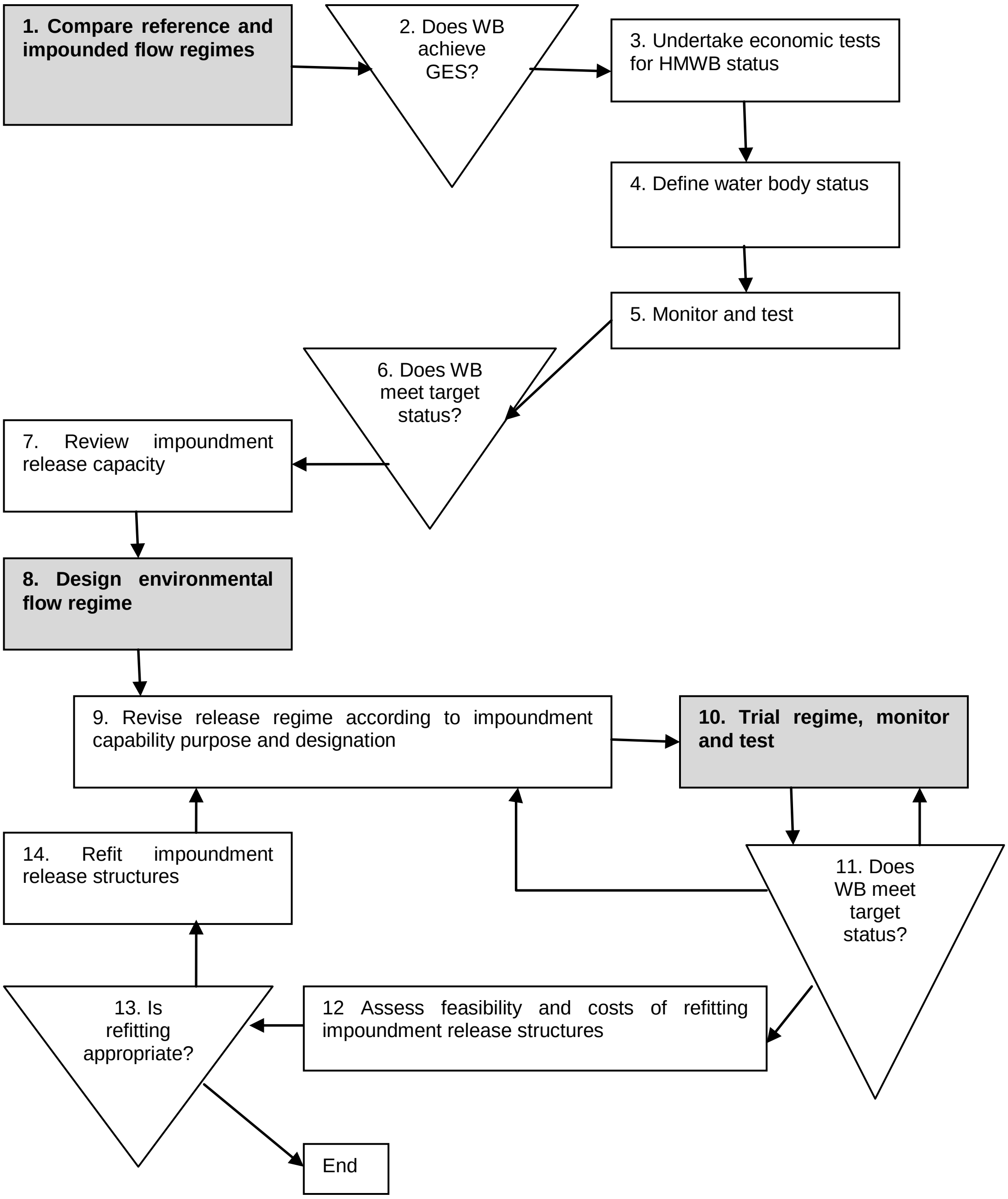
WFD 82 set out a 13-step process for setting flow releases from impoundments (Figure 4.), based on the Building Block Methodology. The optimisation framework focuses on Step 8 - Design environmental flow regime.

The intention of Step 8, for which WFD 82 also provided advice on implementation, is expressed as follows:

“For GES (GEP in the case of SNIFFER WFD 21D, see below) , the key activity of this step is to determine which elements of the natural flow regime (Floods, freshets, medium flows low flows) are important for the river ecosystem downstream. The selected elements need to be specified in terms of their magnitude duration timing and frequency and combined to give an ecological flow release. Ideally this will be achieved from knowledge of the species that are present (or should be present) and their flow and associated habitat requirements in terms of, for example, temperature, sediment concentrations and oxygen levels.”

WFD 82, following the WFD, recommends “For GEP, a flow regime that will achieve an ecological status similar to the best examples of similar reference conditions...” but notes that it may not be appropriate simply to transfer the flow regime from the reference site but (the flow design) will involve an iterative process of determining the elements of the flow regime (that) are important for the river ecosystem downstream”.

Figure 4.1 - Flow chart for setting flow releases from impoundments, Redrawn from WFD 82; SNIFFER, 2007). Grey shading indicates where the Optimisation Framework links with the WFD 82 process

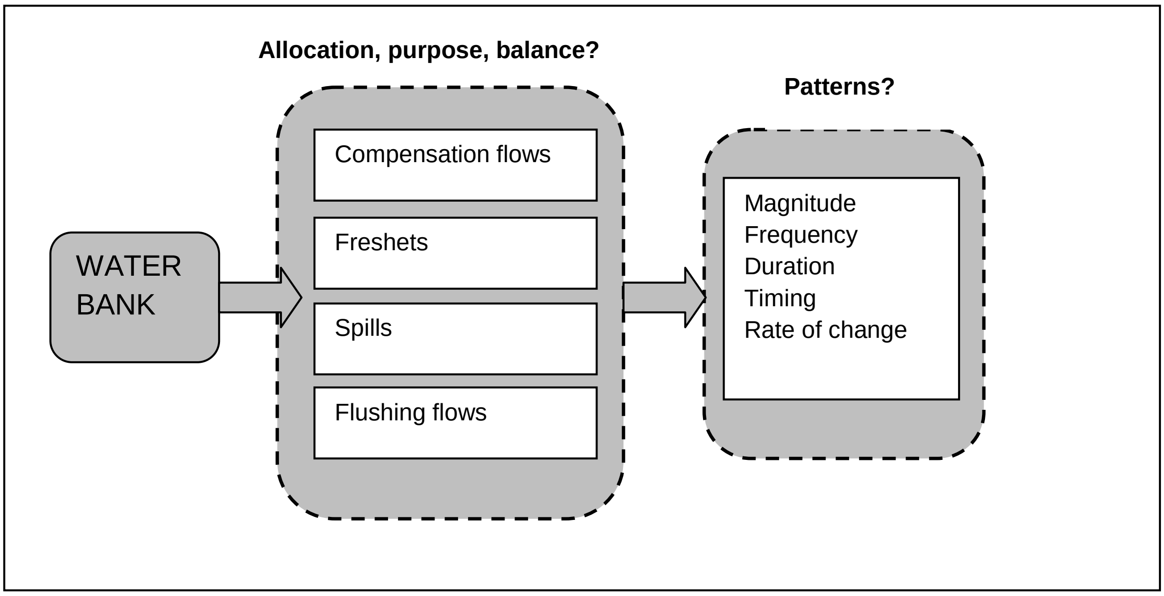


4.1.2 Scope of the optimisation framework

The optimisation framework sets out a generic decision framework with which to meet the requirements of Step 8. It is based upon WFD 82 (SNIFFER, 2007; Acreman et al. 2009) and the Building Block Methodology that underpins it, but is expressly designed to use the conceptual models developed in Section 2, and considers further work undertaken since WFD 82. It does not include for the use of ecological indicators developed in Section 3, because the usefulness of these indicators has not yet been demonstrated in real applications.

The optimisation framework describes a framework for defining how the release of water from impoundments can be optimised to reduce adverse ecological impacts and to enhance the ecological potential on HWMBs. The work leads directly from the recommendation in report WFD 82 (SNIFFER, 2007; Acreman et al. 2009) which outlined a broad approach that is followed and developed here. The aim can be interpreted simply as how to allocate water releases in order to optimise the ecological benefits of limited water banks in the face of competing ecological demands (Figure 4.2).

Figure 4.2 - The principal operational decision questions to optimise ecological benefits



This necessarily limits the scope of the optimisation framework.

- Ideally, goals for optimisation should define both the environmental and social conditions that, when achieved, would constitute success (Richter and Thomas, 2007). This is beyond the scope of this report.
- The optimisation framework considers flow changes downstream of reservoirs only. It does not explicitly consider changes to water quality, thermal or sediment regimes caused by the impoundment, or any impacts on or upstream of the reservoir.
- Water release profiles should be defined for scenarios considered achievable given the water bank or use of the impoundment, physical constraints of release structures and scope for varying supporting legislation. This implies some understanding of the operation of the impoundment, and its use with other, linked sources. Scope for changes at the impoundment(s), however, is taken as having been assessed at Stage 7 of the WFD 82 process, and is beyond the scope of this report. Potential re-engineering strategies are elaborated upon by Richter and Thomas (2007).

4.1.3 International practice

In seeking how to best define the water flow needs for rivers in HMWBs, Bradford et al. (2011) identified two broad schools of thought. First is the paradigm of a monotonic relationship between the degree of hydraulic alteration and the disturbance to the ecology (Richter et al. 1997).

The primacy of the natural flow regime has intuitive, theoretical appeal and is the logical starting point when faced with a lack of information (Arthington et al. 2006). Some recent literature also supports an approach of constraining reservoir releases to metrics of hydrological variability, either through selection and explicit relationship to biotic data, or maintaining aspects of flow variability as an implicit means of maintaining aquatic ecology. For example, presenting a consensus approach amongst numerous international river scientists, Poff et al. (2010a), offer a framework for selecting from a range of hydrological indices and relating these to biotic variables. Shiau and Wu (2004) constrain flow releases to maintain variability to within hydrological limits based on the Range of Variability Approach (Richter et al. 1997). Yin et al. (2011) report on impoundment operation to synchronise reservoir releases and constrain loss of variability using telemetered data in real time. Nevertheless, the hydrological indices are more often used to assess flow changes than to design releases; generally, "the process of determining environmental flows does not involve attempting to devise a regulated flow regime that has a statistically defined variability across all time scales identical to that of the (natural) flow regime" Gippel (2001).

The second school of thought begins with relatively specified management goals (e.g. abundance of key or valued species), and uses knowledge of their life histories and habitat needs to build up a skeleton flow regime to meet these goals (e.g. Tharme, 2003, Enders et al. 2009; Acreman and Ferguson, 2010). The second approach has become known as the Building Block Methodology (King, 2008), and forms the basis of the methodology described in SNIFFER research project WFD 82 (SNIFFER, 2007; Acreman et al. 2009).

The two methods are not founded on different science; both can be traced back the natural flow paradigm (Poff et al. 1997) and they are not mutually exclusive. Indeed, Petts (2011) demonstrates the use of these alternative strategies in the operation of a single reservoir system (using a 'preferred' strategy of maintaining natural flow variability during periods of high storage, a 'basic' strategy of meeting specific ecological functions in normal years, and a 'minimum' strategy of ecosystem protection during dry periods.)

Accordingly, both approaches offer value in optimisation routines. However, the second approach offers a more pragmatic, widely applicable way to specify artificial flow regimes in the face of heavy water demands, competing ecological and other usage, restricted release flexibility, absence of real-time control, sparse biological data and the need for transparency in dealing with diverse user groups. This is especially so as the benefits of maintaining the full range of natural flows have been questioned by the few studies that have measured the benefits (e.g. Jowett and Biggs, 2006; Bradford et al. 2011).

The consistent recommendation in modern river regulation studies is for local solutions to be based upon local information coupled with effective monitoring and adaptive management (Souchon et al. 2008; Poff et al. 2010b).

Typically (e.g. King et al. 2008), are allocated to ensure:

- a minimum low flow component to maintain habitat throughout the year;
- higher 'maintenance' flows to meet ecological flow needs at some times of year;
- freshets to stimulate fish migration; and
- and flood flows to flush river sediments, maintain floodpain connectivity and esure continued evolution of channel form.

Recently, however, (e.g. Petts, 2009), increasing emphasis has also been placed upon maintaining flow variability, at least between seasons and years.

Because of the uncertain and inter-disciplinary nature of the problem, involvement of interested parties and relevant specialisms throughout the process is generally advocated in a holistic appraoch, following the 'expert panel' approach of the influential Building Block Methodology. Notably, the Downstream Response to Imposed Flow Transformations (DRIFT, King at al. 2010) methodology, an alternative to the influential Building Block Methodology (King, 2008) also developed in South Africa, differs from its predecessor in that it emphasises an exploratory, scenario-driven approach in preference to the derivation of a single prescriptive solution.

4.1.4 Approach

The optimisation framework is broadly consistent with modern international practice cited in the literature, for example King et al. (2010), Souchon et al. (2008), and Petts (2009), and represents an evolutionary development of the guidance set out in WFD 82, rather than a revolutionary change in direction.

The Building Block Methodology assumes dominance in the approach, being used to derive an initial condition whilst undertaking the monitoring that in the longer term may allow explicit relations between flow and biotic data (e.g. Poff et al. 2010b) to be derived or for the success of management intervention to be established in biotic terms. More natural flows are a secondary target, to be worked towards through iterations of adaptive management, but it is accepted that flows downstream of some impoundments may never achieve a near-natural condition, however that is defined.

Given the uncertainties in the underpinning science, the approach identifies and prioritises risks and flow needs for the chosen habitat or ecological element, and rather than focussing upon formal objectives (e.g. WFD standards), identifies risk areas resulting from potential flow modifications.

Locally-specific, bottom up solutions are recommended, based upon scenario exploration with the current release flow regime as a starting point from which to attempt targeted augmentation or reallocation from the existing water bank. A 'top down' approach, in which existing features of a natural hydrograph are selectively removed, was considered to be of more limited application, although is perhaps more appropriate where a new impoundment is proposed.

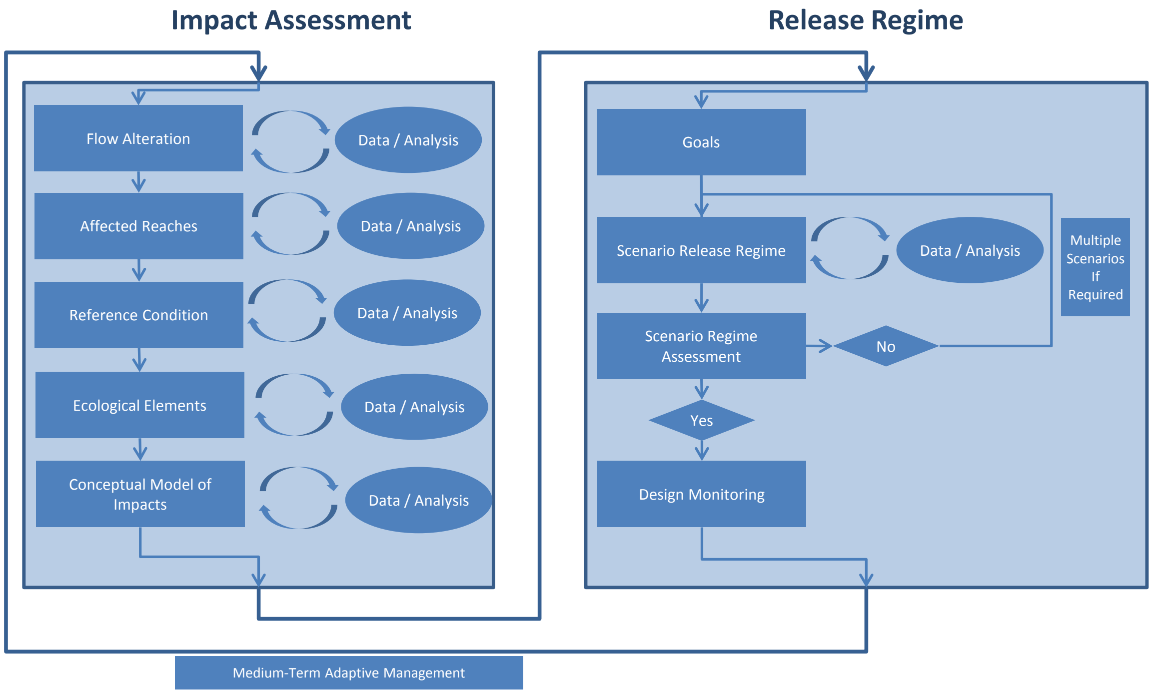
Features that are not present in the optimisation framework are an explicit mechanism for stakeholder engagement and issue resolution, or consideration of wider issues, such as the social and economic aspects included in methods such as the Building Block Methodology. The optimisation framework is not intended as a standalone process in this respect. Rather, it is anticipated that it would be used with existing procedures of the UK and Northern Ireland regulatory agencies, with priorities between ecological elements and scope for issue resolution determined as appropriate to the driver of the study.

4.2 The optimisation framework

4.2.1 Overview

The optimisation framework is illustrated in Figure 4.3.

Figure 4.3 - The optimisation framework



The framework provides a means by which the conceptual models can be used to inform decision-making regarding enhancements to the flow regime downstream of impoundments.

Design of an initial release regime is divided into two stages; development of a local conceptual model to determine potential causes of impact, and use of this conceptual model to design scenario releases. Each stage is further subdivided into a number of steps.

The optimisation framework operates at two timescales; scenario exploration to derive a best initial estimate (potentially with iterations to make use of more data – intensive tools); and progressive refinement of the release regime over river basin cycles.

The framework is presented as a linear process for clarity, but in practice steps might be undertaken in parallel, or in an alternative sequence, or the process used iteratively to refine initial qualitative estimates in the light of quantitative data as best fits local circumstances Therefore it is not intended as a prescriptive 'step by step' procedure.

The framework permits the use of a variety of hydro-ecological methods with which to determine the flow needs of the ecological elements. It does not attempt to prescribe a fixed order for the use of these, as for example, presented in DRIFT (King et al. 2010). This is because, whilst some solutions rely upon making explicit linkages between the components of the conceptual model (hydrology – hydraulics – geomorphology – ecology), other solutions (e.g. ELOHA (Poff et al. 2010a)) seek more direct linkages between hydrology and ecology.

In general, greater accuracy in determining and mitigating ecological impacts is achieved along two axes:

- desk studies are of lower accuracy than detailed quantitative modelling/ monitoring; and
- hydrological standards are of lower accuracy than hydraulic techniques, which are in turn of lower accuracy than estimates based on biological data.

Further detail is given in Table 4.1, below.

Table 4.1 Advantages and disadvantages of hydro-ecological tools of potential use within the Optimisation Framework

Method	**Advantages**	**Disadvantages**
Hydrological (general)	Expressed in terms of volume or discharge, which facilitates use in water management.	Flow is considered a proxy for biotic effects. There are few well-established quantitative links between hydrological effect and ecological impact. Changes needs to be considered in terms of natural differences in flow regime, often necessitating derivation of an estimated natural flow regime.
Desktop Natural flow percentiles (scaling, Low Flow 2000) and time series (scaling, CERF) can be readily estimated from catchment characteristics, allowing derivation of Richter IHAs (or alternative descriptors) or LF2000 proxies.	Inexpensive. Can be widely applied for screening.	Short records or uncertainties in hydrological time-series can have a significant effect on derived Richter IHAs. There is no universally accepted scheme for summarizing flow variability in an ecologically meaningful way.
Site-specific Greater accuracy in determining actual or scenario flows can be achieved through locally-specific measurement / modeling.	Reduces the uncertainty in hydrological characterization.	Uncertainties in scenario timeseries can still have a significant effect on IHAs. More accurate characterization of the hydrological regime does not necessarily result in improved definition of ecological effects.

Hydraulic/ geomorphological (general)	Representation of hydraulic response to hydrological change. Bed response can be included with hydraulic formulations.	Additional expense. Assumes that biological communities are adapted to meso-scale habitats. Reliant upon inferences of habitat suitability that are often narrowly defined and of limited transferability between rivers. Complexity of linkages and of temporal and spatial scale relationships between hydraulic/ geomorphological behaviour and biotic response not represented. Substrate often assumed static and longer term channel/ macrophyte responses typically excluded.
Desktop Hydraulic variables and habitat suitability can be estimated using generalized statistical relationships (e.g. RAPHSA) and combined with flow regimes to produce habitat regimes.	Least expensive form of hydraulic analysis. Can be built upon with local data.	Large uncertainties in generalizing hydraulic response. Type-dependence may restrict application. Geomorphology/ bed response would require separate investigation or assumed static
Site specific (I) Hydraulic rating using transect data can be used to augment/ replace desktop estimates.	Includes local detail, reduces uncertainty.	Difficult to adequately represent spatial variability within and between reaches. Geomorphology/ bed response would require separate investigation or assumed static
Site specific (II) Hydraulic modeling	Detailed description of hydraulic response. 2D solutions can achieve broad spatial representation between and across transects.	Expensive and application therefore tends to be confined to short reaches, which may not be representative. Would require a full 3-D solution to fully characterize differences within the water column/ hydraulic behaviour at the bed. May underestimate habitat diversity as smaller/ temporary habitat patches

		(from woody debris, etc.) not represented. Geomorphological responses uncertain, especially over longer timescales.
Biotic	Directly measures impact rather than surrogates of it.	Requires local data that are typically restricted to a few biological quality elements from which overall ecological status is inferred. Existing standard methods for sampling biological quality elements are designed for water quality assessments and are not always sensitive to measuring hydromorphological effects.
Assessment of current status	Directly measures impact rather than surrogates of it.	Requires biological monitoring, allowance for temporal variability, spatial representativeness and river type. Specific uncertainties depending upon the biotic element considered.
Prediction of scenario status	Directly estimates impact rather than surrogates of it.	Few generalized models/ relationships available. Relies upon establishing valid relationships for predictive use, which may require long-term data, pooled across many sites, and an improvement in the knowledge base. Many site specific solutions based on conceptually-led empirical relationships rather than representation/ understanding of underlying hydro-biological mechanisms.

4.2.2 Impact assessment

4.2.2.1 Flow alteration at the impoundment(s)

The magnitude, duration, timing, sequencing, frequency and rate of change of hydrologic alterations should be identified, and expressed in terms of the ecological flow components:

- Extreme and extended low flows
- Enhanced and stabilised flows
- Loss of small floods (<= 1yr)
- Loss of large floods (>1yr)
- Extreme or untimely High flows
- Rapid Flow change

Grade the effect of the impoundment on each ecological flow component:

- Highly significant impact – e.g. low flows or floods missing or greatly reduced
- Significant impact – e.g. low flows or floods significantly reduced or enhanced
- Minor alterations – relatively small changes from natural
- Unaffected – no change

Any differences in the degree of effect at different times of year should be noted to assist with prioritisation of flows at the design stage. For example, downstream of a reservoir with a steady compensation flow that also frequently spills in late winter, flow variability might be considered missing or highly affected during months when spills are unlikely, and significantly reduced during late winter.

Note that Richter indicators of hydrological alteration - or LowFlows2000 proxies of them (SNIFFER, 2008) - can be used to support this categorisation. Although only offered as putative thresholds, SNIFFER (2008) offered boundaries of:

- <40% in any Richter indicators of hydrological alteration - low risk of failing to meet GES;
- 40% - 80% change in any Richter indicators of hydrological alteration – medium risk of failing to meet GES; and
- >80% change in any Richter indicators of hydrological alteration – high risk of failing to meet GES.

Ecological flow components that are unaffected, or only affected to a minor degree, might be disregarded, to better focus on the main issues.

4.2.2.2 Affected reaches

In some cases the downstream limit of hydrological alteration may be easily defined, for example where hydrological changes at the impoundment are minor compared to downstream watercourses.

Where the downstream limit is not easily identifiable, Richter's indicators of hydrological alteration or their LF2000 proxies (SNIFFER, 2007) offer an objective starting point, using the WFD82 standard of <40% change in any of the indices or their proxies. However, given the limitations of the LF2000 proxies and the 40% standard (SNIFFER, 2008, and Section 2.5.1.2), the downstream limit should be corroborated against gauged flows. If available, biotic data, and a conceptual understanding of the functioning of the river basin in which the impoundment is sited, should also be used to corroborate the extent of potential impacts.

Where hydrological impacts extend far downstream, the affected river length may be subdivided where the hydrological effect, channel morphology or species or ecosystem composition create significant changes in the degree or type of impact. Bottlenecks (e.g. barriers to migration) and distinct features (e.g. salmon spawning sites) should also be identified, if known.

4.2.2.3 Reference condition

The conceptual models should be referenced to natural (or naturalised) variation, either from a catchment history (Mika et al. 2010) or with reference to an appropriate comparator. This may include comparison with reservoir inflows.

Identify the ecological elements present in the reference waterbody. This should include any designated species, the flow-sensitive WFD biological elements and any particularly important species which may be targeted when re-designing the flow regime.

4.2.2.4 Ecological elements

Identify the ecological elements in the affected reach(es) and, by comparison with the reference condition.

Final prioritisation is not necessary at this stage, and may be counter-productive, but initial priorities will help reduce the range of ecological elements to be considered in subsequent stages. Registering the importance of different elements to various stakeholders might also be undertaken at this stage.

4.2.2.5 Conceptual model of impacts

Assess how changes to ecological flow components affect habitat, referencing the process diagrams for relevant ecological flow components. Differentiate, where possible, between the main effects, and subsidiary effects. Specifically (if possible with available information), prioritise between the relative significance of in-channel, riparian and hyporheic effects, and (again, if there are data to support this), whether reductions in discharge result chiefly in a loss of depth and maintenance in velocity (i.e. a miniaturisation of habitat), a loss in velocity (a change in the character of habitat), or a combination of the two.

Consider the degree to which the different habitat effects are likely. For example, for a site with leakage but no compensation flow, consider whether drying is likely, or whether leakage will sustain some flow in the downstream watercourse.

Consider also how differences in the timing of hydrological changes affect the timing of habitat effects.

Consider how changes in habitat are likely to affect relevant ecological elements. The generic process descriptions map habitat effects to relevant ecological elements and provide an estimate of the sensitivity of biota to the habitat alteration.

Potential risks to relevant ecological elements can now be identified by combining the degree of hydrological and habitat change and the sensitivity of the biota to these effects. As a default, categorise these risks according to Table 4.2. This can be overridden in the light of local judgement as required.

Note that the assessment of habitat effects can be supported by site data collection, for example with the ecological indicators, transect data or (a high detail, high cost solution), hydraulic modelling.

Table 4.2 Risk matrix for combining the magnitude of change in ecological flow components with the sensitivity of biotic receptors

	Ecological Flow Component Missing or highly affected	Ecological Flow Component Significantly reduced	Minor alterations to Ecological Flow Component	Ecological Flow Component Unaffected
Sensitive (-ve change)	Very High impact	High impact	Moderate impact	Negligible
Moderately sensitive (-ve change)	High impact	Moderate impact	Low impact	Negligible
Neutral	Negligible	Negligible	Negligible	Negligible
Moderately Sensitive (+ve change)	Beneficial	Moderately Beneficial	Slightly beneficial	Negligible
Sensitive (+ve change)	Very Beneficial	Beneficial	Moderately beneficial	Negligible

Consider whether risks vary at different times of year. The impact tables for the relevant ecological elements can be used for this purpose, and coloured according to the scheme above to define the timing of risks.

Tables can be used for each reach as appropriate, though it is cautioned that too fine a breakdown of the river network may not be justified by the precision of the assessment, and may also result in unmerited complications in balancing needs across impacts.

Note that the assessment of biotic impacts can be supported by site data collection, for example with the ecological indicators, or more comprehensive biotic survey and analysis. Where biotic, habitat or hydraulic data has been used risk tables might be replaced by more quantitative analysis. This should include the spatial extent of timing of impacts.

4.2.3 Release regime

4.2.3.1 Goals

The impact tables identify a range of competing, and potentially conflicting, flow needs. Priorities should be targeted at achieving the best ecological return, but defining this is a matter of policy and only general guidance can therefore be offered in this report.

Priorities should be expressed qualitatively (Gippel, 2001). For example, “to restore salmon populations in named reaches”, and may include subsidiary priorities, for example, whilst “maintaining the condition of riparian wetlands at a specified location”.

Priorities need to be established between species, other taxonomic groups or ecosystem level functions and processes. There is a balance to be achieved, for example, where the release of freshets in late spring/early summer to encourage the downstream migration of salmon smolts may cause negative impacts on rare invertebrates on exposed riverine sediments.

Where species are targeted, priorities also need to be established between life stages. For example, salmon life stages (eggs, fry, parr) occupy a range of meso-habitats and in typical rivers the absolute and relative abundance of these will change differentially as flow changes. The degree of obstruction presented by barriers for migrating adults will vary at yet other flows. Therefore a given flow release pattern will benefit some life stages more than others, and flows need to be targeted at those life stages that are a barrier to recovery.

Priorities also need to be expressed between reaches. This may arise between different reaches downstream of an impoundment, or may balance the needs of completely different catchments where water transfers are made, or where multiple sources are linked in a network.

An appropriately risk-averse strategy might simply recognise:

- the need to protect the most sensitive life stages of the various biota, or key ecosystem functions, if they can identified';
- the need to avoid the risks of major impacts from certain obviously damaging practices such as extended extreme low flows, extreme high flows and rapid changes in flow rate, without specifying tightly what they should be; and
- the opportunities where flow releases may produce mutual benefits for multiple ecological components where there are obvious interactions between species that require consideration.

Priorities should also be accompanied by the intended means of achieving them. For example, "salmon populations will be improved by restoring longitudinal connectivity during salmon migration and re-mobilising channel bars". The assumption at this stage is that the above will be achieved by reversing the impacts of the flow alterations.

4.2.3.2 Scenario release regime

Compensation and freshet releases should be scaled according to general guidance in the conceptual model (Section 2.7), or to species specific guidance; for example, guidance for freshet releases for salmonids (Appendix I.9).

Compensation and freshet releases should be designed for an average year, and variability achieved by varying this base regime for wet years and dry years. A trigger is also required for switching between flow patterns.

Where possible, the timing and scaling freshets should be coincident with natural inflows, but this requires considerable flexibility at the impoundment, and knowledge of inflows in real-time. It is therefore likely to be applicable to only a minority of sites. Even where applied, it is also likely that an element of flow design would be required to ensure ecosystem or target species viability in the downstream reaches.

Where hydraulic approaches have been used, local optima in a flow weighted useable area relationships or secondary breakpoints in a flow - wetted perimeter relationship may provide a basis for wet or dry year flows. Alternatively, variability might be established by adopting different strategies during wet, normal and dry years, following the approach of Petts, 2011 (Section 4.1.3), or with reference to the natural variability in monthly flows. Note, however, that an element of design will be needed to ensure that 'dry' year flows do not result in extinction flows. Note too that if a flow measure is used, the variability may not correspond to equivalent variability in habitat or proxy variables due to non linearity and discontinuity in the underlying relationships.

Once building blocks are identified they should be assembled into a proposed skeleton flow regime.

The scenario regime should be assessed using the impact tables and conceptual models to determine likely improvements to the target ecological elements. Explicit reference to abiotic requirements should also be made – for example, determining whether the pattern of freshets and spills will affect any mobilisation of fines considered necessary. Note that reinstating flow conditions to their pre-impact state may not necessarily reverse geomorphological changes or ecological impacts and consider any long-term morphological changes required for sustainable recovery, as well as the shorter–term ecological responses that may be achieved within the existing channel structure.

The potential for flow releases causing unintended consequences, and limitations presented by other anthropogenic influences in the catchment should also be considered.

Process linkages between corrective flow measures, habitats and ecological elements by reference to, in order:

1. The process diagrams (Appendix II.1.2).
2. The description of the targets' (in this case salmon and trout) life history and links with flow-related habitat (e.g. Appendix I.9).
3. The impact tables. (e.g. Appendix II.1.3).
4. The detail of the habitat requirements (e.g. Appendix I.9).

The scenario regime should be also assessed to examine any unforeseen consequences of the new regime. This should make reference to the impact tables, in the first instance, and if necessary the conceptual model evidence base, for relevant ecological elements.

Consideration should also be made of whether the new flow regime is more natural than the old. This may require a simulation of the effect of changes to compensation flows on spills.

The water requirement for this regime should be calculated for average, wet and dry years, and (if appropriate) compared with any existing waterbank. This may identify the need for reductions in this regime, or any spare volume available for further augmentation.

Once a scenario is assessed, it may need refinement, for example, to keep releases within the overall water bank, or to balance effects at one time of year against those in another.

4.2.3.3 Choose a preferred release regime

Numerous scenario releases can be defined. The final regime should be the one which is judged to best meet the goals outlined at the outset of the process. This is a water management, not a technical decision, and no further guidance is offered here.

4.2.3.4 Design monitoring

The procedures outlined above are intended to be carried out, wherever possible, in an adaptive management framework (Hilborn and Mangel, 1997; Hilborn and Walters, 2001; King et al. 2010). Monitoring to properly assess the benefits is essential and needs to be done a level that provides relevant, useable (that is, scientifically robust), results and conclusions. Souchon et al. (2008) give a comprehensive and recommended framework for designing such schemes. In the context of ecological flow optimisation, two levels of monitoring can be envisaged:

Compliance monitoring: carried out when the aim is simply to be sure that the flow regimes intended are in fact delivered and that the ecological status is not suffering deterioration. This is de minimus monitoring, that should be undertaken on all schemes.

Ecological target monitoring: carried out to measure and assess the intended ecological responses. This is the monitoring that offers the information benefits that are achievable with adaptive management. It should be designed to meet clear scientific objectives and to advance collective understanding.

The value of good monitoring has been stressed many times in the context of river flow impacts (Souchon et al. 2008; Sabaton et al. 2008; Arthington et al. 2006; Milner et al. 2011; Bradford et al. 2011) and without it the iteration needed to achieve optimisation cannot proceed in any informed way. Not all schemes may be appropriate for monitoring. Reasons not to carry out investigative monitoring might include:

- the scheme outcome is too small to justify costs;
- the scheme is one of many similar schemes, some of which are monitored, and further replication would be wasteful; and
- the resources available or the logistics of the site will not provide data of sufficient quality.

Monitoring is not a trivial task and the recommended approach is to promote collaborative projects involving scientists from government agencies, consultancies or universities. Moreover, the benefits of the work will be enhanced if they can be combined across contrasting types and sizes of schemes and rivers, and if the methods used and aims can be expressed in common form, enabling later meta-analysis.

Monitoring is aimed at operational applications that may legitimately tightly constrain the aims. However, that should not erode the design principles of the surveys, which will need to consider the issues of replications and controls. Before-after-control impact (BACI) (Stewart-Oaten, Murdoch and Parker, 1986; Stewart-Oaten and Bence, 2001) provides an optimised framework for undertaking monitoring within the adaptive management context (Downes et al. 2002). There are many accounts of how to design such studies (see for example Downes et al. 2002; Hilborn and Mangel, 1997; Sedgwick, 2006; Quinn and Keough, 2002) which will not be repeated here; but the key message is that when it is done it needs to be done well: economy surveys are almost always of poor value in the long run, because the data interpretation is compromised.

However, monitoring in adaptive management contexts may not lend itself to conventional statistical design, replication is clearly difficult where large dams are concerned and more innovative approaches involving modelling, aimed at setting out uncertainties unambiguously may be more appropriate or complementary (see Walters, 2007; Keith, et al. 2011).

4.3 Case Study

4.3.1 Introduction

This case study applies the optimisation framework to an imaginary assessment typical of upland reservoir management. It does so in order to illustrate how information in this report might be used in practice to develop an optimal flow regime for a site subject to flow modification from a dam, and is entirely hypothetical; no data have been used to support this example.

The case study envisages flow regime changes downstream of a small upland impoundment which is considered to cause significant impacts to the river downstream, but this assessment has been made in the absence of local biological data.

There is considerable flexibility for changing the pattern of flow releases at the impoundment, but the reservoir is operated as a standalone source, offering lower flexibility in operation that would be the case with a more integrated system.

4.3.2 Impact assessment

4.3.2.1 Flow alteration

The reservoir is a direct supply reservoir, water being diverted to another catchment. There is no compensation flow, but in the absence of spills, a minimal flow is maintained downstream by leakage from the dam and catchment inflows that join the watercourse almost immediately downstream. Spills are restricted to the winter months, with reduced likelihood of overtopping during the late autumn and early winter, and no summer spates. A scour valve is also operated for testing, mostly in the winter months, but otherwise the dam is a barrier to sediment and nutrient transport downstream.

The alterations to flow are extreme/ extended low flows and a loss of freshets and small floods. Low flows are not absent, but they are considered highly affected during extended periods from late spring to early winter, and significantly reduced during the late winter, when leakage and catchment contributions tend to be greater. Freshets and small floods are considered significantly reduced during late winter and highly affected at other times.

Given that the reservoir makes no compensation releases, the possibility of enhanced and stabilised flows, or of extreme or untimely high flows, can be discounted. Note also that although the effect of the dam on sediment transport provides useful context, the effects of this are beyond the scope of the optimisation framework.

4.3.2.2 Affected reaches

The reservoir is located on a tributary which joins the (much larger) main river 5 km downstream. LF2000 proxies suggest that the main river is unlikely to be affected by >40% changes on the tributary.

The tributary is a steep, cobble bedded channel likely under natural flows to contain riffle-pool sequences. Longitudinally, there are no obvious barriers to fish movement and there are no changes in morphological character along the 5 km reach. Extensive exposed bed, channel features and margins are apparent.

There are few biological data on the reach, but anecdotal evidence from anglers suggests that fish communities are limited to small brown trout (*Salmo trutta*) in isolated pools, connected by minor trickles.

The affected reach is therefore considered to be a single 5 km reach between the impoundment and the confluence with the main river.

4.3.2.3 Reference condition

Rivers in neighbouring catchments are similar in character, have a relatively natural flow regime, and are not thought to be significantly affected by other influences. Like the reach downstream of the impoundment, there is little natural surface water storage, and synthetically produced Baseflow Index (BFI) and flow duration curves suggest that baseflow contributions are low. The estimated naturalised average daily mean flow is $3m^3s$-1.

There are no biological data on these reference rivers, but expert opinion of local specialists considers that special features of Atlantic salmon (*Salmo salar*), brown trout and freshwater pearl mussels (*Margaritifera margaritifera*) may be present. Otherwise the aquatic fauna and flora are typical of unpolluted, stony rivers in upland Britain.

4.3.2.4 Conceptual model of impacts

It is considered that the flows downstream of the impoundment are impacting salmon and trout populations. It is suggested by the conceptual model that if conditions are suitable for maintenance of salmon and trout populations they are likely to be suitable for freshwater pearl mussels; optimised flows are therefore likely to be targeted at salmon, trout and freshwater pearl mussel.

The process diagrams in the conceptual model for extreme/extended low flows and a loss of small floods both confirm that impacts on salmon, trout and freshwater pearl mussels are likely, given the potential effects of reduced flows on habitat, and the sensitivity of salmon and trout.

To substantiate effects further, sampling is undertaken to corroborate impacts.

Fish surveys on the reach downstream of the impoundment confirm that trout are present, but indicate low numbers, low recruitment and low biomass. Surveys also confirm the presence of potential habitat for spawning and rearing of Atlantic salmon and trout.

Macroinvertebrate surveys indicate a few taxa characteristic of high velocity rivers with coarse substrate and clean waters, although diversity appears low.

The channel was rectangular in cross-section, and bed armouring was observed immediately downstream of the impoundment. The valley form allowed for only a narrow floodplain, with riparian wetland absent or very limited in extent.

Surveys in the reference river confirmed healthy populations of salmon and trout. There was no indication of freshwater pearl mussels. Macroinvertebrate communities were characteristic of high velocity rivers with coarse substrate and clean waters. Surveys also indicate that the reference river is naturally prone to low flows, with exposed bed, channel features and margins during the summer months. The rivers are also typified by narrow floodplains, with no, or very limited riparian wetland.

The survey findings broadly corroborate the initial conceptual model, but suggest that freshwater pearl mussel need not be considered further. Effects on riparian wetlands can also be discounted. For illustration purposes, the impact tables for salmon are reproduced below, with cells coloured to indicate the impacts that require addressing (Table 4.3).

Table 4.3 Summary of main risks to salmonid fish posed by principal types of modified flows

Expected response, if known as likely and significant:
G↑ Growth increase
G↓ Growth decrease
N↑ Number increase (mortality decrease)
N↓ Number decrease (mortality increase)

Life stage	(1) Extreme & extended Low Q	(2) Enhanced & stabilised Q	(3) Loss of small floods (<=1yr), inc. freshets	(4) Loss of large floods (>1yr)	(5) Extreme or untimely High Q	(6) Rapid Q change	Water Temperature
Egg incubation (Oct-Mar)	Desiccation loss of gravel flushing N↓		Loss of gravel flushing N↓		Washout N↓		Incubation rate reduced at low temps
fry swim up (Mar-Apr)	Area/habitat loss predation increased competition increased displace-ment to deeper water N↓				Displace-ment N↓	Strand-ing acute for trout due to pref. for margins N↓	Mismatch with 2^{o} production N↓
0+ May-Nov	Area/ habitat loss predation increased competition displace-ment to deeper water N↓ G↓	Increased area/ habitat & production G↑ N↑			Displace-ment N↓	Strand-ing acute for trout due to pref. for margins N↓	Growth rate reduced at low temps from hypol. discharge
0+ & >0+ (winter)	Loss of depth shelter N↓	Increased shelter G↑ N↑			High metabolic costs G↓		
>0+ (inc adult residents)	Area/habitat loss food loss predation increased displacemen-t to deeper water N↓ G↓	Increased area/ habitat G↑ N↑			High metabolic costs (displace-ment) G↓		Growth rate reduced at low temps
Smolting (not applicable BT or grayling) April-June			Lack of cues N↓				Lack of/ or mixed stimuli NB temp. AND flow and daylength

Life stage	(1) Extreme & extended Low Q	(2) Enhanced & stabilised Q	(3) Loss of small floods (<=1yr), inc. freshets	(4) Loss of large floods (>1yr)	(5) Extreme or untimely High Q	(6) Rapid Q change	Water Temperature
adult passage all yr mainly May-Oct	Obstructed passage **N↓**		Lack of stimuli and directional cues **N↓**				Loss of/ or mixed cues
spawning (Oct-Dec)	Access restricted **N↓**		Lack of stimuli **N↓**			Spawn-ing disrupt-tion **N↓**	
Kelt (Nov – April)	(Likely barriers, and greater energy demand) **N↓**		Slow or delayed DS passage				

(Brackets) = less important or, likely but unsubstantiated

4.3.3 Release scheme

4.3.3.1 Goals

The primary goal of re-designing flow releases is to re-establish salmon spawning and recruitment in order to supplement downstream fisheries.

A secondary aim is to improve trout recruitment, population size and biomass in order to offer a local trout rod fishery (also by providing flow-habitat goals as outlined for salmon)

A tertiary aim is that biological metrics of the wider ecology should be improved, or at least maintained, with macroinvertebrate communities of a character equivalent to achieving GES. Note that invertebrate production is vital for fish production, so is implicit in goals 1 and 2.

Finally, the flow regime should become more, not less natural as a result of management intervention. Specifically, variability should be introduced into the low flow regime.

The flow requirements are checked by reference to, in order:

1. The process diagrams (Appendix II.1.2).
2. The description of the targets' (in this case salmon and trout) life history and links with flow-related habitat (e.g. Appendix I.9).
3. The impact tables. (e.g. Appendix II.1.3).
4. The detail of the habitat requirements (e.g. Appendix I.9).

For clarity, in this example flows to meet the priority goal, relating to the priority ecological element (salmon in this case) are considered to meet the needs of lower priority ecological elements (trout and invertebrates), the exception in this case being that the flow needs for spawning trout are required earlier in the autumn, a point that will be evident from reference to the two species' life histories (Sections I.8.1 and I.9.1) and should also be confirmed by local knowledge.

This will be achieved by offering a low flow regime that makes available (a) the quantity and (b) the variety of meso-habitats (riffles, runs, pools etc) that constitute the missing functional habitats (spawning, rearing and holding areas). An important consideration will be the spatial distribution and relative abundance of meso-habitats in the affected reach. Section I.8.3 recommends flows that maintain a balanced mosaic of functional salmonid habitats: for spawning, juvenile rearing and adult holding, that is appropriate to the local channel morphology.

Freshets will take the place of lost natural small floods of biological significance. The biological role of these freshets is to assist with the stimulation of smolt downstream emigration (increasing water temperature is also necessary), the downstream dispersal of kelts and the upstream migration of adults. Of these the latter is the priority, because adults are vital for eggs and the start of each generation. Because this site is in the upper reaches of the main catchment, salmon would naturally arrive late in the season, say late August. Freshets are needed to attract fish upstream, to orientate them to their natal stream to help them to pass barriers, even natural partial barriers such as shallow riffle sections or some waterfalls.

4.3.3.2 Scenario release regime (1)

In the absence of local specific information, then following Baxter (1961) flows of 0.125x and 0.25x the local naturalised average daily mean flow are considered as an estimate to maintain de minimus conditions (see Appendix I.9). A modest inter-annual variability is allowed for with reference to hydrological behaviour in wet and dry years in the reference watercourse.

There is no tested advice in the literature on frequency of freshets. As a starting point, for salmon, from September onwards weekly releases of 4 – 8 hrs duration preferentially at night, particularly during spawning time (which is between mid-November and mid-December in this sub-catchment) is considered appropriate.

Size of the freshets should be based on guidance in Section 2.8.5 or mimic those on adjacent tributaries and if possible they should be timed to match the natural spates, in order to avoid problems of miscued and misdirected migrations on the whole catchment scale (i.e. avoid attracting fish from other sub-catchments). However, the impoundment is remote and telemetered flow monitoring and dam release operation may not be straightforward, making sequencing of releases with natural catchment inputs from upstream of the impoundment difficult.

In the absence of any other data, following Baxter (1961) freshets of 0.3 the local average daily mean flow are considered potentially appropriate. Rates of increased and decreased flows are scaled to mimic the rates observed in similar adjacent catchments, determined by analysis of hydrographs.

Smolts move in April - June period. As a minimum, weekly freshets will be required in May to 1st week in June. There is no guidance on thresholds, but it is suggested that 0.3 average daily flow would be a starting minimum, with releases made at night.

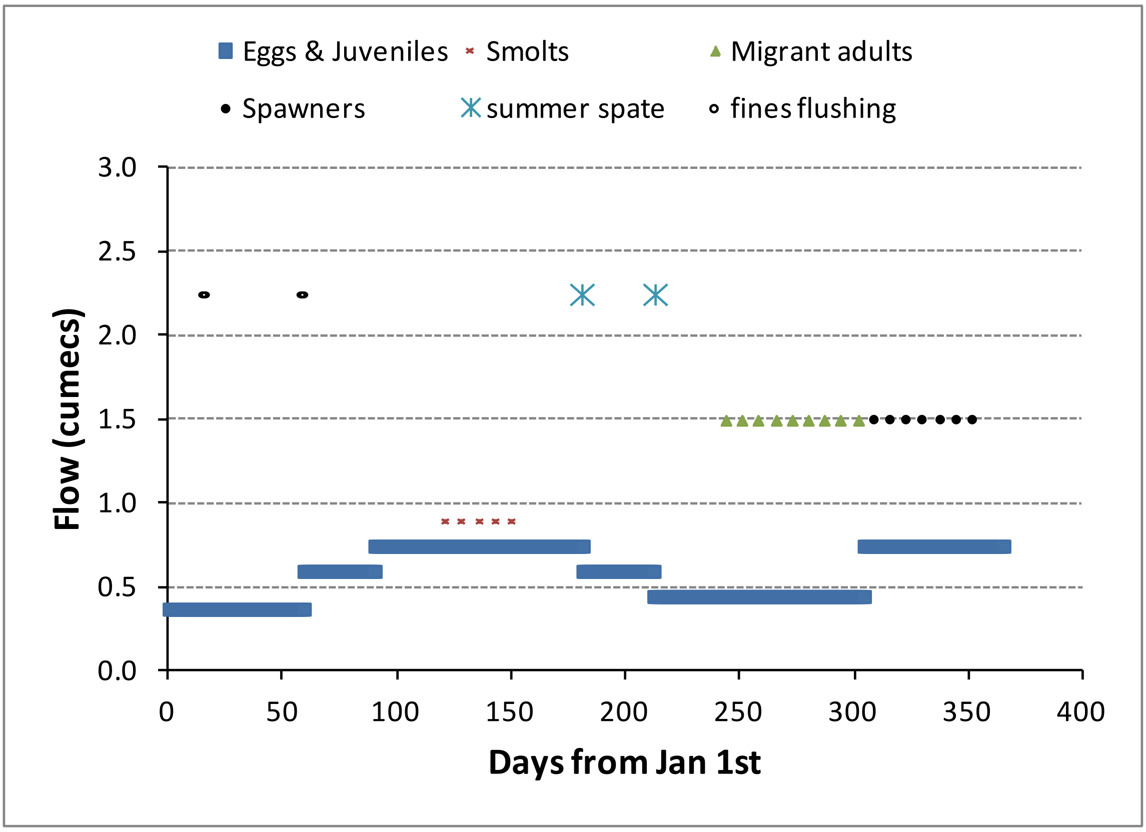
Trout freshet requirements are less than those of salmon, but they will spawn earlier, say late October through November at such a sub-catchment. Therefore consideration to twice-weekly freshets in late October would cover this need, as an initial procedure.

Similarly, in order to convey the occurrence of summer floods that serve to redistribute biological and substrate material in the reach, three summer spates should be released between June and September at 0.75 ADF. Two spates during the winter egg incubation period should be released for periods of 1 to 3 days to clean fines from sediments, but it is considered that these may be met by natural spills. The scour valve operation is timed to provide this.

For freshets, variability is achieved by randomising the timing of freshets between years, within the seasonal requirements noted above.

The composite artificial flow regime is illustrated in Figure 4.4.

Figure 4.4 Simulated artificial flow regime to maintain salmon population on the case study scenario river, with a naturalised ADF of 3 m^3s^{-1}. Red, green and black symbols show weekly freshet releases for smolts, migrating adults and spawners respectively. The blue crosses and open circle symbols are summer “maintenance” and fine flushing spates respectively. Note that inter-annual variability and spills are not shown



4.3.3.3 Assess the scenario release regime

The improvements achievable by the revised release regime are assessed and summarised in impact tables, an example of which is given below. Note that it is expected that the proposed flow regime will moderate, but not eliminate impacts. Indeed, against the criteria offered in 4.2.2.1, and given the sensitivity of salmonids to low flows, salmonids in the reach might still be considered at high risk. However, the criteria offered in 4.2.2.1 are indicative, and have no specific relevance to salmonids. Risks in Table 4.4 are therefore based on flows meeting the prescribed criteria for salmonids.

Table 4.4 Summary of main risks to salmonid fish posed by principal types of modified flows

Expected response, if known as likely and significant:

G↑ Growth increase
G ↓ Growth decrease
N↑ Number increase (mortality decrease)
N↓ Number decrease (mortality increase)

Life stage	(1) Extreme & extended Low Q	(2) Enhanced & stabilised Q	(3) Loss of small floods (<=1yr), inc. freshets	(4) Loss of large floods (>1yr)	(5) Extreme or untimely High Q	(6) Rapid Q change	Water Temperature
Egg incubation (Oct-Mar)	Desiccation loss of gravel flushing **N↓**		Loss of gravel flushing **N↓**		Washout **N↓**		Incubation rate reduced at low temps
fry swim up (Mar-Apr)	Area/ habitat loss predation increased competition increased displacemen-t to deeper water **N↓**				Displacem-ent **N↓**	Stranding particular-ly acute for trout due to preferen-ce for margins **N↓**	Mismatch with 2^{o} production **N↓**
0+ May-Nov	Area/habitat loss predation increased competition displacemen-t to deeper water **N↓** **G ↓**	Increased area/ habitat & production **G↑ N↑**			Displacement **N↓**	Stranding particular-ly acute for trout due to preference for margins **N↓**	Growth rate reduced at low temps from hypol. discharge
0+ & >0+ (winter)	Loss of depth shelter **N↓**	Increased shelter **G↑ N↑**			High metabolic costs **G ↓**		
>0+ (inc adult residents)	Area/ habitat loss food loss predation increased displacemen-t to deeper water **N↓** **G ↓**	Increased area/ habitat **G↑ N↑**			High metabolic costs (displace-ment) **G ↓**		Growth rate reduced at low temps

Life stage	(1) Extreme & extended Low Q	(2) Enhanced & stabilised Q	(3) Loss of small floods (<=1yr), inc. freshets	(4) Loss of large floods (>1yr)	(5) Extreme or untimely High Q	(6) Rapid Q change	Water Temperature
Smolting (not applicable BT or grayling) April-June			Lack of cues **N↓**				Lack of/ or mixed stimuli NB temp. AND flow and daylength
adult passage all yr mainly May-Oct	Obstructed passage **N↓**		Lack of stimuli and directional cues **N↓**				Loss of/ or mixed cues
spawning (Oct-Dec)	Access restricted **N↓**		Lack of stimuli **N↓**			Spawnin-g disruptio-n **N↓**	
Kelt (Nov – April)	(Likely barriers, and greater energy demand) **N↓**		Slow or delayed DS passage				

(Brackets) = less important or, likely but unsubstantiated

Impact tables for other ecological elements indicate that the proposed changes to the flow regime may have impacts on coarse fish, amphibians and bryophytes (if present). These risks are accepted on the basis that these ecological elements would not be more impacted than if flow regimes were natural.

A high-level reservoir simulation also indicates that the increased releases affect the reservoir storage regime and reduces spill frequency, but a further check on relevant ecological elements does not indicate significant impacts.

Impact tables for the target species indicate that the success of flow releases in establishing and sustaining salmonids may be limited by water temperature. It is decided that this will be checked by ongoing monitoring.

A further limit on the achievement of the primary goal is that freshets will encourage salmon into upper river section, which as flows decrease (and depending on its structure) may become unsuitable for holding fish as they need deep, slow sections with substantial cover such as under-cut banks, tree-root systems and in-stream large woody debris. This risk will need to be managed by provision of such areas or by maintenance of flows until spawning is completed and the kelts move downstream.

The success of artificial freshets may be limited because the impoundment is remote and telemetered flow monitoring and dam release operation may not be straightforward. Sequencing of releases with natural catchment inputs from upstream of the impoundment is therefore difficult.

The conceptual model process descriptions indicate that, without allowing for passage of sediment, increased freshets may result in armouring immediately downstream of the impoundment. This may over time cause a gradual loss of habitat availability, but the risk is accepted given the likely gains achieved over the five km reach over which flows will benefit.

The revised provisional flow regime is considered more natural than the previous regime, as measured by departure from adjusted LF2000 proxies of the Richter indicators of hydrological alteration.

However, annual and seasonal water requirement are calculated and found to be in excess of what can be achieved without significant cost to use.

4.3.3.4 Scenario release regime (2)

A second scenario is developed in which compensation flow requirements are estimated hydraulically. A relatively simple hydraulic approach is adopted, using transects surveyed at representative sections on the channel downstream of the impoundment to estimate the flows necessary to maintain spawning habitat sufficient to incubate eggs and provide fry recruitment to the available nursery area. Trout spawning would be in smaller isolated marginal patches, compared to salmon which will spawn more collectively in the main stem, using large substrate size (Appendix I.9). Similarly, in the potential parr nursery areas the flows are calculated that should ensure hydraulic variables commensurate with stage and species.

The average year compensation flow is reduced to meet habitat requirements at what is considered a lower, but viable level. This is still in excess of the current flows, and, on the basis of the new information, is considered to have potential to meet the rehabilitation goals.

Lower dry year compensation flows and higher wet year compensation flows are varied in a similar way to before, and the effect of the reduced releases benefits spills to show an acceptable pattern of freshet releases and spills. This regime is considered to achieve a similar degree of impact to that estimated by the initial scenario, but a lower water requirement. It is considered an acceptable scenario, but still a provisional one, being based on hydraulic and not biotic data.

4.3.3.5 Monitoring

Monitoring is designed to assess the recovery of salmon and trout. Macroinvertebrate surveys are also undertaken to provide a biological quality assessment and to provide an ecological index of hydraulic conditions. Monitoring of physico-chemical parameters, such as water temperature, conductivity and pH might also be considered to help diagnose any failure to recover, and an occasional check is made at the time of other surveys to monitor bed armouring and ensure that freshets are not allowing a build up of fine sediment.

4.3.3.6 Adaptive Management Cycle (Phase II)

Biological monitoring confirms a limited recovery of salmon, but suggests that fish passage may be inhibited by freshet releases. Rather than increasing the number of freshets, the dam operator invests in the capacity to make releases coincident with catchment inputs, thus increasing their efficiency. The success of this new regime is then monitored in a new cycle of adaptive management.

5 SUMMARY AND RECOMMENDATIONS

5.1 Conceptual Models

The report has presented working descriptions illustrating (a) the adverse ecological effects on rivers and associated, river-dependent, wetlands expected to result from changes to river flow regimes and (b) the changes to flow regimes that are expected to cause those effects. The conceptual models have covered and, where relevant, differentiated the environmental effects of abstraction and impoundment of water in rivers. There is a large body of scientific literature describing the different effects of human water use on the environment and many recent studies have provided reviews of this literature. Rather than providing another literature review, the purpose of the conceptual models was to summarise existing knowledge to provide route map of the main pathways linking water resource pressures to ecological impacts in a form that river managers can use to help protect the environment.

The conceptual models have adopted the natural flow paradigm (Poff et al. 2010b) as a basic principle; but along with contemporary thinking recognises that environmental river flow management in the face of incomplete knowledge involves pragmatism. The conceptual models have described the ecologically important components of the river flow regime that should be the focus for management effort and the basis of a framework for optimising water releases from impoundments in rivers:

- extreme or extended low flows;
- enhanced and stabilised low flows;
- loss of high flow pulses (return period<1yr) or small floods (2-10 year events);
- loss of large floods (>10yr events);
- extreme high or untimely discharge; and
- rapidly changing flows.

Alterations to these ecological flow components changes hydrological, hydraulic and geomorphological parameters in rivers, which combine to create the habitat state – the conceptualisation of the physical environment that supports aquatic organisms. Emergent properties of the habitat state have been identified that are important to allow aquatic organisms to reproduce and progress through their life-cycles and form the basis of identifying abiotic Ecological Indicators of the severe effects of river flow alteration:

- size of the habitat (area/volume of aquatic habitat space);
- connectivity and juxtaposition of habitat; and
- character and diversity of the habitat (ecological ‘quality’ of the habitat).

Recognising these three properties of the habitat state in rivers is of ecological importance as the hydraulic effect of reduced discharge in certain channel forms is to miniaturise the size of the wetted habitat space, whilst maintaining the overall hydraulic character of the habitat state.

The conceptualised response of aquatic organisms to reduced discharge is best described for macroinvertebrates and involves initially increases in the density of organisms as individuals become concentrated into smaller habitat spaces whilst hydraulic character is maintained. Further reductions in discharge alter hydraulic character (velocity and water depth) such that physico-chemistry (e.g. dissolved oxygen concentration and water temperature) directly affects the survival of different species and biotic effects (predation and competition) have a major indirect effect on species abundance and composition. As prey species become concentrated in smaller habitat spaces, predation rate increases, the relative abundance of predators to prey increases and eventually the habitat state becomes intolerable for all aquatic species. This conceptual model is space invariant and at larger spatial scales, which are relevant for the response of larger, more mobile organisms, such as fish, habitat connectivity and juxtaposition must also be considered.

These simple conceptual models are relevant to ecological river flow management as they form the basis of the new management tools identified in this report and also provide an understanding of the reasons why there is not always a good relationship between the assessment of river flows and the biological classification of water bodies using existing tools.

5.2 Ecological Indicators of the severe effects of abstraction and impoundment in rivers

The conceptual models have described a suite of biotic and abiotic ecological elements from which 54 candidate ecological indicators have been identified. The ecological indicators are mostly able to be measured easily in the field and are accessible to workers without extensive specialist expertise.

Ecological indicators will be subject to local influences and their behaviour is likely to be river type-specific. Specific combinations of indicators are likely to apply to certain river types and situations. However, when taken together it is expected that the ecological indicators will be able to provide a weight of evidence approach to identify river sites that are most severely affected by river flow alterations. This in turn will improve the certainty of classification of Poor and Bad status and improve the weight of evidence for prioritising mitigation measures in the most severely impacted water bodies.

The expert workshop agreed that the strength of the ecological indicators is in the combination of biotic, abiotic, multi-taxa and multi-trophic level indicators. However, some groups of ecological indicators offer greater certainty and potential for further development. These included: freshwater macroinvertebrate indices (Lotic invertebrate Index for Flow Evaluation [LIFE] and Proportion of Sediment-sensitive Invertebrates [PSI]), combinations of hydraulic measures and fine sediment deposition, bryophytes and terrestrial plants on exposed mid channel substratum and depositional features, and diatom indicators.

Key recommendations:

- Develop specific survey methodologies and undertake field trials in a range of water bodies (confirmed at Poor/Bad status and where flow standards suggest Poor/Bad status).

- Refine the diagnostic capabilities of different combinations of indicators and generalities within river types.

- Develop the LIFE methodology for use in Scotland and Northern Ireland and for diagnosing the severe ecological effects of river flow regulation downstream of impoundments across the UK. Using local reference sites rather the predictions obtained from RICT might improve the diagnosis of severe impacts in specific water bodies that are currently classified with high uncertainty. PSI used in conjunction with LIFE might improve the diagnostic power of LIFE, especially at locations that are most severely affected by altered river flows. In terms of the current report objectives in the

UK-wide context, current methodologies for LIFE and RIVPACS/RICT are considered sensitive enough of identifying sites that are most severely affected by abstraction and at Poor or Bad status. LIFE and associated macroinvertebrate indices offer potential for distinguishing class boundaries above Poor status, but are subject to river type specificity, particularly for groundwater fed rivers on chalk geologies compared to other types. It is possible that the sampling methodology will need to be changed for hydromorphological assessments so that it reflects the size of aquatic habitat space in separate meso-habitats and not just the composite character or quality of the habitats.

- Remote sensing techniques have advanced rapidly over the past few years and high resolution digital photography combined with advanced GIS methods and automated image analysis have released the constraints of scale associated with ground-based visual surveillance of riverine habitats. Remote sensing could provide a solution to mis-matches of spatial scale of observation relative to the scale of environmental impact described in the conceptual model and enable combinations of Ecological Indicators to be assembled from larger spatial scales that might offer more powerful diagnostic and classification capability than can be achieved at smaller scales on the ground. Remote sensing offers visual surveillance of ecological indicators in inaccessible locations and a permanent record of ecological indicators and other river features that can be interrogated at any time and shared among others. In addition, remote sensing offers a highly cost-effective solution for data capture and analysis at larger spatial scales and previously inaccessible locations.

5.3 Optimisation Framework

The optimisation framework sets out a generic decision framework. It is based upon SNIFFER research project WFD 82 (SNIFFER, 2007; Acreman et al. 2009) and the Building Block Methodology that underpins it, but is expressly designed to use the CMs developed in this report, and considers further work undertaken since WFD 82. It describes a framework for defining how the release of water from impoundments can be optimised to reduce adverse ecological impacts and to enhance the ecological potential on heavily modified water bodies. The work leads directly from the recommendation in report WFD 82 (Acreman et al. 2009) which outlined a broad approach which is followed and developed here. The aim can be interpreted simply as how to allocate water releases in order to optimise the ecological benefits of limited water banks in the face of competing ecological demands.

Given the extreme uncertainty in quantifying river flow-ecology relationships at the scales appropriate for the water release optimisation framework, a risk-based approach is therefore offered. This identifies and prioritises risks and flow needs for the chosen habitat or ecological element, and rather than focussing upon formal objectives (e.g. WFD standards), identifies risk areas resulting from potential flow modifications.

Consistent with the recommendation in modern river regulation studies, the water release optimisation framework is designed for local solutions to be based upon local information coupled with effective monitoring and adaptive management. Given the uncertain and inter-disciplinary nature of the problem, involvement of interested parties and relevant specialisms throughout the process is also advocated, following the ‘expert panel’ approach and an exploratory, scenario-driven approach.

Key recommendations:

- The water release optimisation framework should be used in a true adaptive management context in that its implementation should be treated as deliberate, large-scale experiments. In this way, uncertainty can be embraced by decision makers in making policy choices.

- The water release optimisation framework should be trialled at a number of key sites and monitoring data collected. It is apparent in the literature that few studies have implemented this kind of framework and have collected data suitable for informing scientific-based decision making.

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7 GLOSSARY

Compensation Flow/Release – Release of water from impoundment to supplement residual base flows in HMWBs. (See Water Release Optimisation Framework).

Drivers, Pressures, State, Impact and Response framework (DPSIR) - Approach used across Europe by the European Environment Agency to link socio-economy with ecology and in ecological research to support the implementation of the WFD.

Ecological Indicator – Biological community or taxon that can be used to measure the impact of a pressure.

Ecological Status – Banding system used to describe the quality of a biological community and, subsequently, the waterbody. Can be relevant at multiple scales. Consists of Bad, Poor, Moderate, Good and High bands. Termed Ecological Potential for HMWBs

Heavily Modified Water Bodies (HMWB) – Waterbodies containing structural elements which cannot be removed. Includes flood defence, potable water supply, navigation and other critical functions.

Physical Indicator – Hydromorphological feature which can be used to measure the impact of a pressure. Often the mechanism through which pressure is delivered to an ecological indicator.

Pressure – Any event or process which causes a disturbance to biological communities or taxon. Often refers to an anthropogenically derived source.

Reference Condition – The (often theoretical) ecological and physical state of a waterbody in the complete absence of anthropogenic pressure.

RIVPACS/RICT – River InVertebrate Prediction and Classification System/River Invertebrate Classification Tool. Macroinvertebrate community prediction and classification tool used to classify the extent of pressure at the reach scale.

UK Technical Advisory Group (UKTAG) – Partnership between the UK administration and conservation agencies created to provide coordinated advice on the science and technical aspects of the European Union's Water Framework Directive (WFD).

Water Release Optimisation Framework (WROF) – Framework for defining how the release of water from impoundments can be optimised to reduce adverse ecological impacts and to enhance ecological potential.

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APPENDICES

Appendix I Evidence base of the Conceptual Model: response of key ecological elements to changes in water flow

Appendix II Water Release optimisation Framework: summary of main risks to ecological elements posed by principal types of modified flows

I APPENDIX I EVIDENCE BASE OF THE CONCEPTUAL MODEL: RESPONSE OF KEY ECOLOGICAL ELEMENTS TO CHANGES IN WATER FLOW

I.1 Wetted Perimeter

I.1.1 Overview

The hydraulic rating method is an established approach (Gordon et al. 2004) in which discharge is related to simple hydraulic parameters rather than combinations of more complex parameters, as used in explicit habitat modelling.

Wetted perimeter has been widely used to define minimum flows internationally (Gippel and Stewardson, 1998). In a related approach, Atkins et al. (2005) also demonstrate the use of flow per unit width in studies of salmon (*Salmo salar*) and trout (*Salmo trutta*) in NW England. The width measurement used in this approach is unclear, but is thought to be the average of six or more measurements taken at the bank foot taken where the river channel is fairly straight, banks vertical and flow downstream.

In hydraulic rating approaches, breakpoints in the relationship between discharge and wetted perimeter or width define the point at which increasing flows offer diminishing returns, as defined by these measures of total benthic habitat space. The most important such breakpoint corresponds to the wetting of the channel bed, The approach might also be extended to define the onset of inundation of riparian land, which has been shown to be important to both riparian and in channel habitat (Poff and Zimmerman, 2010), and could utilise a further breakpoint at bankfull.

Wetted perimeter methods are reviewed by Gippel and Stewardson (1998), who offer an objective approach to defining breakpoints in the discharge-wetted perimeter relationship from transect data. Numerous transects should be located, either at random, to meet the needs of a hydraulic model, or targeting specific habitats to avoid undue weight being placed on single cross-sections. Typically, transects are located on riffle sections, which are ecologically important and are the first sites to become exposed by drying (Gordon et al. 2004). The assumption here is that if riffle sites are protected, other sites should be, although the diversity offered by other habitats should not be neglected (e.g. Mainstone, 2010).

I.1.2 Spatial scale

Wetted perimeter and similar hydraulic parameters measured across a transect vary longitudinally.

I.1.3 Temporal scale

Wetted perimeter and similar measures will vary with discharge. Measurements at different discharges may be needed, and these will need to make reference to an appropriate comparator.

I.1.4 Temperature

Wetted perimeter and similar measures can be taken as independent of temperature.

I.1.5 River type variation

Leopold and Maddock (1953) make generic use of a power law relationship. Gippel and Stewardson (1998), however, demonstrates variation according to the shape of the channel cross section: In the absence of backwater effects, depths in roughly triangular geometries (such as u-shaped headwater channels) reduce in proportion to reductions in discharge, and that wetted perimeter to decreases in a power law, without obvious breakpoints. In rectangular or trapezoidal channels wetted perimeter varies logarithmically with discharge, with a definable breakpoint where the bed is reached. Losses in wetted width or perimeter per unit discharge are relatively small between bed and bankfull, but once the bed is reached, even small changes in discharge cause significant loss of wetted perimeter.

In their review, Gippel and Stewardson (1998) reported only varied success in typing discharge – wetted perimeter relationships, reporting little consistency between studies in the flow measures at which break points occurred, and the tendency for clearly defined breakpoints in individual transects to be smoothed out when many transects, even on the same river, were combined.

In the UK context, Newson (pers comm) concludes from field research on hydraulic biotopes (Padmore, 1997) that there is no evidence of a single threshold in either quality or quantity, and that, if any and depending upon river type, flow exceedences between Q60 and Q85 may mark a fuzzy boundary between diversity and reduced hydraulic opportunity for biota.

Subsequently, Booker and Dunbar (2008) applied linear multilevel models to transects collected for hydrometric purposes in England and Wales, establishing a framework with which to predict hydraulic geometry and associated uncertainty from little or no site-specific data. However, despite a very extensive dataset – a total of 35 000 gaugings at 3600 sites - the relatively coarse resolution of hydrometric data, and a strong tendency for gaugings to be undertaken in straight sections of glide character, raise questions over the ability to generalise from these hydraulic geometry relationships to more varied hydraulic environments.

Some general physical habitat-discharge relationships have also been reported for a sample of 63 rivers in the UK to which PHABSIM (Bovee, 1982) has been applied - Booker and Acreman, 2007) reporting a breakpoint at or close to the Q95 for sites on the River Tavy and River Kennet (Acreman et al. 2009). However, although the sample set ranged in type from steep upland coarse-grain rivers to lowland chalk-streams, only one site was included from Scotland and none from Northern Ireland. It has also been recognised by the authors that application to many UK rivers would require extrapolation beyond the calibration data (Acreman et al. 2009).

I.1.6 Ecosystem relations

Geomorphological processes and macrophyte growth have feedbacks into wetted perimeter and other hydraulic parameters.

I.1.7 Ecological Indicator potential

Good. Wetted perimeter describes the total bed habitat space along a transect or reach, and is linked through hydraulic geometry to wetted width (which can be used as a proxy) and cross sectional area or volume of water in a reach (which defines the aquatic space in the channel). It is less complex, and costly, than hydraulic habitat modelling, and does not require the use of Habitat Suitability Curves, one of the major sources of error in PHABSIM type studies.

Implicitly, by preserving habitat space, maintaining wetted perimeter might also be assumed to maintain longitudinal connectivity. However, this need not be the case for species where a particular depth or flow across the habitat space is required. Note also that no attempt is made to describe habitat character, and thereby useable area for particular species. Gippel and Stewardson (1998), applying the method in Australia, found that wetted bed space was poorly correlated to, and underestimated blackfish habitat, and proposed the flowing water wetted perimeter as an alternative measure. This makes some allowance for habitat quality, and offered a better proxy for blackfish habitat.

However, there is some support for wetted perimeter as a measure of habitat character more generally. Van der Nat et al. (2002), Sommer et al. (2004) and Larned et al. (2010) report that habitats progressively coalesce and homogenise between bed coverage and bankfull, a conclusion that may offer some support to the fuzzy boundary between Q60 and Q85 discussed above. Studies on regulated rivers in the UK for setting compensation releases from reservoirs have further reported that once the full channel width is wetted, further increases in water level homogenise aquatic habitats, reducing the diversity of functional habitats for aquatic organisms (Environment Agency, 2009).

I.1.8 Suggested field indicators of Poor and Bad status

- Loss or absence of wetted channel relative to an unimpacted control site (1a)
- Fragmentaion of aquatic habitat (1b)
- All mid channel substratum submerged during March – June (1h)

I.2 Surface flow types

I.2.1 Overview

That flow types have a spatial correspondence with biotic organisation is well established, both for the water column and the benthos. It is further acknowledged that differences in hydraulic conditions in the water column and at the bed manifest in appearance at the water surface; higher Froude numbers, are associated with broken water (waves) and lower Froude numbers with a smooth water surface. Likewise, whilst most river habitat is 'turbulent' (in the hydraulic sense), the degree of turbulence (measured by the Reynolds number) is also an important distinguishing property that is manifest at the surface.

This linkage through channel hydraulics underpins the biotic relevance of various classifications of flow type. The surface flow patterns have been used to distinguish between slackwater and flowing waters (defining the lentic and lotic environments), between flowing water types (riffles, runs, glides and pools, etc.) and distinct 'patches' within them, such as riffle crests and riffle margins). This information can then be further combined with other hydraulic or morphological information to distinguish more finely, differentiating, for example, shallow glides from deep, or shallow marginal slackwaters. This has resulted in a wide range of different flow type classifications based upon surface appearance (compared in Heritage et al., 2009).

Some authors (e.g. Jowett, 1993; Statzner & Higler, 1986) note scaleable ranges of hydraulic variables. However, in a re-examination of published data, Clifford (2006) notes only broad and sometimes inconsistent associations between biotopes and classifications based upon hydraulic variables, considered that neither physical biotope nor functional habitat categories can be easily delimited by flow type alone, and concluded that linkages between hydraulically defined biotopes and biologically defined functional habitats could not be considered established.

Harvey (2006) further notes inconsistency in thresholds and overlap in flow type categorisations, and found that Froude number (and because of this surface flow type), may discriminate only between broad 'low' and 'high Froude' habitats, albeit with some gradients within type. Harvey (2006) identified broad 'assemblages' of flow biotopes, which indicate associations between reach-scale morphological preferences and functional habitats; a rough, high Froude combination of chute flow, unbroken standing waves and rippled flow representing higher-energy step-pool morphologies; an intermediate riffle-pool assemblage of smooth boundary turbulent flow, rippled flow and unbroken standing waves; and a 'slower' glide-pool assemblage with no perceptible flow, smooth boundary turbulent flow or rippled flow.

I.2.2 Spatial scale

In river management terms, 'biotopes' define meso-scale physical habitat structures, typically at the sub-reach scale (riffle, pool etc.). These units are comprised of, and comprise, habitat at both smaller and larger spatial scales.

I.2.3 Temporal scale

Hydraulic biotopes vary with discharge at sub-daily timescales, although variability in flow type may be limited during periods of steady flow. Physical (morphological) properties of biotopes integrate over longer timescales and may be stable, at the meso-scale, over years to decadal timescales.

I.2.4 Temperature

N/A (with the exception of ice cover).

I.2.5 River type variation

There is a large natural variation in dominant surface flow types and the mix of sub-dominant surface flow types, both between rivers of different morphological type and within rivers of similar type. Reach-scale morphological units and their respective flow type assemblages are organised along an energy gradient from high to low altitude and slope with distance from the river source. Thus, steep, coarse-bedded upland rivers tend to be associated with rapid, turbulent flow types, and a greater diversity of surface flow types, and also often greater temporal variability because of their typically more flashy nature. Slower flowing, lowland river types are more likely to exhibit lower flow intensity, lower spatial diversity and tend to be more stable.

I.2.6 Ecosystem relations

- Depth/ velocity criteria define functional habitat preferences for macroinvertebrates, macrophytes, fish and other biotic elements.
- Physical flow stress exerted on macroinvertebrates, macrophytes and other biotic elements.
- Substrate size and homogeneity available for benthic biota, macrophyte attachment, fish spawning.
- Interaction and feedback with macrophyte growth can be important, particularly in chalk streams. Macrophyte growth is also affected by a range of other factors, suggesting and are associated with different spatial distributions, complicating the distribution of habitat.

I.2.7 Ecological Indicator potential

Good, but relationships between surface flow types and hydraulic, physical and biologicallly-defined biotopes are complex, and may only be capable of broad-scale discriminatory power. Mapped data may therefore be most useful to determine relative changes.

Survey problems include temporal variability of hydraulic behaviour with changing discharge, high cross sectional variation causing transect-based surveys to overlook secondary biotopes and marginal features (Padmore, 1998), the attribution of some surface flow types to more than one hydraulic or morphological feature, differences in the various surface flow classification schemes, natural differences in river type, reach-representativeness and interactions with macrophyte growth and direct morphological alterations.

I.3 Volume of fine sediment in channel bed

I.3.1 Overview

Particles of sand, silt and clay are termed fine sediment. These tend to be found in relatively low quantities in gravel bed rivers, because they can be transported by flows of relatively low velocity. Fines are likely to occur in greater quantities in the subsurface of the bed, where they are protected from entrainment by the coarser particles above (see channel bed armouring section).

The amount of fine sediment in the channel bed is a function of the rate of supply from the catchment and the capacity of the flow in the channel to transport it (Lisle and Hilton, 1992, Lisle and Hilton, 1999), and is also linked to groundwater flushing. Where the fine sediment supply remains unchanged (e.g. because catchment runoff and tributary inputs are the main sources), but in-channel flows are reduced, a greater amount of fine sediment may accumulate.

When high flows occur, fines are flushed from the bed, preventing long-term accumulation of excessive levels of fines (Acornley and Sear, 1999). Fine sediment may accumulate in the river bed during periods of low flow, when it commonly infiltrates the interstices between gravel particles (Wood and Armitage, 1997). Where high levels of fine sediment are found in the bed, this may be an indication of a prevailing low flow condition or that a flushing flow has not occurred for a long time (Sear, 1995). Flushing flows are considered by King et al. (2008) to occur seasonally, two or three times a year.

The proportion of the bed that is composed of fine sediment depends on river type, catchment characteristics and catchment management practices and is therefore highly variable between rivers (Lisle and Hilton, 1992). Assessment of any impact of changes to the flow regime on substrate fine sediment levels must, therefore use a control reach as an indication of baseline conditions.

I.3.2 Spatial scale

Deposition of fine sediment depends on local flow velocities and is therefore locally spatially variable (e.g. it is more likely to occur in pools and in backwaters (Wood and Armitage, 1997)). Levels of deposition may also be higher close to sources of fine sediment, such as eroding banks or field drains. Assessment must take into account this reach-scale spatial variability and target sensitive biotopes such as riffles and glides.

I.3.3 Temporal scale

Fine sediment levels within the bed are likely to be subject to temporal variability, in relation to high flows that flush sediment from the bed (Owens et al. 1999), episodes of significant delivery from the catchment and channel banks (Acornley and Sear, 1999) and long periods of low flows that allow build up of fine sediment in the bed (Wood and Armitage, 1997). Assessment must therefore take into account the recent flows and likely sediment dynamics in the catchment

I.3.4 Temperature

N/A. However, the impact of fine sediment is greater where upwelling groundwater is low in dissolved oxygen.

I.3.5 River type variation

The typical proportion of the bed composed of fines is variable between river types (but will also show variation within river types, in relation to catchment factors). Slower-flowing, lowland river types are more likely to contain larger amounts of fine sediment. Any changes in fine sediment levels due to flow changes may be less apparent in these river types. Steeper channels (e.g. plane bed, plane-riffle and actively meandering channels) that have gravel or cobble beds are less likely to contain large amounts of fine sediment under normal flow conditions and may, therefore be more sensitive to fine sediment accumulation if flows are reduced.

I.3.6 Ecosystem relations

- Loss of salmonid spawning habitat
- Smothering interstices for sediment-sensitive invertebrates, loss of 'riverflies'
- Eliminating submerged aquatic plants, such as *Ranunculus* spp.
- Homogenisation of instream habitat
- Severe impact on juvenile stages (and hence recruitment) of freshwater pearl mussels
- Increased sediment oxygen demand, reduced dissolved oxygen concentration in water
- Increased production of methane

I.3.7 Ecological Indicator potential

Excessive levels of fine sediment may occur as a result of high levels of supply from the catchment or localised high rates of bank erosion as well as from a modified flow regime. Assessment should take the likely supply of fine sediment from the catchment into account and use a representative control reach to determine fine sediment levels under natural flow conditions.

Levels of fine sediment are likely to vary in time, in relation to changes in flow. Therefore observation at a single point in time may not provide an accurate indication of the long-term characteristics of the system.

Assessment needs to also take into account the spatial heterogeneity of the phenomenon and focus on sensitive biotopes such as riffles and glides.

I.3.8 Suggested field indicators of Poor and Bad status

- Fine sediment covering sensitive habitats (1d)
- Dense plume of sediment occluding water column when distrubed (1e)

I.4 Channel bed armouring

I.4.1 Overview

An armoured bed is the term used to describe a river bed where smaller particles are absent from the surface, leaving a layer of coarser material with a high threshold for mobilisation. This protects finer subsurface material from mobilisation by lower discharges and results in a low rate of increase in sediment transport with increase in discharge (Bathurst, 2007). Armoured beds occur naturally in gravel bed rivers, but the armouring may be exaggerated by changes to the flow and/or sediment regime.

Armouring occurs where flows are competent enough to transport finer gravels from the bed surface, but are unable to mobilise larger particles (Vericat et al. 2006; 2008). Under a natural flow regime, periodic floods with a high competence would break up the surface armour layer, releasing finer sediment from underneath and replenishing the surface layer with finer particles (Vericat et al. 2006). Where impoundments prevent large floods, break up of the armour layer does not occur and the armouring effect becomes more extreme, creating a more permanent armour, or 'pavement'. The bed becomes coarser and stable (Sear, 1995) and sediment supply to downstream reaches is reduced. Without active sediment transport, the pool-riffle sequence becomes stagnated, maintaining reasonable flow diversity but not good spawning habitat.

Newson (pers comm.), investigating the break up of armoured and highly structured gravel beds, found random (very minor) movements of bed material at low flows and selective entrainment – (enough to release some fines and be considered a 'flushing flow') at half bankfull. 'Equal mobility' - i.e. full movement of the bed, was found at bankfull.

These rules of thumb are being considered for incorporation into the abstraction licence for Haweswater at Swindale Beck, and would appear to be corroborated by experience with hydropower releases from Kielder (Newson, pers comm.), which at c.a quarter of bankfull level are sufficient only to create random mobility, and are considered to harden the riffles in the reach down to Bellingham. Carling (1988), however, is reported in King et al. (2008), as showing evidence that in some coarse gravel bed rivers, flows of greater than bankfull are required before substrate is fully mobilised.

A further factor affecting the degree of armouring is the supply of sediment from upstream. The presence of a dam upstream prevents the downstream transport of sediment, meaning that where finer particles are removed from the bed by competent flows, they are not replenished by sediment from upstream (Vericat et al. 2008). The implication of this is that a high degree of armouring may occur downstream of a dam, even where the flow regime is natural.

The severity and extent of armouring below a dam is also dependent on the amount of sediment that is supplied to the channel downstream of the dam (for example by bank erosion or inputs from tributaries). If this is significant, sediment supply may be replenished and the effects of the dam would not propagate a long distance downstream (Carling, 1988).

I.4.2 Spatial scale

The downstream persistence of the armouring is affected by the rate of sediment input downstream of the dam (Carling, 1988), as well as the locations of significant, unregulated tributary flow inputs, that may augment flows.

I.4.3 Temporal scale

Severe armouring downstream of a dam develops over medium to long timescales and may increase in downstream extent over time (depending on the locations of downstream sediment inputs). Armouring may be broken up during extreme floods and then re-formed during flows of lower magnitude (e.g. Vericat et al. 2006).

I.4.4 Temperature

N/A

I.4.5 River type variation

Bed armouring is a phenomenon found widely in most types of gravel and cobble bed rivers. River types with a high rate of coarse sediment transport are likely to be most sensitive to the effects of flow regulation on bed composition. These include rivers with plane bed, braided, wandering and plane-riffle flow types.

I.4.6 Ecosystem relations

Some degree of armouring is normal and may be beneficial for freshwater pearl mussels, providing a stable protected habitat with good flushing of fine sediments.

Extreme armouring possibly creates poor habitat conditions for other aquatic biota.

I.4.7 Ecological Indicator potential

A high degree of armouring downstream of a dam may be a result of the reduction in downstream sediment conveyance caused by the dam, rather than, or in addition to, changes to the flow regime.

The severity and downstream persistence of the armouring is affected by the rate of sediment input downstream of the dam – this may need to be taken into account when assessing the degree of armouring.

Given that some degree of armouring is normal, the armour ratio and D_{50} of the regulated reach should be compared with a set of control measurements from a representative, unregulated reach.

I.4.8 Suggested field indicators of Poor and Bad status

- Absence of gravel from bed surface (1f)
- Uniform cobble particle size on bed surface (armouring or paving) (1g)

I.5 Stability of channel bed

I.5.1 Overview

Bars are a common bedform in gravel bed rivers and may occur in a range of positions within the channel (mid-channel, point, lateral, tributary confluence). In a gravel or cobble bed channel, the presence of active bars indicates that the process of sediment mobilisation and transport through the system is occurring. Active bars are characterised by clean gravels with a low level of vegetation cover. Where the sediment transport capacity of the river has been reduced, due to a reduction in the magnitude of high flows, bars and bed may become stable, allowing colonisation by vegetation (Sear, 1995; Gilvear, 2004). Extent and type of vegetation cover is a key indicator of the stability of in channel bars and therefore prevailing river flow conditions that might be low and/or stable due to severe abstraction or regulation from impoundments. Where bars are not present in the channel, moss and lichen on any exposed particles indicates a recent lack of movement. Other signs of a stable bed include severe armouring, as described above and a coarser particle size (compared with an unregulated reference reach).

In some cases colonisation of bars by vegetation may occur, despite active gravel transport. These cases include situations where lateral channel migration or widening is occurring and in these cases bar stabilisation would occur in combination with erosion on the opposite bank.

I.5.2 Spatial scale

I.5.3 Temporal scale

Colonisation by vegetation may occur naturally during summer when flows tend to be lower and vegetation growth is more prolific, but this tends to be ruderal or annual species that germinate quickly and are not resistant to the effects of high flow events. The presence of perennial species, ferns, mosses, trees, bushes and mature stands of terrestrial grasses is a key indicator of highly stable channel bars due to chronic low and/or stable flows.

I.5.4 Temperature

N/A

I.5.5 River type variation

Most likely to occur in gravel and cobble bedded rivers. A change in bed stability is most likely to be noticeable in rivers that would naturally (prior to flow regulation) have a high bedload and active bed, where stabilisation of previously mobile depositional features is obvious.

I.5.6 Ecosystem relations

Exposed bars provide exposed riverine sediment (ERS) habitat for a range of invertebrate species that are highly specialised to live in this habitat and have high conservation value (see ERS invertebrates detail). Species richness of ERS specialists is negatively related to the stability of the ERS habitat and they require a variable regime of flooding and drying to maintain the loose substratum, moisture variation and to prevent competition from other species that can colonise stable habitats. Stable channel bars that become vegetated, indicating chronic, stable flow conditions due to flow regulation or severe abstraction pressure are not suitable for most ERS specialists.

I.5.7 Ecological Indicator potential

There are several features of stable channel beds that have potential to be used as indicators, especially when used in conjunction with other indicators, such as armouring, and bar deposition downstream of tributary confluences.

I.5.8 Suggested field indicators of Poor and Bad status

- No active channel bars (1i)
- Presence of stable channel bars without active bars (1j)

I.6 Stability of channel banks

I.6.1 Overview

Erosion of channel banks is a complex process that incorporates a wide range of variables. However, the effect of flow shear stress is a major factor influencing the rate of erosion. High flows exert the highest levels of shear stress on the banks and therefore cause the greatest rate of bank erosion. Where high flows are eliminated by regulation, bank erosion activity may be reduced. Conversely, where flows are increased, or where sediment loads are reduced (e.g. by the dam, or by armouring), bank erosion may increase (Germanoski and Ritter, 1988; Gilvear et al. 2002).

Eroding channel banks are typically steep or undercut, showing signs of disturbance, such as a lack of vegetation cover, slumping and eroded material at the base (Thorne, 1997). If channel banks are of a low gradient and are well vegetated this would suggest that bank erosion is not occurring. This could be because the high flows that cause bank erosion have been eliminated by regulation.

Rapid and sudden fluctuations in flows, e.g. from hydropower or scour valve releases, can saturate river banks on the rising limb and them leave the alluvium unsupported on the recession. Impact zones in rivers with structurally weak bank sediments often exhibit ‘slumped’, rotational shear failures on an extensive scale.

Factors other than flow also influence rates of bank erosion. These include the bank material and structure (certain types are more susceptible to erosion), bank height, role of sub-aerial processes and external pressures, such as livestock activity. Therefore, a lack of bank erosion may not necessarily indicate a lack of competent flows and conversely, the presence of bank erosion may not indicate that fully competent flows are occurring.

I.6.2 Spatial scale

I.6.3 Temporal scale

Extent and type of vegetation cover is a key indicator of the stability of in channel banks and therefore prevailing river flow conditions that might be low and/or stable due to severe abstraction or regulation from impoundments. Note that colonisation by vegetation may occur naturally during summer when flows tend to be lower and vegetation growth is more prolific, but this tends to be ruderal or annual species that germinate quickly and are not resistant to the effects of high flow events. The presence of perennial species, ferns, mosses, trees, bushes and mature stands of terrestrial grasses is a key indicator of highly stable channel banks due to chronic low and/or stable flows.

I.6.4 Temperature

N/A

I.6.5 River type variation

Bank erosion tends to be an important process in alluvial channels with floodplains and may be less prevalent in upland channels where banks are likely to be lower and composed of bedrock or boulders.

I.6.6 Ecosystem relations

Channel banks provide exposed riverine sediment (ERS) habitat for a range of invertebrate species that are highly specialised to live in this habitat and have high conservation value (see ERS invertebrates detail). Species richness of ERS specialists is negatively related to the stability of the ERS habitat and they require a variable regime of flooding and drying to maintain the loose substratum, moisture variation and to prevent competition from other species that can colonise stable habitats. Stable channel banks that become vegetated, indicating chronic, stable flow conditions due to flow regulation or severe abstraction pressure are not suitable for most ERS specialists.

I.6.7 Ecological Indicator potential

Bank erosion rates have potential as an indicator when used in conjunction with other indicators, such as stability of the channel bed, and when interpreted in the context of the expected rate of bank erosion for the channel in question. However, owing to the complexity of the response, careful interpretation by an experienced geomorphologist is likely to be necessary.

I.6.8 Suggested field indicators of Poor and Bad status

- No exposed substrate on channel banks (1l)
- Gradient of channel banks less than vertical (1m)
- Widespread gravitational bank collapse (1s)

I.7 Adjustment at tributary confluences

I.7.1 Overview

Where unregulated tributaries join a regulated river, disparities in their flow and sediment transport regimes may result in the formation of certain geomorphological features. Degradation of the main channel below dams may induce base level lowering in tributaries (Brandt, 2000). This in turn can cause degradation of tributaries, resulting in deepening and/or widening (Germanoski and Ritter, 1988). However, in most UK systems the degree of flow lowering in the main channel is not of sufficient magnitude to result in noticeable incision of tributaries. Increased bank erosion, or changes to the planform of tributaries may provide signs that this is occurring.

Where flows in a regulated river are reduced, there may not be sufficient capacity to transport sediment inputs from unregulated tributaries. This results in deposition of sediment in bars downstream of tributary confluences (Sear, 1995), which may develop into benches and become vegetated (Gilvear, 2004). Significant channel narrowing may occur downstream of tributary confluences, where competence in the regulated main channel is too low to mobilise the calibre of sediment deposited by tributaries (Curtis et al. 2010).

Careful interpretation may be needed to be sure that any indications of tributary degradation, or deposition of sediment downstream of confluences, are a result of changes to the flow regime and not a result of other catchment changes (such as changes to the sediment supply regime in the tributary, or changes to the bed level of the main channel as a result of other factors).

I.7.2 Spatial scale

Effects are likely to be greatest nearer to the point of flow alteration, while more distant tributary confluences would be unlikely to show an impact.

I.7.3 Temporal scale

The extent of deposition and accumulation of sediment downstream of tributary confluences is likely to vary in relation to the flow hydrographs of both the regulated main stem and the unregulated tributary inflows. For example, extensive deposition may occur following a high flow event in a tributary, while deposits may be eroded if flows in the main stem increase. Assessment must, therefore, take into account recent flows in both the main stem and the tributary. Channel adjustment following impoundment and flow regulation typically occurs over decades. Assessment of geomorphological forms must, therefore, take into account the length of time since flow regulation commenced.

I.7.4 Temperature

N/A

I.7.5 River type variation

Effects on tributary base level, bank erosion and planform are more likely to occur in unconfined channels with a gentle gradient, where tributaries are also unconfined and where tributary base level is controlled by the height of the main channel.

Confined channels with steep tributaries and bedrock control will be unlikely to show changes.

Deposition of sediment in the main channel downstream of unregulated tributary confluences, with associated narrowing and aggradation, occurs in a wide range of river types, but is likely to be easier to observe in shallow, gravel or cobble bedded rivers, where bars are exposed during normal flows.

I.7.6 Ecosystem relations

- Reduced salmonid spawning habitat
- Elimination of aquatic plants
- Elimination of sensitive invertebrates

I.7.7 Ecological Indicator potential

Tributary incision in response to lowered flows in the main channel is minor in the UK and therefore not a good indicator. Deposition of sediment in the main channel downstream of tributary confluences is a more widespread phenomenon and has better potential as an indicator. However, it is dependent on there being tributary inflows in the regulated reach. The indicator must be used in the context of the sediment transport regime of the tributary and with the recent temporal variability in flows taken into account.

I.7.8 Suggested field indicators of Poor and Bad status

- Low width to depth ratio (1n)
- Steep undercut or eroding tributary banks (1o)
- Tributary terraces (1p)
- Exposed tree roots in bottom of tributary channels (1q)
- Presence of active bars downstream of tributary confluences (1r)

I.8 Spacing of riffles

I.8.1 Overview

Riffles are a common bed formation in gravel and cobble-bed rivers and are areas of shallower flow, where the substratum is coarser and bed gradient is steeper. They are usually interspersed with pools, which are areas of deeper, slower flow and typically finer substrate, with international literature settling on a spacing of c.7x bankfull width.

The presence of riffles is an indication that the channel has an active sediment transport regime. Where the flow regime is altered riffles may become degraded, while pools may become aggraded (Sear, 1995), along with other changes to the structure of the channel bed, including stabilisation and armouring, as described above. Further research is needed to substantiate the effect of changes to the flow regimes on riffle and pool spacing, but the ecological importance of riffles is well established (e.g. Mainstone, 2010), and departures to the seven times bankfull width may provide a useful measure even if diagnosis of causes remains difficult.

I.8.2 Spatial scale

I.8.3 Temporal scale issues

The effects of changes to the flow regime on riffle and pool morphology is likely to occur over the long-term (decades), and be related to the timing and frequency of floods, which have the capacity to alter bedforms.

I.8.4 Temperature

N/A

I.8.5 River type variation

Changes to the spacing and frequency of riffles is only applicable on rivers that have a riffle-pool typology

I.8.6 Ecosystem relations

I.8.7 Ecological Indicator potential

Speculative. There is currently little research into the effects of flow regulation on riffle-pool sequences. Further investigation would be necessary before this could be used as a reliable indicator.

I.8.8 Suggested field indicators of Poor and Bad status

- Loss of riffles/ runs, preponderance of pools (1d)

I.9 Atlantic salmon (*Salmo salar* L.)

I.9.1 Overview

In the UK, to which this account applies, native, river-dwelling salmonids include the Atlantic salmon (*Salmo salmo* L.), brown trout (*Salmo trutta* L.) the latter occurring in two main forms, "resident" brown trout and sea-going sea trout; and, in a separate but related genus, the European grayling (*Thymallus thymallus* L.). The common characteristics of all salmonids are extensive up-river spawning migration, spawning in gravel beds and downstream dispersal of juveniles. Therefore, (a) river geomorphology and flow regimes are very influential on the survival and distribution of salmonids and (b) conditions across the full range of spatial scales (microhabitat to catchment) relevant to populations need to be considered in identifying and mitigating flow-related impacts on populations.

I.9.2 Overview of salmon life cycle and flow needs

Salmon are migratory, anadromous fish that enter rivers (after sea feeding and maturation) throughout year. They spawn in autumn (typically Nov-Dec), with some latitudinal variation), laying their eggs in stream gravels having specific size range and texture and water columns with specific velocity and depth ranges (size–specific) (see Table I1). Incubation occurs typically during November to March and requires good intra-gravel flows and no drying out of gravels, although some dewatering of gravels is tolerable providing humidity and temperature are maintained, but this is a site–specific qualification, depending on local hydromorphology and sediment structures (Becker and Neitzel, 1985). Emergence of fry in April-May, requires avoidance of high flows because they cause displacement and mortality (Ottaway and Clarke, 1981; Jensen and Johnsen, 1999) and the avoidance of low flows which restrict food supply (small invertebrates). An adaptive evolutionary point: egg burial is typically at depths below scour depth associated with typical bankful discharges (Gibbins and Acornley, 2000).

Juveniles (termed parr) remain in the river for 1 to 3 years (usually 2), reaching 10 to 20cms length. As they grow, they have increasing space and shelter dimension requirements to allow them to feed, grow and avoid predators. Local habitat patch quality is strongly influenced by flow and local hydraulic features (depth, velocity, substrate size), However, their preferences are not independent; moreover they are influenced by the presence and behaviour of other competing fish of their own and other species (especially trout, which tend to outcompete salmon) and predators (see Armstrong et al. 2003; Finstad et al. 2011; Milner et al. 2011; Nislow and Armstrong (in press) for recent reviews).Thus responses to flow variation are complex, dynamic and interdependent upon other factors. There are many accounts of hydraulic preferences, that enable broad guidance on limit conditions; but simple transferable models of quantitative responses of juveniles to changing flow have so far proved elusive (see Table I1). Site specific information and interpretation are essential. Juveniles migrate to sea as smolts in April–May, stimulated by temperature and flow combinations (location-specific), downstream migration requires free passage and moderate flows; if flows are too low delays and increased predation risk can arise. The response of juveniles to rapid flow variation from hydro-peaking is size-specific, because swimming to deeper refuge areas is important; but this and the resultant larger territory sizes entails energy costs that have been suggested to increase winter mortality (Scruton et al. 2004; Nislow and Armstrong, in press). Enhanced, but stabilised flows, (e.g. some compensation flows) are thought to be beneficial to salmonid production through increased food availability, more stable territory sizes and less variable temperatures (Milner et al, 2011).

Spawning migrations of adult salmon (at 40-140cm length) back from sea occur throughout the year on large British rivers, on which large spring run salmon enter early in the year when flows are naturally high. On the majority of rivers, which are smaller, run timings vary, but tend to be between April and November with most in July-September (see EA catch statistics). Sea trout tend to have more restricted seasonal runs usually peaking in June-July. Natural flow regimes also vary from large Alpine, snowmelt-influenced rivers of Scotland to the groundwater rivers of southern England (Lewin, 1981). The entry timing of salmon appears to match that variety, reflecting their natural adaptations to maximise fitness in the face of environmental circumstances (Poff et al. 2007).

Salmon are attracted to their natal rivers and spawning areas through odours from some unknown combination of chemicals including kin recognition substances (e.g. pheromones) and geochemical signals from sub-catchments. Return migration is strongly influenced by flow-related factors, through (1) direction finding and homing to chemical cues and (2) stimulation and maintenance of upstream movements and (3) passage past barriers. These last two responses although being apparent reactions to “flow” are probably mediated by hydraulic and correlated variables. However, while there is strong literature support for this (Banks, 1969; Alabaster, 1970; Milner, 1992; Thorstad et al. 2008), there is substantial variation in reported flow-movement responses for reasons including: the size of rivers, the variety of conditions that fish experience moving up each river, the diversity of flow regimes between rivers and the variation on the individual fish physiologies and their migration intentions.

Under natural regimes, increases in river flows attract salmon from the sea into estuaries, through estuaries into rivers and thereafter encourage movement up rivers. Low flows in estuaries appear to be particularly problematic for salmon migration and can, possibly in association with high temperatures, lead to significant delays or displacement back to sea and permanent loss through mortality (Clarke et al. 1999; Solomon and Sambrook, 2004). Salmon entry into larger rivers (e.g. >20m^3s^{-1}) appears to be less flow-dependent than into smaller rivers, the latter more often requiring spates to encourage entry and upstream passage (Jonsson et al. 1991).

Once in the main river, small spates generally provide stimulus and the conditions to move upstream and pass barriers, which may be impassable at low flows; but this is very site-specific (e.g. Solomon et al. 1999). On large rivers the need for spates appears to be less critical in lower main stem reaches, possibly because larger, deeper channels can be passed under low flows. However, extended low flows in summer can cause in-river mortality through low dissolved oxygen concentration and high water temperatures (Brooker et al. 1978). Salmon movements in smaller coastal rivers are more dependent on higher flows and in that respect such rivers may simply be analogs of similar sized tributaries in the upper reaches of large catchments. There may be antecedent effects, for example following a period of low flows smaller spates are usually sufficient to stimulate salmon movement (Tetzlaff et al. 2008; Malcolm et al. in press). Critical flows vary along length of rivers (Solomon et al. 1999), such that in upstream reaches higher proportions of the local ADF are required to meet critical thresholds, probably because the channels are progressively smaller, steeper and the barriers harder to negotiate. Age, maturation status and size of fish (Trépanier et al. 1996) also affect response to flows.

However, and crucially, reported migration responses to flow are inconsistent and variable between rivers and still poorly understood (Trépanier et al. 1996; Thorstad et al. 2008).

Variable behaviour in main stem rivers culminates in a final phase – the spawning run to breeding locations (in main stem, or in tributaries), requiring at least moderate flows and is typically initiated by small spates. Variable flows within and between years are important to enable full distribution of egg deposition across available spawning sites of a catchment (Malcolm et al. in press). After spawning (which incurs high mortality) the surviving post spawners, termed kelts, drop downstream to the sea. This downstream migration can be impaired by extended or exceptionally low flows, which can impede passage or increase predation risk, but is normally achieved without flow related problems.

Key review refs (Klemetsen et al. 2003; Armstrong et al. 2003; Cowx et al. 2004; Finstad et al. Crisp 2000; Thorstad et al. 2008; Nislow and Armstrong, in press; Malcolm, in press).

I.9.3 Spatial scale and connectivity

Salmon use the whole of the river system downstream of their spawning grounds to complete their life cycle, thus passage to and from the sea is essential for up and downstream migrations. Salmon will not be present in inaccessible areas unless they have been artificially stocked.

The distribution and proportion of meso-habitats influences salmon abundance and biomass production; but no clear guidance has been published on this and precise optimal conditions (in terms of flow or channel structure) cannot be specified. However, each life stage requires specific functional habitat features (see above and Table I.1), with spatial requirements increasing with age (size), and each stage must be present in a reach, so a balance is required of each functional habitat (i.e. spawning, rearing habitat for fry to older parr, winter parr habitat and holding location for adults). Downstream dispersal from spawning sites to parr rearing areas is typically in range 10 to 500m (i.e. the minimum habitat unit is probably the upper end of this scale).

The redds of spawning salmon (at mean weight of 2.8Kg) are about 1m x 3m and the average space to avoid competition with other spawners is 9.5m^2 (Bjorn and Reiser, 1991). Juvenile densities are highly variable depending upon food delivery rates and flow (Armstrong et al. 2003). Grant et al. (1993) noted that Atlantic salmon density dependent effects were only began to occur at Percent Habitat Saturation of above 27%, concluding that most habitat is actually unsuitable or suboptimal for juvenile salmonids in streams. Symons (1979) suggested that nursery area for parr up to two years old should represent about 30% of the stream wetted area in optimal conditions. A balanced mosaic of habitats is required providing the attributes for the different life stages (Table I.1). Thus river flows suitable for maintaining salmon and trout populations should support this habitat diversity.

Relative upstream migration flow thresholds for adult passage (Solomon et al. 1999) and minimum flows for spawning (Gibbins et al. 2008) tend to increase, relative to local ADF, in upstream direction.

I.9.4 Temporal scale

Annual lag effects can be important in the response time of indicators and the nature (e.g. mortality or growth) of the impact. For example, egg mortality may affect recruitment the following spring; but flow-related mortality of early fry may be absorbed by density dependent effects in remaining months of 1^{st} year. First year parr after a few weeks are less vulnerable to displacement by high flows (Jensen and Johnsen, 1999); but subsequent flow effects on older fish mainly act through reduced growth and consequent smolt age and survival. Four to seven year generation times determine the lag between egg/fry losses and returning adults. Run timing of adults returning from sea is river-specific. They may be all year in big East coast Scottish rivers, with a summer peak; but in smaller rivers – salmon mainly arrive in summer and early autumn.

I.9.5 Temperature

Because they are cold-blooded salmon are strongly influenced by water temperature through metabolic rates affecting developmental rates, such as egg incubation and growth. Main issues regarding HMWBs are: temp regime shifts from hypolimnetic (cold) releases and long term decreases in water volume that can lead to warming. NB cooler release water can also be a mitigating agent at times of hot weather. See Crisp (2000).

Upper lethal temp for eggs >12 ^{o}C, range for 50% hatch 0-<12^{o}C.

Egg incubation time is strongly temp dependent e.g. $\log_{10}$(Date of median hatch) = [-2.6562$\log_{10}$(T + 110)] +5.1908.

Juvenile growth range 6.0 – 22.5^{o}C, opt 15.9 ^{0}C.

Salmon movements are inhibited at low (<5^{o}C) and high (>22-25^{o}C) temperatures (Crisp, 2000).

I.9.6 River type variation

Salmon are found in a wide range of rivers from surface fed “spate” to groundwater fed “chalk” rivers. However, they are not found in lowland, low gradient rivers, unless those are passed by adults en route to upstream spawning grounds. Rivers with substantial spring runs (i.e. fish entering pre-June) tend to be larger than those without.

Temperature stimulus for smolting may be more important in chalk rivers, where flows more stable. Also, in groundwater rivers where flows are seasonally more stable, adult passage appears less responsive to small spates (Hellawell, 1974).

I.9.7 Ecosystem relations

Pearl mussel synergies. Juvenile salmon act as vectors for juvenile PM stages

Food requirements dependent upon adequate 1^{o} and 2^{o} production and flow delivery (NB salmon are primarily drift feeders, if drifting prey are reduced through Q effects, so might salmon be, if there are no alternative food sources. Poaching can increase at obstructions especially at low flows. Angling success and economic values are also flow-related (Alabaster, 1970; Gee 1980; Potts and Malloch, 1991).

I.9.8 Ecological indicator potential

Mostly low to moderate at present, due to (a) field sampling costs and (b) effects of confounding factors; but growth and biomass production may offer options in future. Requires electrofishing surveys, or redd counts and adult stock assessment. But the species is responsive to flow. Redd counts may be useful on cost and effectiveness grounds where observation conditions permits. Presence/absence of redds or juveniles may reflect spawner access, marine recruitment or state of gravel quality (egg mortality).

For juveniles: flows cause mainly mortality (abundance) effects in early stages (egg to emergence + 3 months), thereafter mainly growth/size effects, although many confounding factors.

I.9.9 Suggested field indicators of Poor and Bad status

- Trout and salmon absent in otherwise suitable and accessible habitat (2a)
- Absence of adult salmon or migratory trout in autumn (2d)

I.9.10 Angling

Most studies show that angling catches of salmon are positively related to river flow (Millichamp and Lambert, 1966; Alabaster, 1970; Gee, 1980; Potts and Malloch, 1991, Smith 1994), but there appears to be inconsistency between rivers in flows that maximise catches. There is evidence that upper reaches of rivers require higher proportions of their local ADF to meet angling requirements (Gee, 1980). The mechanisms are not fully understood, but are probably some combination of flow-related variables acting on fish catchability, availability, accessibility (Harden Jones, 1968) and fishing effort.

I.9.11 Guideline flow standards for salmonids (adult salmon and sea trout)

Stream flow requirement for salmonids have a long history of development, important early reviews were by Giger (1973) and Fraser (1975). More recent accounts of flow standards have explained and emphasised the difficulty in setting generic standards (Acreman and Dunbar, 2004; Poff et al. 2010b, Poff and Zimmerman, 2010).

In spite of the difficulties, flow guidelines (standards) for various life stages have been set by several authors and agencies in UK (see review by Solomon in Milner et al. 2011). They vary even for specific life stages because of the different criteria applied and the different river sizes and types for which they are intended (see above).

Flow guidelines for salmonids, mainly for migration, have been specified by Stewart (1969), Baxter (1961) and Cragg-Hine (1985) based on counter observations. Solomon et al. (1999) have made recommendations for specific rivers and locations based on tracking data. Flows to maintain angling have also been studied and have sometimes been taken as approximations to migration flows (see above); although most authors recognise that this identity may be tenuous because of factors such as angler and fish behaviour and the different nature of the relationships in various parts of a river (Crisp, 2000). Stewart's recommendations in particular are often quoted, perhaps because they are simple to use and have intuitive appeal. Unfortunately, the methodology, rationale and data analysis supporting most of the recommendations of Stewart, and indeed many of the other guidelines, have not been published in reviewed sources and it has not been possible to trace their derivation and justification from the grey literature.

Freshet characteristics are summarised below, under general headings of their ecological flow components and a note on water quality. In addition, the location of the release relative to the intended benefits of the receiving channel and adjacent tributaries needs to be considered, particularly in the case of linked supply networks – for example, if salmonids are attracted differentially away from their intended spawning sites, this could be to the detriment of the overall catchment production.

Magnitude

Generally higher flows are needed following higher antecedent average flows. Thus small freshets will not increase fish passage rates at times of high average flow; but even small freshets may work at times of low flow. Generally, flow thresholds to pass river sections increase in an upstream direction, expressed as proportions of the local ADF. Freshet impacts on hydraulic variables will reduce downstream as the flow peak attenuates, therefore the intended aim should be clear and the freshet fit for that purpose.

Frequency

A single freshet may bring fish into a river, but multiple freshets are required to maintain movements and draw them through the river system. However, multiple freshets may be wasteful of resources if fish are not present, or if ambient flows are already high. Weekly freshets have been recommended to maintain fish movements (Baxter, 1961).

Duration

Reported freshets for upstream adult migration have ranged from a few hours to 48 hours. Baxter (1961) recommended freshets <18hr of which 12 hr should be at full rate, the remainder tapering to base level. Enders et al. (2009) suggested that freshets of 3-5hrs duration at night would be sufficient to enable downstream smolt migration.

Rates of change

Most reported studies on adult movements have failed to specify ramping rates, but from the accounts they appear to be very rapid. Change rates may not be important for the adult migration, but rapid flow reduction can lead to stranding of juveniles, particularly the early, less mobile stages and if frequent, such as in HEP hydro-peaking contexts, can also cause loss or reduced growth of larger parr. Stranding risk will be site specific according to the bed profile. The best advice is to try to mimic the rising and receding limbs of the natural flood hydrograph as closely as practicable with the facilities. An important risk hazard with ramping is the safety of anglers wading in rivers, particularly at night.

Timing (seasonal and diurnal aspects)

Freshets need to be seasonally relevant. Because of seasonal run timings the availability of fish to respond will vary greatly during a year. If no fish are present there will obviously be no benefit and in small rivers, or the upper tributaries of large rivers, salmon and sea trout may only arrive naturally in late season (e.g. August or later). The spawning time is particularly sensitive and salmonids in small spawning tributaries (e.g. <8m wide) may require flows higher than Q50 to enter and distribute eggs effectively over stream areas. Salmon move preferentially at night on low to average flows in clear water and when they have to pass barriers (e.g. falls, shallow rapids), but this pattern breaks down early and late in the season or in high turbidity, when they will move in the day. Freshets may need to be timed to be in the appropriate areas at night in order to maximise movements. However, for salmon fisheries purposes day time flows suitable for angling (see Table I 2) are required.

Water quality and temperature

Artificial freshets may involve chemically modified water from impoundments. Consequently they may be half as effective as natural floods in stimulating upstream movements. Moreover, hypolimnetic releases may significantly reduce water temperatures and inhibit or reduce movements.

The various guidelines are set out in Tables I1 to I3, but are difficult to compare because they are not all expressed in the same terms, are not based on the same type of data (Stewart's were based on counters and Solomon's on telemetry) and not all are applicable to locations of common channel morphology or hydrology. However, they do reflect the general principle that flow needs increase progressively upstream relative to local hydrographs. The radio-tracking studies, which are well-documented and relate to specific sites, seem to indicate generally higher flow requirements for migration than the other methods. However, when the guidelines are expressed in common terms, such as proportions of local ADFs or some flow percentile, there is overlap amongst them, which is encouraging. What is presently lacking is a similar treatment of the wide range of reported tracking and other studies; this might reveal other areas of apparent consistencies for some flow criteria, which could be structured by location within the river.

Table I 1 Recommended river flows for resident salmonid (salmon and trout) life stages (Baxter, 1961). Note that 'small' and 'large' rivers were not explicitly defined, but that the same source text refers to small rivers as having an ADF < 5cumecs. Large rivers are variously referred to as having an ADF of > 20cumecs or 40 cumecs. Guidance is not given for flows in the intervening (mid) range of 5 – 20 cumecs

Salmonid survival	**Proportion of ADF**	
Month	**Small rivers**	**Large rivers**
Oct	0.125 -0.15	0.125-0.15
Nov	0.25	0.15
Dec	0.125-0.25	0.1-0.15
Jan	0.125	0.1
Feb	0.125	0.1
Mar	0.2	0.15
Apr	0.25	0.2
May	0.25	0.2
Jun	0.2- 0.25	0.15-0.20
Jul	0.15-0.20	0.125-0.15
Aug	0.15	0.125-0.15
Sept	0.125-0.15	0.125-0.15

Salmon spawning	0.125-0.30
Salmon egg incubation	0.10 -0.17

Table I 2 Recommended river flows for salmon survival, migration and angling. From Stewart (1969) for spate rivers in general, and Gee (1980) for salmon angling on the River Wye, Wales

Criterion **1-4 (Stewart 1969)**	**Recommended minimum flow as $m^3s^{-1}m^{-1}$ (cumecs per metre of bankful width)**
1. Survival (of adults)	0.03
2. Start of Migration	0.08
3. Peak of migration	0.20
4. Angling peak	0.29
Angling peak (Gee 1980)	0.43 to 0.76 of ADF

Table I 3 Recommended river flow thresholds for salmon passage, expressed as proportion of local Q95 and ADFs, adapted from data in Solomon et al. (1999). Based on radio-tracking at specific locations in a chalk river (Hampshire Avon) and five surface water dominated rivers (Exe, Tamar, Taw, Torridge and Tavy)

River type	**Lower river**		**Upper river**	
	Prop (Q95)	Prop (ADF)	Prop (Q95)	Prop (ADF)
Chalk	1.10	0.39	1.30	0.46
Surface water, Min.	1.00	0.11	2.50	0.26
Surface water, Max.	2.50	0.26	6.00	0.63

(min /max refers to range across the five rivers)

Freshet effectiveness

The reported inconsistency in the effectiveness of natural floods and artificial freshets (e.g. Banks, 1969; Thorstad et al. 2008) needs comment, in addition to the factors outlined above which lead to variation in flow responses. Salmon, and probably also sea trout, do not enter rivers continuously. On smaller rivers, particularly in the lower reaches, there is circumstantial evidence that they arrive in pulses, in clumped distributions, possibly reflecting arrivals at the coast from marine migrations, or the combined effects of tide and weather including antecedent freshwater flows to the estuary. This has two consequences: 1) one flood/freshet may not bring fish far upstream, but may serve to attract fish into river or to a holding area; 2) artificial freshet efficiency may decrease if they are delivered too frequently for too long; because the pulse of fish available to respond may be used up and further releases will have little benefit until the pool of fish available rebuilds.

Table I 4 - Indicative values for some hydraulic and related variables for key salmon life stages and effects of flow modification. NB (1) these values are highly variable and sensitive to location, fish size, acclimation and to other environmental variables. (2) some of the velocity variability results from differences in the position in the water column of measurements. See e.g. Armstrong et al. (2003) for a review of factors to consider

Life stage process	Habitat criteria	Comments	Source
Spawning and egg incubation	Burial depth: size related, typically 5-30cm Depth = bL+a, where L is length(cm),a = 2.4, b=0.262. Water depth for spawning: mean 38cm, range 17-50cm. Fish size (L,cm) specific, thus: Depth(cm) = 0.176L + 0.76 (Crisp and Carling,1989). Velocity: >15 – 20cm s-1 (Armstrong *et al*., 2003), mean 46cm, upper limit may be x2 body length (typically 40-120cm) cover. Froude number: approx 0.3.	Washout: burial depth adapted to avoid this, but >10yr flood caused 40% washout at 15cm. Intra-gravel flow conditions: flows are vital to maintain intragravel conditions which can be compromised by accumulation of fine material that reduces oxygen delivery to eggs. Dewatering: rare because humidity *etc.* can be maintained in gravel. NB sediment composition appears more important for spawning site selection and is optimised in high sediment supply to transport capacity ratio characterised by riffle-pool sequences. Unnatural, rapid Q variation can disrupt spawning **.**	Crisp (2000); Becker and Neitzel (1985) Armstrong *et al.*, (2003); Klemetsen *et al.*,2003) Moir *et al.*, (2006) Crisp 2000 Hendry et al 2003
	Substrate size: mean range 21-100mm, median range 5.4 – 78. Fines (1mm) <10 -15% by wt	Armouring: avoid prolonged low flows. Periodic flushes needed to remove fines.	Armstrong *et al.*, 2003
Fry emergence,	Critical displacement velocity: (swim-up) 15cm s^{-1} at 6-8^{o}C, 19cm s^{-1} at 12-14^{o}C. Substrate cover. SAC advised V<100 cm s^{-1} for salmon eggs alevins (this is high). Depth: <40cm, prefer shallower e.g. <10cm.	**Displacement** and mortality of swim-up fry faced with high flows, or stranding from dewatering is a major concern. Bed armouring reduces interstitial habitat.	Jensen and Johnsen, 1999; Heggenes and Traan (1988) in Crisp 2000 Armstrong *et al* 2003 Nislow and Armstrong (in press)

Life stage process	Habitat criteria	Comments	Source
0+ rearing (summer/autumn)	Depth: 5-65cm, prefer 20-30cm (but 25-60cm in Symonds and Heland (1978)). Velocity: (mean column) 10- <100 cm s^{-1}, pref 20-30 cm s^{-1} Substrate: range 16-256 mm. Cover: substrate, macrophytes displacement velocity (@8weeks): >50cm s^{-1} (Crisp2000), Crit displacement Vels are temp dependent: e.g. 13.5 -14.6 cm s^{-1} @ 7-9 ^{o}C, vs 32 -34 cm s^{-1} @ 13-18 ^{o}C (Gibbins *et al* 2001).	Salmon behaviour changes, with movement away from stream bed as V decreases e.g. <10 cm s^{-1} (Kalleberg, 1958). **Stranding and dispersal from hydropeaking** is a demonstrated risk for young 0+ and fry, leading to mortality. In contrast, older fish swim better, can avoid stranding, but suffer growth rate reduction through low summer flows (food supply and bank shelter effects), with knock-on life history and production effects. More DS dispersal at night. See Ugedal et al, 2008; Hvidsten 1985;Saltveit et al 2001. Scruton et al., 2008.	Gibbins *et al* (2001) Crisp 2000 Kalleberg, 1958 Armstrong *et al*, 2003 Nislow and Armstrong (in press) and refs therein.
1++ rearing (summer/autumn)	Depth: range 20-70cm. Velocity: (mean column) <20-120 cm s^{-1}; pref 10 – 65 cm s^{-1}. Substrate 64 – 512mm. Cover. Substrate, macrophytes, woody debris, roots etc	Hydro-peaking increases utilised home range and energy expenditure which may reduce survival.	Armstrong et al., 2003
1++ winter habitat	Depth. Velocity ***see next column.*** Substrate. Cover.	Winter habitat is crucial. Generally, parr move to deeper, slower water when temp <8 ^{o}C. For 0+ - 1++ stages, flow stabilisation (more wetted area, velocity, food supply) in summer likely to benefit production at time of high growth potential. Elevated flows in winter likely to increase metabolic costs (Nislow and Armstrong in press).	Armstrong et al., 2003 Klemetsen et al., 2003 Nislow and Armstrong (in press)
Smolt migration	Velocity: no specific vals, but small freshets needed, emigration inhibited by low flows. Temperature: locally determined.	The combination of temp and velocity is important.	Klemetsen *et al* (2003); Armstrong *et al*, (2003). McCormick et al., (1998)

Life stage process	Habitat criteria	Comments	Source
Adult upstream migration	Flow: normally discharge is the specified variable in migration studies, because proximate D and V are not known for individual fish. Flow criteria available, but variable (see tables I1-3). Spate duration may vary with location. Conflicting values reported. 3-5hrs at night (Enders et al., 2009), 12-18hrs (Baxter 1961). Cover: require shelter in deep pools (Armstrong et al 2003). Temperature: migration rate impaired by low temp, <5.5 to 8.5 (Gowans et al 1999).	NB Highly variable, between and within rivers. Fish may move on low flows, and on rising or falling hydrographs. NB in ground water rivers, tend to show less flow –movement responses. BUT 4 points: 1. typically fish move on higher than normal flows i.e. small floods; 2. Flows as proportion of average increase further upstream; and 3. Critical response decreases during season summer to autumn 4. Artificial freshets work, but are less attractive than natural (e.g. x2 less) and have been reported as ineffective (site –specific), effectiveness may be enhanced if coincide with natural.	Banks, (1969); Alabaster, (1970); Crisp (2000), Solomon *et al*., (1999), Thorstad *et al*., (2008). Hellawell *et al* (1974); Trépanier et al., (1996) Archer *et al* 2008; Thorstad and Heggberget (1998); Milner et al (2011) Enders et al. (2009)
Adult spawning, tributary entry	Tributary entry stimulus.		Webb et al. (2001), Gibbins et al. (2008)
Kelt downstream migration	No specific values, but probably require protection (e.g. depths < 100cm for recovery) and free passage over barriers (so, not extreme low flows).	Kelt downstream migration.	

I.10 Brown trout (*Salmo trutta* L.) resident and seagoing (sea trout)

I.10.1 Overview

The resident form of trout is ubiquitous in most water types, their primary requirement being access to spawning areas of fast flow over gravel beds. The sea going form of trout (sea trout) are often sympatric with residents and interbreed, sea trout runs have higher proportions of females. Sea trout behave in ways similar to salmon and many of the comments on salmon apply to trout. They enter rivers over narrower seasonal range than salmon (after sea feeding and maturation) mainly between May and October, peaking June/July. Both forms spawn in autumn, slightly earlier than salmon (e.g. October to November), laying eggs in stream gravels having specific size range and texture and water of specific velocity and depth ranges (fish size–specific) (Crisp, 2000; Baglinière and Maisse 1999; Armstrong et al. 2003). Incubation (October – March) requires good intra-gravel flows and no drying out (but see dewatering for salmon, likely to be same for trout). If the spawning habitat is reduced for more than 80% optimum for continuous period of 20 days then recruitment was reduced (Capra et al. 1995).

Emergence of fry in April-May (slightly earlier than salmon), requires avoidance of high flows (washout leads to mortality) and presence of food (small invertebrates). As they grow, the juveniles become less vulnerable to displacement by high flows (Jensen and Johnsen, 1999) and have increasing space and shelter requirements. Local patch quality is strongly influenced by flow and local hydraulic features (depth, velocity, substrate size – but preferences are not independent) and also, because of their competitive behaviour, by presence of other competing fish and the presence of predators. Trout tend to be more prevalent in smaller channels than salmon, often being the dominant salmonid in channels <6m width (Milner et al. 2006). Trout juveniles remain in river for up to three years (15-25cm length); at which point they make a choice based on energetic status and growth trajectories to mature and pass the rest of their lives in freshwater or go to sea.

Sea trout migrate to sea as smolts in April – May, stimulated by temperature and flow combination (location-specific), downstream migration requires free passage and moderate flows (if too low that delays migration and increases predation risk).

Return sea trout migration (at 30-100cm length, on average smaller than salmon, but sizes overlap) in rivers is influenced by flows, because they home to chemical cues and, probably, flow rate changes (e.g. Crisp, 2000; Finstad et al. 2005). As for salmon the response of trout movement to flow variation is inconsistent, with the balance of evidence in favour of an increase in probability of movement initiation and rate of upstream progress as flow increases (Evans, 1994; Jonsson and Jonsson, 2002; Svendsen et al. 2004; Finstad et al. 2005; Rustadbakken et al. 2004). Small spates, relative to ambient provide such stimulus and the conditions to pass barriers which may be impassable at low flows. As for salmon but there are probably antecedent effects. In small systems high water discharge appears to be more important than in larger rivers (Jonsson and Jonsson, 2002). Trout are thought to have lower migration flow needs than salmon (Banks, 1969) and display more variable behaviour in main stem rivers than salmon (Finstad, et al. 2005). Water temperature confounds the effects of flow (as a covariate), leading to faster speeds, thus dispersal through rivers (Svendsen et al. 2004) and aids barrier passage (Rustadbakken et al. 2004). The spawning run into breeding locations (in main stem or tributaries) requires at least moderate flows and is typically initiated by small flow increases. Post spawning mortality of sea trout is less that salmon, many fish returning to spawn for 2 to 5+ times. Limited evidence suggests that sea trout post-spawning migration is not limited by flows (Svendsen et al. 2004). Although excessive low flows are likely to be detrimental through delays and increased predation.

Like salmon, trout populations are flow-dependent and exceptional droughts have major negative effect on numbers of resident juvenile trout (Elliott, 1997).

I.11 Spatial scale and connectivity

Migration access is essential, trout migrate upstream to spawn many km in case of residents and from sea to spawning ground in case of sea trout; up and downstream (smolts and kelts) migration.

Dispersal from spawning sites to parr rearing areas are less well known than salmon, but are likely to be similar, e.g. 10-500m (i.e. the minimum habitat unit is this scale).

Habitat diversity needs to be maintained, along with connectivity required between spawning, nursery and rearing areas (see I.8.3, the same concepts apply to trout)); but winter shelter needs are less well known than salmon. Barriers formed by low flows can prevent passage through rivers.

I.11.1 Temporal scale

For same reasons as salmon lag effects are important: mortality effects in egg stage may be evident following spring; loss of early fry may be absorbed by density dependent effects in early post-emergent weeks (Elliott, 2006). Run timing of sea trout is river- specific, but more confined to summer months than salmon, with mid-summer peak; NB need local knowledge. Generation times of resident trout are, normally three to eight years in rivers (typically up to 4 or 5 years in small upland streams); sea trout may be up to 10 years. NB in rivers having lakes in the system, trout may adopt anadromous-like behaviour, migrating to and from lakes to feed and mature.

I.11.2 Temperature

See salmon. See Crisp (2000). Egg incubation time is strongly temp dependent: e.g.
Date of median hatch = 218 $T^{-0.84}$
Where T = daily temp
Upper lethal limit for eggs 15.5, temp range for 50% Hatch 0-11.0

Juvenile growth within 3.6 – 19.5°C, optimum 15.9°C. (NB varies with acclimation, lower preference and tolerated range than salmon). Growing season has been defined as days water temperature >7°C (Power, 1981).

Temperature preferenda for trout are generally about 2-3 °C lower than for salmon.

I.11.3 River type variation

Trout are generally widespread across most river types in British Isles, where suitable spawning conditions occur, but are less abundant in slow flowing large rivers. Sea trout occur in most coastal streams and rivers, and may extend more than 100 km upstream; but this appears to be moderated by balance between migration risks (distance to sea) and sea feeding opportunities. Thus, there are river specific influences on distribution, requiring local knowledge to assess significance.

I.11.4 Ecosystem relations

Pearl mussel synergies (as salmon)

Food requirements dependent upon adequate production and flow delivery, but trout show greater usage of benthos and riparian food sources than salmon. Sea trout poaching may increase at obstructions and at low flows.

NB Sea trout angling often occurs at night; therefore there may be local safety issues for hydro-peaking and ramping rates.

I.11.5 Ecological indicator potential

Moderate, because the lack, or major reduction (*cf.* against habitat model predictions) of, trout in wetted channels with suitable habitats is evidence of severe to major impact. However reduction may not be necessarily a local flow-specific response, because of confounding factors affecting recruitment and juvenile survival.

Requires electric-fishing surveys, or redd counts and adult stock assessment. Presence/absence may reflect spawner access or gravel quality. For juveniles: Mortality effects in early stages (egg to emergence + 3months), thereafter mainly growth/size effects, although not flow specific – many confounding factors.

There is also the potential to use growth indices. See salmon.

I.11.6 Suggested field indicators of Poor and Bad status

- Trout and salmon absent in otherwise suitable and accessible habitat (2a)
- Increased growth rate of trout (2b)
- Decreased growth rate of trout (2c)
- Absence of adult salmon or migratory trout in autumn (2d)

Table I 5 - Indicative values for some hydraulic and related variables for trout life stages and effects of flow modification. NB (1) these values are highly variable and sensitive to location, fish size, acclimation and to other environmental variables. (2) some of the velocity variability results from differences in the position in the water column of measurements. See e.g. Armstrong et al. (2003) for a review of factors to consider

Life stage / process	Habitat criteria	Comment on flow modification	Source
Spawning and egg incubation	Burial depth: size related, typically 5-30cm Depth = bL+a, where L is length(cm),a = 2.4, b=0.262.	<u>Washout:</u> burial depth adapted to avoid this, but >10yr flood caused 40% washout at 15cm.	Crisp (2000); Becker and Neitzel (1985)
	Water depth for spawning: mean 38cm, range 17-50cm. D Is fish size (L,cm) specific, thus: Depth(cm) = 0.176L + 0.76 Assume same for Sea Trout as for salmon.	<u>Intra-gravel flow conditions:</u> vital to maintain. <u>Dewatering:</u> rare because humidity etc. can be maintained in gravel.	Armstrong *et al.*, (2003); Klemetsen *et al.,2003;* Bagliniere and Maisse (1999) Moir *et al.*, 1998
	Substrate size: mean range 21-100mm (Armstrong et al 2003); mean range 70-140mm median range 5.4 – 78	<u>Armouring</u>: avoid prolonged low flows.	Armstrong *et al*, 2003 Bagliniere and Maisse (1999).
Fry emergence	Displacement critical velocity: swim-up) 15cm s^{-1} at 6-8^{o}C, 19cm s^{-1} at 12-14^{o}C.	Displacement and mortality of swim-up fry is a major concern.	Hegennes and Traan (1988), in Crisp 2000
	SAC advised V<100 cm s^{-1} for salmon eggs/alevins. Alevin displacement inhibited at T <4.5^{o}C DS dispersal of new emerged trout minimal at 2.5cm s-1, high at >25cm s^{-1}. Substrate cover needs.	Bed armouring reduces interstitial habitat.	Nislow and Armstrong (in press) Ottaway and Clarke (1981)

Life stage / process	Habitat criteria	Comment on flow modification	Source
0+ rearing (summer/autumn)	Depth: 5-35 cm. Velocity: (mean column) 20-50 cm s^{-1} (Armstrong *et al*., 2003) but B&M, 1999 give 2 -30 cm s^{-1} approx. range. Displacement Velocity (@8weeks): >50cm s^{-1} Substrate: 8-128mm. Cover: substrate, macrophytes and banks. NB in winter 0+ move to slow flow (2-5 cm s^{-1} and depth 30-40cm	Generally, size for size, trout occupy deeper water than salmon. More DS dispersal at night. See salmon for dewatering stranding, high/low flow effects. Trout may be more susceptible than salmon (Hvidsten *et al*., 1985).	Armstrong et al, 2003; Baglinière and Maisse, (1999) Nislow and Armstrong (In press and refs therein)
1++ rearing (summer/autumn)	Depth: range 5-120cm; mainly opt >50cm. Adult generally 20 - >50cm Velocity: range 10-70, pref < 25 cm s^{-1}. Substrate: coarse gravel to boulders. Cover: substrate, macrophytes, woody debris, tree roots, undercut banks.	Trout prefer stream margin habitat. Generally, move to deeper slower water with incr size. Depth limiting for larger trout. Trout of 21.3cm actively avoid <5cm D. For 0+ - 1++ stages, flow stabilisation (more wetted area, velocity, food supply) in summer likely to benefit production at time of high growth potential. Elevated flows in winter likely to increase metabolic costs (Nislow and Armstrong in press).	Armstrong *et al*., (2003); Crisp (2000). Nislow and Armstrong (In press).
1++ winter habitat	Depth. See next column Velocity. Substrate. Cover.	No info, but likely move to deeper slower more sheltered sites at temps < 10 ^{o}C.	Armstrong *et al*., 2003.
Smolt migration	Velocity. Temperature: may be size selective: small fish moving at higher temps (e.g. 7.5-12 ^{o}C, vs large fish at <7.5 ^{o}C.	Little data for sea trout, but thought similar to salmon i.e. combined high flow and temps required to trigger emigration.	Klemetsen *et al*., 2003.

Life stage / process	Habitat criteria	Comment on flow modification	Source
Adult upstream migration	Cover needs of large boulders, undercut banks and deep water.	Less flow demand than salmon.	Armstrong *et al*., 2003.
Adult spawning, tributary entry	Stimulus can be high flows and or temperature.		Campbell (1977)
Adult spawning	Depth: 27-50cm velocity: av 20-40 cm s^{-1} (adapted from B&M, 1999). cover need close to give protection; night time spawning. Q variation probably disruptive.	Broadly, as salmon, size for size; but probably less flow- dependent.	Baglinière and Maisse (1999).
Kelt downstream migration	As salmon, probably		

I.12 Grayling (*Thymallus thymallus* L.)

I.12.1 Overview

A member of the family Salmonidae, grayling is a free-swimming, mid-water, shoaling species living in the faster sections of rivers and overlapping in distribution with trout and salmon, but generally tend to prefer deeper slower flowing water than trout. Habitat requirements have been less extensively studied than trout or salmon and on a more restricted range of river types.

Spawning occurs in March April, and they can make extensive movements to reach suitable spawning habitats. Eggs are laid at shallow depths (e.g. 5 cm, and therefore vulnerable to washout) in fine gravel, e.g. 2-8 cm, with admixture of finer material (Ibbotson et al. 2001) and hatch rapidly the same spring. Velocities and depths utilized by spawning grayling have been reported to be between 20 and 90 cm s^{-1} and 10-40cms. The newly emerged fry prefer marginal, slower flowing habitats (e.g. $0.1 ms^{-1}$). Marginal habitats are left after the fish reach around 6 cm. Utilized velocities of older fish vary across study sites, but lie with the range 0.2 to 1.1 m s^{-1}, depending upon the substrate (apparently tolerating faster water over larger (e.g. boulder) substrates (Riley and Pawson, 2010, Ibbotson et al. 2001). They seek out deeper water in winter. Flow tolerance increases with size and age; for example in a chalk stream 1+/2+ fish occupied 40-70 cm depth with velocities 30 - 50 cm s^{-1}, whereas 0+ juveniles preferred shallower 30-40 cm and slower (10 -20 cm s^{-1}) (Sagnes et al. 1997; Ibbotson et al. 2001; Lucas and Bubb, 2005; Riley and Pawson, 2010). Their swimming ability is rather less than other salmonids and they may be blocked by fishpasses designed for trout and salmon (Lucas and Bubb, 2005).

I.12.2 Spatial scale

Home ranges on the Welsh Dee have been found to be mostly (76%) within 1 km (Wooland, 1972). However, longer migrations occur at spawning time, when they move upstream to spawn in shallow gravel beds. The linear extent of habitat use by grayling varies considerably, tracking studies have shown that spawning migrations may be 0.2 to 3.5 km (Lucas and Bubb, 2005).

As for the salmonids, habitat diversity needs to be maintained, along with connectivity required between spawning, nursery and rearing areas.

Barriers formed by low flows can prevent passage through rivers.

I.12.3 Temporal scale

Strong diurnal shifts on habitat preferences have been reported with fish moving to slower or still water at night (Bardonnet and Gaudin, 1991).

I.12.4 Temperature

Critical survival temperatures depend upon acclimation but Crisp (1996), quoted in Ibbotson et al. 2001) suggests: minimum: s 0-4 ^{o}C, maximum: >18 ^{o}C, and optimum around 18 ^{o}C.

I.12.5 River type variation

Grayling are present in many rivers of the British Isles and Europe, although it has been extensively introduced beyond its natural distribution. Grayling occur in the overlap zone between trout and rheophilic coarse fish and have been awarded their own nominal zone, the “grayling zone” (Huet, 1959). Need local knowledge to establish occurrence.

I.12.6 Ecosystem relations

Food requirements overlap with trout and salmon, but there appears to be effective resource partitioning and no strong evidence of inter-specific competition (Ibbotson et al. 2001).

I.12.7 Ecological indicator potential

Their value is dependent upon their local introduction status and fish surveys required. However, they require deep slow water at night and juveniles are vulnerable to washout or marginal dewatering through hydro-peaking, which has been explicitly identified as a limiting factor for grayling (Valentin et al. 1994). In contrast, artificially stabilised flows on the Welsh Dee have been shown to be beneficial (Ibbottson et al. 2001).

I.13 River lamprey (*Lampetra fluviatilis* L.)

I.13.1 Overview

The river lamprey typically enters European rivers in the late summer and autumn (Winter and Van Densen, 2001), after which they can spend several months in fresh waters prior to spawning between March and May (Kelly and King, 2001; Kearn, 2004). Eggs are laid in crude gravel beds (presumably the gravel would need to be a certain size range/texture with water of a specific velocity and depth range, although these are unavailable in the current literature search) and hatch after 2 weeks and larvae remain in the gravel for a further 1-3 weeks. On emergence from the gravel beds they move downstream and require silt beds in sheltered areas (low discharge thus required to allow fish to settle and avoid removal of fine particle substrate. No drying out should be permitted. Flows should not exceed 0.01-0.5 ms^{-1} over the bed (Potter, 1980a, b), although this reduces as larval concentrations increases (Kelly and King, 2001)), with particle sizes between 0.5 and 3.8 mm to create burrows. During the burrowing life stage they are filter feeders thus low flows are required to provide a constant food source of diatoms etc. Larvae move downstream mainly during the night and this is seasonal and temperature dependent. Rivers with long shallow longitudinal profiles with limited flow leads to relatively little downstream displacement. Where the stream has a high or logarithmic profile and average gradients tend to be greater, there is often a marked gradation in the proportions of larvae of different size groups according to the distance below the spawning areas, with the older larvae becoming increasingly predominant in the downstream regions (Hardisty et al. 1970). Passive migration during flooding is also a major factor in the redistribution of larvae. Conversely, the movement of larvae can also be produced by a reduction in water levels during periods of low rainfall.

They migrate to the sea as Macrophthalmia after approximately 4.5 years. This is nocturnal and occurs during the winter, triggered by a marked increase in freshwater discharge (if flow is too low then migration can be delayed; Potter, 1980b) and a decrease in temperature. The exact timing of seaward migration varies depending on a combination of these stimuli.

The return migration (approx. 30cm in length) during late summer and autumn occurs when there is a high river discharge and low light intensity, being exclusively nocturnal (Kelly and King, 2001).

I.13.2 Spatial scale

Migration access essential, sea to spawning ground; up and downstream migration.

Dispersal from spawning sites to larval burrowing site = Unknown, but presumably depends on river type and availability of suitable substrate (see overview). Habitat diversity maintenance: connectivity required between spawning and burrow/residential habitat.

Barriers formed by low flow prevent passage through rivers, and where flow is too large distance between suitable burrowing substrate may be too great (due to removal of fine bed material).

I.13.3 Temporal scale

Run timing very river- specific (generally are later at more northern latitudes) requiring local knowledge.

I.13.4 Temperature

Spawning occurs when temperature reaches approximately 10-11^{o}C.

I.13.5 River type variation

Present in all river types throughout Western Europe. Absence in many rivers is due to pollution, the presence of migratory barriers (e.g. dams and weirs), and river engineering (Maitland, 2003).

I.13.6 Ecosystem relations

Food requirements dependent upon adequate production and flow delivery (primarily drift feeders).

I.13.7 Ecological indicator potential

Requires electric-fishing surveys, or nest counts and adult stock assessment. But responsive to flow. Presence/absence may reflect spawner access, gravel quality or silt quality for juveniles to create burrows.

I.14 Sea lamprey (*Petromyzon marinus* L.)

I.14.1 Overview

The sea lamprey typically enters European rivers in the spring to early summer after which they can spend several months in fresh waters prior to spawning between May and June (Kelly and King, 2001).

Eggs are laid in crude gravel beds (presumably the gravel would need to be a certain size range/texture with water of a specific velocity and depth range, although these are unavailable in the current literature search) and hatch after 2 weeks and larvae remain in the gravel for a further 1-3 weeks. On emergence from the gravel beds they move downstream and require silt beds in sheltered areas (low discharge thus required to allow fish to settle and avoid removal of fine particle substrate. No drying out should be permitted. Flows should not exceed 0.6-0.8 ms^{-1} over the bed (Thomas, 1962), although this reduces as larval concentrations increases (Kelly and King, 2001), with particle sizes between 1.8 and 3.8 mm to create burrows. During the burrowing life stage they are filter feeders thus low flows are required to provide a constant food source of diatoms. Larvae move downstream mainly during the night and this is seasonal and temperature dependent. Rivers with long shallow longitudinal profiles with limited flow leads to relatively little downstream displacement. Where the stream has a high or logarithmic profile and average gradients tend to be greater, there is often a marked gradation in the proportions of larvae of different size groups according to the distance below the spawning areas, with the older larvae becoming increasingly predominant in the downstream regions (Hardisty et al. 1970). Passive migration during flooding is also a major factor in the redistribution of larvae. Conversely, the movement of larvae can also be produced by a reduction in water levels during periods of low rainfall.

They migrate to the sea as Macrophthalmia after approximately 5-6 years. This is nocturnal and occurs during the winter, triggered by a marked increase in freshwater discharge (if flow is too low then migration can be delayed; Potter, 1980b) and a decrease in temperature. The exact timing of seaward migration varies depending on a combination of these stimuli.

The return migration (approximately 30 cm in length) during spring and early summer occurs when there is a high river discharge and low light intensity, being exclusively nocturnal (Kelly and King, 2001).

I.14.2 Spatial scale

Migration access essential, sea to spawning ground; up and downstream migration.

Dispersal from spawning sites to larval burrowing site = Unknown, but presumably depends on river type and availability of suitable substrate (see overview).

Habitat diversity maintenance: connectivity required between spawning and burrow/residential habitat.

Barriers formed by low flow prevent passage through rivers, and where flow is too large distance between suitable burrowing substrate may be too great (due to removal of fine bed material).

I.14.3 Temporal scale

Run timing very river- specific (generally are later at more northern latitudes) requiring local knowledge.

I.14.4 Temperature

Spawning occurs when temperature reaches approximately 15°C.

I.14.5 River type variation

Present in all river types throughout Northern and Western Europe and Eastern North America. Absence in many catchments is due to pollution, the presence of migratory barriers (e.g. dams and weirs), and river engineering (Maitland, 1980 and 2003).

I.14.6 Ecosystem relations

Food requirements dependent upon adequate production and flow delivery (primarily drift feeders).

I.14.7 Indicator potential

Requires electric-fishing surveys, or nest counts and adult stock assessment. But responsive to flow. Presence/absence may reflect spawner access, gravel quality or silt quality for juveniles to create burrows.

I.15 Brook lamprey (*Lampetra planeri* L.)

I.15.1 Overview

The brook lamprey is a resident fresh water species requiring similar habitats to the juvenile river lamprey. They spawn between March and May (Kelly and King 2001).

Eggs are laid in crude gravel beds (presumably the gravel would need to be a certain size range/texture with water of a specific velocity and depth range, although these are unavailable in the current literature search) and hatch after 2 weeks and larvae remain in the gravel for a further 1-3 weeks. On emergence from the gravel beds they move downstream and require silt beds in sheltered areas (low discharge thus required to allow fish to settle and avoid removal of fine particle substrate. No drying out should be permitted. Flows should not exceed 0.3-0.5 ms^{-1} over the bed (Maitland, 1980), although this reduces as larval concentrations increases (Kelly and King, 2001)), with particle sizes between 1.8 and 3.8 mm to create burrows. During the burrowing life stage they are filter feeders thus low flows are required to provide a constant food source of diatoms. Larvae move downstream mainly during the night and this is seasonal and temperature dependent. Rivers with long shallow longitudinal profiles with limited flow leads to relatively little downstream displacement. Where the stream has a high or logarithmic profile and average gradients tend to be greater, there is often a marked gradation in the proportions of larvae of different size groups according to the distance below the spawning areas, with the older larvae becoming increasingly predominant in the downstream regions (Hardisty et al. 1970). Passive migration during flooding is also a major factor in the redistribution of larvae. Conversely, the movement of larvae can also be produced by a reduction in water levels during periods of low rainfall.

The return upstream (approximately 7 cm in length) after approx. 6.5 years just prior to spawning when there is a high river discharge and low light intensity, being exclusively nocturnal (Kelly and King, 2001).

I.15.2 Spatial scale

Migration access essential, up and downstream migration between residential and spawning habitat is required.

Dispersal from spawning sites to larval burrowing site distance = Unknown, but presumably depends on river type and availability of suitable substrate (see overview).

Habitat diversity maintenance: connectivity required between spawning and burrow/residential habitat.

Barriers formed by low flow prevent passage through rivers, and where flow is too large distance between suitable burrowing substrate may be too great (due to removal of fine bed material).

I.15.3 Temporal scale

Run timing very river- specific (generally are later at more northern latitudes) requiring local knowledge.

I.15.4 Temperature

Spawning occurs when temperature reaches approximately 10-11 ^{o}C.

I.15.5 River type variation

Present in all river types and in a number of lakes throughout North West Europe.

Absence in many catchments is due to pollution, the presence of migratory barriers (e.g. dams and weirs), and river engineering (Maitland, 2003).

I.15.6 Ecosystem relations

Food requirements dependent upon adequate production and flow delivery (primarily drift feeders).

I.15.7 Ecological indicator potential

Requires electric-fishing surveys, or nest counts and adult stock assessment. But responsive to flow.

Presence/absence may reflect spawner access, gravel quality or silt quality for juveniles to create burrows.

I.16 European eel (*Anguilla anguilla* L.)

I.16.1 Overview

European eels migrate from the sea and enter freshwater as elvers between April and September (Solomon and Beach, 2004) under primarily the nocturnal period. Elvers require a low attraction flow to stimulate upstream movement, if flow is too high they are unable to swim against it. Elvers cannot swim against flows > 0.5 ms^{-1}, but can swim freely at flows < 0.4 ms^{-1} (McCleave, 1980). During upstream movement shallow depths can be tolerated over short distances as long as the substrate is moist. Upon reaching their residential habitat upstream they remain resident for several years before their spawning run to the sea as silver eels during August to November (Tesch, 2003). This is triggered by increased flow and coinciding with low illumination during the night, i.e. the dark of the moon, and if conditions are correct the vast majority of fish can migrate to the sea in one or two nights during the potential migratory months (Tesch, 2003).

I.16.2 Spatial scale

Migration access essential, river headwaters to spawning ground; up and downstream migration.

Access of resident adult (yellow eels) to several KM of feeding habitat is required for survival (Parker, 1995).

I.16.3 Temporal scale

Spawning run timing very river- specific. Dependent on antecedent conditions in the area (i.e. rainfall, cloud cover during the night and moon phase at time of freshets occurring).

I.16.4 Temperature

The optimum temperature for juvenile growth is between 22-23 ^{o}C (Sadler, 1979). Upstream migration of elvers is triggered when temperature reach 10-11^{o}c or above, and increases in rate at temperatures above 15-16 ^{o}C (White and Knights, 1997).

I.16.5 River type variation

Present in all river types throughout Europe and The Mediterranean and Northern Africa.

I.16.6 Ecosystem relations

Poaching increases at obstructions and at low flows.

Catch success and economic values are also flow-related.

I.16.7 Ecological indicator potential

Requires electric-fishing surveys and adult stock assessment. But responsive to flow. Presence/absence may reflect elver access.

I.17 Bullhead (*Cottus gobio* L.)
I.17.1 Overview

Bullhead is listed in Annexe II of the EU Habitats Directive; however there is no other conservation legislation in place and it is not a UK BAP species. Its IUCN Red List status is "Least Concern".

Being morphologically adapted to thriving in rheophylic habitats, bullhead demonstrates considerable geographic range with populations recorded from the majority of freshwater system types in Britain and continental Europe. However, lotic habitat requirements for the species can be quite specific with populations heavily dependent on the availability of suitable substrate and instream habitat features within a heterogeneous flow environment.

Maintenance of populations in lakes demonstrates that bullhead distribution is not limited directly by flow (however, depth is limiting with a lower threshold of 5 cm (Perrow et al., 1997)). Nevertheless altered flow regimes in rivers may limit population sizes indirectly via a reduction in the abundance and density of instream habitat features. These structures are critical for bullhead populations as their cryptic behavior, high predation potential and poor swimming ability results in their reliance on behavioral as opposed to physical adaptations to survive extreme disturbances and predation. Minimum acceptable flows are also likely to impact reproductive capacity as an indirect driver for sedimentation and water quality. Ultimately these variables impact on the clean substrates on which bullhead rely for spawning and refuge. Previous studies conducted on a Southern chalk stream, reported significant relationships between flow and population performance, with positive relationships between flow, carrying capacity and 0+ growth observed.

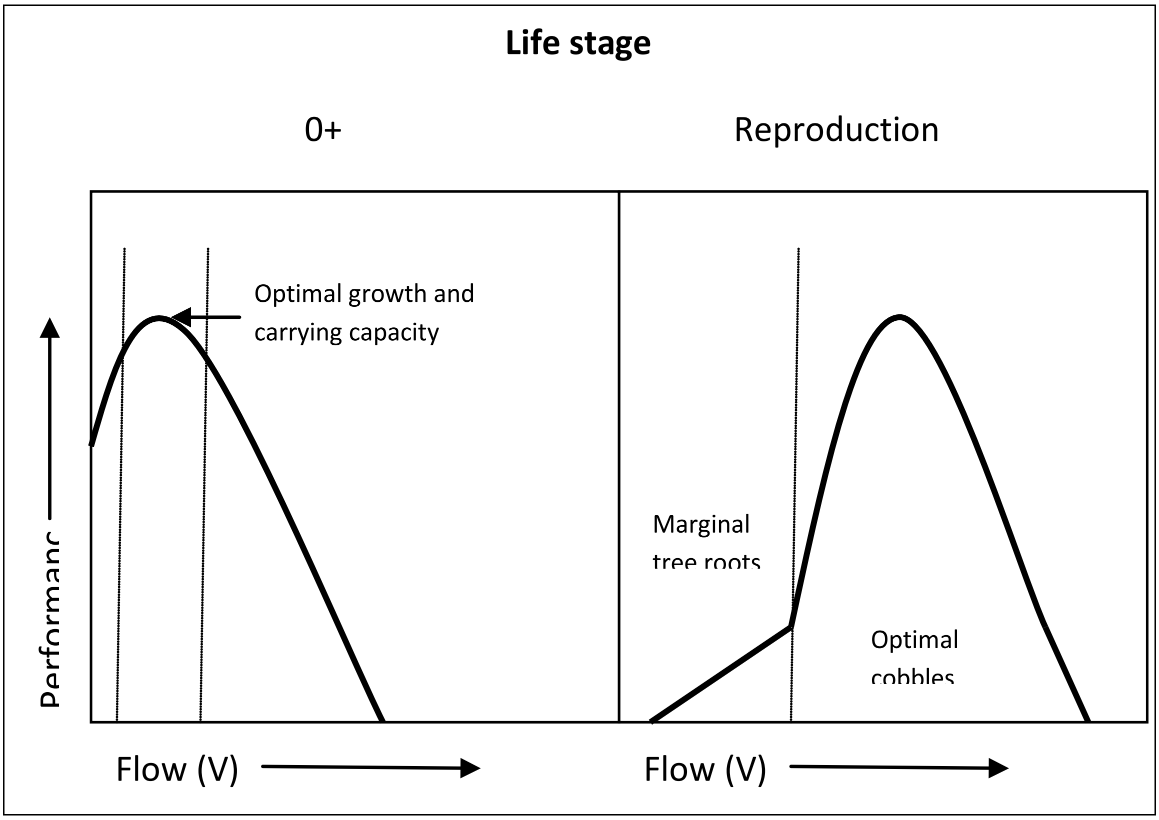
Egg deposition and fertilization occurs in nests excavated in the substrate below larger rocks. The micro eddy created by the large rock may facilitate fertilization of the eggs by retaining sperm in the nest area (Tomlinson & Perrow, 2003). Although oxygenation and waste removal of the eggs is facilitated by "fanning", alterations to flow rates will reduce the instances of suitable nest sites as well as increase the risk of nest exposure in shallower areas. Pulses of water released from impoundments may also increase the risk of washout.

Habitat requirements are specific to life history stages. Young of year (YOY) fish are often found in interstitial spaces in riffles, while older fish prefer deeper water with high levels of instream habitat features. However, during disturbances, all age groups require slack water refuges (Perrow et al. 1997). Stony substratum is not just utilized during reproduction; a preference is also shown outside of the reproductive period. Bullhead populations can persist at quite low densities ($0.002/m^2$ – $0.41/m^2$) (Utzinger et al. 1998); therefore, any factors which reduce in-channel connectivity could impact negatively on population performance.

Because bullhead populations can persist in stillwaters lacking any discharge, there is no lower threshold of velocity requirement (see intercept with Y-axis). Within rivers, however, optimal habitat quality and availability will be governed by flow, with a range of velocities to support an optimal balance of food availability and bioenergetic expenditure. As velocities increase beyond this optimal range, there is a concomitant decrease in the carrying capacity and growth of juveniles as habitat suitability and energy budgets become compromised.

With respect to reproduction, bullhead demonstrate successful recruitment within still waters, but some water movement (even mediated by wind within the margins of lakes) is likely to be necessary to maintain clean oxygenated substrates within which eggs can incubate. As such, the recruitment performance versus velocity curve demonstrates that increases in velocity at lower flows results in a gradual increase in the performance of egg incubation. This is until a lower threshold velocity is breached at which point, velocities within the lotic channels become adequate to maintain areas of clean substrate. This allows access to a greater diversity of spawning habitats and an optimal range of velocities to aid the successful fertilisation and incubation of eggs.

Figure I 1 - Flow and life stage relationship - bullhead



I.17.2 Spatial scale

In part due to a largely sedentary life history and low mobility (Downhower et al. 1990), any factors which reduce instream habitat and refuge can have serious consequences for bullhead populations. Low flows result in habitat homogeneity through sedimentation of substrata, uniformity of flow and a reduced rate of creation and incorporation of habitat structures. This leaves bullhead susceptible to increased rates of predation and an inability to resist disturbances such as drought and periods of high flows. At the edges of the bullheads' range, habitat fragmentation caused by low flows may completely prevent reproduction in widely dispersed populations.

I.17.3 Temporal scale

Adults are typically sedentary and do not necessarily move too far to complete reproduction. However, density of suitable nest sites and distribution of post-hatch life stages can be negatively impacted by flow alterations through sedimentation and connectivity with juvenile habitat. Deviations from the natural flow regime can also exacerbate poor instream habitat and morphological quality resulting in inter-annual declines in population numbers.

I.17.4 Temperature

Thermal limits -4.2 and 27.7 ^{o}C (Elliot & Elliott, 1995).

I.17.5 Ecosystem relations

There is a strong negative relationship between the distributions of bullhead and the invasive signal crayfish, *Pacifastacus leniusculus*. As invasions are often facilitated by changes to natural regimes, it is possible that changes to flow patterns and volumes could encourage crayfish invasions into bullhead rivers.

Changes in flow patterns could also alter the dynamic between bullhead and brown trout, as under natural conditions bullhead can co-exist with brown trout (their main predator) at quite high densities (Prenda et al. 1997). This co-occurrence is likely made possible by the presence of suitable refuge (Perrow et al. 1997).

I.17.6 Ecological indicator potential

As bullhead distribution depends primarily on preferred habitat, even over prey availability (Welton, 1991), they should be considered good indicators of hydrologic alterations. They could be considered especially good indicators in rivers of poor morphological quality.

Specialist sampling methods are required to accurately quantify 0+ abundance.

I.18 Coarse fish – general

I.18.1 Overview

Typically occupying the mid to lower reaches of river systems, coarse fish are can be characterized as a collection of potadromous species which lack an adipose fin. Species are highly variable in their sensitivity to flow, with ontogeny being the key driver of temporal flow and physical habitat requirements. Consequently habitat diversity and floodplain connectivity are of key importance to satisfying the conflicting environmental requirements of species and indeed intra-species life stages across the same timescales.

The spawning season extends between spring and early summer and spawning is stimulated through a combination of elevating water temperature and photoperiod. Egg incubation times are temperature dependant and considerably shorter than salmonids, with hatching occurring between 4 and 15 days. Newly hatched larvae (free embryos) of most species have poor locomotive capabilities and depend on the availability of low flow habitats in order to absorb initial yolk reserves and exploit suitable dietary resources. Early ontogeny is complex, with larvae often requiring access to a range of microhabitats during the first summer.

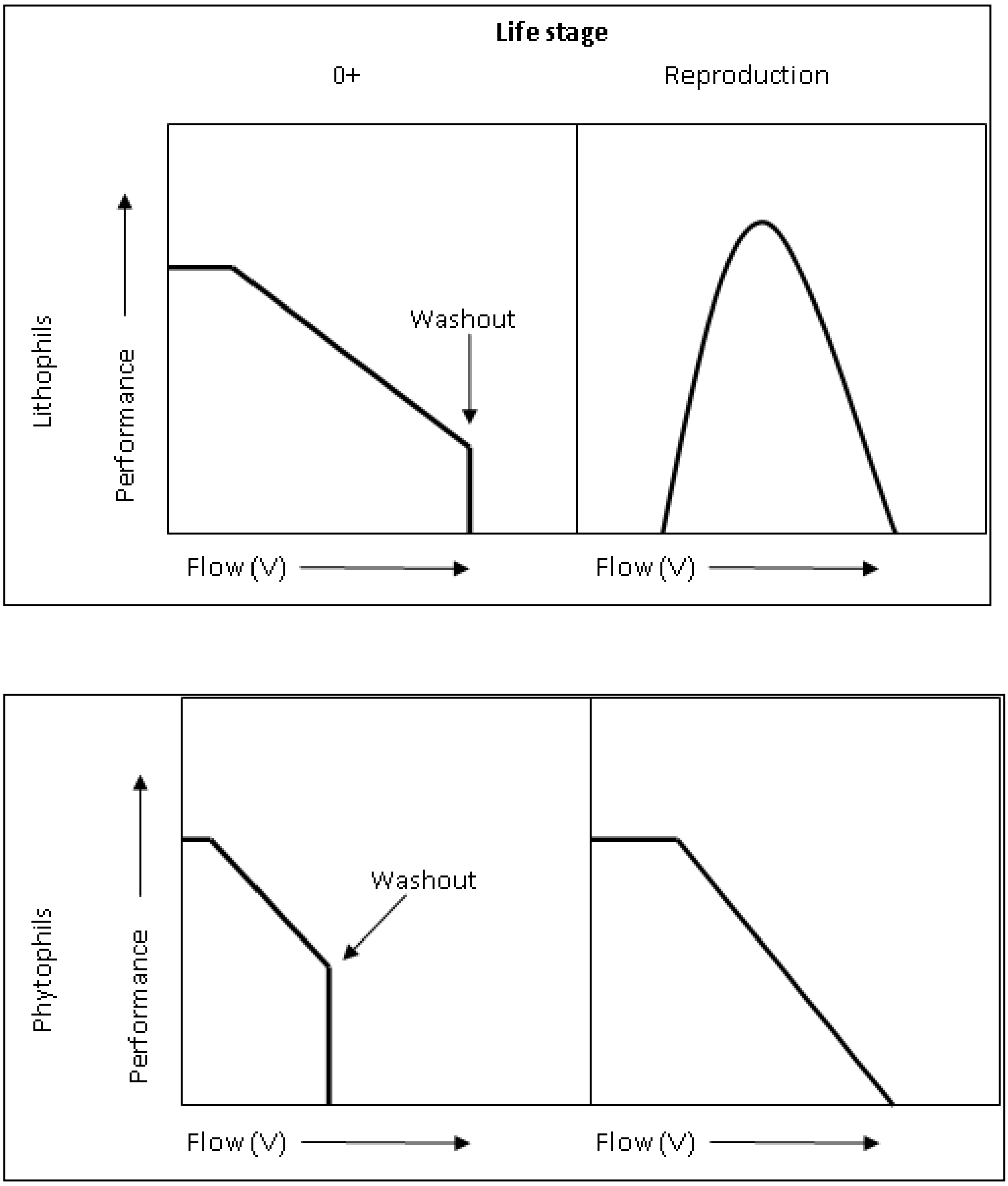
While there is considerably greater plasticity in the environmental requirements of most coarse fish than salmonids, coarse fishes can be usefully separated into two major reproductive guilds which assists in defining grouped consistencies in life history traits and flow requirements. These are the lithophils (spawning on stones and typically favoring running water) and phytophils (spawning on plants and favoring slow/still water). This is a gross oversimplification of the guild system proposed by Balon (1975, 1981) and summarised by Mann (1996) and there are species which overlap between these guilds (phytolithphils). With respect to flows, lithophilic species are more sensitive to river hydraulics and depend on the availability of adequate flows to maintain areas of clean well oxygenated gravels; and indeed migratory access to such habitats in synchrony with their physiological readiness to spawn. Provided that physiological processes are not compromised by water quality, populations of phytophilic species perform better under consistently low or no flow, hence their relative abundance in the lower reaches of river systems and in particular lowland catchments. With the exception of a small number of species, a consistent requirement across the spawning guilds is the availability of low (or nil) flow habitats in which larvae and juveniles can develop, grow, avoid predation and overwinter. Consequently, in addition to the limited temporal flow requirements of the lithophils, all species are strongly dependent on water level throughout the year, as this controls connectivity and access to the greater diversity of habitats supported within the wider floodplain. As a general rule, coarse fish demonstrate greater sensitivity to high flows than salmonids, with poor annual recruitment often linked to above average discharge years.

Due to the spatial and temporal dynamics of the environmental requirements of coarse fish, previous studies on flow requirements have acknowledged that in order to determine the long term influence of hydrograph characteristics on the performance of coarse fish populations, further work needs to be undertake. Hence, studies have tended to focus on defining velocity and depth requirements independently of discharge as a proportion of annual Q (Cowx et al. 2004). Indeed, it is considered that it may not be possible to definine generic flow requirements for coarse fishes and it is likely that flow rules would need to be developed for individual species and adjusted for specific reach/river types (Cowx et al. 2004).

I.18.2 Spatial scale

Longitudinal migratory access is more important to lithophylic species. This is to allow larger scale upstream migrations which allow access to higher gradient habitats (riffles) and compensate for downstream drift/dispersal of larvae. With the exception of bullhead, all species benefit from lateral connectivity which facilitates access to the floodplain. Juveniles and adults of many species also utilize tidal reaches for both nursery and feeding habitats.

Figure I 2 - Flow and life stage relationship – coarse fish



I.18.3 Temporal scale

Adequate flows and levels which allow access to spawning and nursery habitats need to be synchronized with species temporal requirements. Ontogeny during early development is highly complex and swimming capacity, prey capture and predator avoidance performance are thought to develop as a series of salutatory steps (thresholds) (Balon, 1979, 1984) which correspond with sensitivities to high flows in particular (Mann & Bass, 1997, Gozlan et al. 2005). Conflicting flow requirements of species and life stages can only be maintained through the provision of lateral connectivity, habitat diversity and resource partitioning.

Incubating eggs (of phytophils in particular) are temporally critically sensitive to fluctuations in water level. This is due to eggs being deposited on macrophytes just below the water surface, where they are vulnerable to sudden falls in water level. This can be caused by natural flow variations, weed cutting or the operation of sluices (Mann, 1996). Consequences of egg drying invariably result in mortality and potential loss of an entire year class.

I.18.4 Temperature

Positive relationship between embryonic development and growth with temperature. Upper tolerance temperatures varying with species, but considerably higher than salmonids. Cumulative degree-days >12 ^{o}C considered to provide the best correlation with cyprinid growth (Mills and Mann, 1985).

I.18.5 River type variation

Coarse fish performance and community structure strongly influenced by catchment geomorphology.

Lithophils dominate in the mid to downstream reaches of unimpounded upland catchments.

Phytophils dominate large lowland catchments.

Catchment gradient/geology and river engineering are also important considerations, with overwintering success of the 0+ cohort and ultimately recruitment being more sensitive to discharge in rivers which are prone to large and rapid fluctuations in flow (Nunn et al. 2007).

Angling performance and economics are also influenced by flow.

I.18.6 Ecosystem relations

Phytoplankton/zooplankton dynamics and food availability – retention and washout governed by discharge.

Prey for otters and several species of bird. Flows determine fish distribution, local densities and ultimately predation mortality.

I.18.7 Ecological indicator potential

Larval and juvenile fish surveys required during summer to assess spawning success (thus flow dependent habitat functionality). Adult surveys and population demographic analysis required to assess overwintering success and relate year class strengths to discharge.

Hybridization indicates a lack of microhabitat partitioning and potentially flow driven limitations

High ratio of phytophilic to lithophilic species indicates inadequate flows to either promote migration and/or maintain the availability of clean gravel habitats.

I.18.8 Suggested field indicators of Poor and Bad status

- Increased ratio of plant-spawning to gravel spawing coarse fish (2e)
- Poor summer recruitment of phytophilic coarse fish (2f)
- Poor winter survival of phytophilic and lithophilic coarse fish (2g)
- Poor summer survivial of lithophilic phytophilic coarse fish (2h)

I.19 Freshwater pearl mussel (*Margaritifera margaritifera* L.)

I.19.1 Overview

Viable populations in the UK are restricted to a handful of Scottish highland rivers. These populations represent up to half of the world's known populations with active recruitment. Life cycle comprises a larval (glochidial) stage living attached to the gills of salmon or trout, a juvenile stage living interstitially in the river bed, and an adult stage, living as filter feeder. Adults are more tolerant of a wider range of in-river conditions than juveniles (Skinner, Young and Hastie, 2003).

- Larvae (glochidia) depend on the host species for survival (salmon and trout), so flow and habitat requirements for the host species are most important.
- Juveniles are most sensitive to river flow and habitat. Minimum water depth = 0.1m; Maximum depth = 2 m. Minimum water velocity = 0.1 m/sec; maximum water velocity = 2 m/sec. Optimum water depth in Scottish rivers = 0.3-0.4m; optimum water velocity = 0.25-0.75 m/sec (Hastie, Boon and Young, 2000).
- Highly sensitive to fine deposited sediment, particularly with high organic content. Moderate flooding thought to be beneficial to flush light, fine sediment from interstices and to promote fairly stable, coarse sand and gravel substratum (Hastie, Boon and Young, 2000).
- The UK's largest population of FWPMs is on the River Kerry, which is regulated by a hydroelectric dam. Recent research suggests that river regulation has benefited this population by creating the ideal hydraulic conditions: a) dampening peak flows – promoting substratum stability and b) removing very low flows – preventing fine sediment deposition (Thomas and Hoey, unpublished data).

Summary of abstraction effects:

- Low flows and increased fine sediment deposition is highly detrimental to pearl mussels.

Summary of river regulation effects:

- Flow regimes that increase substratum stability by dampening peak flows are beneficial.

Flow regimes that maintain base flows to eliminate very low flows – preventing fine sediment deposition and maintaining good water flow through substratum is beneficial

I.19.2 Spatial scale

Once settled, spatial movement is limited. Dispersal and spatial distribution depends on salmonid hosts. See salmon and trout requirements.

I.19.3 Temporal scale

Potentially highly sensitive to flow and habitat conditions for the first 10-15 years of life until maturity. After maturity, more resistant to sub-optimal in-river conditions at the individual level, but sub-optimum conditions will prevent recruitment (Skinner, Young and Hastie, 2003).

I.19.4 Temperature

No information

I.19.5 River type variation

Restricted to oligotrophic upland rivers. Viable populations are now restricted to a handful of Scottish highland rivers (Skinner, Young & Hastie, 2003).

I.19.6 Ecosystem relations

Reproduction depends on salmon and trout as hosts for larvae (glochidia).

I.19.7 Ecological indicator potential

Absence of pearl mussels may not automatically indicate Poor or Bad status. This should be discussed in workshop. Presence of adults does not necessarily indicate ideal conditions, as they are tolerant of sub-optimal conditions. No potential as direct indicators.

I.20 White clawed crayfish (*Austropotamobius pallipes* L.)

I.20.1 Overview

Crayfish distribution is determined largely by geology and water quality. Populations are mostly restricted to relatively hard, mineral rich and calcareous water.

Requirements in terms of water quantity are water depths of 5 cm to 125 cm. It will occur in small streams of 0.5 m wide (Holdich, 2003).

Summary of abstraction effects:

Persistent low water levels due to drought or over abstraction can be devastating to local crayfish populations, increasing their vulnerability to predation. Crayfish can persist under stones in ephemeral watercourses (there are no details on how long) (Hodich, 2003).

Summary of river regulation effects:

Crayfish can occur in deeper rivers up to 2.5 m deep. Crayfish can survive in rivers with strong flows (no details provided on flow/velocity thresholds for this) as long as there are adequate refuges, such as weirs and boulders (Holdich, 2003).

I.20.2 Spatial scale issues

Movement of crayfish in a stream is generally limited to within reaches and not more than around 80 m in a given year. Crayfish populations can easily become fragmented in streams and are unlikely to expand rapidly to colonise new or favourable habitat. Populations impacted by abstraction or regulation are unlikely to recover quickly.

I.20.3 Temporal scale issues

June to September is the time when juvenile crayfish are released and when moulting occurs, and when they are most sensitive to environmental stresses. Low flows from abstraction or high flows from impoundments at this time of year can be damaging to crayfish.

I.20.4 Temperature

High mortality occurs when water temperatures exceed 28^{o}C (Firkins & Holdich, 2003). Minimum lethal temperatures are not available.

I.20.5 River type variation

Crayfish distribution is determined largely by geology and water quality. Populations are mostly restricted to relatively hard, mineral rich and calcareous water.

I.20.6 Ecosystem interactions

They are vulnerable to displacement by invasive crayfish, especially the signal crayfish. When setting regulated flow regimes, consideration must be given to preventing invasion by alien species.

I.20.7 Ecological indicator potential

Crayfish are unlikely to persist in watercourses that are severely affected by human water use and of Poor or Bad status, so presence will confirm status as above Poor or Bad. Absence does not indicate Poor or Bad status. Not a direct indicator of the severe effects of abstraction or river flow regulation consistent of Poor or Bad status.

I.21 Aquatic macrophytes

I.21.1 Overview

Submerged and floating plant communities occurring in UK rivers are of international importance in their own right and are listed on Annex II of the Habitats Directive. Additionally, riverine plant communities provide important habitat and food for aquatic animals, particularly invertebrates and fish. Submerged and floating aquatic plants have a finely balanced relationship with the flow regime: aquatic plants are influenced by the flow regime, but the flow regime is also influenced by them.

Flow regime and velocity have been highlighted as primary factors influencing the abundance and condition of *Ranunculus* in UK rivers (Environment Agency, 2001). Yet despite this, far more attention has been paid to how aquatic plants effect the dynamics of water flow in rivers by their growth (e.g. Chambers et al. 1991; Marshall & Westlake, 1990; Owens and Edwards, 1961, 1962; Kondolf et al. 1987; O'Hare et al. 2007), rather than how the hydraulics of water flow effect the growth and survival of aquatic plants.

The influence of water flow on macrophyte growth in rivers has been most extensively studied in lowland chalkstreams in England (e.g. Westlake, 1967; Holmes, 1996; Westwood et al. 2006).

Water velocity is of particular importance for determining the growth and survival of aquatic plants, for example, Chambers et al. (1991) suggested that when current velocity exceeds 0.01 m/sec, plant biomass decreases and macrophytes are rare when velocity exceeds 1.0 m/sec.

The effect of the quantity and dynamics of water flow has been most extensively studied for water crowfoots (*Ranunculus* spp.), for example, the growth pattern of *Ranunculus penicillatus* subsp. *psuedofluitans* coincides with the maximum flow in chalk streams. Spring and early summer are thought to be the critical seasons when some aquatic macrophytes are most sensitive to flow change and protection of flows (Acreman et al. (2008). Additionally, 0.1 m/sec is often quoted as the maximum velocity to maintain the growth of *Ranunculus penicillatus* subsp. *psuedofluitans* (Cranston and Darby, 2004). *Ranunculus penicillatus* subsp. *psuedofluitans* prefers water depths of between 50 and 100 cm, whereas *R. peltatus* prefers shallower depths of 0 to 30 cm (Newbold and Mountford, 1997).

For aquatic plants in general, submerged fine leaved macrophytes and mosses often occur in fast water (>0.5 m/sec); submerged broad leaved macrophytes in deeper moderately fast water (~0.4 m/sec); and emergent macrophytes in slower water (0 – 0.05 m/sec) (Hatton-Ellis, Greive and Newman, 2003).

Low flow/abstraction effects: Referring to line colours in the conceptual illustration below (Fig. I 3) (using river types described in Hatton-Ellis, Greive and Newman, 2003).

- Increased deposition of fine sediment in all river types can smother submerged plants which cannot alter their rooting depth. In chronic low flow conditions can cause die back.

- Stable exposed substratum due to persistent low or regulated stable flows are associated with increased vegetation cover of perennial species relative to annual species (Holmes et al. 1972; Holmes and Whitton, 1977).
- Increase in emergent plants in the channel replacing submerged and floating species in river types CB1, CB2, CB4 and CB6a (orange line) (Table I 3; Fig. I 3; Holmes, 1996; Hatton-Ellis, Grieve and Newman, 2003; Westwood at al. 2006)
- Increase in terrestrial plant species in river margins in river types CB3, CB4, CB5 and CB6b (red line) (Table I 3; Fig. I 3; Holmes, 1996; Hatton-Ellis, Grieve and Newman, 2003; Westwood at al. 2006)
- Decrease in aquatic plant cover in river types CB4, CB5 and CB6b (Hatton-Ellis, Grieve and Newman, 2003).
- Increase in filamentous algae which can smother submerged macrophytes in all river types. Generally, this is temporary in high energy river types, but its growth and negative effects are prolonged in chronic low flow conditions (purple line)
- Persistent still water conditions in rivers are dominated by pondweeds, free-floating and non-rooted macrophytes, which are useful indicators (Janauer et al. 2010).
- Increase in abundance of *R. peltatus* (blue line) relative to *Ranunculus penicillatus* subsp. *psuedofluitans* (green line) (Hatton-Ellis, Grieve and Newman, 2003) (Fig. I 3).

High flow/river regulation effects:

- Macrophytes in rivers downstream of reservoirs are generally more productive because of lack of scouring flows (Acreman et al. 2008).
- Elevated flow velocity >1 m/sec is likely to be damaging to submerged macrophytes.
- Elevated flows are likely to clean macrophyte stands of old growth (Acreman et al. 2008).

Figure I 3 - A simple conceptual model describing the effect of decreasing water flow on macrophyte abundance/cover

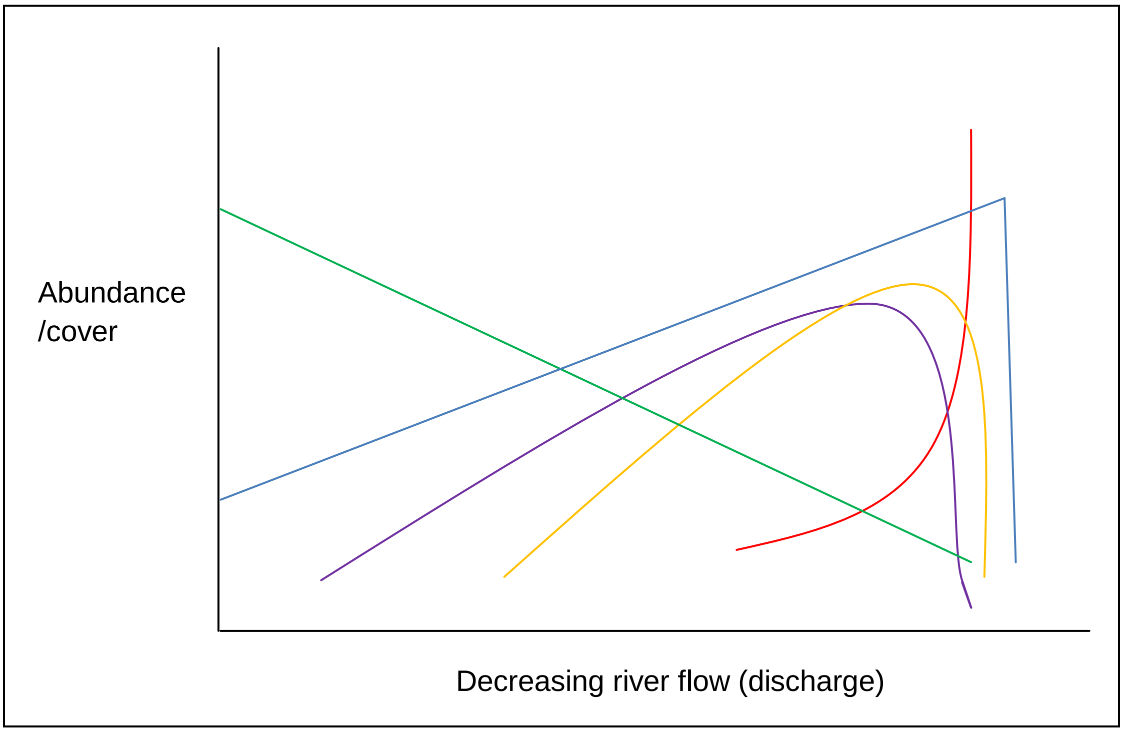


Table I 6 - Effect of periodicity of flow on key macrophyte species in headwaters and winterbournes, based on survey of >120 sites in 1992-95 (Holmes 1996)

	Months dry in summer						
Species	**>6**	**4.5-6**	**3-4.5**	**1.5-3**	**0.5-1.5**	**± Perennial**	**Always perennial**
Non-aquatic grasses	5	5	4	3	1		
Non-aquatic herbs	4	3	1	1	1		
Alopecurus geniculatus	4	5	5	2	1		
Stachys palustris	3	3	1				
Mentha aquatica	3	3	2	1			
Myosotis scorpioides	3	3	2	1			

	Months dry in summer						
Species	**>6**	**4.5-6**	**3-4.5**	**1.5-3**	**0.5-1.5**	**± Perennial**	**Always perennial**
Glyceria fluitans/plicata	1	1	4	5	5	1	1
Apium nodiflorum		1	3	5	5	5	5
Rorippa nasturtium-aquaticum		1	3	5	5	5	5
Rhynchostegium riparioides	2	2	2	2	2	2	2
Fontinalis antipyretica	1	1	1	1	2	2	3
Veronica anagallis-aquatica		1	3	5	5	5	5
Ranunculus peltatus			3	4	4	2	1
Catabrosa aquatica						1	4
Callitriche obtusangula						2	4
Verrucaria spp.						4	5
Hildenbrandia rivularis						3	4
Ranunculus penicillatus. subsp. pseudo.						3	4
Berula erecta						3	4

Key: 5 = expected, 4 = very likely, 3 = typically found, 2 = occasional, 1 = rare on streambed

I.21.2 Temporal scale

March – June is the critical time when flows need to be protected and optimized for aquatic macrophytes (Hatton-Ellis, Greive and Newman, 2003; Acreman et al. 2008).

I.21.3 River type variation

River types based on seven community types in Hatton-Ellis, Greive and Newman (2003).

I.21.4 Ecosystem relations

Increased geomorphological stability due to persistent low or stable river flows are associated with increased macrophyte growth (Holmes et al. 1972; Holmes and Whitton, 1977).

Provide important habitat and food resource for fish and invertebrates. Submerged macrophytes can influence water levels in groundwater fed rivers, maintaining water levels during summer when discharge decreases.

I.21.5 Indicator potential

Flow velocity is thought to be the single most important control in the condition of *Ranunculus* spp. The type and extent of vegetation cover of depositional features in river channels is a key indicator of the level of impact of natural processes in rivers, due to flow modification, according to river type. Non-rooted, free floating macropyhtes and filamentous algae are key indicators of persistent slow/still flows in rivers. Aquatic plants are recognized and recorded in the field, so provide ideal Ecological Indicators within the scope of this report.

I.21.6 Suggested indicators of Poor and Bad status

- Dominance of emergent plants in the channel replacing submerged and floating species in river types CB1, CB2, CB4 and CB6a is a potential indicator of prolonged low flows or very stable low flows downstream of impoundments (4b).
- Dominance of terrestrial plant species in the channel replacing submerged and floating species in all river types is a potential indicator of very extreme and prolonged low flows or very stable extreme low flows downstream of impoundments (4c).
- Dominance of terrestrial plant species in river margins in river types CB3, CB4, CB5 and CB6b is a potential indicator of extreme and prolonged low flows (4d).
- >10% cover of perennial terrestrial vegetation colonizing bars (4e)
- >10% cover of perennial terrestrial vegetation colonizing channel banks (4f)
- Increase in filamentous algae which can smother submerged macrophytes in all river types. Generally, this is temporary in high energy river types, but its growth and negative effects are prolonged in severe or chronic low flow conditions, consistent with Poor and Bad status (4g).
- Increase in abundance of *R. peltatus* relative to *Ranunculus penicillatus* subsp. *psuedofluitans*. This is a potential indicator of extreme low flows and temporary drying of the channel, consistent with Poor and Bad status in river types CB1, CB2, CB4 and CB6a (4h).

- Absence of submerged aquatic macrophytes in river types CB4, CB5 and CB6b, especially in combination with shallow water depth, fine sediment covering the substratum or extensive filamentous algae/diatom growth is a potential indicator of severe abstraction or severe low/stable flows from impoundments (4i).
- Presence of non-rooted, free floating species such as duckweed and floating filamentous algae in the river channel (4j)
- Dominance of rooted species that are usually confined to still backwaters in mainriver channel (4k)

I.22 Riparian vegetation

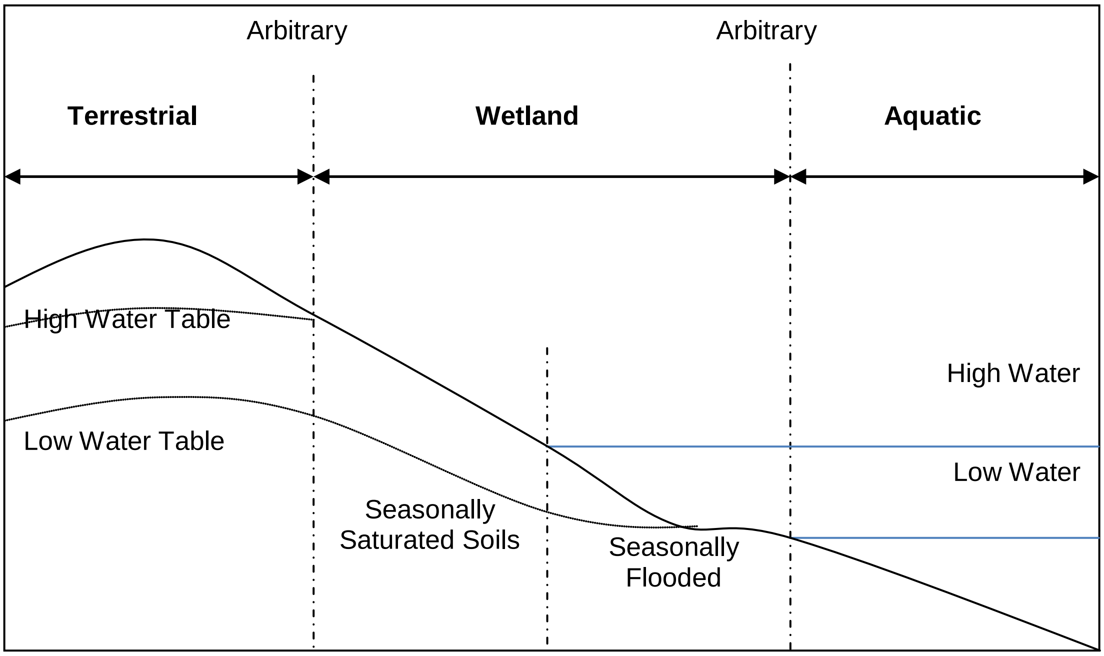
I.22.1 Overview

Figure I 4 shows how natural wetlands lie on a continuum between terrestrial and aquatic systems, where the exact upper and lower limits are often arbitrary. At their terrestrial margin, land remains saturated for less than 30 days per year, a short enough period so that oxygen and other soil conditions do not limit plant growth. At the aquatic margin, wetlands grade into systems which are flooded to a depth and duration where emergent, rooted plants cannot survive. The average water depth which typically separates wetlands from adjacent aquatic ecosystems is 1-2 m.

The most easily recognisable diagnostic feature of wetlands is the presence of hydrophytic (water-loving) vegetation.

Hydrophytes possess anatomical and physiological adaptations that allow them to survive and thrive in saturated or inundated soils, where oxygen depletion is the primary factor limiting vegetation occurrence. In addition to there being species present able to survive these conditions, water level is also considered to be a major determinant of the composition of the vegetation. Which of these are most important in relation to plant distribution remains to be established (Wheeler and Shaw, 1995), though Grime et al. (1988) were able to present a classification of 281 common British plant species according to general affinity with different hydrological states. Newbold and Mountford (1997) also aimed to combine elements of wetland function by presenting known water-level requirements for a range of wetland plants, birds, amphibian and dragonflies.

Figure I 4 - Schematic to show wetland characteristics



One of the challenges in definition and delineation of wetlands is in classifying plant communities. The variety of wetland types is enormous and all wetland classifications must impose subjective boundaries on types, which is further complicated by the successional gradient along which they are found. In the UK, the National Vegetation Classification (NVC), describes the smallest easily recognisable plant community as a unit. These can then be grouped into wetland habitat types by species domination (Rodwell, 1995). In fen and marsh habitat types, wetlands can be classified as, for example, S4 (*Phragmites australis* dominates) or S12 (*Typha latifolia* dominates), or S5 (*Glyceria maxima* dominates). The freshwater and wetland habitat classifications for the UK including the NVC types are included in Table I 7. The NVC has the advantage that it covers a wide range of named communities and their variants, so most vegetation can be fitted in, and it also gives information on a wide range of features, physiognomy, floristics, habitat and zonation (Haslam, 2003).

Table I 7 - Freshwater and wetlands habitats classifications including NVC types

Wetland Habitat Type	NVC Wetland Categories	UK BAP Broad Habitat categories	UK BAP priority habitats	Habitat Directive Habitats
Ponds, Lakes, Canals and Ditches	A1-A16 A19-A24 and a continuum to swamp and mire communities S1-S3 S5-S8, S12-S18, S20, S22-S23	Standing open water and canals	Mesotrophic and Eutrophic standing waters Aquifer fed naturally fluctuating water	22.11 22.12 22.13 22.14 22.31 22.32 22.34 22.44
River Streams and Riparian Habitats	A2 A8-A9 A11-A20 and other riparian habitats including swamps, mires and wet grassland	Rivers and streams	Chalk rivers	24.4
Fens	S1-S3 S5-S21 S23-S24 S25b S26-S28 M4-M14 M21 M22 M24 M27-M38	Fen, marsh and swamp	Fens	53.3 54.12 54.2 54.5
Reedbeds	S4 S25b S26	Fen, marsh and swamp	Reedbeds	
Bogs	M1 M2 M3 M17 M18 M19 M20 M25	Bogs	Lowland raised bog Blanket bog	51.1 51.2 54.6

Wetland Habitat Type	NVC Wetland Categories	UK BAP Broad Habitat categories	UK BAP priority habitats	Habitat Directive Habitats
Wet Heaths	M15 M16	Dwarf shrub	Lowland heathland Upland heathland	21.11 31.12 54.6
Wet Grasslands	MG4-MG6 MG9-MG13 MG7 MG8 M23 M25 S22	Neutral and improved grassland Fen, marsh and swamp	Lowland Meadows Coastal and floodplain grazing marsh	38.2
Wet Woodlands	W1-W7	Broad leaved mixed and yew woodland	Wet Woodlands	44.A1-44.A4 44.3

The UK research experience on water requirements of key wetland vegetation communities (wet grasslands, fens/mires and swamps/ditches) has been brought together as Ecohydrological Guidelines that can be applied directly to impact assessment (Wheeler et al. 2005) an example in Figure I 5 below for MG13 type wet grasslands; water levels in the green zone are "desirable" for this plant community, water levels in the amber are "tolerable for limited periods", whilst water levels in the red are "unacceptable".

Figure I 5 - Water table depth zones for MG13

Seasons and Variable	Tolerable Amber Range	Not Tolerable beyond Red limit
Spring (Mar-May)		
A Mean Water Table Depth (maximum) / m	0.3 - 0.55	0.55
B Mean Water Table Depth (minimum) / m	0.3 - ?	-
C Max duration of surface water flooding episode covering ›10% of ar	-	-
D Cumulative duration of flooding during season/days	-	-
Summer (June-Aug)		
A Mean Water Table Depth (maximum) / m	0.8 - ?	0.55
B Mean Water Table Depth (minimum) / m	0.3 - 0.1	0.1
C Max duration of surface water flooding episode covering ›10% of ar	8 - 20	20
D Cumulative duration of flooding during season/days	30 - 60	60

Figure 1.3 Variables for MG13 Grassland

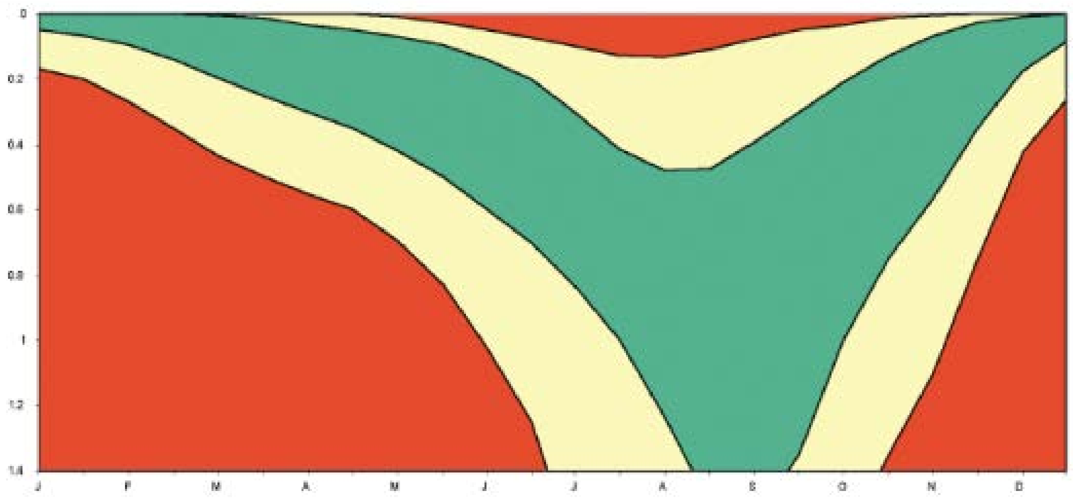


Figure 1.4 Water Table Depth Zones for MG13 Grassland
Water Table zones, illustrating the range of depths that are "desirable" (green) and "tolerable for limited periods" (amber). Values are based on the mean of at least three readings, taken at least 7 days apart, but all within a four-week period.

These guidelines are designed to assist with the ability to find out whether a vegetation community on a site is at risk of moving out of regime in terms of its water needs.

Communities for which guidelines have been produced to date:

Figure I 6 - Communities with guidelines

Box 2: Communities for Which Guidelines Have Been Produced to Date

Wet Grassland

- MG4 *Alopecurus pratensis-Sanguisorba officinalis* grassland
- MG5 *Cynosurus cristatus-Centaurea nigra* grassland
- MG7 *Lolium perenne-Alopecurus pratensis-Festuca pratensis* grassland
- MG8 *Cynosurus cristatus-Caltha palustris* grassland
- MG9 *Holcus lanatus-Deschampsia cespitosa* grassland
- MG13 *Agrostis stolonifera-Alopecurus geniculatus* grassland

Fen and Mire

- M13 *Schoenus nigricans-Juncus subnodulosus* mire
- M24 *Molinia caerulea-Cirsium dissectum* fen meadow
- S2 *Cladium mariscus* swamp
- S24 *Phragmites australis-Peucedanum palustre* swamp
- PPC *Peucedano-Phragmitetum caricetosum* (Wheeler, 1980)

Swamp and Ditch Communities

- S4 *Phragmites australis* reedbed
- S5 *Glyceria maxima* swamp
- A3 *Spirodela polyrhiza-Hydrocharis morsus-ranae* community
- A4 *Hydrocharis morsus-ranae -Stratiotes aloides* community
- A9 *Potamogeton natans*

I.22.2 Ecological Indicator Potential

The following table (Table I 8) gives some examples of the types of ecological variables that can be measured to assess the condition and status of wetland habitats.

Table I 8 - Types of ecological variable for monitoring wetlands

Variable Type	Variable	Examples
Habitat		
Quantity	Wetland Habitat	Total habitat area
Habitat Composition	Communities	Presence/absence Area occupied by NVC communities
	Richness/diversity	Total species richness Diversity indices
	Definitive, keystone, indicator species	Presence/absence Frequency/occurrence Number/density Total cover/percentage cover Total biomass/percentage biomass
Habitat dynamics	Succession/Cyclical change	Total area of scrub Rate of silt accumulation Litter depth % cover of key successional stages

Variable Type	Variable	Examples
Species		
Quantity	Species	Presence/absence Range Number/density Population/size Frequency/occurrence Total cover/percentage cover
Population dynamics	Recruitment	Mean/total number of eggs/births Mean/total number surviving to maturity
	Mortality	Total deaths Percentage of deaths to named causes Number dying before breeding age
	Immigration/Emigration	Mean/total numbers immigrating and emigrating
Population structure	Age and sex ratio	Mean age of population, modal age of population Mean age of breeding
	Fragmentation/Isolation	Distance to nearest population
	Genetic diversity	Rate of colonization Genetic diversity indices

I.22.3 Assessment Techniques

Existing rapid techniques to assess wetlands, particularly in the US, are primarily used as a planning tools to assess the status and functioning of wetlands created specifically to compensate for the loss of other wetland areas or function, known as mitigation wetlands. There is now an array of over 40 assessment techniques which have been developed and regionally modified to perform the same general function of evaluating sites for ecological functioning or for compliance with regulatory permits (Table I 9).

Table I 9 - Summary of core rapid assessment techniques

Name	Summary	Comments	Reference
WET	Method for rapid wetland functional assessment	USA rapid assessment method. Mainly concerned with diversity, groundwater and sediment. All given 'value ratings'	(Adamus, 1988)
FAEWE	Functional Analysis of European Wetland Systems	Focuses on river marginal wetlands and predicting hydrogeomorphic units. A classification system.	(Maltby *et al.*, 1994)
HGM	Hydrogeomorphic Units	Sites classified by location, source of water, hydrodynamics	(Brinson, 1993)
WRAP	Wetland Rapid Assessment Procedure	Developed by South Florida water management. Rely on structural indicators to imply function.	Miller and Gunsalus (1997 and 1999) in (Cohen *et al.*, 2005)
WEA	Wetland Ecological Assessment	As WRAP but with specific changes. Rapid tool but can also be used over time.	(Breaux *et al.*, 2005)
RCA	Rapid Condition Assessment of wet grassland	Environment Agency work in progress. Draft has not been tested for repeatability.	

These include quick approaches which could be adopted e.g. assessment based on ratings and 4-point scales (rapid but subjective).

I.22.4 Suggested field indicators of Poor and Bad status

Ideally would need site assessment both after prolonged rainfall and after a prolonged dry spell when vegetation patterns may indicate areas where the wetland is reliant on groundwater during the dry season or droughts.

- Loss of more aquatic Sphagna and perhaps transition to a different NVC community (e.g. M4 to M6) (6a)
- Loss of wetland species and increased represenatation of more terrestrial species. (6b)
- Depth and extent of water in the wetland during wet months (6c)

For long-term monitoring:

- Establishment of a reference quadrat in each distinct ecological feature (NVC survey)
- Measurement of elements of the water budget (Rainfall, Dipwells, Piezometers, surface water).

I.23 Aquatic macroinvertebrates

I.23.1 Overview

Aquatic macroinvertebrates are routinely used as indicator organisms for determining the biological quality of watercourses and diagnosing environmental pressures. A range of biotic indices can be derived from semi-quantitative samples of macroinvertebrate communities, with each one characterising different stressors affecting the ecology of freshwaters. The Lotic-invertebrate Index for Flow Evaluation (LIFE) (Extence et al. 1999) is accepted as the most useful tool for assessing the effects of drought and abstraction. However, LIFE scores cannot be interpreted in isolation. It is important to interpret water quality indices as well, because pollution impacts can confound the interpretation of LIFE scores as an index of water velocity. The water quality indices used are the Biological Monitoring Working Party (BMWP) index, the Average Score per Taxon (ASPT) and the number of taxa (NoT), which provide a standard measure of biological quality and indicate background levels of organic pollution. A new index, Proportion of Sediment-sensitive Invertebrates (Extence et al. 2011), follows the same principles as LIFE but is calibrated to detect the ecological impacts of excessive fine sediment deposition in rivers.

Currently only ASPT and NoT are used by UKTAG in the classification of surface waters using the macroinvertebrate quality element because the reference condition model (River Invertebrate Classification Tool; RICT) is currently unable to predict reference LIFE scores accurately to generate an EQR. LIFE is used as a diagnostic index for hydromorphological pressure in England and Wales, but is not currently used in Scotland.

The response of LIFE to flow variation depends on the degree of modification of the river channel. LIFE scores are more responsive to flow change in artificially modified channels than natural channels; reflecting the fact that macroinvertebrate communities are perhaps more resistant to altered flows in more heterogeneous habitats, which provide more refuges from high and low flows (Dunbar et al . 2010 a,b).

LIFE describes the mean current velocity requirements for British aquatic macroinvertebrate species and families, which are categorized into Flow Groups (in this case flow = velocity) (Extence et al. 1999).

Table I 10 - LIFE flow groups for British aquatic macroinvertebrates

Group	Ecological flow association	Mean current velocity
I	Taxa primarily associated with rapid flows	Typically > 100 cm s^{-1}
II	Taxa primarily associated with moderate to fast flows	Typically 20 - 100 cm s^{-1}
III	Taxa primarily associated with slow or sluggish flows	Typically < 20 cm s^{-1}
IV	Taxa primarily associated with flowing (usually slow) and standing waters	
V	Taxa primarily associated with standing waters	
VI	Taxa frequently associated with drying or drought-impacted sites	

Presence of taxa from Flow Groups V and VI in the river channel could indicate the severe effects of abstraction of low flows from impoundments, consistent with Poor or Bad status, although LIFE does not define quality status, and this must be obtained by consultation with experts.

The Environment Agency used class intervals of LIFE O/E to describe low flow impact based on Clarke et al. (2003) during the Environment Agency's Resource Assessment and Management framework 3 (RAM). Expected reference condition values of LIFE were predicted by RIVPACS III+ and compared to observed LIFE at each site. Indicative ecological status class intervals can be applied to LIFE O/E which might be able to define the severe effects of flow alteration, consistent with Poor and Bad status. This system has not been formally adopted by the Environment Agency or UKTAG, and its application across a wide range of rivers remains to be tested, especially for Scottish rivers. However, studies in the West Midlands on groundwater fed headwaters have indicated that the application of this system has resulted in significant relationships between LIFE O/E and the impact of abstraction at low flows (Q_n75) (Bradley et al. in prep; APEM & ESI, 2010). These are the first studies to measure the effect of abstraction impact on macroinvertebrates and to provide a test using real data of the current UKTAG flow standards, as described in WFD 48. Whilst there remains debate about whether RIVPACS III+ provides accurate enough expected values for LIFE under reference conditions, these data suggest that this system is likely to be sensitive enough to distinguish sites that are severely impacted by abstraction or low flows from impoundments, consistent with Poor and Bad status. However this is a point for expert consultation and discussion.

Table I 11 - LIFE O/E ratios and the indicative classification of impacts from low flows and indicative ecological status class intervals

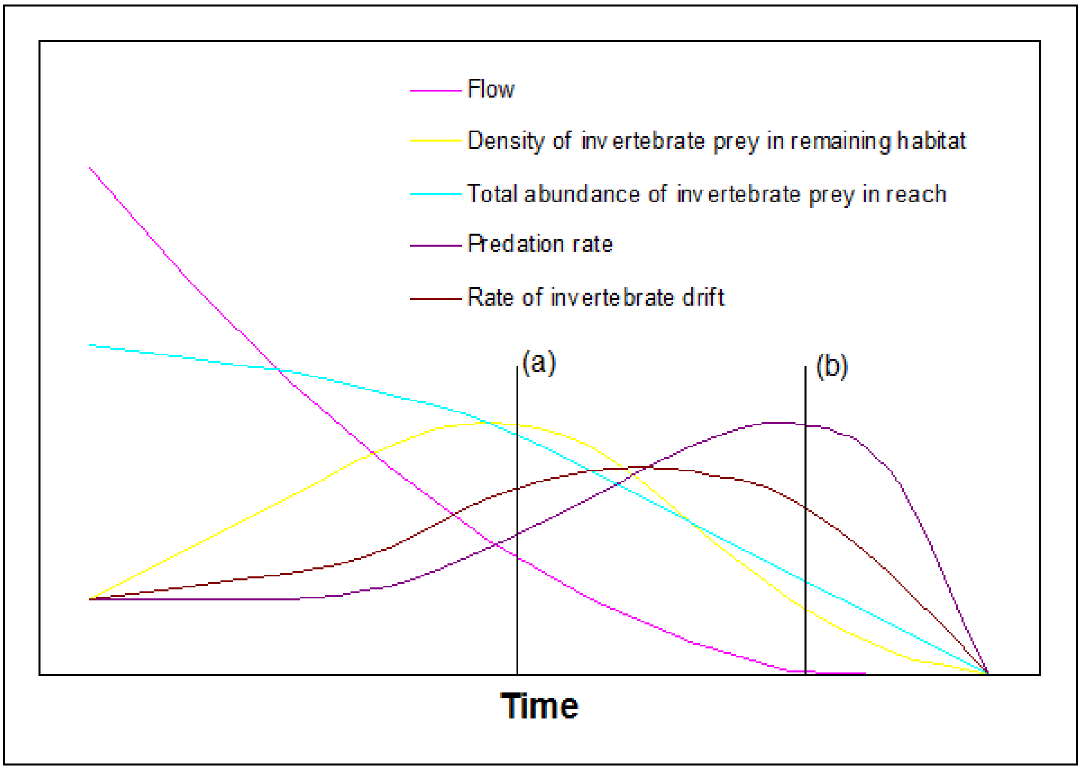
LIFE O/E ratios	Flow	LIFE O/E ratios	Indicative Ecological Status
>0.974	Unimpacted	>0.974	High
0.925-0.974	Mildly impacted	0.945-0.974	Good
0.875-0.924	Moderately impacted	0.915-0.944	Moderate
<0.874	Severely impacted	0.885-0.914	Poor
		<0.885	Bad

LIFE and PSI are calibrated to individual species preferences for fast velocities/clean gravel/cobble substrata to slow/still velocities/silty substrata. The strengths of these metrics are in their sensitivity to the habitat effects of water abstraction and flow regulation. However, the sensitivity of these indices to hydromorphological impacts in rivers depends on the method used to collect the samples from which the indoces are calculated.

Standard sampling methods for macroinvertebrates to support routine biological quality assessments in the UK are not amenable to evaluating reductions in habitat size (space) across river reaches (Armitage and Pardo 1995; Armitage and Cannan 1998; Mainstone, 2010). The standard three-minute kick/sweep sampling method that integrates all instream meso-habitats was designed to factor out the effect of habitat size for the purpose of water quality assessment. Reductions in habitat size can generate large reductions in the total abundance of species, whilst often concentrating individuals into the remaining space (e.g. Extence, 1981; Wright and Berrie 1987; Suren and Jowett 2006; other references are given in Table I 12). Mainstone (2010) suggested that over time, however, this makes populations more subject to density-dependent mortality and movement (in drift) associated with intra- and inter-species competition and predation (McIntosh et al. 2002, Peckarsky et al. 1990). Depending on the timing of observation, therefore, either an increase or decrease in apparent relative abundance of predators and prey, and all invertebrates may be observed (Figure I 7). This model might at least partially explain the diverging findings of various studies of flow depletion summarised by Dewson et al. (2007) in Table I 9 and the low certainty in the sensitivity of existing biological classification and assessment tools to hydromorphological pressures.

Mainstone's (2010) conceptual model illustrates the importance of biotic interactions in determining the observed ecological response to reduced flow and shrinking habitat size in rivers – especially when flows are severely low at time b in the model. The effects of both habitat size and biotic interactions are not incorporated in the standard biotic indices that are calibrated on organism's responses to the abiotic environment. This model enables the identification of ecological indicators of the severe effects of reduced flows due to abstraction or impoundment of water in rivers – corresponding to time b in the model (Figure I 7).

Figure I 7 - Conceptualised macroinvertebrate responses to summer flow recession during drought. Routine observation at time (a) – high invertebrate prey density, low predation rate; routine observation at time (b) - low prey density, high predation rate (reproduced from Mainstone, 2010)



Aquatic macroinvertebrates living in different river habitats might be more sensitive to flow change than others. Macroinvertebrates (and other organisms) specialising in marginal wet habitat can be particularly affected by loss of habitat space (Ormerod, et al. 1987; Wright 1992; Rose et al. 2008). Rose et al. (2008) found that, when standard benthic macroinvertebrate biotic scores were generated separately for riffle and edge habitats in Australian streams, riffle scores were far less affected by drought than edge scores. This was because high flow velocity requiring taxa were out-competed in edge habitats by lentic taxa. In the UK, marginal aquatic habitats are included in integrated 'sweep' samples of the macroinvertebraqte community during routine monitoring, but the values of biotic indices will be sustained during low flows flow by rheophilic taxa surviving in remaining riffle habitat.

Boulton (2003) summarises these effects in a simple conceptual model of the stepped changes that occur as water levels decrease in rivers; describing the impacts on macroinvertebrate communities when water levels drop below successive instream habitats (Figure I 8). This conceptual model supports the suggestion that macroinvertebrate communities in river margins are the first to be affected by reduced river flows and that riffle communities might only be impacted after Stage 2, despite experiencing reduced habitat space before being impacted.

Figure I 8 - Conceptual model of 'stepped' changes in macroinvertebrate assemblage composition in response to declining water levels in a river (reproduced from Boulton, 2003)

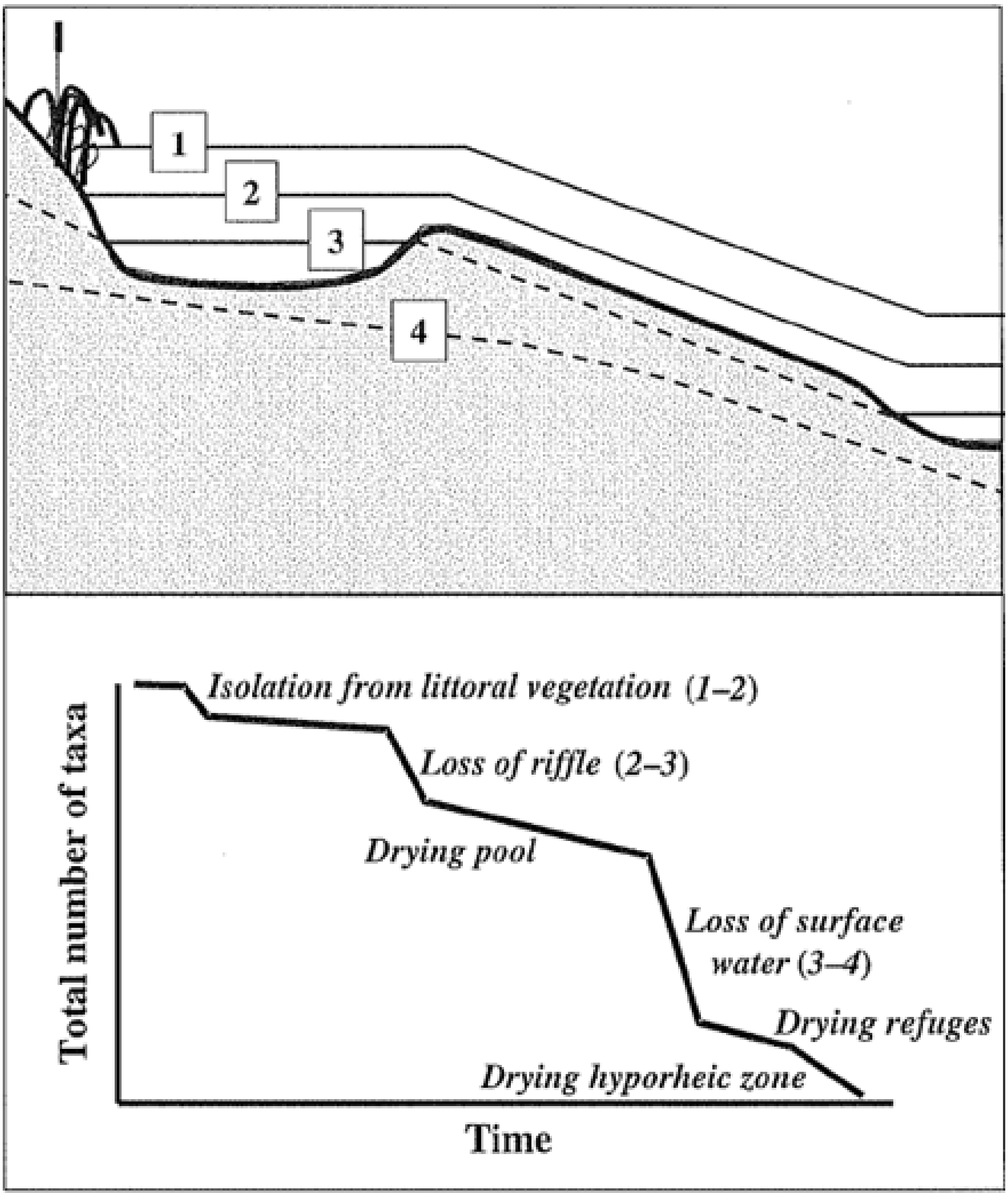


Table I 12 - Summary of effects reported for decreased stream flow on invertebrate communities (from Dewson et al. 2007). For full citations see Dewson et al. (2007)

Variable	Increase	No Change	Decrease
Density	Gore 1977	Cortes et al. 2002	Cowx et al. 1984
	Extence 1981	Suren et al. 2003a	Hooper and Ottey 1988
	Wright and Berrie 1987		Wood and Petts 1994
	Rader and Belish 1999		Englund and Malmqvist 1996
	Wright and Symes 1999		Malmqvist and Englund 1996
	Dewsen et al. 2003		Cazaubon and Giudicelli 1999
	Suren et al. 2003a		Rader and Belish 1999
			Wood and Petts 1999
			Wood et al. 2000
			McIntosh et al. 2002
			Wood and Armitage 2004
Taxon richness		Armitage and Petts 1992	Englund and Malmqvist 1996
		Cortes et al. 2002	Rader and Belish 1999
		Dewsen et al. 2003	Wright and Symes 1999
			Cazaubon and Giudicelli 1999
			Wood and Armitage 1999
			Wood et al. 2000
			McIntosh et al. 2002
			Wood and Armitage 2004

Studies by Wessex Water Services Ltd. in the upper reaches of the River Avon (Wiltshire) have shown that there are characteristic macroinvertebrate communities associated with different periodicities of drying of the river channel in winterbourne reaches (Punchard and House, 2009) . These communities can be used to predict the severe effects of abstraction in the upper reaches of chalk rivers, when abstraction will extend the length of the winterbourne downstream. The presence of the species listed in Table I 13 in previously perennial headwater reaches can indicate the severe effects of abstraction. Conversely, the sudden disappearance of other mayfly species from headwater streams in the absence of major changes in water quality or mechanical habitat disturbance can indicate the effects of temporary drying of the channel, consistent with the severe effects of abstraction (D. Bradley Pers. Obs).

Table I 13 - Invertebrate 'winterbourne specialists' in the Wiltshire Avon catchment 2002-2007 (Punchard and House, 2009)

Planariidae	Helophoridae
Phagocata vitta	*Helophorus aequalis*
Lymnaeidae	*Helophorus grandis*
*Galba truncatula**	*Helophorus brevipalpis**
*Stagnicola palustris**	Hydrophilidae
Planorbidae	*Hydrobius fuscipes*
*Anisus leucostoma**	*Anacaena limbata**
Sphaeriidae	*Anacaena lutescens*
*Pisidium personatum**	Limnephilidae
Niphargidae	*Glyphotaelius pellucidus**
*Niphargus aquilex**	*Limnephilus auricula*
Leptophlebiidae	*Limnephilus bipunctatus*
Paraleptophlebia werneri	*Limnephilus centralis*
Nemouridae	*Limnephilus vittatus*
Nemoura cinerea	Simuliidae
Perlodidae	*Metacnephia amphora*
*Isoperla grammatica**	*Simulium latipes***
Dytiscidae	*Simulium vernum gp.***
Hydroporus discretus	*Simulium aureum gp.***
Hydroporus marginatus	*Simulium ornatum gp.***
Hydroporus nigrita	
Agabus biguttatus	* Occasionally found at perennial sites but more common in winterbournes
Agabus didymus	
Agabus guttatus	
Colymbetes fuscus	**Very common in winterbourne samples but may also be at perennial sites

Armitage (2006) reported the long-term (32 years) effects of river flow regulation in the upper River Tees on macroinvertebrates immediately downstream of Cow Green Reservoir. This study indicated that stable river flows and physical habitat downstream of the reservoir has resulted in a dynamically fragile macroinvertebrate community that is very susceptible to physical perturbations because it has developed in its absence. This also indicates that biotic interactions (predation, competition) may be exerting a dominant influence on community structure and dynamics. Increased periphyton and aquatic bryophyte growth on the stable substratum is thought to create more microhabitats and food for macroinvertebrates, further promoting biotic interactions and opportunities for colonization by a wide range of organisms (presumably increased risk of invasion by competitively dominant and alien species). *Gammarus pulex* is one such species that has become increasing dominant downstream of Cow Green dam and this is consistent with other studies that have reported increases in amphipods at regulated sites (Armitage, 2006).

Baetid mayflies (olive mayflies) are ubiquitous in rivers throughout the UK, are among the most abundant invertebrates in river where the habitat and chemical conditions are suitable and are important food for juvenile and adult salmonid fish. Baetid mayflies require emergent rocks in the river channel (especially at riffles) for oviposition. Lancaster, Downes and Arnold (2010) showed that egg supply might be a limiting factor to baetid populations in rivers. They reported that egg supply was positively related to the density of emergent rocks in the river channel over 30 m reaches. A further study indicated that egg supply controls the local density of baetid larvae and that baetid larvae do not disperse very far from the natal habitat (Lancaster, Downes and Arnold, 2011). This research has major implications for the management of regulated river flows from impoundments. Controlled flows that are set too high and cause in channel substrata to be submerged at the crucial time for mayfly oviposition (March – June) could have major impacts on local baetid populations, especially given in stream nymphal dispersal might be limited or blocked by an impoundment upstream. Reduced abundance of baetid mayflies can have major knock-on effects to salmonid fish by reducing the food supply for juvenile fish.

I.23.2 Spatial scale

Many invertebrates have aerial dispersal phases and can quickly colonise new sites that offer suitable conditions, such as after an environmental disturbance has ceased or if the flow regime is made more suitable. Limitations to this will occur in isolated river channels, such as upland headwaters separated by high ground or long distances. Invertebrates can disperse in the water downstream by drifting, swimming and crawling. Recent studies have suggested that some invertebrate groups that were thought to be quintessential in-stream dispersers, baetid mayflies, probably disperse far short distances in rivers than previously thought, most not drifting beyond the natal riffle (Lancaster, Downes and Arnold, 2011). The implications for this work are that the severe effects of water resource pressure might impact ecological communities over longer distances than expected and the ecological benefits of an optimized river flow regime might extend over longer distance than expected.

I.23.3 Temporal scale

Macroinvertebrates are often described as not resistant to environment disturbance, but resilient. In other words, they generally recover quickly from impacts. The assessment of whether the effects of abstraction or regulated flows are ‘severe’ and if the ecological status is Poor or Bad must be made in relation to whether the resilience of the communities has been damaged, and not just reporting short-term severe impacts and low status during droughts or low flow periods.

The expert workshop suggested that impacts on aquatic macroinvertebrates arising from water resource pressure should only be considered ‘severe’ and consistent with Poor and Bad status if they are detected outside of short-term natural low flow periods.

I.23.4 Temperature

Changes to water temperature as a result of flow regulation downstream of impoundments can affect the growth of larval insects in rivers (Webb and Walling, 1993).

I.23.5 Ecosystem relations

Reduced macroinvertebrate abundance and diversity can have knock-on consequences to fish (especially juvenile salmonid), river birds and bats by reducing their food supply.

I.23.6 Ecological indicator potential

Very good and well established.

I.23.7 Suggested indicators of Poor and Bad status

- LIFE O/E >0.914 using RIVPACS III+ or IV (RICT) and family LIFE in all rivers except chalk rivers where it might have to be adjusted upwards (3b)
- Increase in the abundance of large bodied predatory invertebrates, such as beetles and odonata in all river types (3c)
- Increase in the abundance of LIFE Flow Group V and VI species (3d)
- Absence of LIFE Flow Groups I-III species in rivers with no water quality impacts (3e)
- Presence of species described as winterbourne specialists in normally permanently flowing reaches near abstractions or downstream of impoundments in chalk streams (3f)
- Absence of baetid mayflies in stony/gravelly rivers with no water quality impacts (3g)
- Dominance or monopoly of *Gammarus* spp. downstream of impoundments (3h)

I.24 Invertebrates of exposed riverine sediments

I.24.1 Overview

Exposed riverine sediments (ERS) support unique communities of specialized invertebrates (particularly beetles and spiders) that live in the terrestrial-aquatic interface and depend on the disturbance regime of seasonal water level changes.

A large number of rare and scarce invertebrates are associated with ERS (Eyre and Lott, 1997, Sadler and Bell, 2002). In the UK, there are 131 specialist ERS beetles, 86 (66%) have either Red Data Book, or Nationally Scarce status (Bates et al., 2005). Disturbance through inundation and flooding is a key factor in creating, maintaining and redistributing sediments. Patches of ERS with greater habitat heterogeneity support more species rich assemblages and contain larger numbers of rare and specialist invertebrates (Sadler et al. 2004). Stable river flows due to river regulation of prolonged low flows due to abstraction do not provide the required disturbance regime needed for ERS communities. At low levels of disturbance or greater intervals between disturbances, more competitive organisms (generalists) will optimise favorable conditions and dominate habitats (Dial & Roughgarden, 1988). This is indeed the case with less disturbed ERS sites, habitats become degraded (i.e sediment becomes compacted and be stabilised by vegetation) the invertebrate fauna becomes less specialised and dominated by more generalists (Sadler and Bell, 2000). Stablilisation of ERS by lack of high flows and vegetation is a major threat to these communities (Henshall, 2011).

Plachter & Reich (1998) suggested that floods may actually favour some species by providing an influx of food (via drift), and reduce competition by removing generalist species. However, if the flooding disturbance is too high (prolonged inundation or scouring of habitats) it can have negative effects on ERS communities, most probably though direct mortality (Hering et al. 2004; Henshall et al. 2011).

The timing of flood events is critical to ERS communities which have attuned their lifecycles to cope with living in highly disturbed habitats. The vulnerable egg and larvae stages are not present during the typical high flow periods (Andersen, 1969; 1983a Manderbach and Platcher, 1997). Their lifecycles are also characterised by extreme outliers that allow continued survival when a large flood disturbance event occurs (Platcher and Reich 1998, cf Stelter et al. 1997). Winter floods are less damaging because many species overwinter away from the water's edge and individuals are in the adult stage so can elicit avoidance behavior, such as adult Bembidion that can burrow in the sediment and persist during floods (Andersen, 1968). Adult ERS invertebrates will also move ahead of the rising flood water levels to avoid drowning (Anderson, 1968). Summer floods can be damaging to ERS invertebrates when eggs and larvae are present, especially if they are prolonged or frequent in quick succession.

It is expected that the rate of change of water levels during floods is critical to allowing avoidance behaviours to be initiated and preventing direct mortality of ERS communities, particularly during the summer, when flows are normally low and vulnerable lifestages are present. Sudden flash floods might not allow enough time for avoidance cues to be picked up. This has major implications for the design of optimized freshet flows from impoundments.

Summary of abstraction effects:

- Prolonged and stable low flows will cause habitats to dry out and competitors to colonise.

Summary of river regulation effects:

- Lack of high flows will cause habitat stabilization and reduce ERS abundance and diversity, particularly of the rare species.
- Sudden high flows due to hydropeaking or freshet release can cause direct mortality if river levels rise too quickly and destroy ERS by scouring.
- ERS invertebrates are adapted to resist the effects of winter flooding.
- ERS invertebrates are more sensitive to the impacts of summer flooding.

I.24.2 Temporal scale

- ERS invertebrates depend on regular inundation to maintain the habitat in a intermediately disturbed state.
- However, too much flood disturbance, particularly in summer can be damaging.
- ERS invertebrates are adapted to resist the effects of winter flooding.
- ERS invertebrates are adapted with avoidance behavior for natural flood events. If the rate of change of water levels of too high, they might be able to undertake avoidance behavior.

I.24.3 River type variation

Cobble/gravel-bedded rivers with depositing features.

I.24.4 Ecosystem relations

Reduced macroinvertebrate abundance and diversity can have knock-on consequences to fish (especially juvenile salmonid), river birds and bats by reducing their food supply.

I.24.5 Ecological indicator potential

Not direct indicators.

Indicators of the stability of ERS habitat as described by the geomorphological indicators: stable channel banks and stable channel substratum which will be surrogate indicators for ERS invertebrates.

I.25 Diatoms

I.25.1 Overview

Diatoms (Bacillariophyceae) are microscopic algae that are abundant within the periphyton (algae that grow on surfaces) in rivers and streams, and contribute to the phytoplankton in larger rivers. Diatoms have rapid reproductive rates (on the order of hours to days) and are known to respond to water quality changes within the aquatic environment over a timescale of 3-4 weeks. Diatoms are sensitive to a variety of environmental factors including nutrients, pH, temperature, light and flow (e.g. Dixit et al. 1992). Diatom-environment models exist for nutrients and pH, in both lakes and rivers, but no model yet exists for diatom-flow relationships. While the importance of current as a dominant physical factor influencing algal population and community structure has long been recognised in lotic systems (Whitton, 1975 in McCormick and Stevenson, 1991), the fact that diatoms show complex responses to current changes (e.g. Stevenson and Peterson, 1989 in McCormick and Stevenson, 1991) is likely a contributing factor to the lack of an existing index for current velocity.

All diatoms present within rivers and streams utilise the same nutrients and are subject to flow disturbance and grazing, but species vary in their adaptations to these disturbances and resource pressures. These differences between species are ultimately displayed along various temporal and spatial gradients (Passy, 2007). Periphytic communities undergo both taxonomic and structural changes during succession. Pioneer biofilms, frequently composed of one or two diatom species (and bacteria) develop in thickness and complexity, to include larger, stalked diatoms, as well as loosely attached diatoms (e.g. *Melosira* sp.) that are more caught in the organic matrix than attached to the substratum *per se*, and motile species that actively move through the matrix.

Well-developed biofilms that have not been disturbed (i.e. via flows high enough to scour) can occur that are visible to the naked eye, these are likely to contain high numbers of loosely attached taxa (e.g. *Melosira*). In biofilms that become exposed, aerophilous taxa (e.g. *Diadesmis* or *Luticola*) frequently increase (Kelly et al. 1998). High current speeds have been linked with changes in growth rate and relative abundance and a decrease in diversity (Antoine and Benson-Evans, 1982; Wendker, 1992; Lindstrøm and Traaen, 1984; Rolland et al. 1997 in Kelly et al. 1998).

Working in the USA, Passy (2007) found that three diatom ecological guilds could be distinguished on their potential to tolerate nutrient limitation and physical disturbance. A low-profile guild was favoured in low-nutrient high disturbance environments, a high-profile guild was favoured in nutrient-rich, low flow disturbance environments while the motile guild increased along the nutrient gradient but decreased along the disturbance gradient. Guild distribution was also habitat specific, the low-profile guild dominated the episammon (sand), the high-profile guild was more common on epilithon and epiphyton (rock surfaces and attached to plants respectively) while the motile guild occurred more frequently on the epipelon (surfaces of the deposit such as mud or sand. Growns and Growns (2001) studied the effects of flow regulation on macroinvertebrates and periphytic diatoms in the Hawksbury-Nepean River system in Australia, and found differences in the periphytic diatoms between regulated and unregulated sites.

I.25.2 Spatial scale

A range of processes that can be placed into two broad types operates to generate diversity within and between periphyton communities in streams. Some processes influence the whole reach (e.g. nutrient availability, pH and hydrology) and finer scale processes applicable at substratum size (Yallop and Kelly, 2005). These variable processes mean that several potential trajectories for biofilm development are possible from the pioneer state at any one site, depending upon the microhabitat. Indeed, where there is a mix of substratum sizes (with a corresponding mix of sensitivity to disturbance) the heterogenic stream reach may have several stages of 'the trajectory' co-existing. The consequence here would be a very patchy and diverse flora, the description of which (and any interpretation against environmental condition) would accordingly be highly sensitive to the sampling strategy used. Peterson (1987) found that diatom communities from more sheltered habitats were less resistant to desiccation stress than communities that had developed in more rigorous flow conditions. This was taken to indicate that resistance to disturbance within the periphyton varies as a function of localised flow regime.

I.25.3 Temporal scale

Diatom communities have rapid immigration rates, and most species have rapid growth rates as well. It is likely that diatom communities would respond rapidly to changes in flow (or flow-induced habitat changes).

I.25.4 Temperature

Diatoms are autotrophic, and their rate of photosynthesis is temperature dependent to the degree that they have slower responses during the winter months. Some individual species are adapted to cooler temperatures, and often dominate spring samples (e.g. *Navicula lanceolata*). No information could be found on the response of biofilm development and temperature.

I.25.5 River type variation

Diatoms are ubiquitous in the aquatic environment, being found wherever there is sufficient water, light and nutrients. Some species are widely distributed and found globally in many different environment, other species are specialised, and have narrow optima and tolerances for certain environmental parameters. Very little information is available on biofilm composition by river type.

I.25.6 Ecosystem relations

Diatoms are typically a welcome component of an ecosystem, being key primary producers underpinning many other trophic levels. One freshwater species, *Didymosphenia geminata*, is regarded as an invasive species. It can develop into thick mats within rivers and streams, and can potentially decrease habitat quality (changes to water quality). In dammed rivers, intentional water release of sufficient magnitude, frequency and duration can be used as a management control (Larned et al. 2007).

I.25.7 Ecological indicator potential

Flow regime will have an influence on the composition and structure of diatom periphytic communities, and there is the potential for a diatom-based index to be developed. Given that diatoms are sensitive to a wide-range of environmental factors, that studies report complex responses of communities to changes in flow, combined with the issues highlighted in the spatial scale box above it is considered that realisation of a usable tool is unlikely to occur in the near future.

It is considered that none of the potential diatom indicators would provide sufficient evidence on their own, but taken in conjunction with other indicators of low flow could provide supporting evidence of impact.

I.25.8 Suggested field indicators of Poor and Bad status

- Where low flow is considered to be an issue (e.g. where low water is apparent, or mossy cobbles/boulders exist) there is the potential to assess diatom community structure for the presence of aerophilic taxa such as Luticola and Diadesmis. Aerophilic genera are likely to increase in relative abundance if surfaces have been exposed for a prolonged period of time (4l).

- The occurrence of long filamentous biofilms visible to the naked eye offers is another potential indicator of low flow or stable flow conditions. These filamentous mats may also occur in slow-flowing waters, so their presence in an unusual habitat (e.g. where faster water would be expected) could be used as a visual indicator that low flow conditions prevail (4m).

- Where low flow results in increased deposition of fine sediment an increase in motile diatom species is likely to result. Proportion of motile taxa is potential indicator. (NB: Lack of motile taxa may indicate lack of fine sediment or early stage biofilm development) (4n).

I.26 Amphibians

I.26.1 Overview

British amphibians (frogs, toads and newts) prefer to breed in ponds but, frogs and toads will breed in very slow flowing habitats, including river margins.

Summary of abstraction effects:

- Severe low flows and ponding in early spring will create suitable habitat for frogs and toads to breed.
- Chronic and severe effects of abstraction, especially in river headwaters might maintain still or slow moving water to allow tadpoles to survive and grow to adults during the spring and early summer.
- Ponding might also create suitable habitat for newts.

Summary of river regulation effects:

- Lack of compensation flows from impoundments can lead to temporary ponds in downstream reaches which are ideal habitats for amphibians to breed (D. Bradley, Pers. Obs).

I.26.2 Temporal scale issues

Amphibians require constant still of very slow moving water from early spring to early summer for successful breeding.

I.26.3 Temperature

Higher water temperatures increase the rate of growth of tadpoles.

I.26.4 River Type variation

Any river type with chronic ponding in channel.

I.26.5 Ecological indicator potential

The presence of frog or toad tadpoles in ponded or very slow reaches, especially in late spring – summer, can indicate the severe and chronic effects of abstraction and severe low flows from impoundments. Newts are even less prone to inhabiting flowing water and the presence of adult newts in still or ponded reaches can indicate chronic conditions related to the severe effects of abstraction and/or impoundment of water.

I.26.6 Suggested field indicators of Poor and Bad status

- Presence of frog or toad tadpoles in river channel, especially in late spring – summer indicates chronic and severe low flows from abstraction and/or impoundment of water (5a).
- Presence of newts in river channels indicates long-term still water conditions due to the severe effects of abstraction and/or impoundment of water (5b)

I.27 Bryopyhtes (mosses and liverworts)

I.27.1 Overview

Bryophytes are commonly associated with wet habitats – including truly aquatic species that live submerged below water and terrestrial species that are associated with riparian habitats. Although much is known about the environmental factors influencing the distribution of aquatic and riparian bryophytes worldwide, relatively few such studies have been conducted in the UK (Scarlett and O'Hare, 2006; Lang and Murphy, 2012).

Disturbance, either as substratum movement or water level fluctuation is a major factor determining the species richness and standing crop of bryophytes in rivers (Muotka and Virtanen (1995).

Substratum stability is a major factor determining the taxon richness and standing crop of bryophytes in rivers. Species richness is typically highest in streams with intermediate levels of substratum stability and discharge (Suren and Duncan, 1999). Interspecific competition is thought to reduce species richness at the most stable sites and physical disturbance is thought to reduce species richness at the most unstable sites (Suren and Duncan, 1999). Standing crop of bryophytes, however, is often highest at the most stable sites. The most stable sites are often characterized by high abundance of a single weft-forming competitive species, such as *Fontinalis antipyretica* in Scottish streams (Lang and Murphy, 2012), *Fontinalis* spp. in Finnish streams (Muotka and Virtanen, 1995) and thalloid liverworts in New Zealand streams (Suren and Duncan, 1999). Regulation of river flow from impoundments often creates stable water and substratum conditions. Increase in the abundance of aquatic mosses (and periphytic algae) has been reported in the River Tees a short distance downstream of Cow Green Reservoir, since impoundment (Armitage, 2006).

Water level fluctuation is important in determining the vertical zonation of bryophytes on river channel substratum. On exposed substratum, species composition of bryophytes shifts from obligate aquatics to facultative aquatic and semi-aquatic species along a gradient of permanently submerged to continuously exposed conditions (Muotka and Virtanen, 1995; Virtanen, Muotka and Saksa, 2001). Species richness is typically highest at or just above the water line (Muotka and Virtanen, 1995).

Observations of rivers in the UK have suggested that highly stable flows downstream of impoundments often result in increased bryophyte growth on the substratum (Holmes et al. 1972; Holmes and Whitton, 1977; Armitage, 2006). In these conditions, bryophytes are often noticeably abundant on the exposed surface of substratum; especially on smaller substrata, such as cobbles and small boulders that might be easily moved by small freshets or floods (D. Bradley, Pers. Obs.). These observations have also indicated distinct zonation on exposed substratum above and below the water line.

The evidence base suggests that aquatic and terrestrial bryophytes are ideal candidate for field based ecological indicators of severe impacts from modified flow regimes.

I.27.2 Temporal scale

Examination of the species of bryophytes both above and below the water level might provide a refined indicator of the flow history of the site and a useful diagnostic of severe river flow alterations.

I.27.3 Spatial scale

The size of exposed substratum covered by terrestrial bryophytes might give an indication of the stablility of low flows. Exposed pebbles, cobbles and small boulders covered by bryophytes suggests they are highly stable indicating chronic low flows.

I.27.4 Temperature

N/A

I.27.5 River type variation

Stony rivers.

I.27.6 Indicator potential: Good

Moss cover indicates stability of channel substratum, due to chronic stable flows and lack of high flow events to mobilise substratum. Of particular importance is smaller exposed substratum (cobbles, pebbles and small boulders) covered by moss as these are most easily moved by high flows.

I.27.7 Suggested field indicators of Poor and Bad status

- Exposed cobbles, pebbles and small boulders in river channels covered by mosses and/or liverworts indicates chronically stable flows and the severe effects of impoundment (4a).

Figure I 9 - Examples of moss-covered river substratum in a stable, low compensation flow regime (River Sett, Derbyshire – downstream of Kinder Reservoir. D. Bradley)

II APPENDIX II PROCESS DIAGRAMS AND SUMMARY OF MAIN RISKS TO ECOLOGICAL ELEMENTS POSED BY PRINCIPAL TYPES OF MODIFIED FLOWS

II.1 Abbreviated conceptual models

II.1.1 Overview

The abbreviated conceptual models are intended to illustrate the derivation of the ecological indicators and facilitate the application of the water release optimisation framework. They comprise;

- process diagrams that link flow changes to habitat state, and through this, to biotic impacts; and
- impact tables that summarise the nature and timing of risks to selected ecological elements.

II.1.2 Process diagrams

The process diagrams and impact tables illustrate the derivation of the ecological indicators and facilitate the application of the water release optimisation framework, but are intended as adjuncts to, not replacements for, the conceptual model text, as they lack the latter's treatment of scale and complexity.

The process diagrams describe all the effects of particular types of pressure. This is because pressures cannot always be considered to be the same within each type and therefore do not map directly onto the ecological flow components. Instead, the hydrological effects of an abstraction or an impoundment can be summarised by a 'pick and mix' of ecological flow components. As a simple example, the main effect of abstraction - reducing flows during, and prolonging, natural low flow periods – is 'extreme or extended low flows'. By contrast, the more complex effect of a direct supply reservoir might be described by several of the ecological flow components.

The process diagrams adopt different approaches to describing physical and biological aspects, reflecting the complexities in defining biological responses, and also enduring differences in approach from the 'hydro' and 'ecology' traditions in hydroecological science:

- The physical processes are mapped out and can be prioritised. They are coloured to differentiate the different physical environments, and the usefulness of a change in habitat state as a physical indicator.
- The biotic responses represent different biotic processes, species and levels of biotic organisation. Colours denote the sign and degree of response, and therefore the sensitivity of biotic response to flow change, with further descriptive detail given for selected ecological elements in the evidence base.

Figure II.1 Generic conceptual model of impacts arising from extreme and extended low flows

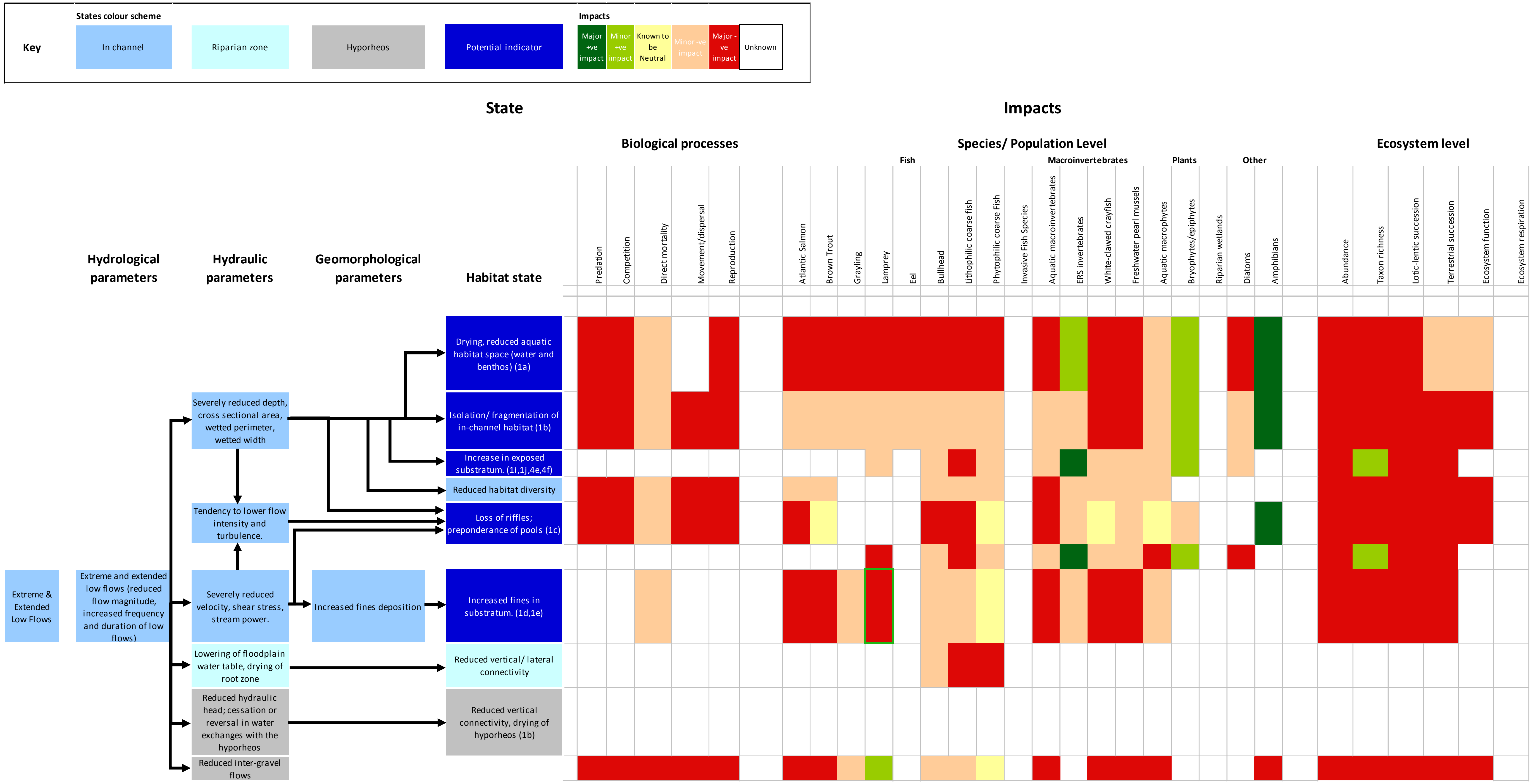


Figure II.2 Generic conceptual model of impacts arising from enhanced low flows

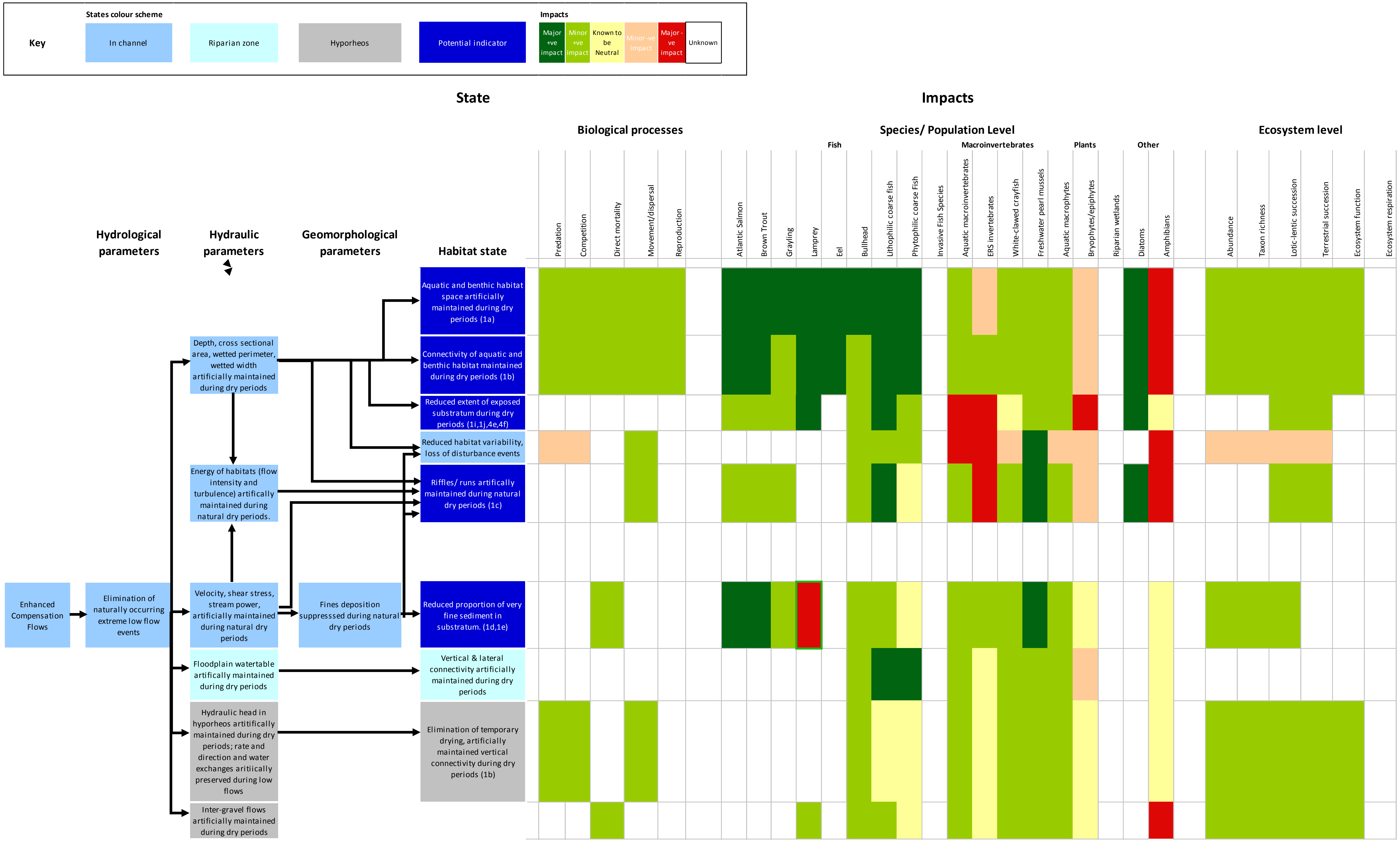


Figure II.3 Generic conceptual model of impacts arising from stabilised low flows

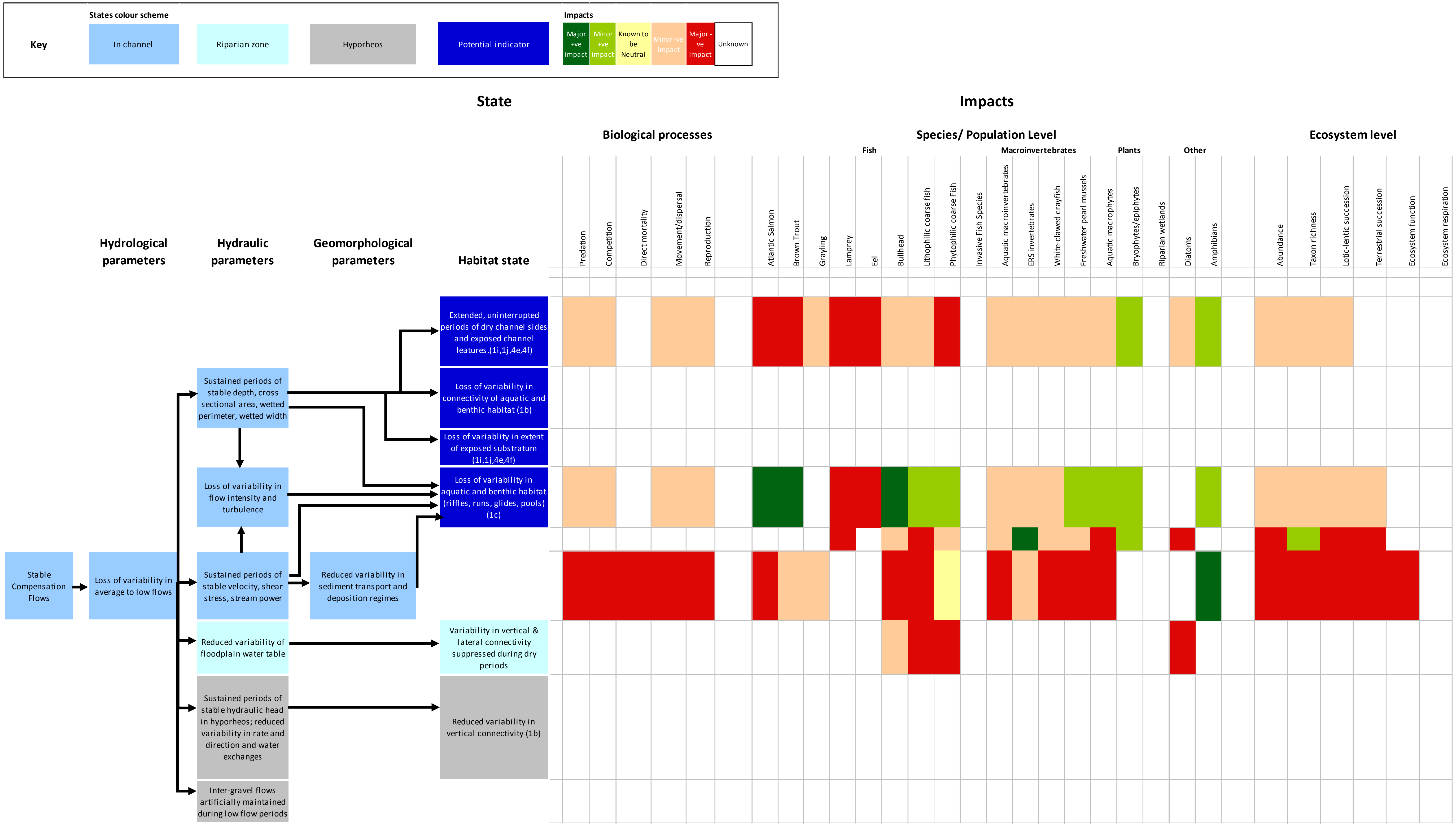


Figure II.4 Generic conceptual model of impacts arising from reductions to magnitude and frequency of freshets and small floods

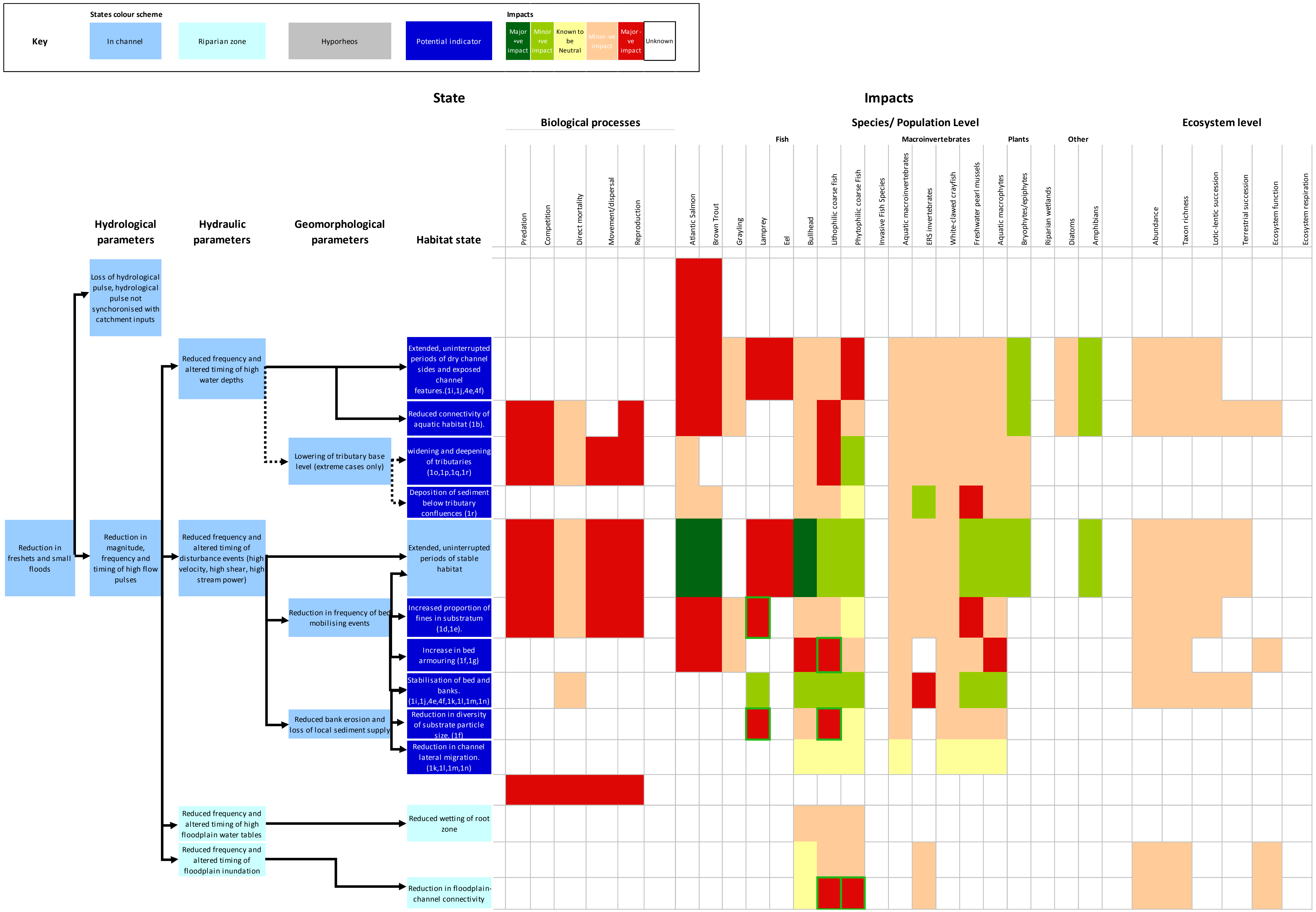


Figure II.5 Generic conceptual model of impacts arising from reductions to magnitude and frequency of large floods

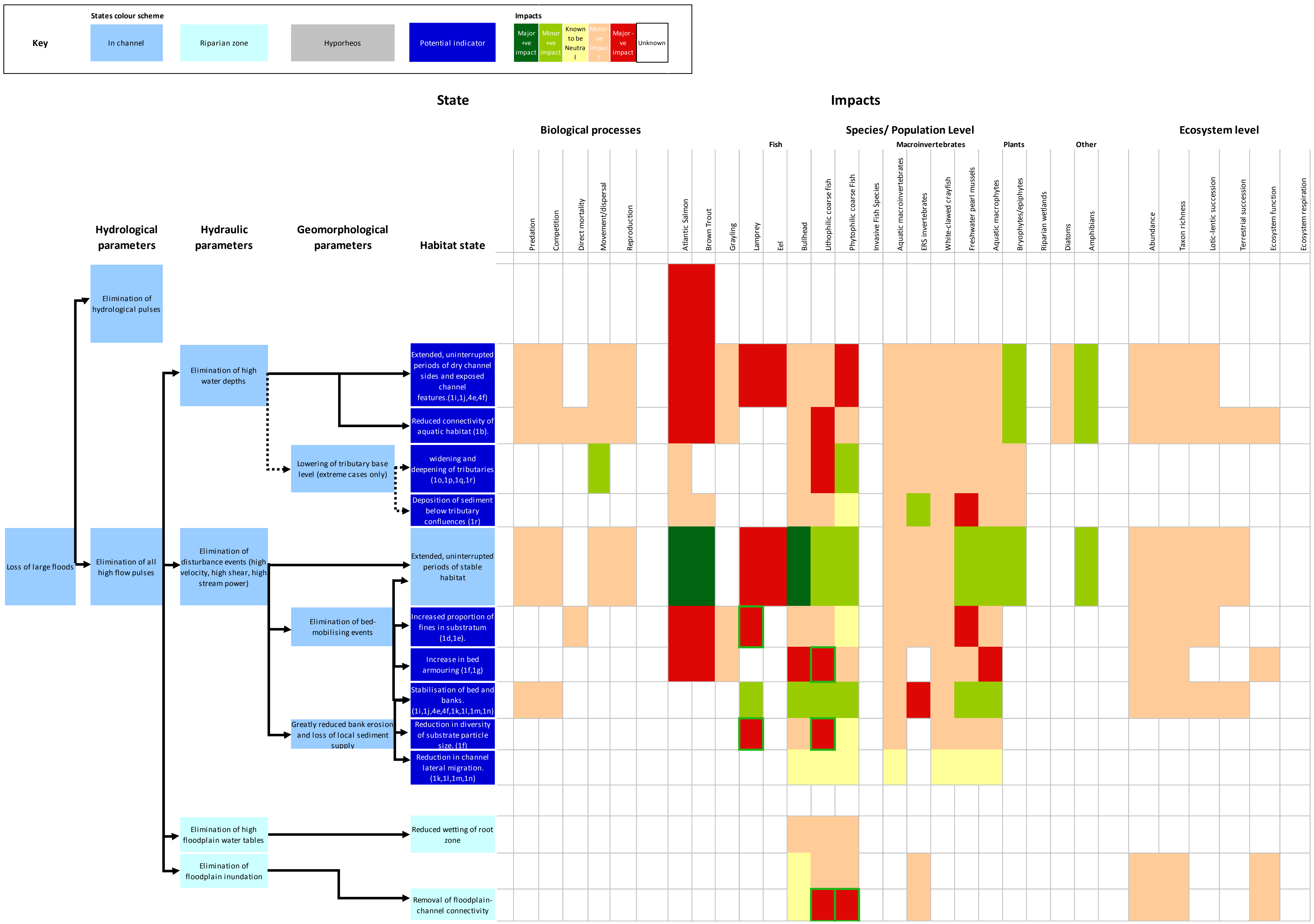
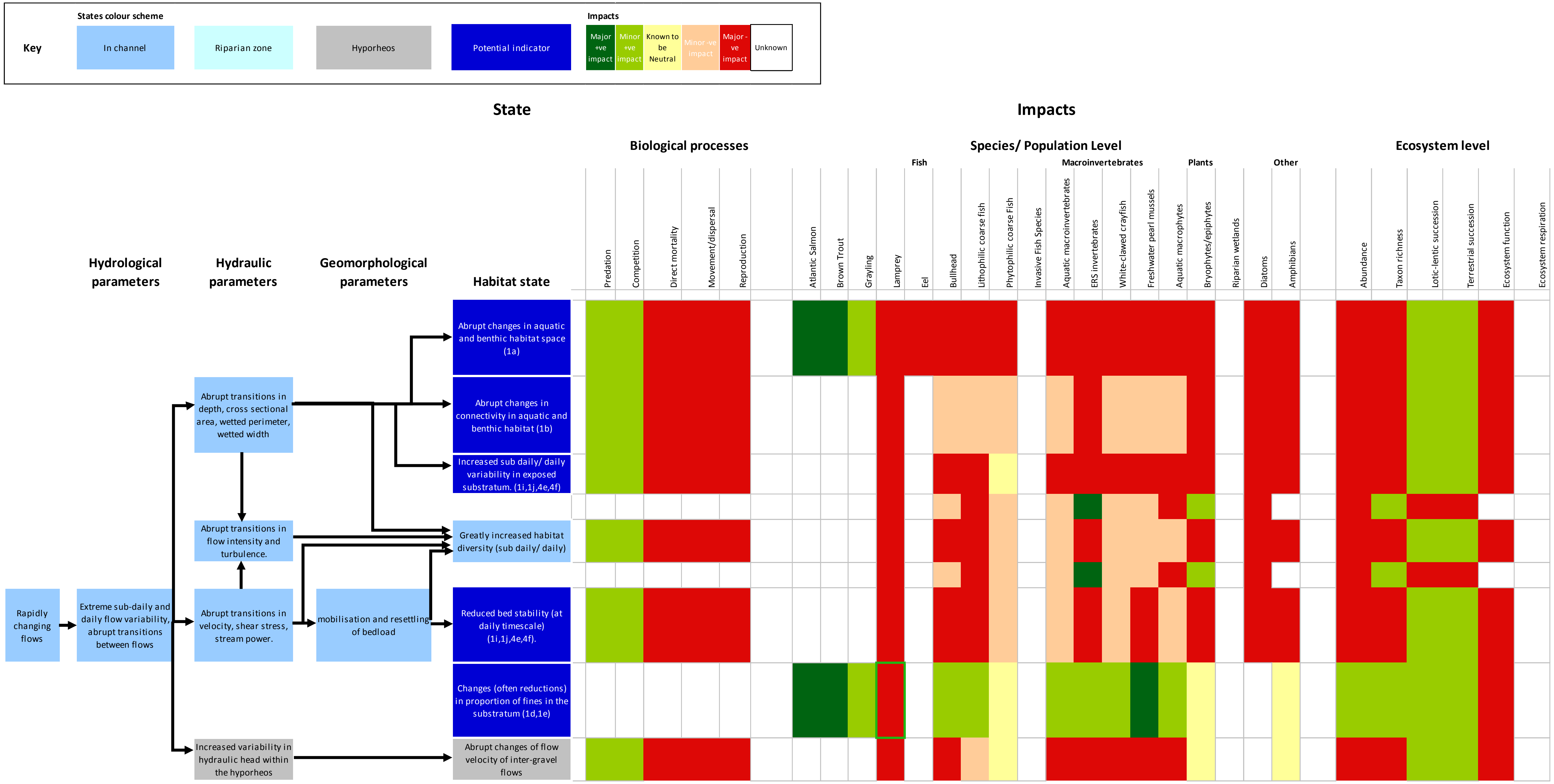


Figure II.6 Generic conceptual model of impacts arising from rapidly varying flows



II.1.3 Impact tables

Table II.1 Summary of main risks to salmonid fish posed by principal types of modified flows

Expected response, if known as likely and significant:

G↑ Growth increase
G↓ Growth decrease
N↑ Number increase (mortality decrease)
N↓ Number decrease (mortality increase)

Life stage	(1) Extreme & extended Low Q	(2) Enhanced & stabilised Q	(3) Loss of small floods (<=1yr), inc. freshets	(4) Loss of large floods (>1yr)	(5) Extreme or untimely High Q	(6) Rapid Q change	Water Temperature
Egg incubation (Oct-Mar)	Desiccation loss of gravel flushing **N↓**		loss of gravel flushing **N↓**		washout **N↓**		Incubation rate reduced at low temps
fry swim up (Mar-Apr)	Area/ habitat loss predation increased competition increased displace-ment to deeper water **N↓**				Displace-ment **N↓**	Strand-ing acute for trout due to pref. for margins **N↓**	Mismatch with 2° production **N↓**
0+ May-Nov	Area/ habitat loss predation increased competition displace-ment to deeper water **N↓ G↓**	Increased area/ habitat & production **G↑ N↑**			Displace-ment **N↓**	Strand-ing acute for trout due to pref. for margins **N↓**	Growth rate reduced at low temps from hypol. discharge
0+ & >0+ (winter)	Loss of depth shelter **N↓**	Increased shelter **G↑ N↑**			High metabolic costs **G↓**		
>0+ (inc adult residents)	Area/ habitat loss food loss predation increased displace-ment to deeper water **N↓ G↓**	Increased area/ habitat **G↑ N↑**			High metabolic costs (displace-ment) **G↓**		Growth rate reduced at low temps

Life stage	(1) Extreme & extended Low Q	(2) Enhanced & stabilised Q	(3) Loss of small floods (<=1yr), inc. freshets	(4) Loss of large floods (>1yr)	(5) Extreme or untimely High Q	(6) Rapid Q change	Water Temperature
Smolting (not applicable BT or grayling) April-June			Lack of cues **N↓**				Lack of/ or mixed stimuli NB temp. AND flow and daylength
adult passage all yr mainly May-Oct	Obstructed passage **N↓**		Lack of stimuli and directional cues **N↓**				Loss of/ or mixed cues
spawning (Oct-Dec)	Access restricted **N↓**		Lack of stimuli **N↓**			Spawn-ing disrupted **N↓**	
Kelt (Nov – April)	(Likely barriers, and greater energy demand) **N↓**		Slow or delayed d/s passage				

(Brackets) = less important or, likely but unsubstantiated

Table II.2 Summary of main risks to bullheads posed by principal types of modified flows

Expected response, if known as likely and significant:

G↑ Growth increase
G ↓ Growth decrease
N↑ Number increase (mortality decrease
N↓ Number decrease (mortality increase

Life stage	(1) Extreme & extended Low Q	(2) Enhanced & stabilised Q	(3) Loss of small floods (<=1yr, inc. freshets	(4) Loss of large floods (>1yr)	(5) Extreme or untimely High Q	(6) Rapid Q change	Water Temperature
Egg incubation (March - April)	Reduced infiltration of oxygen rich water/ siltation N↓	Stability of habitat, reduced risk of displace-ment N↑		Reduced risk of mechan-ical damage and displace-ment of eggs N↑	Washout of eggs and substrate, elevated predation risk N↓	Displace-ment and elevated predat-ion risk N↓ G ↓	Incubation rate reduced at low temps G ↓
Larvae (March-May)	Reduced area of optimal habitat N↓ G ↓	Stability of habitat availability and enhanced growth prospects G↑ N↑		Reduced risk of displace-ment and predat-ion G↑ N↑	Washout, instability of habitat and increased predation risk N↓	Displace-ment and elevated predat-ion risk N↓ G ↓	Reduced growth at low temps G ↓
0+ April-Sept	Reduced area of optimal habitat N↓ G ↓	Stability of habitat availability and enhanced growth prospects G↑ N↑		Reduced risk of displace-ment and predatio-n G↑ N↑	Washout, instability of habitat, increased predation risk and compromi-sed energy budgets N↓ G ↓	Displace-ment and elevated predat-ion risk N↓ G ↓	Reduced growth at low temps G ↓
0+ (winter)	Reduced area of optimal habitat N↓ G ↓	Stability of habitat availability and enhanced growth prospects G↑ N↑		Reduced risk of displace-ment and predatio-n G↑ N↑	Washout, instability of habitat, increased predation risk and compromi-sed energy budgets N↓ G ↓	Displace ment and elevated predat-ion risk N↓ G ↓	
>0+ (inc adult	Reduced area of optimal	Stability of habitat availability		Reduced risk of displace-	Washout, instability of habitat,	Displace ment and	

Life stage	(1) Extreme & extended Low Q	(2) Enhanced & stabilised Q	(3) Loss of small floods (<=1yr, inc. freshets	(4) Loss of large floods (>1yr)	(5) Extreme or untimely High Q	(6) Rapid Q change	Water Temperature
residents)	habitat N↓ G ↓	and enhanced growth prospects G↑ N↑		ment and preda-tion G↑ N↑	increased predation risk and compromi-sed energy budgets N↓ G ↓	elevated predat-ion risk N↓ G ↓	
spawning (March-April)	Reduced area of optimal habitat N↓ G ↓	Stability of spawning habitat availability N↑		Increas-ed area of suitable spawn-ing habitat N↑	Reduced availability of spawning habitat N↓		

Table II.3 Summary of main risks to lithophilic coarse fishes posed by principal types of modified flows. Note that EFC category 1 has been split in to two, to accommodate contrasting level /water height combinations

Expected response, if known as likely and significant:

G↑ Growth increase

G↓ Growth decrease

N↑ Number increase (mortality decrease

N↓ Number decrease (mortality increase

Life stage	(1a) Extreme & extended Low Q and low level	(1b) Extreme & extended Low Q and high level	(2) Enhanced and stabilised Q	(3) Loss of small floods (<=1yr, inc. freshets	(4) Loss of large floods (>1yr)	5) Extreme or untimely High Q	(6) Rapid Q change	Water Temper-ature
Egg incubation (March - June)	Desiccation poor infiltration of oxygen, siltation N↓	Low level, desiccation poor infiltration of oxygen, siltation N↓			Reduced risk of mechanical damage to eggs and deposited eggs being washed out N↑	Mechanical damage to eggs, physical transport of spawning substrate and washout of eggs N↓	Washout N↓	Incubation rate reduced at low temps G↓
Free embryos and larvae (Mar-June)	Lack of access to marginal nursery habitats G↓ N↓	Optimal nursery conditions and retention of important phyto/zoo-plankton food resources G↑ N↑		Reduced risk of displacement and flushing of phyto/zoo plankton blooms G↑ N↑	Reduced risk of washout of larvae and flushing of phyto/zoo-plankton blooms G↑ N↑	Washout N↓	Washout N↓	Reduced phyto/ zoo-plankton available and reduced growth at low temp G↓
0+ April-Sept	Lack of access to marginal nursery habitats G↓ N↓	Optimal nursery conditions and retention of important phyto/zoo-plankton food resources G↑ N↑		Reduced risk of displacement and flushing of phyto/ zoo-plankton blooms G↑ N↑	Reduced risk of washout of larvae and flushing of phyto/zoo-plankton blooms G↑ N↑	Displacement and/or washout N↓	Washout N↓	Reduced phyto/ zoo-plankton available and reduced growth at low temp. (Reduced recruitment potential over winter through lower lipid reserve)

Life stage	(1a) Extreme & extended Low Q and low level	(1b) Extreme & extended Low Q and high level	(2) Enhanced and stabilised Q	(3) Loss of small floods (<=1yr, inc. freshets	(4) Loss of large floods (>1yr)	5) Extreme or untimely High Q	(6) Rapid Q change	Water Temper -ature
								G ↓ N↓
0+ (winter)	Loss of marginal refuge habitat and floodplain connectivity N↓	Optimal nursery conditions and access to floodplain as refuge from sudden elevations in Q G↑ N↑		Reduced risk of displace-ment N↑	Reduced risk of washout G↑ N↑	Displace-ment and or (washout) N↓	Washout N↓	
>0+ (inc adult residents)	Congrega-tion of shoals, (increased competition and predation pressure) N↓ G ↓	Broad diversity of habitat availability (and reduced competition/ predation pressure) G↑ N↑			Reduced risk of displace-ment and more profitable energy budgets G↑ N↑	High metabolic costs (displace-ment) G ↓		
Adult spawning migration (Feb-June)	Obstructed passage of weirs N↓	Potential easement of passage over weirs N↑		Potential negative impact on longitudin-al migration and physiology -ical cues	Reduced access to floodplain/ off river habitats	*(Potential easement of passage over weirs. Use of river margins/ floodplain as migratory conduit) N↑		(Low winter temperat -ures may be import-ant to stimulate gonad develop ment)
spawning (March-June)	Reduced habitat quality through siltation and poor infiltration of clean well oxygenated water, access restricted N↓	Reduced habitat quality through siltation and poor infiltration of clean well oxygenated water N↓	Guaranteed availability of spawning habitat N↑		Reduced risk of mechanic-al damage to eggs and deposited eggs being washed out levels N↑	Reduced availability of optimal spawning habitat N↓	Spawn-ing disrupt-ion N↓ (can result in multiple cohorts) N↑	Temp increase import-ant for stimulat-ing courtship N↓

(Brackets) = less important or, likely but unsubstantiated

Table II.4 Summary of main risks to phytophilic coarse fishes posed by principal types of modified flows

Expected response, if known as likely and significant:

G↑ Growth increase
G↓ Growth decrease
N↑ Number increase (mortality decrease
N↓ Number decrease (mortality increase

Life stage	(1a) Extreme & extended Low Q and low level	(1b) Extreme & extended Low Q and high level	(2) Enhanced and stabilised Q	(3) Loss of small floods (<=1yr, inc. freshets	(4) Loss of large floods (>1yr)	5) Extreme or untimely High Q	(6) Rapid Q change	Water Temper -ature
Egg incuba-tion (April - June)	Reduced availability of spawning habitat results in high egg densities and elevated predation risk **N↓**	Eggs deposited over increased spatial scale. Predation risk reduced **G↑ N↑**		Reduced risk of deposited eggs becoming desiccated from rapidly receding levels **N↑**	Reduced risk of deposited eggs being washed out on macrophyt es or becoming desiccated from rapidly receding levels **N↑**	Washout of spawning substrate and/or eggs **N↓**	Washout of spawn-ing substrate and/or eggs **N↓**	Incubat-ion rate reduced at low temps **G↓**
Free embryos and larvae (April-June)	Lack of access to marginal nursery habitats **G↓ N↓**	Optimal nursery conditions and retention of important phyto/ zooplankton food resources **G↑ N↑**		Reduced risk of displacem ent and flushing of phyto/zoo plankton blooms **G↑ N↑**	Reduced risk of washout of larvae and flushing of phyto/zoo plankton blooms **G↑ N↑**	Washout **N↓**	Washout **N↓**	Reduced phyto/zo oplankto n availabilit y and reduced growth at low temp **G↓**
0+ May-Sept	Lack of access to marginal nursery habitats **G↓ N↓**	Optimal nursery conditions and retention of important phyto/zoopl ankton food resources **G↑ N↑**		Reduced risk of displacem ent and flushing of phyto/zoo plankton blooms **G↑ N↑**	Reduced risk of washout of larvae and flushing of phyto/zoo plankton blooms **G↑ N↑**	Displace-ment and/ or washout **N↓**	Washout **N↓**	Reduced phyto/ zoo-plankton available and reduced growth at low temp. (Lower recruit-ment potential over winter through lower

Life stage	(1a) Extreme & extended Low Q and low level	(1b) Extreme & extended Low Q and high level	(2) Enhanced and stabilised Q	(3) Loss of small floods (<=1yr, inc. freshets	(4) Loss of large floods (>1yr)	5) Extreme or untimely High Q	(6) Rapid Q change	Water Temper -ature
								lipid reserve) G ↓ N↓
0+ (winter)	Loss of marginal refuge habitat and floodplain connectivity N↓	Optimal nursery conditions and access to floodplain as refuge from sudden elevations in Q G↑ N↑		Reduced risk of displace-ment N↑	Reduced risk of washout G↑ N↑	Displace-ment and or (washout) N↓	Washout N↓	
>0+ (inc adult residents)	Congrega-tion of shoals, (increased competition and predation pressure) N↓ G ↓	Broad diversity of habitat availability (and reduced completion/ predation pressure) G↑ N↑			Reduced risk of displace-ment and more profitable energy budgets G↑ N↑	High metabolic costs (displacem ent) G ↓		
Adult spawning migration (April-June)	*(Obstruct-ed passage of weirs) N↓	*(Potential easement of passage over weirs) N↑		Reduced access to floodplain/ off river habitats	Reduced access to floodplain/ off-river habitats	*(Potential easement passage over weirs. Use of river margins as migratory conduit) N↑		(Low winter temperat -ures may help stimulate gonad develop-ment)
Spawning (April-June)	Reduced availability of spawning substrate through lack of access to marginal macrophyt-es and floodplain N↓	Enhanced access to spawning substrate within river margins and floodplain N↑		Reduced risk of deposited eggs becoming desiccated from rapidly receding levels N↑	Reduced risk of deposited eggs being washed out on macro-phytes or becoming desiccated from receding levels N↑	Reduced availability of optimal spawning habitat N↓	Sudden reduction in water level can leave eggs stranded above water line N↓	Temp. increase assists to stimulate courtship N↓

(Brackets) = less important or, likely but unsubstantiated
* = of lower importance than lithophilic guid

Table II.5 Summary of main risks to European eels posed by principal types of modified flows

Expected response, if known as likely and significant:

G↑ Growth increase
G ↓ Growth decrease
N↑ Number increase (mortality decrease
N↓ Number decrease (mortality increase

Life stage	(1) Extreme or extended Low Q	(2) Enhanced and stabilised Q	(3) Loss of small floods (<=1yr, inc. freshets	(4) Loss of large floods (>1yr)	(5) Extreme or untimely High Q	(6) Rapid rate of Q change	(7) Water temperature
Glass eel/elver upstream migration (Apr-Sep)	Access restricted and reduced cues **N↓**	Increased area/ habitat **G↑ N↑**	Lack of stimuli **N↓**		Unable to swim against flows **N↓**		
Yellow and silver eels (resident all year)	Area/ habitat loss food loss **N↓ G ↓**	Increased area/ habitat **G↑ N↑**					
Adult silver eel downstream migration (Aug-Nov)	Lack of stimuli **N↓**		Lack of stimuli **N↓**		Increased cue for migration **N↑**		

(Brackets) = less important or, likely but unsubstantiated

Table II.6 Summary of main risks to river and brook lampreys posed by principal types of modified flows

Expected response, if known as likely and significant:

G↑ **Growth increase**

G ↓ **Growth decrease**

N↑ **Number increase (mortality decrease**

N↓ **Number decrease (mortality increase**

Life stage	**(1) Extreme or extended Low Q**	**(2) Enhanced and stabilised Q**	**(3) Loss of small floods (<=1yr, inc. freshets**	**(4) Loss of large floods (>1yr)**	**(5) Extreme or untimely High Q**	**(6) Rapid rate of Q change**	**(7) Water temperature**
Egg incubation (Mar-May)	Desiccation loss of gravel flushing **N↓**		Loss of gravel flushing **N↓**		Washout **N↓**		
Larvae (Apr-Jun)	Area/ habitat loss **N↓**	Increased area/ habitat **G↑ N↑**			Unable to settle **N↓**	Stranding due to pref. for margins **N↓**	
Ammocoetes (resident all year)	Area/ habitat loss food loss displacement to deeper water **N↓ G ↓**	Increased area/ habitat **G↑ N↑**			Loss of fine substrate for burrowing **N↓**		
Macrophthalmia downstream migration (not applicable for Brook Lamprey) **(Jan-Mar)**			Lack of stimuli **N↓**		Increased cue for migration **N↑**		
Adult upstream migration (Sep-Jan)	Access restricted and reduced cues **N↓**		Lack of stimuli **N↓**		Unable to swim against flows **N↓**	Spawning disrupted **N↓**	

(Brackets) = less important or, likely but unsubstantiated

Table II.7 Summary of main risks to sea lampreys posed by principal types of modified flows.

Expected response, if known as likely and significant:

G↑ Growth increase
G ↓ Growth decrease
N↑ Number increase (mortality decrease
N↓ Number decrease (mortality increase

Life stage	(1) Extreme or extended Low Q	(2) Enhanced and stabilised Q	(3) Loss of small floods (<=1yr, inc. freshets	(4) Loss of large floods (>1yr)	(5) Extreme or untimely High Q	(6) Rapid rate of Q change	(7) Water temperature
Egg incubation (May-Jun)	Desicca-tion loss of gravel flushing **N↓**		Loss of gravel flushing **N↓**		Washout **N↓**		
Larvae (Jun-Jul)	Area/ habitat loss **N↓**	Increased area/ habitat **G↑ N↑**			Unable to settle **N↓**	Strand-ing due to pref.for margins **N↓**	
Ammocoetes (resident all year)	Area/ habitat loss food loss displace-ment to deeper water **N↓ G ↓**	Increased area/ habitat **G↑ N↑**			Loss of fine substrate for burrowing **N↓**		
Macro-phthalmia downstream migration (Oct-Dec)			Lack of stimuli **N↓**		Increased cue for migration **N↑**		
Adult upstream migration (Apr-May)	Access restricted and reduced cues **N↓**		Lack of stimuli **N↓**		Unable to swim against flows **N↓**	Spawn-ing disrupt-tion **N↓**	

(Brackets) = less important or, likely but unsubstantiated

Table II.8 Summary of main risks to diatoms posed by principal types of modified flows.

Expected response, if known as likely and significant:

G↑ Growth increase
G↓ Growth decrease
N↑ Number increase (mortality decrease
N↓ Number decrease (mortality increase

Life stage	**(1) Extreme or extended Low Q**	**(2) Enhanced & stabilised Q**	**(3) Loss of small floods (<=1yr), inc. freshets**	**(4) Loss of large floods (>1yr)**	**(5) Extreme or untimely High Q**	**(6) Rapid rate of Q change**	**Water temperature**
Diatoms	Desiccation **N↓** (Aerophilic taxa e.g. *Luticola* and *Diadesmis* potentially increase in abundance)	Potential for increase in habitat availability **N↑** Decreased disturbance could lead to develop-ment of large biofilms easily visible by eye			Scour of biofilms **N↓**		Diatoms typically show an increase in growth rate as temperature increases. Some species (e.g. *Navicula lanceolata*) are low temperature specialists and typically dominate early spring biofilms. Increase in water temp may result in **G↑** overall, but could change species composition.

(Brackets) = less important or, likely but unsubstantiated

Table II. 9 Summary of main risks to aquatic macroinvertebrates posed by principal types of modified flows

R↑ Taxon richness increase
R↓ Taxon richness decrease
N↑ Number increase (mortality decrease)
N↓ Number decrease (mortality increase)

Season	(1) Extreme & extended Low Q	(2) Enhanced & stabilised Q	(3) Loss of small floods (<=1yr), inc. freshets	(4) Loss of large floods (>1yr)	(5) Extreme or untimely High Q	(6) Rapid Q change	Water Temperature
March - May	Area/ habitat loss predation increased competition increased density in deeper fast flowing refuges N↓ R↓	Loss of baetid oviposition sites N↓	Loss of gravel flushing N↓ R↓			Stranding, particularly acute for macro-invertebrat-es in marginal habitats N↓ R↓	
June - August		Loss of baetid oviposition sites N↓		Preven-tion of washout of eggs/ larvae R↑ N↑	Washout od eggs/ larvae N↓	Stranding, particularly acute for macro-invertebrat-es in marginal habitats N↓ R↓	
August - September		Loss of habitat hetero-geneity if at full channel width R↓		Preven-tion of washout of eggs/ larvae R↑ N↑	Washout of eggs/ larvae N↓	Stranding, particularly acute for macro-invertebrat-es in marginal habitats N↓ R↓	
October - February	Area/ habitat loss predation increased competition increased density in deeper fast flowing refuges N↓ R↓	Loss of habitat hetero-geneity if at full channel width R↓	Loss of gravel flushing N↓ R↓			Stranding, particularly acute for macro-invertebra-tes in marginal habitats N↓ R↓	

Table II.10 Summary of main risks to aquatic macrophytes posed by principal types of modified flows

R↑ Taxon richness increase
R↓ Taxon richness decrease
N↑ Number increase (mortality decrease)
N↓ Number decrease (mortality increase)

Season	(1) Extreme & extended Low Q	(2) Enhanced & stabilised Q	(3) Loss of small floods (<=1yr), inc. freshets	(4) Loss of large floods (>1yr)	(5) Extreme or untimely High Q	(6) Rapid Q change	Water Temperature
March - May	Area/ habitat loss conditions suitable for algae growth N↓ R↓	Enhanced flows reduce algae growth and increase wetted area; stable flows suitable for macrophyte growth N↑		No washout N↑	Washout of early plant growth N↓		
June - August		Enhanced flows reduce algae growth and increase wetted area; stable flows suitable for macrophyte growth N↑		No washout R↑ N↑	Washout N↓		
August - September		Enhanced flows reduce algae growth and increase wetted area; stable flows suitable for macrophyte growth ↑		No washout R↑ N↑	Washout N↓		
October - February	Loss of clearing of dead macro-phytes and fine sediment N↓ R↓	loss of clearing of dead macro-phytes and fine sediment N↓ R↓	Loss of clearing of dead macro-phytes and fine sediment N↓ R↓	Loss of clearing of dead macro-phytes and fine sediment N↓ R↓	Clearance of dead macro-phytes R↑ N↑		

Table II.11 Summary of main risks to invertebrates of exposed riverine sediments (ERS) posed by principal types of modified flows

R↑ Taxon richness increase
R↓ Taxon richness decrease
N↑ Number increase (mortality decrease)
N↓ Number decrease (mortality increase)

Season	(1) Extreme & extended Low Q	(2) Enhanced & stabilised Q	(3) Loss of small floods (<=1yr), inc. freshets	(4) Loss of large floods (>1yr)	(5) Extreme or untimely High Q	(6) Rapid Q change	Water Temperature
March - May	Loss of habitat disturbance by water level fluctuation **N↓ R↓**	Loss of habitat disturbance by water level fluctuation **N↓ R↓**	Loss of habitat disturbance by water level fluctuation **N↓ R↓**	Prevention of complete washout during sensitive life stages **R↑ N↑**	Adapted to floods in winter/ spring	Adapted to floods in winter/ spring	
June - August	Loss of habitat disturbance by water level fluctuation **N↓ R↓**	Loss of habitat disturbance by water level fluctuation **N↓ R↓**	Loss of habitat disturbance by water level fluctuation **N↓ R↓**	Prevention of complete washout during sensitive life stages **R↑ N↑**	Complete washout of habitat and communities **N↓ R↓**	Water levels rising too fast prevents avoid-ance behav-iours **N↓**	
August - September	Loss of habitat disturbance by water level fluctuation **N↓ R↓**	Loss of habitat disturbance by water level fluctuation **N↓ R↓**	Loss of habitat disturbance by water level fluctuation **N↓ R↓**	Prevention of complete washout during sensitive life stages **R↑ N↑**	Complete washout of habitat and communities **N↓ R↓**	Water levels rising too fast prevents avoid-ance behav-iour **N↓**	
October - February	Loss of habitat disturbance by water level fluctuation **N↓ R↓**	Loss of habitat disturbance by water level fluctuation **N↓ R↓**	Loss of habitat disturbance by water level fluctuation **N↓ R↓**	Adapted to floods in winter	Adapted to floods in winter	Adapted to floods in winter	

Table II.12 Summary of main risks to freshwater pearl mussels posed by principal types of modified flows

G↑ Growth increase
G ↓ Growth decrease
N↑ Number increase (mortality decrease)
N↓ Number decrease (mortality increase)

Season	(1) Extreme & extended Low Q	(2) Enhanced & stabilised Q	(3) Loss of small floods (<=1yr), inc. freshets	(4) Loss of large floods (>1yr)	(5) Extreme or untimely High Q	(6) Rapid Q change	Water Temperature
March - May	Stranding and increased fine sediment deposition N↓ G↓	Stable habitat and reduced deposition of fine sediment G↑ N↑	Reduced fine sediment flushing N↓ G↓	Preven-tion of complete washout G↑ N↑	Adapted to floods in winter/ spring		
June - August	Stranding and increased fine sediment deposition N↓ G↓	Stable habitat and reduced deposition of fine sediment G↑ N↑		Preven-tion of complete washout G↑ N↑	Complete washout of habitat and organisms during sensitive life stage N↓ G↓		
August - September	Stranding and increased fine sediment deposition N↓ G↓	Stable habitat and reduced deposition of fine sediment G↑ N↑		Preven-tion of complete washout G↑ N↑	Complete washout of habitat and organisms during sensitive life stage N↓ G↓		
October - February	Stranding and increased fine sediment deposition N↓ G↓	Stable habitat and reduced deposition of fine sediment G↑ N↑	Reduced fine sediment flushing N↓ G↓	Preven-tion of complete washout G↑ N↑	Adapted to floods in winter		

Table II.13 Summary of main risks to white-clawed crayfish posed by principal types of modified flows

G↑ Growth increase
G↓ Growth decrease
N↑ Number increase (mortality decrease)
N↓ Number decrease (mortality increase)

Season	(1) Extreme & extended Low Q	(2) Enhanced & stabilised Q	(3) Loss of small floods (<=1yr), inc. freshets	(4) Loss of large floods (>1yr)	(5) Extreme or untimely High Q	(6) Rapid Q change	Water Temperature
March - May	Stranding in early spring in margin refuges N↓ G↓	Increased risk of invasion by non-native species N↓ G↓	Reduced fine sediment flushing and increased risk of invasion by no-native species N↓ G↓	Preven-tion of complete washout G↑ N↑	Adapted to floods in winter/spring		
June - August	Loss of habitat space, density dependent mortality N↓ G↓	Increased risk of invasion by non-native species N↓ G↓		Preven-tion of complete washout G↑ N↑	Complete washout of habitat and organisms during sensitive life stage N↓ G↓		
August - September	Loss of habitat space, density dependent mortality N↓ G↓	Increased risk of invasion by non-native species N↓ G↓		Preven-tion of complete washout G↑ N↑	Complete washout of habitat and organisms during sensitive life stage N↓ G↓		
October - February	Stranding in early in margin winter refuges N↓ G↓	Increased risk of invasion by non-native species N↓ G↓	Reduced fine sediment flushing and increased risk of invasion by no-native	Preven-tion of complete washout G↑ N↑	Adapted to floods in winter		

			species N↓ G↓				

Table II.14 Summary of main risks to amphibians posed by principal types of modified flows

↑ Growth increase
G ↓ Growth decrease
N↑ Number increase (mortality decrease)
N↓ Number decrease (mortality increase)

Season	(1) Extreme & extended Low Q	(2) Enhanced & stabilised Q	(3) Loss of small floods (<=1yr), inc. freshets	(4) Loss of large floods (>1yr)	(5) Extreme or untimely High Q	(6) Rapid Q change	Water Temperature
March - May	Potential breeding habitat if ponded or very slow flowing G↑ N↑	Amphibians do not favour flowing water N↓ G↓					
June - August	Potential breeding habitat if ponded or very slow flowing G↑ N↑	Amphibians do not favour flowing water N↓					
August - September		Amphibians do not favour flowing water N↓					
October - February		Amphibians do not favour flowing water N↓					

Table II.15 Summary of main risks to bryophytes posed by principal types of modified flows

R↑ Taxon richness increase
R ↓ Taxon richness decrease
N↑ Number increase (mortality decrease)
N↓ Number decrease (mortality increase)

Season	(1) Extreme & extended Low Q	(2) Enhanced & stabilised Q	(3) Loss of small floods (<=1yr), inc. freshets	(4) Loss of large floods (>1yr)	(5) Extreme or untimely High Q	(6) Rapid Q change	Water Temperature
March - May	Stable exposed substratum in river channels and riparian habitats G↑ N↑	Enhanced flow submerges bryophyte habitat N↓ G↓	Stable exposed substrat-um in river channels and riparian habitats G↑ N↑	Stable exposed substrat-um in river channels and riparian habitats G↑ N↑	Extremely high flow submerges and scours bryophyte habitat N↓ G↓		
June - August	Stable exposed substratum in river channels and riparian habitats G↑ N↑	Enhanced flow submerges bryophyte habitat N↓ G↓	Stable exposed substrat-um in river channels and riparian habitats G↑ N↑	Stable exposed substrat-um in river channels and riparian habitats G↑ N↑	Extremely high flow submerges and scours bryophyte habitat N↓ G↓		
August - September	Stable exposed substratum in river channels and riparian habitats G↑ N↑	Enhanced flow submerges bryophyte habitat N↓ G↓	Stable exposed substrat-um in river channels and riparian habitats G↑ N↑	Stable exposed substrat-um in river channels and riparian habitats G↑ N↑	Extremely high flow submerges and scours bryophyte habitat N↓ G↓		
October - February	Stable exposed substratum in river channels and riparian habitats G↑ N↑	Enhanced flow submerges bryophyte habitat N↓ G↓	Stable exposed substratum in river channels and riparian habitats G↑ N↑	Stable exposed substrat-um in river channels and riparian habitats G↑ N↑	Extremely high flow submerges and scours bryophyte habitat N↓ G↓		