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# Incorporating climate change in river typologies for the Water Framework Directive

Science Project Record SC030301



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**Authors:**
Chris G. Kilsby, Andrew R.G. Large, Malcolm D. Newson
Harriet G. Orr, Claire L. Walsh

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**Research Contractor:**
Chris G Kilsby, School of Civil Engineering and Geosciences, University of Newcastle, Newcastle upon Tyne, NE1 7RU
Tel: 0191 222 5614

**Environment Agency's Project Manager:**
Harriet G Orr, Environment Agency Research Fellow (Climate Change), Hysed Geography Department, Lancaster University, Lancaster LA1 4YB
Tel: 01524 593 923

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Steve Killeen

Steve Killeen

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# Executive summary

The EU Water Framework Directive (WFD) does not explicitly consider the implications of climate change in detail. However, there is the potential to explore these implications by carefully defining ‘ecological status’ and through the adaptive management guidelines of the Directive. Incorporating climate change in operating the Directive, requires climate-sensitive integration of geomorphological, hydrological and ecological functional links through new research at the catchment scale. The starting point for such research is the existing range of river typologies and classifications from all three disciplines.

Assessments of ecological status will be reported at river basin district scales, within which individual water bodies will be characterised. Water bodies are currently defined using simple ‘top-down’ typologies. Typologies make assumptions about our knowledge of driving, process and variables; classifications, by contrast, are empirical. There is a need to incorporate better process information, or substitutes for process, in water body typologies to enable climate change impacts to be included in the workings of the WFD. This would not only provide a more accurate characterisation of water bodies and sharper definition of ecological status, but it could also lead to a programme of measures to adapt to climate change. Crucially, the currently divergent process information on geomorphology, hydrology and ecology would be integrated within a spatial context; in other words, both upstream and downstream interactions at the catchment scale.

Characterising channel types by integrating geomorphology, hydrology and ecology requires the careful review of existing approaches along with a practical test of those most likely to serve the needs of the WFD under climate change scenarios. Existing approaches to determining hydrological, biological and geomorphological typologies in rivers, as well as those that have the potential to measure or incorporate climate change impacts, are reviewed in this report. Whilst the breadth of the review is international, the selection of the most promising approaches pays specific attention to UK conditions, applications and relevance.

None of the typologies reviewed in this report are directly transferable to rivers in England and Wales and their level of integration of geomorphology, hydrology and ecology is generally poor. There are problems with the theories within each discipline and with the empirical database available for England and Wales. A minority of typologies are process-based but have not been widely applied and may need to be adapted for the rivers of England and Wales. Users may also need to find a compromise between the excessive regulatory workload implied by local conditions and the high levels of uncertainty in ‘universal’ typologies. Some variables from existing typologies (such as slope, stream power and flow characteristics) can be used to create datasets suitable for climate change assessment.

A significant research gap, widely identified as a potential flaw in the delivery of the WFD, is in the workable concept of ‘hydromorphology’ and its impact on long-term ecosystem functioning. The current assumption is that diverse channel and flow conditions boost biodiversity. As a starting point, users should employ process-based geomorphological typologies within which local variability in channel morphology can be compared with biological and ecological habitat and species population data. The

authors recommend that this be carried out via a catchment-scale test outlined in this report. Further concepts such as hydraulic modelling can then be used to explore changes in available aquatic habitat and scaled up to assess catchment implications for habitat loss or gain and impacts on ecological status from different climate scenarios.

Geomorphological typologies can be improved by including variables for climatically-driven responses such as the dominant characteristics of morphological change (styles of adjustment), sediment transport and morphological diversity (rate, location of channel adjustment). Rivers adjust to change but current definitions of typologies do not permit users to adequately evaluate the complex response of catchment and water body phenomena, such as 'memory' and feedback, or to monitor and manage change.

Following the review of current theoretical and empirical approaches, this report builds on research by the Environment Agency to explore the complex interactions within a catchment. A catchment platform, the Eden in Cumbria, was chosen for this study because of its physiographic variability and biodiversity; it was used to collect data over a range of spatial and temporal scales.

The proposed approach and catchment test involves developing tools to describe geomorphological structure at the catchment, basin, reach, and bedform scale, with ecologically relevant definitions of fluvial habitats. The field study in this project provides an example of how to produce 'ecological status maps' and explore the potential impact of climate change on both status and interactions between hydromorphology and biological quality.

This report will help users who are implementing the WFD to consider the potential impacts of climate change. The report explains how to carry out catchment-scale tests and develop measures to adapt to climate change. The results of this study provide useful data for a river basin management plan, including a programme of measures (POM), for the Eden along with approaches that could be applied elsewhere.

# Contents

# 1 Introduction

## 1.1 Climate change in context

There is a growing acceptance that humankind is currently experiencing a period of rapid climate change. This, coupled with awareness that the surface of the planet increasingly bears the signs of human activity, has brought the concept of landscape sensitivity to the forefront of recent geomorphic research (Thomas, 2001). There is further growing acceptance of the need for adaptive tools aimed at management change under a range of different climate scenarios. From the wealth of existing information on the classification, assessment and monitoring of river system status, users must select and develop methods which offer a predictive capability rather than a simple taxonomy. The aim of this report is to outline the range of characterisation tools available, their capacity for incorporating or responding to climate and land use change and the data collection required for each approach. However, this report does not necessarily provide an exhaustive list, nor does it attempt to develop new approaches.

This report reviews existing approaches to hydrological, biological and geomorphological typologies in rivers, highlighting approaches that have the potential to detect or incorporate climate change. Evidence thus far suggests that climate change will affect UK river systems through changing flow regimes, increased sediment yields, channel instability and morphological change (Sear, 1994).

It is not within the scope of this study to develop new methods to assess potential system perturbation from climate change. However, new initiatives in policy legislation for Europe's rivers have derived new terminology, such as 'hydromorphology', within the EU Water Framework Directive (WFD).

## 1.2 Climate change and system response

River patterns and morphology result from complex patterns of sediment erosion, transport and deposition within the constraints imposed by the geology and terrain of the surrounding landscape (Greig, 2004). These processes results in an intricate and diverse continuum of channel patterns and forms (Powell, 1998; Lewin and Brewer, 2001; Harvey, 2001). UK rivers have been shown to adjust to climatic drivers in a number of ways. In a discussion of the sensitivity of Scottish rivers and upland valley floors over the period 1750-2000, Werritty and Leys (2001) highlight the alternation of 'flood-rich' and 'flood-poor' periods, an analysis repeated in north east rivers and other UK sites by Macklin *et al*. (2002). This alternation occurs as a result of changes in global-scale circulation patterns, a feature mirrored in fluvial systems further afield and over much longer timescales (Macklin *et al*. 2002; Viles *et al*. 2003).

Long-term planning for rivers in England and Wales should include the potential impact of climate change on river system hydrology, geomorphology and ecology. Landforms react to environmental change in one of two ways (Schumm, 1979: Werritty and Leys, 2001).

Robust systems, as defined by Werritty and McEwan (1997), have the capacity to absorb change with only minor adjustments.  While individual landforms are formed and reformed in new locations, they retain their generic identity as part of a landform assemblage.  Examples include channel planform migration with characteristic pools, riffles, eroding banks and bar formation.  Responsive systems are those which, under certain drivers, undergo a fundamental and persistent change in their morphology after crossing a threshold.  The result is a new landform assemblage, generically different to that which existed before.  Werritty and Leys (2001) give an example of a responsive-mode system as the phase shift from meandering to braided planform as a result of extreme flooding. The latter form is uncommon and very few UK river systems show evidence of such abrupt changes in river behaviour or channel metamorphosis.

The Environment Agency (2004) highlights a number of management goals relating to the river-climate system:

- System *stability* – a state whereby the system is not easily changed or affected;
- System *resilience* – system resumes its form and function after stress;
- System *flexibility* – state where the system can easily adjust without collapsing;
- System *adaptivity* – whereby the system can easily adapt to prevailing conditions

The Environment Agency (2004) recommends management approaches which ensure system resilience to climate change in the short term and system adaptivity in the long term. Resilient management seeks to maintain the current balance of the system into the future, whilst adaptive management recognises that the system drivers may have changed fundamentally and a new balance may be necessary.

### 1.2.1 Linking fluvial geomorphology and ecology river system scale

As Newson and Newson (2000) point out, there have been several periods of productive interaction between the concepts and practices of fluvial geomorphology and those of freshwater ecology, especially at the scale of river systems.  The River Continuum Concept or RCC (Vannote *et al*., 1980) stresses an orderly change in biotic functioning downstream in relation to nutrient levels and biotic processes  - a change termed ‘synchronised species replacement’.  The RCC uses geomorphological stream ordering as a (somewhat arbitrary) indication of position within the catchment unit, as does the serial discontinuity model of Ward and Stanford (1983).  Until recently, however, the strength of the RCC and its large-scale explanatory seemed to frustrate interdisciplinary work.  The evolution of the hydraulic stream ecology approach (Statzner *et al*., 1988) and new concepts of the space-time links in physical habitat patterns (Ward and Stanford, 1988; Townsend, 1989) would appear to offer some scope for integrated assessments.  Coincidentally, there is growing frustration amongst geomorphologists and ecologists with the routine application of hydraulic habitat simulation models such as PHABSIM (for example, King and Tharme, 1993).  This frustration arises from limitations in the models’ ability to represent patterns of interaction between channel morphology and a range of flows, along with the limited biological calibration to predict ecological responses to flow regulation (and potentially system response to wider hydroclimatic change).

## 1.3 Hydromorphology and the Water Framework Directive

The term 'hydromorphological' was introduced by the WFD to describe the overall quality of water bodies in addition to their physico-chemical and ecological properties. It does not figure in any recognised English dictionaries (unlike the words 'fluvial' and ‘geomorphology’, which together can describe the hydromorphological quality of rivers). The word is defined by NTNU (2002) as “the hydrological characteristics of rivers together with the physical structure that they create”. A semantic extension from 'morphology' (of river channels) to 'hydromorphology' has occurred because of the need to consider the many natural and anthropogenic variants of river flow regime as well as fluvial geomorphology in the description of river physical habitat.

Under the WFD, each surface water category – rivers, lakes, transitional waters and coastal waters – are assigned specific hydromorphological quality elements. However, as Figure 1.1 shows, these elements only contribute to status classification for water bodies at high ecological status/potential, or when downgrading to good ecological status. For other ecological status/potential classes, status is based solely on the biological and water quality classification, resulting in an incongruity. In order to fully assess the status of water bodies under changing hydroclimatic conditions, as well as make predictions of the impacts of such changes, monitoring techniques and classification schemes will need to incorporate the complete suite of hydrological, geomorphological and ecological attributes.

As Greig (2004) points out, hydromorphological quality elements of the WFD are defined in terms of supporting the biological quality elements, although the reverse may be the case (Wilby *et al.* in press) (Figure 1.1).

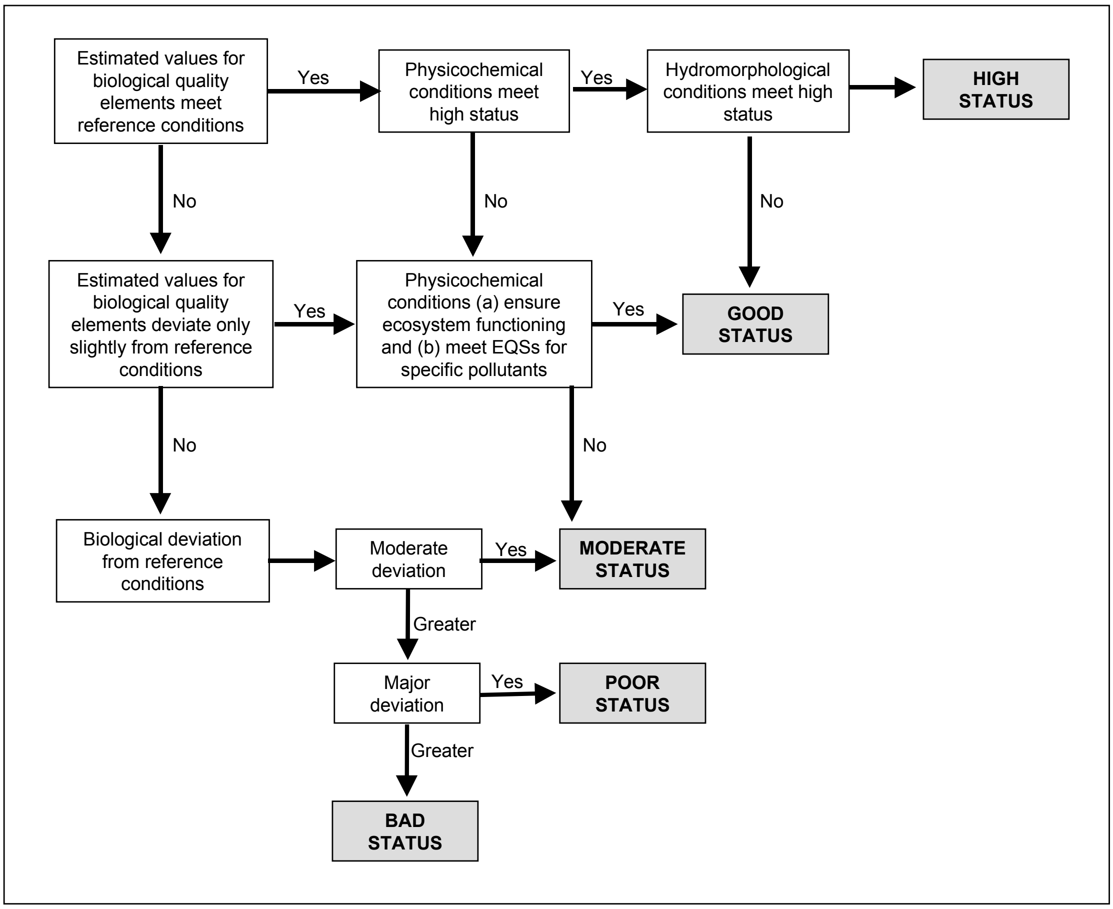


**Figure 1.1 Schematic diagram illustrating the definition of 'ecological status' under the European Water Framework Directive (WFD)**

Table 1.1 lists the range of ecological quality to be assessed for rivers and other water bodies under the WFD, including elements of hydromorphological quality. Whilst the latter applies only to the highest class, there are strong drives, particularly under Scottish legislation which wholly adopts the WFD, to gain some control over physical alterations to river channels, including restoration. In this case, a full range of hydromorphological conditions and programmes of measures (POMs) to secure or improve ecological status will need to be defined, assessed and monitored, although climate change may seriously undermine these efforts (Wilby *et al*., in press). The Environment Agency has a land drainage asset tool which could be modified as a policy tool.

At the lower extreme of hydromorphological quality, work is proceeding to define the Directive's aspiration for the designation of heavily-modified water bodies (HMWBs).These bodies (typically standing waters rather than rivers) will potentially be subjected to less stringent regulation to enable their ecological potential to be realised. While HMWBs typically represent standing water habitat, lentic habitats within fluvial hydrosystems (such as oxbows, floodplain ponds and backwaters) are potentially at risk here if located adjacent to HMWBs.

**Table 1.1 Quality elements for the classification of ecological status in rivers under the European Framework Directive (WFD). Source: CEN 2004**

| Quality element | Description |
|---|---|
| Biological elements | Composition and abundance of aquatic flora<br>Composition and abundance of benthic invertebrate fauna<br>Composition, abundance and age structure of fish fauna |
| Hydromorphological elements supporting the biological elements | Hydrological regime<br>Quantity and dynamics of water flow<br>Connection to groundwater bodies<br>River continuity<br>Morphological conditions<br>River depth and width variation<br>Structure and substrate of the river bed<br>Structure of the riparian zone |
| Chemical and physico-chemical elements supporting the biological elements | Thermal conditions<br>Oxygenation conditions<br>Salinity<br>Acidification status<br>Nutrient conditions<br>Specific pollutants<br>Pollution by all priority substances identified as being discharged into the body of water<br>Pollution by other substances identified as being discharged into the body of water |

Users of this report should bear in mind that the WFD – the main driving force for water resource legislation and policy in Europe over coming decades - has little inbuilt and implicit recognition of the potential implications of wider hydroclimatic change on elements listed in Table 1.1.

## 1.4 Scope of this review

The literature review in this report covers four main themes:

- Geomorphological assessment techniques
- Hydrological assessment techniques
- Biological assessment techniques
- Techniques that integrate the above.

The following critical considerations were covered:

- Which techniques allow the identification of key parts of the fluvial system which (a) are sensitive; (b) have ‘feedback implications’ for other aspects of the system (such as sediment pulses)?

- What are the key variables that will indicate change in geomorphology, hydrological and biological state?

- What tools, techniques and approaches are best placed to detect and predict trajectories of change?

- Are techniques based on restrictive assumptions or do they have ranges of tolerance, suitability and optima? If such flexible techniques exist, are they the ‘best of the best’?

- Are typologies used in the UK only suitable for current conditions? Are they process-based or taxonomic?

- Which techniques are transferable to other water body types?

# 2 Geomorphological typologies

## 2.1 Background to geomorphological classification

Geomorphological classifications of the landscape date from the 19th century, although models of landscape evolution have come and gone. At present the only consensus is that landforms and landscapes change over time - there is no consensus on the sequence of events or rates of change. Although there is broad agreement that landscapes differ in different parts of the world, it is not always understood why this is so (Thorn, 1988). Clearly the long timescales for geomorphological evolution, persistence of memory effects, spatially and temporally-variable sensitivity to change and difficulty in monitoring sediment regimes has hindered the development of process understanding to a large degree. In the UK there has been no formal monitoring of either geomorphological change or sediment transport (cf. North America).

There is general acceptance that river management must take into account processes that operate over longer than engineering timescales, along with understanding the response of rivers to climate and land use changes. However, there is profound ignorance about how climate change affects channel processes and responses (Macklin and Lewin, 1997; Wilby *et al*., in press).

Early morphological classification of rivers systems generally focused on drainage network, stream order and planform patterns (see Montgomery and Buffington, 1998, and Thorne, 1997 for reviews). Many of these have limited applicability and may oversimplify important processes. In general, such approaches are useful for a high level classification at landscape (river basin or segment) scales.

Fluvial geomorphology in the UK is a young subject which in its early years focused on small-scale process studies. Notable developments in process understanding have been made (for example, Harvey, 1991; Harvey *et al*., 1979) and at larger spatial scales attempts have been made to identify process dominance or intensity zones in catchments (Lawler, 1992). There is general acceptance of the processes and controls on changes in channel planform (for example, Brice, 1974; Hooke, 1995), but in reality, meandering, braided and straight patterns only develop in a predictable way where channels are unconfined and working their own alluvium rather than glacial or historic flood deposits (Newson, 2002). In UK rivers, the degree of modification (often extending back to monastic times but certainly for hundreds of year) adds a further complication. Newson (2002) summarised UK river channels as sediment supply-limited, polycyclic in profile, confined within glacial and periglacial sediments and extensively modified by engineering.

Nevertheless, unifying principles that describe basic controls on downstream morphological change are extremely useful along with models or styles of fluvial adjustment. Generic process-based models for typologies are discussed below and examples of applications given (see Goodwin, 1999 for review). They offer more of an explanation rather than merely a description lacking extrapolation and predictive capacity. Homogeneous units or reaches can be readily determined at large scales but where modification or particular geological history exists, significant reaches may only be determined after extensive study (Church, 1992).

Modelling advances at the small scale show considerable promise (for example Reid *et al*. in review a and b). However, scaling up from very small catchments to larger ‘main river’ or segment and catchment scales is not well advanced (Downs and Priestnall, 2003). Models have tended to consider on-site erosion (some have sediment delivery algorithms), but relatively few have any verification data. Such models may eventually help to investigate water storage potential in key landforms, such as periglacial and soliflucted slopes that may be important for mitigation of climate change effects but are hard to observe in the field. Catchment-scale models based on coarse resolution data (for example, 1 km land use data) are more suitable for large-scale strategic assessments and may not help operational scale management (see Figure 5.2). Pragmatic attempts to link these types of risk-based models to more detailed local-scale field assessments have been shown to have some management use. Similar approaches are being adopted to assess risks of pollution from of particulate and phosphorous supply (Heathwaite, 2003). For example, the PSYCHIC project aims to build a Decision Support System on this basis (http://www.psychic-project.org.uk/).

A common theme in modelling papers and process-based typologies is that local controls on geomorphic condition, response and memory or history render models unsuccessful (for example, de Vente and Poessen, in press). The need for information at the reach scale for operational management and at the catchment scale for strategic decision-making has resulted in a proliferation of techniques that are largely based on reconnaissance-type surveys. Most of these collect data on fluvial channel and in some cases floodplain morphology; some may be amenable to repeat survey and potential to detect change. Most of the available techniques and many of the developed typologies do not necessary include measurements of sensitivity to change, although they may pick up on geomorphic activity.

Typologies most able to detect change will be those that include some spatial connectivity up and downstream, thus incorporating system memory and long timescales of response to disturbance. Making the upstream downstream link is important because the WFD requires that all water bodies reach good ecological status, which marks a change from primarily protecting the good whilst removing the bad to a system of ecological process which links up and downstream effects (Boon, 2000; see also Section 1).

The purpose of typology is to enable type-specific reference conditions to be established. Such conditions then become the basis for classification schemes, with consequences for all subsequent operations when implementing the WFD (including monitoring, assessment and reporting) (Borja et al., 2004). River managers will be particularly interested in the ability to describe and predict the adjustment of channel form in three dimensions, which requires knowledge of changes in the rate or style of adjustment with changes in driving variables, in this case climate change.

## 2.2 Generic process-based typologies

Many of the published fluvial classification approaches based on geomorphic processes focus on specific parts of the river systems, especially floodplain areas. Few cover all three of Schumm's (1977) geomorphic zones (that is, headwaters, mid-reaches and lowland rivers). A selection of these are briefly described here. The following section (Section 2.3) describes typologies that have been applied to river management problems at larger scales.

The channel morphology of headwater streams has been associated with debris flow impacts, channel substrate size and processes and rates of sediment transport (Whiting and Bradley, 1993). Planform adjustment classifications are generally limited to unconfined channels (for example, Leopold and Wolman, 1957). Nanson and Croke (1992) describe an energy-based floodplain classification based on stream power and cohesiveness of floodplain sediments, which requires detailed information on sediment types that can usually only be gained by reconnaissance survey. Similar data gathering enabled bar patterns to be related to changes in channel gradient and sediment supply (Church and Jones, 1982).

Church (1992) offers a typology based on downstream change in certain variables. Uniquely amongst classification schemes, this typology addresses the issue of size and scale, including changes in width/depth ratios, bed roughness to flow depth and channel slope. Changes in morphological adjustment characteristics and modes of sediment transport are also described. Church (1992) defines approximate quantitative thresholds for changes from one type to another (mainly slope and discharge-based) but cautions that these are largely based on unmodified channels in the Pacific North West. These thresholds have considerable use, particularly in outlining potential channel changes, and may be useful in modelling However, there is no information on whether threshold based typologies have been used effectively to aid strategic decision-making or reach-scale management.

A number of attempts have been made to characterise types of fluvial adjustment which may be helpful in indicating trajectories of change. Many are focused on changes in meander development and lateral change in channel planform (for example, Kellerhals and Church, 1989; Hooke and Redmond, 1992). Downs (1995a) proposes a generalised channel adjustment typology which relies heavily on expert judgement and interpretation of field reconnaissance information.

It is argued that although local changes in geology, geomorphology, vegetation and modification will create local discontinuities, the three dominant variables that determine local channel morphology and behaviour are: channel gradient, degree of channel confinement and catchment hydrology (Reinfelds *et al*., 2003). Using total stream power or specific stream power as a guide to channel morphology is relatively practical when combined with geographic information systems (GIS) and detailed digital elevation models (DEMs), and hydrological catchment estimates of discharge (at least in the UK). Data is still required on channel width and most applications of the stream power approach have relied on spot measurements of width from representative reaches (such as Lawler *et al*., 1999). Although cross-section surveys are not routine in monitoring, they may be available from river corridor surveys (RCS) and river habitat surveys (RHS) but are unlikely to be sufficiently regular.

The stream power approach, essentially a measure of discharge and slope, has been applied to the River Trent in the UK (Knighton, 1999). This approach has the potential to be used for climate change scenarios since the stream discharge variable can be altered, although this exercise has not yet been undertaken and may simply indicate locations where the potential for increased geomorphic activity is most likely or sensitivity is greatest. Some attempts to identify indices of sediment mobility and the likelihood of erosion or deposition based on excess stream power and stream power change have been applied in the catchment flood management plan (CFMP) for the River Calder in the UK (Environment Agency, 2004). These have been based on reach average conditions. The erosion indices do not match field evidence, but reaches were already heavily reinforced, which may indicate a response to erosion in the past. It is unclear whether further verification with field data has been undertaken.

Arguably these types of approaches may have more predictive potential and may even be useful in determining the potential for natural geomorphic processes in heavily modified or degraded reaches. However, questions must be raised about the usefulness of such approaches since modified river channels extend to two-thirds of all UK sites assessed by RHS (stratified random sample of 3.5 per cent of UK channels). Middle and lower reaches of rivers may be more intensely modified; for example, surveys of 40 km of main river channels in the Bassenthwaite catchment, English Lake District, found 85 per cent to be modified (Orr, 2003).

The principal benefit of process-based typologies is that they highlight the drivers of morphological variability. Where data is available, such analysis at catchment scale of, for example, stream power may help to guide sampling and reconnaissance surveys and thus would increase observation-only type classifications. However, system complexity (Philips, 2003), singularity (Schumm, 1991) and limited data have so far prevented detailed analysis of the sensitivity of British rivers to a range of different driving variables (Downs, 1995b). Data availability may improve with the advent of more readily available, remote-sensed, high resolution channel morphology data, although currently these remain limited or need considerable data extraction and analysis. More could potentially be determined from existing survey data and the need for a central data resource for geomorphological data has been recognised (Jim Walker personal communication, 2005).

Process-based approaches will primarily be useful in a GIS environment to define homogeneous reaches. Their use in more detailed assessments will depend on the availability of data on channel cross-section, incision and bed material size. Key variables are channel slope (approximate critical thresholds can be outlined); bed material size; channel width and depth (thresholds defined at critical width/depth ratios); degree of incision or confinement (channel to floodplain width threshold). For larger channels, the degree of channel activity can be determined from historic maps, which can also determine planform and styles of adjustment.

## 2.3 Published large-scale typologies

This section outlines some of the best known typologies that have been applied in some way to river management, where most of them rely to a greater or lesser extent on field reconnaissance survey data.

### 2.3.1 Rosgen

This system of classification of rivers based in North America has been widely applied to river management and has been used as a template for the redesign of rivers to restore them (Rosgen, 1994). According to Rosgen's definition, “a stream functioning best in its most probable state maintains its dimension, pattern, and profile over time, in the present climate, while moving the watershed's sediment and flow without aggrading or degrading. In a self-stabilized condition, bank erosion and deposition are balanced and the stream at bankfull discharge stays within stable ranges of channel geometry for that stream type.” Rosgen has made significant impacts on environmental stewardship in North America, having provided a user-friendly tool as an alternative to hard channel design. However, Rosgen’s typology has been criticised because it is not process-based and does not identify drivers of change in type, which may mean it lacks sensitivity to disturbances (Montgomery and Buffington, 1998).

Rosgen’s classification is based on dominant slope, cross-section geometry (width to depth ratio), planform pattern (including entrenchment) and predominant bed material size. North America boasts a large database of information on channel dimensions and sediment transport; however, this approach still relies heavily on the collection of field data.

### 2.3.2 Montgomery and Buffington (1998)

Montgomery and Buffington (1997) identified a continuum of channel landforms and processes that predominate in steep upland mountain streams (especially forested systems). In a review of classification typologies they state that few will be suitable for all purposes, but that a hierarchical system of classification enables assessment of interactions across spatial and temporal scales (Montgomery and Buffington, 1998). The authors propose a division into geomorphic province, catchment, segment and reach scales and present seven distinct channel types for mountain drainage basins. These are colluvial, bedrock and alluvial valley types based on valley fill, sediment transport capacity and sediment supply. The types are similar to others proposed for the Pacific North West USA (Frissel and Liss, 1986).

Montgomery and Buffington describe the range of bed mobility variation whereby stream beds are characteristically only active under specific flow ranges, either during extreme events or semi-continuously. This description includes the transition between sand and gravel beds, but also applies throughout the range of gravel to boulder sediment sizes. The types presented can be assessed according to the likelihood of change in some of their basic geometries (Montgomery and Buffington, 1998).

The benefit of this approach is that it focuses on the potential for change in response to different sediment and discharge regimes – in other words, there is plenty of process information for exploring climate or land use change responses in channels, although it is not clear that rates of change can be identified. The approach is dependent on a wide range of field data (variables required are slope, bed material size and valley width). Channel types are not necessarily applicable to other regions and the authors acknowledge the importance of local controls of historic geomorphic features and large woody debris. The typology may be hard to apply in modified reaches where bed material size is controlled by channelisation. In addition, this typology has not been assessed for its ecological relevance.

### 2.3.3 River Styles©

Brierley and Fryirs (2000; 2005) provide a comprehensive view of their River Styles© method whilst acknowledging the more regional relevance of the styles of rivers they describe. Australian landscapes are unique in many ways and prior to European settlement, it is thought rivers were more like a series of ponded reaches. Land use change has resulted in gullied streams, where restoration of these systems requires a clear understanding of how the current systems have evolved.

Despite being focused on Australian rivers, this approach has great potential for river management and has been used for problem-solving, application and participatory management. In essence, this approach can explore the potential for a different set of functions to those currently on offer in any river,; in other words, opportunities for restoration or rehabilitation. Geomorphic condition is assessed by comparing it with a reference condition or via a subjective assessment of channel geometry, planform and bed material character in order to assess the potential for restoration or rehabilitation. Geomorphic condition is thus a measure of how far a system or reach is from its geomorphological potential or naturalness (we might call this transience) and this indicates the potential for improvement in hydromorphological status. The framework has also been developed and applied in a participatory environment and has demonstrated its utility in catchment management in Australia.

The approach uses a framework (Figure 2.1) designed as a learning tool through which geomorphologists can summarise river character, behaviour, condition and recovery potential and convey these to a range of practitioners (Brierley and Fryirs, 2005). Local reach scale issues can also be explored, but its strength lies in using the framework in its entirety.

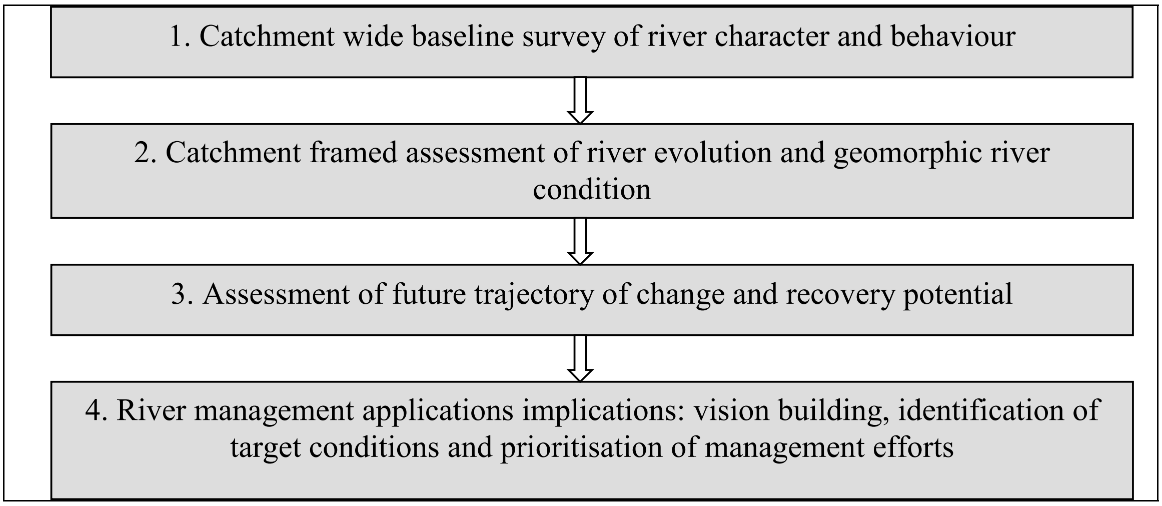


**Figure 2.1 Stages of the River Styles© framework (modified from Brierley and Fryirs, 2005)**

The framework relies on GIS interpretation of geology, soils, vegetation, climate and information on catchment history. Brierley and Fryirs use a nested hierarchical approach to defining catchment, sector, reach (style) and geomorphic unit. River styles are characterised by common combinations of geomorphic units, although individual units may not be unique to that style. River styles can be partly defined with this information together with longitudinal profiles and estimates of stream power. Verification of the allocated styles requires field reconnaissance surveys on bed material size and river behaviour.

Attempts have been made to link river styles with biological parameters (Thompson *et al*., 2004). Invertebrate assemblages were closely related to bed material size and hydraulic characteristics but were not significantly different between styles. The authors suggest that the styles do not take into account large scale drivers of local habitat variability and recommend incorporating stream size, temperature and hydrological regime.

### 2.3.4 Fluvial audit

Fluvial audit has evolved in the UK in a similar way to the Australian River Styles. A major difference is that it has not been applied in a nested hierarchical manner. Early definition of a fluvial audit approach was described by Sear *et al.* (1995 and Figure 2.2) and has been subsequently modified by others, such as Orr *et al*. (2004). This method is a morphological survey but in many of its applications, attempts have been made to incorporate adjustment-based classification and local channel types for local reach-scale management and strategic assessment. Such classifications are based on expert interpretation and are therefore qualitative and subjective. Use of such a system requires rigorous and consistent methods of stream reconnaissance and considerable experience and awareness of process-form linkages (Thorne, 1997).

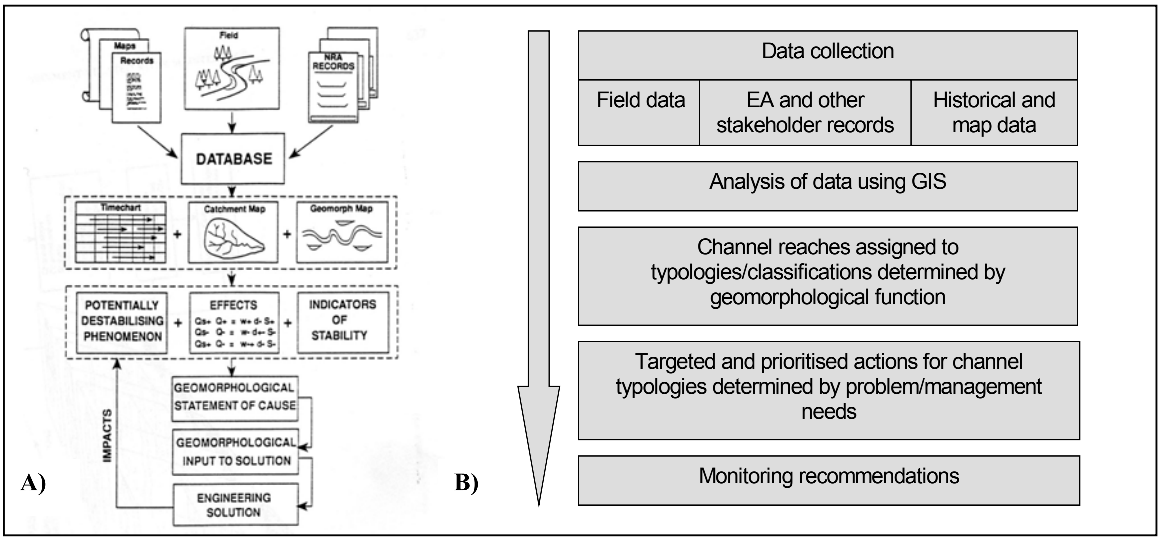


**Figure 2.2 Fluvial audit (A) after Sear *et al*., 1995 and (B) modified by Orr *et al*. (2004)**

This method has evolved primarily to provide river managers with information on the sedimentary system and its apparently chaotic behaviour. It has the advantage over other systems of having been widely applied and continually developed in the UK. Usually problem-driven, this method requires quantitative and qualitative data to be collected to answer specific questions. A common reason for using this approach is to provide enough information to identify critical processes at all points in a catchment and hence guide catchment-scale strategic solutions, but with sufficient detail for reach-scale operational needs. This basic gathering of information of geomorphological forms, history and landscape evolution is then subjected to expert interpretation. An explanation of geomorphological process is provided for individual reaches indicating likely trajectories of change - by necessity a subjective and semi-qualitative assessment. The spatial connectivity of the data is a key strength and the detailed data can be used to monitor change (for example, length of eroding bank). The derived site-specific typologies are process-based and, in general, are aimed at supporting the assessment of land drainage consent applications - the principal regulation of morphological change in England and Wales prior to the WFD.

Examples of the application of fluvial audit to UK catchments are generally contained in contract reports (for example, Newson and Orr, 2003; Orr *et al*., 2004; Geodata, 2001) although discussion of central data collation is ongoing. A review of fluvial audits in the UK to explore the range of sediment yields and controls on sediment delivery reveals that large-scale morphological change is rare and is usually a response to major land use changes. For example, rapid incision may be related to gravel extraction (such as Newson and Orr, 2003); planform change may be related to the stage of development of meanders and limited to larger floodplains; and aggradation may be linked to recent mobilisation of stored mine waste (Orr *et al*., 2001). Most of the problem-orientated investigations over the last decades relate to sediment transport and in particular to changes in the amount and rate of erosion and deposition. It seems likely that increased mobility of sediment is the most likely outcome of climate change in forthcoming decades. The mobilisation of mining spoil in the British uplands and the subsequent impacts on channels on the upland fringe are a notable driver for geomorphological investigations.

### 2.3.5 Broad-scale classification

Classifications using broad-scale information on geology, relief, hydrology and vegetation amongst others, have been used in Germany to establish characteristic regional stream types (Bostelmann *et al.*, 1998). More detailed classification of streams is based on channel flow capacity (bankfull discharge), longitudinal profile (determined by the presence of either riffle pool or step pool sequences) and stream planform. Stream types were found to be slope dependent around a threshold slope of 5 per cent. Reference sites for each stream type are determined in a similar way to RHS and these can be used to build a 'Leitbild' or vision of potential for the site. Such typologies have limited capacity to explore change and sensitivity to climate drivers. The use of bankfull discharge may have potential and as the authors point out, has been most often related to flow frequencies of one to two-year return frequency. However, it is likely that larger, more lowland rivers respond to more frequent formative events, whereas upland systems may respond to much rarer events. For example, the 2004 Boscastle floods had a one in four hundred year return period resulting in substantial mobilisation of tributary sediments, a channel incision of one to three metres, channel migration and large scale deposits of debris across the floodplain (www.environment-agency.gov.uk/commondata/103599/boscastle_findings_945477.doc).

## 2.4 Specific UK-based approaches

A range of fluvial geomorphological tools and techniques for assessing river characteristics have been developed and applied in the UK. These include classification systems (such as Downs, 1995a; Raven *et al.*, 1998), and a range of reconnaissance surveys and standardised approaches for incorporating geomorphology in river management, such as catchment baseline surveys and fluvial audits (for example, Environment Agency, 1998, Newson, 2002). Many of these have attempted to formalise experience-driven field observation, often resulting in qualitative and descriptive tools.

Classifications listed here have already generated – or are likely to generate - relatively large datasets on physical habitat data and geomorphology. Most of these are essentially pro forma-based surveys, useful for assessing habitat and physical wealth and quality.

In general, these approaches record the presence and absence of features within a limited survey area. These features may be colonised or used by biota that thrive within the physical conditions of the area. However, there is no explanation as to why these features exists, how transient or permanent they are, or how they may change under different scenarios. Spatial connectivity – the link between upstream and downstream activity - is also lacking in much of the data.

### 2.4.1 River corridor survey (habitat scale 500 m)

A fore-runner of RHS, river corridor surveys (RCS) can yield descriptive insights into channel form and features. However, RCS data has not been quality controlled to the same degree as RHS data and great care is needed during interpretation. These map-based surveys give some spatial continuity of information but are highly variable in quality. Although they have not been digitised, they nevertheless represent a useful source of historic information. RCS usually contain information on reach average channel dimensions and may cover an extensive area.

### 2.4.2 River habitat survey (habitat scale 500 m)

In England and Wales, river habitat surveys (RHS) have been conducted at 6,000 sites on a stratified random basis within 10 km grid squares. Future surveys may be stratified to ensure adequate sampling across river types rather than simply on a grid basis. The resultant database of sites has been used to compile indices of physical habitat quality and habitat modification which are based on divergence form semi natural reference conditions also found in the UK (Raven *et al*., 1998). The RHS approach collects information on physical form, flow types and features of interest including any modifications within the channel and immediate riparian area. The data essentially covers the presence or absence of the most important forms and structures, including extensive erosion, but it is not sufficiently detailed to monitor moderate changes in the extent of erosion or mobility of gravel deposits. An attempt to re-survey RHS sites following extensive flooding showed that change was difficult to detect (Defra and Environment Agency 2003).

RHS and the System for Evaluating Rivers for Conservation (SERCON) (biological habitat) can evaluate habitat quality scores but do not describe processes or their dominance and hence have no capacity for prediction. Newson *et al*. (1998) sought to derive a river channel typology based on RHS and other datasets and identified stream power as the most significant driving variable explaining the type boundaries. The objective was to explore the nature of changes in the environment against a composite axis of stream power and substrate size, which itself provides a measure of the sensitivity of environments to external environmental change, for example large-scale flooding.

### 2.4.3 GeoRHS (habitat scale 500 m)

The RHS method has recently been extended to collect physical information from the floodplain, again within 500 m reaches (DEFRA and Environment Agency, 2003). This method, known as GeoRHS, will extend habitat quality maps to include valuable floodplain habitats, but is likely to be limited as far as predictive capacity and dominant process identification is concerned. The survey is still in its development phase and has not yet been applied as extensively as RHS.

Thus, RHS and GeoRHS provide valuable, if somewhat indirect, information on channel physical habitat and process. Their strengths lie in the consistency and quality control of data recording. RHS have been conducted by trained surveyors who may have a range of disciplinary backgrounds. GeoRHS will need to be conducted by experienced geomorphologists. These surveys can be used to define relative or strategic habitat quality scores and habitat modification scores and potentially river habitat objectives.

## 2.5 Prospects for defining change in fluvial geomorphology

Geomorphological typologies offer the potential to explore the impacts of climate change in rivers, although classification should only be considered as part of a much larger assessment along with observation, laws, hypotheses, theories and models (Goodwin, 1999). Goodwin (1999) strongly recommends the use of typologies based on the dominant processes or controlling factors from which typical channel forms are derived rather than characteristic channel forms. An important message from this review is that river types and thus most typologies are likely to be locally or at least regionally-specific. However, modes and styles of adjustment and some threshold relationships are more broadly applicable. Understanding the physical principles that underpin fluvial geomorphology must be combined with an understanding of regional differences that drive rates of change and styles of adjustment; this probably requires regional research programs (Montgomery, 2001).

Change can be defined in different ways, such as the difference between two points in time or space, or as the potential for an effective change. The former represents variation that is tangible and detectable (but may not be significant); the latter is of interest in terms of site or system sensitivity and therefore has relevance to monitoring (Sear and Newson, 2003). Understanding trajectories of change, styles and rates of adjustment at individual reach scales and catchment scales may be sufficient to establish the significance of the change, which in turn can ensure that adaptation and mitigation are effective.

Typologies need to be able to characterise sensitivity to different kinds of adjustment in key locations within the catchment. Geomorphic debate has moved on from defining thresholds to recognising trajectories of change and styles of adjustment, born perhaps of a recognition that threshold conditions are rarely sharp and many are transitive. From a management perspective, the ability to detect and anticipate adjustment over decadal timescales may be sufficient. Sear and Newson (2003) suggest three critical types of change and recommend a national typology to guide monitoring. The changes are:

- large-scale synchronous changes in channel geomorphology;
- long-term, persistent changes in channel geomorphology;
- socially significant (risk-defined) changes in channel geomorphology,.

A database that provides information on geomorphic status is as vital as maps of water quality and biological condition. Furthermore, under the WFD the three criteria must be assessed together at all scales. Their interactions are by no means straightforward. Monitoring systems must be capable of detecting change and indicating trajectories of change since long time-series data invariably indicates that the longer we observe a population, the more likely it is to deviate from a preconceived baseline (Burt, 2003).

Debate on the relative importance of the magnitude and frequency of flood events and their impacts on geomorphology is ongoing. Changes in response to climate drivers may be temporary or permanent (Newson, 1992). Transitive or temporary changes could potentially increase sediment mobility largely in-channel; intransitive or permanent changes may lead to a new state from which recovery to the former state is unlikely (such as incision into a floodplain and creation of new terrace systems). Essentially, river systems are highly complex and there is not enough data to explore the relative importance of intrinsic channel adjustments, persistence of memory, response to external drivers and how systems evolve over time from the interaction of their constituent parts (Manson, 2001). Additional data will be required, along with advancements in modelling at the segment scale, to tie in broad-scale models (risk) to localised geomorphological responses.

No model can predict change as such, but some can provide enough information to determine which interpretation should be made. A good typology may then be used to determine the most probable state using some of the models of adjustment or potential within the defined reaches. Werritty (1997) describes geomorphic robustness, responsiveness and sensitivity, and notes that the critical issue is the balance between the size of the disturbance and the system’s ability to resist or accommodate the disturbance.

# 3 Hydrological typologies

## 3.1 Methods for defining flows

A number of terms are frequently used when determining objective-based environmental flow requirements in rivers. 'Compensation flows' are generally used when determining releases from reservoirs. 'Minimum acceptable flows' provide adequate protection for fish and dilution of effluent discharge. 'Ecologically acceptable flows' are flows required to sustain ecological populations. 'Hands-off flows' is a term generally used by the Environment Agency for limiting abstraction licences; that is, a flow at which no more abstraction may occur to preserve ecological integrity.

In 1888, the Halifax Corporation Waterworks Act set the precedent for releasing compensation water as a constant discharge to maintain the quality of a local beauty spot (Petts *et al*., 1995). This Act protected the environment from the deleterious effects of zero flows and initiated environmental awareness of the effects of different compensation policies. Other similar Acts such as the 1890 Bradford Waterworks Act (Gustard *et al*., 1987) followed. The specific needs of fisheries was first raised during the promotion of the Bill by the Corporation of Birmingham (1892) to construct reservoirs in the Elan and Claerwen valleys in Wales. In 1919, an Act to impound Haweswater in the Lake District stipulated for the first time the total quantity of water to be discharged each year, as well as making provisions for the daily compensation flow (Petts *et al*., 1995). However, infrastructure to establish the flows needed to maintain rivers was not in place until the Water Resources Act 1963. This Act created rights to impound and abstract water by licence from the river authorities, and Section 48 of the Act gave the river authorities (now the Environment Agency) the power to determine compensation flow requirements (Petts *et al*., 1995). The Act also introduced the concept of statutory minimum acceptable flow (MAF). However, at the time, although the negative effects of regulation on river ecology were recognised, understanding of the relationships between flow and ecological health was not well developed.

Quantitative instream flow methods can be divided into three main categories: historical flow methods, hydraulic methods and habitat methods. All three aim to maintain the stream environment, yet they focus on different aspects of the stream such as flow, wetted perimeter or physical habitat. Jowett (1997) argues that an instream flow policy requires clear and measurable goals, ideally defining the goal (such as retention of a resource or instream use), the extent to which this is to be achieved (in other words, the level of protection) and the criteria for evaluating its success.

### 3.1.1 Historical flow methods

As the name suggests, historical flow methods are based on the recorded or estimated flow regime of a river and traditionally, minimum acceptable flows have been set using such discharge-based methods. These express the instream flow as a hydrological statistic: commonly either as a flow duration statistic (such as the 95th percentile) or as a fixed percentage of the average daily flow (ADF) defined from a baseline period (Petts and Maddock, 1994).

### 3.1.2 Flow duration curves

Flow duration curves (Linsley and Franzini, 1964) are a fundamental tool used in water resource assessment. For example, indices of the flow duration curve such as Q95 (the flow achieved at least 95 per cent of the time) were traditionally used in hands-off approaches to abstraction licensing. However, flow duration curves do not capture the temporal sequencing of flows; consequently, it is unknown whether high or low flows occur consecutively. Equally, they do not show how quickly the flows change (Acreman, 2005). It is increasingly recognised that the magnitude, duration, frequency and timing of flow regimes of rivers are important to their ecology rather than just low flows; these, however, are not characterised by the flow duration curve (Poff *et al*., 1997; Richter *et al*., 1997).

### 3.1.3 The Tennant Method

The Tennant Method (1976), also known as the Montana Method, is the most widely known historical flow method and is the second most popular method in the USA. The method assumes that a percentage of the mean flow is needed to maintain a healthy stream environment. Using cross-section data from 11 streams across Montana, Nebraska and Wyoming, Tennant discovered that stream width, water velocity and water depth increased rapidly from zero flow to 10 per cent of the mean flow, while at flows exceeding 10 per cent the rate of increase declined. At less than 10 per cent of the mean flow, Tennant considered that water velocity and depth would be degraded and could only ensure the short-term survival of aquatic life. He considered that as a 'baseflow regime', satisfactory stream width, depth and velocity could be provided by 30 per cent of the mean flow. In his study, at 10 per cent of the mean flow, average depth was 0.3 m and velocity 0.25 m/s, Tennant considered such properties to be the lower limits for aquatic life. At 30 per cent of the mean flow or higher, average depths were 0.45 – 0.6 m and velocities 0.45 – 0.6 m/s, levels which he considered would be a good to optimum range for aquatic life (Jowett, 1997).

### 3.1.4 The discharge method

In the UK, two measures of the dry weather flow index (DWF) - the 95th percentile flow or the mean, annual, minimum seven-day flow frequency statistic - have been used most frequently to set prescribed flows (Petts and Maddock, 1994). Justification for a hydrological approach is that over the long term, stream flora and fauna have evolved to survive periodic flow adversities without major population changes. Systems in their natural state can balance stress with recovery mechanisms. However, any human impact that reduces the effectiveness of these recovery mechanisms will affect the level of flow required to meet the primary objective, namely to sustain and perpetuate indigenous aquatic fauna.

Petts and Maddock (1994) criticise the discharge method on two counts:

- there is no explicit consideration of habitat requirements;

- there is a complex array of processes, influenced by flow, that can affect biota.

The authors suggest that considering discharge in isolation of other water quality and geomorphological factors provides only a partial understanding of the potential impacts of flow regulation. Thus, attention should be directed towards a more scientific understanding of the response of species and biological communities to hydrological change (Petts and Maddock, 1994).

When setting flow targets other than downstream of a reservoir, users should pay due regard to:

- the natural flow duration curve, given that only a proportion of the available flow can be assigned;
- a more absolute flow requirement based on ecological needs, including geomorphological and species requirements.

For example, a river may be supported by high base flows and the flow could drop significantly whilst still providing good habitat in terms of wetted perimeter, depth and velocities for the species that live there. However, a different river may naturally have flows that are, at times, close to the limits for certain species and so there would be little scope for abstraction.

### 3.1.5 Hydraulic methods

Hydraulic methods relate various parameters of the hydraulic geometry of stream channels to discharge. Hydraulic geometry is based on surveyed cross-sections or hydraulic models, from which parameters such as channel width, depth, velocity and wetted perimeter are determined. Variation in hydraulic geometry with discharge can be established by measurements at different flows, predictions from cross-section data and stage-discharge rating curves, Manning's or Chezy's equations or calculation of water surface profiles.

Two criteria have been suggested for specifying minimum flow requirements using hydraulic methods. Wetted perimeter usually increases with flow, sometimes showing a point of inflection. Tennant (1976) used the inflection point criterion when he found that depth and width began to decline sharply at flows less than 10 per cent of the mean of his river studies. The other criterion, percentage habitat retention, retains a percentage of the width or wetted perimeter of the river at mean flow.

Whereas the wetted perimeter method has the advantage of being quick and inexpensive to apply, determining inflection points is difficult and subjective. Leathe and Nelson (1989) highlight wetted perimeter curves, which frequently have two or more inflection points. In such cases the upper inflection point is defined as the optimal habitat and the lower as the minimum acceptable flow.

#### 3.1.5.1 Indicators of hydrological alteration (IHA)

Hydrological variation is the major driving force within river ecosystems as it influences both biotic diversity and major environmental conditions (Poff and Ward, 1989; Sparks, 1995; Stanford *et al*., 1996). In recognition of the importance of hydrological variation, the US Nature Conservancy has developed indicators of hydrologic alteration (IHA) (Richter *et al*., 1996, 1997, 1998). Subsequently Richter *et al*. (1997) developed the range of variability approach (RVA), based on 32 hydrological parameters, to help river managers to define and adopt preliminary flow management targets in the absence of long-term ecosystem data. RVA considers relationships between characteristics of river flow and river habitat condition and addresses the critical role of hydrological variability in natural flow regimes. The technique therefore encompasses the magnitude, timing, frequency, duration and rate of change in stream flows assumed to sustain aquatic ecosystems.

The general approach for hydrologic assessment with the IHA has been to define a series of attributes, recognised as biologically relevant, which characterise intra-annual variation in flow conditions. An analysis of the inter-annual variation in these attributes forms the foundation for comparing hydrologic regimes before and after a system has been altered by anthropogenic activity (Richter *et al*., 1996). The IHA is essentially a technique developed to:

- statistically characterise the temporal variability in hydrologic regimes using statistical attributes assumed to be ecologically relevant;
- quantify hydrologic alterations associated with presumed perturbations (such as dam operations, flow diversion or major conversion of land uses in a catchment) by comparing the hydrologic regimes from ‘pre-impact’ and ‘post-impact’ time periods.

Five groups of characteristics have been used by the IHA developers to describe the flow history of a river (Table 2.1). These five groups are defined and characterised by 32 regime and hydrologic parameters (Richter *et al*., 1996, 1997). As such, the method is ideally suited as a tool to assess the impact of invasive vegetation on rivers and their flow characteristics. With the use of computer software developed specifically for the purpose, the user is able to compare flow before and after an impact has occurred, by analysis of pre- and post-impact flow records of a river. Analyses include text tables of the program output as well as graphs of all 32 of the indicators (shown in Table 3.1 below). The IHA method also allows for the determination of observable trends in historical flow records of a river. As well as defined baseline conditions, this allows for a rapid assessment of the flow records if there is no known development along the river. By performing a trend analysis, a change in stream flow characteristics over time may be identified.

**Table 3.1 The main groups of characteristics used in IHA to describe the flow dynamics of a river (after Richer *et al*., 1996)**

| *IHA statistics Group* | *Regime characteristics* | *Hydrologic parameters* |
|---|---|---|
| **Group 1**:Magnitude of monthly water conditions | Magnitude<br>Timing | Mean value for each calendar month |
| **Group 2**:Magnitude and duration of annual extreme water conditions | Magnitude<br>Duration | Annual minimum 1-day means<br>Annual minimum 3-day means<br>Annual minimum 7-day means<br>Annual minimum 30-day means<br>Annual minimum 90-day means<br>Annual maximum 1-day means<br>Annual maximum 3-day means<br>Annual maximum 7-day means<br>Annual maximum 30-day means<br>Annual maximum 90-day means |
| **Group 3**:Timing of annual extreme water conditions | Timing | Julian date of each annual 1-day maximum<br>Julian date of each annual 1-day minimum |
| **Group 4**:Frequency and duration of high and low pulses | Magnitude<br>Frequency<br>Duration | Number of high pulses each year<br>Number of low pulses each year<br>Mean duration of high pulses within each year<br>Mean duration of low pulses within each year |
| **Group 5**:Rate and frequency of water condition changes | Frequency<br>Rate of change | Means of all positive differences Between consecutive daily means.<br>Means of all negative differences Between consecutive daily means.<br>Number of rises<br>Number of falls |

The undeniable fact is that flow is *the* major determinant of physical habitat in streams (see for example Vannote *et al*. 1980; Junk *et al*., 1989; Padmore, 1998). The habitats delivered by the flow regime of a river in turn determine biotic composition and diversity. IHA Groups 1 and 4 (flow magnitude and duration) are concerned with lateral and longitudinal connectivity in the river; here it is clear that maintenance of such patterns are integral to the survival and success of species and communities in river environments. The monthly magnitudes forming Group 1 record the available habitat for aquatic organisms and the soil moisture availability for plants. Furthermore, this group also covers the availability of water for terrestrial animals, as well as the water temperature and oxygen levels of the stream. IHA Group 2 (the timing of occurrence of flows) corresponds with Bunn and Arthington's (2002) 'First Principle' – that the movement of water across the fluvial landscape determines the ecology of a river over a varied range of spatial and temporal timescales (Figure 2.1). The result of the complex interaction of flows and local geology and landforms is a mosaic-like distribution of habitats and communities of varying composition, abundance and biodiversity (such as Poff and Allan, 1995). Altering the magnitude and duration of the annual extreme conditions represented by Group 2 will have a direct effect on the ecosystem of the river. By changing these flows, the creation of sites for plant colonisation is affected along with the structuring of river channel morphology and physical habitat conditions. This is consistent with South African observations (Heritage *et al*., 1996).

IHA Groups 3 and 5 (frequency of occurrence and rate of change of flows) reflect the fact that all aquatic and riparian organisms have evolved life strategies in direct response to the natural flow regime of the river and the timing of flows. Patterns of spawning, germination and recruitment are affected by, and in many cases depend on, rates of water level fluctuation and disturbance (in other words, floods and low flows) and changes in flow velocity and shear stress (Large and Prach, 1999). By changing the frequency and duration of the high and low pulses in the river, nutrient and organic matter exchanges between the river and the floodplain will be affected. In South Africa, the importance of the first "freshes" of the rainfall season is well known (Weeks *et al.*, 1996). This also has an influence on the bedload transport and the texture of the channel sediment. Parameters representing these characteristics are seen in Group 4. The default high pulse level is the 75th percentile of all pre-impact daily flows, and the low pulse level is the 25th percentile. Finally, the rate and frequency of hydrograph changes, represented by Group 5, influence the amount of drought stress on plants (Richter *et al*., 1996; Mackenzie *et al*., 2003).

Some researchers have queried whether the link between the IHA and ecological functioning has been adequately tested, particularly in areas outside of the USA where the method was developed. The general absence of adequate ecosystem, climatic and runoff data in South Africa led Taylor *et al*. (2003) to apply the RVA to set preliminary flow management thresholds for the Mkomazi River. Such an approach allows flexibility and adaptability when further ecological data becomes available. However, the RVA may be open to criticism, given that reliance on the 25th to 75th percentile range of the hydrological parameters used in the analysis has not been tested statistically. In addition, there are limited statistical analyses on the link between flow and organisms. Regardless, in the case of the ecological reserve for the Mkomazi River, identifying critical variations in the magnitude, timing, frequency, duration and rate of change of flows offers a feasible and practical method. In a recent comprehensive review of many statistical approaches to characterising flow regime, Olden and Poff (2003) have concluded that the IHA provides a powerful tool for the "calculation of high information, non-redundant indices describing the major components of the flow regime".

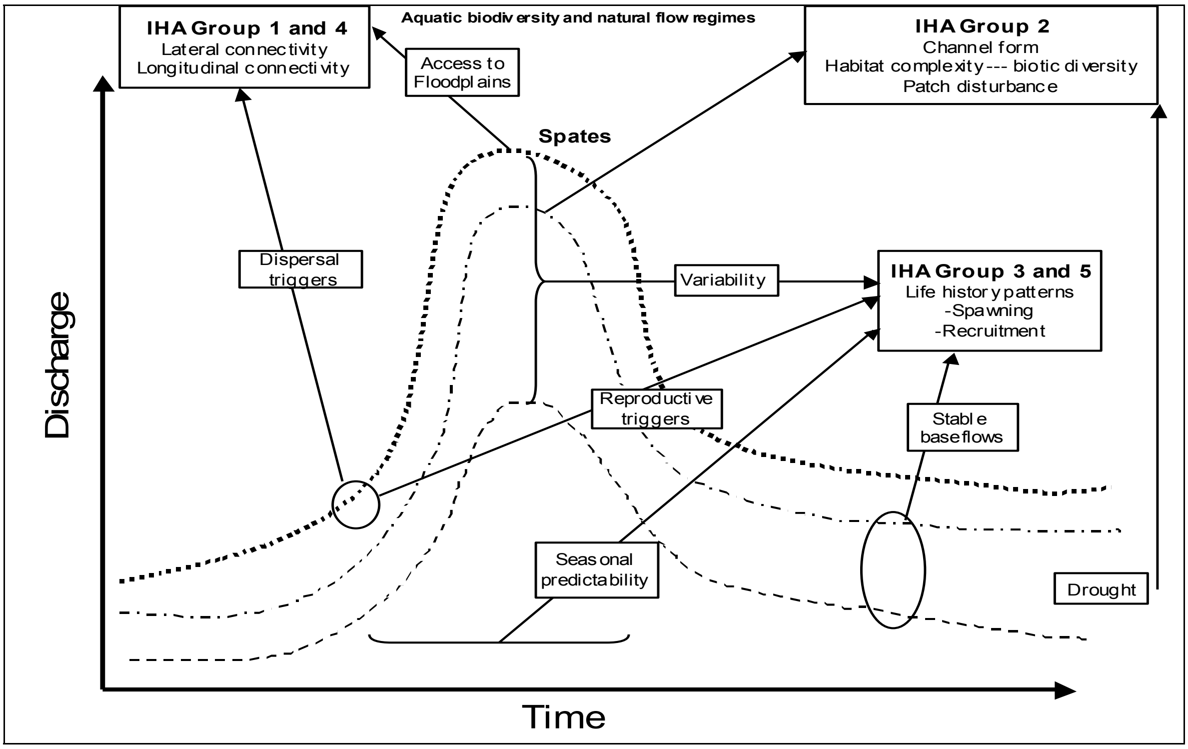


**Figure 3.1 The flood pulse plays a vital role in the natural flow regime of a river that encourages aquatic biodiversity by a number of interrelated mechanisms. Here, high flow events shape the relationship between channel form, physical structure and biodiversity, while droughts and low flows limit habitat availability. Seasonality, predictability and timing influence life history and dispersal patterns as well as connectivity. (After Bunn and Arthington, 2002)**

Olden and Poff (2003) suggested that only a subset of the IHAs should be used in any analyses and that these should always focus on a particular ecological question. This becomes especially important when assessing potential impacts of future climate change on river system structure and function.

### 3.1.6 Habitat methods

Habitat methods are a natural extension of hydraulic methods. The difference is that the assessment of flow requirements is based on hydraulic conditions that meet specific biological requirements rather than the hydraulic requirements themselves. Hydraulic models predict water depth and velocity throughout a reach. These are then compared with habitat suitability criteria to determine the area of suitable habitat for the target aquatic species. When this is done for a range of flows, it is possible to see how the area of suitable habitat changes with flow (Jowett, 1997).

Habitat methods are considered more reliable then other methods because they are quantitative and based on biological principles. Habitat methods were first used in the assessment of flow suitability for spawning salmon (McKinley, 1957 in Jowett, 1997), but since then have been applied to most biological and recreational instream uses (e.g. Collings, 1972; Waters, 1976; White, 1976 in Jowett, 1997). The instream flow incremental methodology (IFIM) was developed by the Aquatic Systems Branch of the US Fish and Wildlife Service to consider ecological demands when recommendations for flow regimes are determined (Bovee, 1982). IFIM relates changes in the extent of habitats available to aquatic species to changes in discharge. This allows ecological demands to be expressed in the same terms as other water resource demands.

#### 3.1.6.1 PHABSIM model

The Physical HABitat SIMulation system (PHABSIM) is a hydro-ecological model designed to assess the impact of changing flow regimes on physical instream habitat. PHABSIM was developed by the US Fish and wildlife Service and has been used throughout the USA since the 1970s (Milhous *et al*., 1984; Jowett, 1997). The PHABSIM model may also be used to assess the impact of changes in channel morphology, such as those arising from flood defence, habitat improvement schemes or future scenarios arising from climate change. Because changes in flow will alter physical habitat in virtually any river, PHABSIM is a valuable tool for water resources investigations (Dunbar *et al*. 1997) and may also be used for simulations of water quality, water temperature or any other characteristic features which could influence habitat. The PHABSIM system simulates the relationship between stream flow and available physical habitat, defined by depth, velocity, substrate and cover. For each life stage of the target species, the model requires expressions of the relative suitability for that species of the full range of values for these variables. These univariate curves are called habitat suitability indices; they may be derived from existing literature, expert opinion or by sampling techniques such as electro-fishing (Bullock *et al*., 1991; Petts and Maddock, 1994; Jowett, 1997).

PHABSIM contains a number of hydraulic models that predict values of depth and velocity for different simulated discharges. These models require calibration using field data collected at two or more calibration discharges. Observations of substrate and cover are recorded using a coding system and are assumed to be independent of discharge. Once calibrated, the model can simulate values of microhabitat variables over the full range of discharge within a river reach. Combining the results with habitat suitability data produces the weighted useable area versus discharge relationship (Bullock *et al*., 1991; Petts and Maddock, 1994; Jowett, 1997).

Simulated values of microhabitat variables (from the calibrated hydraulic model) are combined with habitat preference data for each target species for each stage of life. Combining this with a time series of historical flows yields a time series of available physical habitat for each life stage of the target species. Using PHABSIM it is possible to simulate habitat curves relating to season and complete life cycles of target species. The method is thought to be expensive and it is difficult to generate habitat preference curves, as they need ideally to have been derived from the stream being investigated (Bullock *et al*., 1991; Petts and Maddock, 1994; Jowett, 1997). PHABSIM also assumes static morphology and hence has a very limited ability to detect change.

The Institute of Hydrology carried out the first trials of the technique in the UK (Bullock *et al*., 1991). The Environment Agency has subsequently carried out a range of applied studies (Spence and Hickley, 2000) including the alleviation of low flows (caused by surface and groundwater abstractions). Other Environment Agency studies have covered licensing (determining optimum flow regimes to set restrictions on new abstractions) and drought management (investigating the impact of temporary changes in allowable abstraction rates or reservoir compensation releases). Gibbins and Acornley (2000) used PHABSIM to investigate the impact of releases from Kielder Reservoir on the habitat of Atlantic salmon and brown trout in the River North Tyne and Tyne, North East England. However, others have criticised PHABSIM (see section 3.2).

### 3.1.6.2 Stewart's Method

It could be argued that Stewart's Method includes aspects of all three methods of defining flows, given that it links fish count numbers to flow using a hydraulic parameter (width). Stewart (1969) used data on fish movement and flow data from over 14,000 fish in rivers in North West England to derive an empirical method for setting flow targets for salmon migration and angling, based on the discharge per unit width (q).

The targets, which apply to adult fish, are as follows:

| | |
|---|---|
| Survival flow: | 0.03 cumecs per metre width |
| Start of migration: | 0.08 cumecs per metre width |
| Peak intensity of migration: | 0.20 cumecs per metre width |
| Angling: | 0.29 cumecs per metre width |

### 3.1.6.3 Equivalent flows project

Stewart's targets were investigated in the recent equivalent flows project, along with other hydraulic variables such as flow depth and flow velocity (see Gill *et al*., 2004). The aim of this project was to determine if minimum equivalent flows for the ecological requirements of salmonids could be developed for rivers within Cumbria and South West Scotland. The approach used a 1-D steady state uniform flow hydraulic model to estimate equivalent flows at the sites where electro-fishing surveys were available during low flow conditions.

The equivalent flow results were compared with salmon densities from the electro-fishing sites and minimum equivalent flow requirements were defined from the results with flow data provided by Low Flows 2000 (see Section 3.2.7). Results from the hydraulic model (that is, generated parameters of discharge per metre width, flow depth and flow velocity) were analysed to determine which range of parameter values were most suitable to salmon populations (in other words, electro-fishing sites where salmon population densities were classified as good or excellent). Discharge per metre width was found to be a useful hydraulic indicator because it provided the most consistent correlation with salmon parr densities. In addition to being based solely on width, this parameter does not need to take account of channel slope and roughness compared with parameters such as flow depth and velocity. Discharge per metre width is an attractive and easily applied means of standardising flows.

Equivalent flow requirements were used to create GIS maps of hydraulic suitability throughout the study catchments for salmonids. Agreement between these maps with measured salmon densities was generally good. This research combined a variety of datasets and techniques to develop a method for determining a suitable equivalent flow parameter for Atlantic salmon, with transferability to a range of upland rivers. Such a method may be used to set minimum acceptable flows for catchment abstraction management and other flow objectives. Mapping of the hydraulic suitability of reaches using GIS may also be used to indicate where salmon could be present in unsurveyed areas. Conversely, the absence of salmon in reaches with good hydraulic suitability could be used to guide salmon habitat improvement. The project is now being continued in Cumbria and South West Scotland in further phases studying (a) trout and sea trout and (b) bullheads and crayfish. Future developments in Low Flows 2000 may enable flows expected under future climate change to be investigated. A current alternative using a modelling approach described in the case study below could be employed.

### 3.1.6.4 Case study: Potential impacts of climate change on Atlantic salmon

The UKCIP02 climate change scenarios suggest that by the 2080s, the UK climate will become warmer (an overall increase of 2.5 to 3$^{o}$C), with temperature increases being greater in the summer and autumn compared to spring and winter seasons. In terms of precipitation, winters are expected to become wetter and summers drier throughout the UK. Such changes will inevitably affect river flow regimes and the ecological populations they sustain.

Stewart's approach has also been applied in a recent study based in the Eden catchment (CHASM catchment) in Cumbria, where the aim was to assess the impacts of climate change on Atlantic salmon (Walsh, 2004). The approach involved developing a comprehensive hydrological model of the catchment using the SHETRAN modelling system. This model was used to reconstruct an hourly time series of flows from 1992 to 1999 at a one-kilometre scale. Predictions of future changes in precipitation and potential evaporation (based on the UKCIP02 medium-high scenario for 2070-2100) were applied to the model inputs to estimate how future flows in the catchment may be affected by predicted changes in the climate. Flows provided information for hydraulic analysis across the catchment, to determine flow depths, flow velocities, discharge per metre width and Froude numbers for both current and future climates. This was also done with a 1-D steady state uniform flow hydraulic model. Hydraulic parameters were then compared with those cited in the literature as being suitable for salmonid habitat and survival. Using the time series of flows, analysis determined at what percentage of the time such parameters were met; this was also repeated using the predicted time series of flows under the future climate. The final part of this work produced catchment maps of hydraulic parameters to highlight areas most sensitive to future changes. Figure 3.2 shows where in the catchment the current $Q_{50}$ flow provides a suitable depth for salmonid spawning, and also where such areas would be reduced following a 30 per cent reduction in the $Q_{50}$ flow.

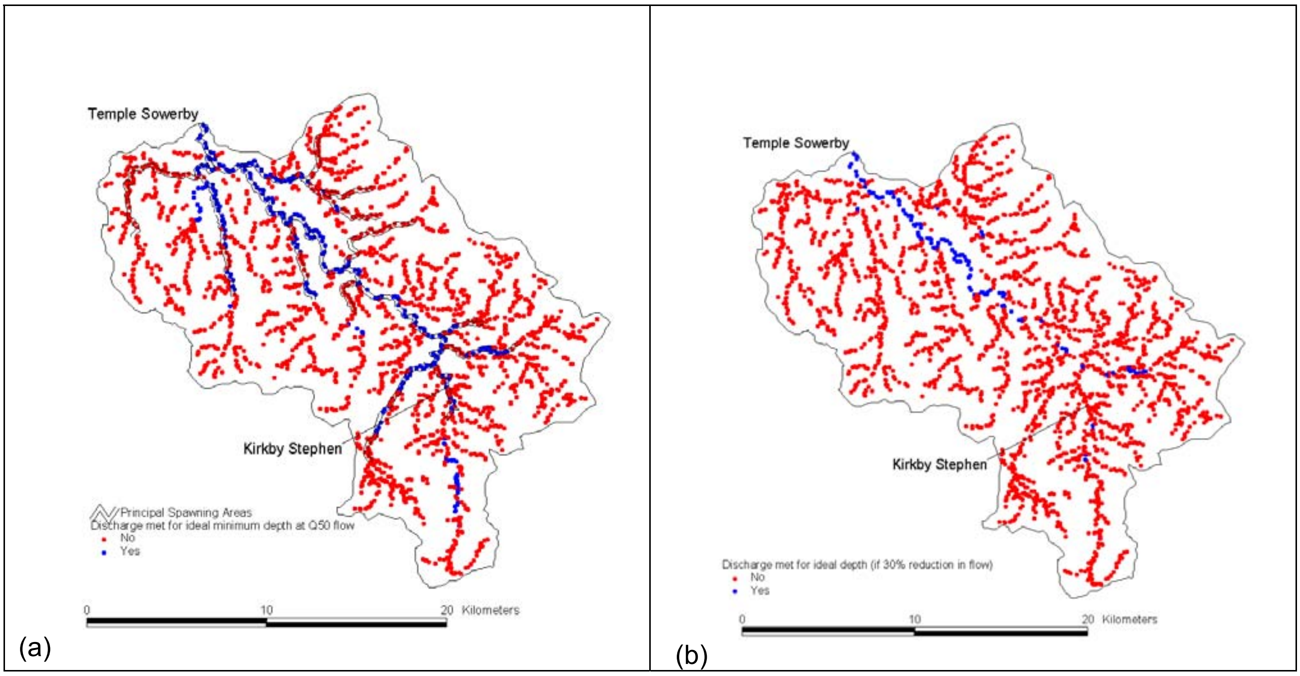


**Figure 3.2 Distribution of vertices at which the discharge is met for an ideal minimum flow depth of 0.17 m for salmonid spawning at the (a) $Q_{50}$ flow and (b) where there is a 30 per cent reduction in $Q_{50}$ flow**

Links between hydrology, geomorphology and ecology are not fully understood under current climate conditions, let alone future climate change. This study does not claim to account for all the potential impacts of climate change on Atlantic salmon, a species protected by the Habitats Directive. Nor does it consider the complex interactions of biotic and abiotic conditions affecting each stage of the salmon's life cycle. However, the study does focus on the main driver of climate change - the flow regime - and provides a conceptual framework that could incorporate other important factors such as channel morphology. It can, therefore, offer a guide for fisheries scientists, water resource managers and supply operators to describe how resources in the Eden catchment may change and how such changes may relate to hydraulic conditions suitable for Atlantic salmon. The GIS approach to extrapolating results can identify suitable spawning sites in terms of flow characteristics. If none were present, this would indicate where management strategies could be used to encourage spawning or highlight the need to investigate and improve potential limiting factors such as substrate or water quality. GIS also readily classifies catchment areas most vulnerable to climate change in terms of catchment flows. These features will inevitably assist classifications required by the WFD.

An obvious extension of this work would be to develop a more sophisticated 2-D river network model to incorporate velocity and depth profiles and geomorphological features such as pool-riffle sequences; this would represent the river environment more accurately.

### 3.1.7 Current techniques within the UK

#### 3.1.7.1 CAMS and RAM

The Environment Agency has developed a number of initiatives to help implement the WFD, including the catchment abstraction management strategy (CAMS). This is a sustainable, catchment-specific approach to water resource management aiming to balance human and environmental water requirements, both in the present and future. The CAMS process aims to make information on water resources and abstraction licensing on a catchment scale readily available to the public, as well as providing a more consistent and structured approach to resource management (Environment Agency, 2001). The resource assessment and management framework (RAM) is central to CAMS, providing a consistent approach to water resource assessment by quantifying both the natural availability of water and the current level of water use within a catchment (Environment Agency, 2002). River flow objectives for catchments are developed from the sensitivity of the riverine environment to changes in the flow regime. Flow duration curves are used to identify portions of the flow regime available for abstraction. However, flow duration curves have limitations, as discussed in Section 3.1.2. Comparing river flow objectives with the impacts of water use and abstraction on the natural flow regime can indicate the resource status of the catchment, along with assessing the effect of abstraction licensing (Holmes *et al*., 2005).

#### 3.1.7.2 Low Flows 2000

Low Flows 2000 is being used to support the CAMS process and the early stages of the RAM process. The Low Flows 2000 system is a decision support tool designed to estimate river flows at ungauged sites and to aid the development of catchment and regional water resources assessment. Scientific techniques for flow estimation and the development of the software system were undertaken as a project jointly funded by the Centre for Ecology and Hydrology in Wallingford (formerly the Institute of Hydrology) and the Environment Agency.

The variability of river flows depends both on climatic variables such as rainfall, temperature and evaporation, and catchment-scale characteristics such as hydrogeology. Trends for these variables are observed for the UK, standardised by their mean flow to remove the impact of scale. Regionalised models employed within the system are based on the region of influence, where a region is constructed from the similarity of gauged catchments to the ungauged one. This is essentially assessed using the distribution of hydrology of soil types (HOST) classes (see Boorman *et al*., 1995), which is used as a surrogate for hydrogeology. Flow statistics for the ungauged basin are then calculated as a weighted average of observed flow duration curves from ten catchments making up the region.

A national programme is underway to validate the natural flow estimates produced by Low Flows 2000 against available data sources, and to include data that quantifies the impact of artificial influences within CAMS catchments. By developing the system, for example by including planned water quality modelling options, Low Flows 2000 has the potential to be a fully integrated water-environment decision support system (Holmes *et al*., 2005).

Low Flows 2000 was used in the equivalent flows project described above, and was found to be appropriate for use in surface water-dominated catchments. Some sites in this study were small catchments. A number of continuous flow gauges exist for large catchments which can be used for validation; however, there are only a few for smaller catchments (less than $20km^2$). A validation exercise was carried out in small catchments within Cumbria, where extracted Low Flows 2000 data from such sites was found to adequately reproduce gauged data (Gill *et al.*, 2004).

**3.1.7.3 RAPHSA**

The ongoing project RAPHSA (Rapid Assessment of the Physical Habitat Sensitivity to Abstraction) will develop a tool to consistently define the physical sensitivity of rivers to abstraction. The tool will complement two existing methods: the default method within the RAM framework, which classifies rivers from photographs of typical channel forms; and the PHABSIM system. It will operate at the catchment scale on the Low Flows 2000 platform and will colour-code any reach of river channel according to its time-varying sensitivity to abstraction.

# 4 Biological typologies

## 4.1 Evolution of approaches

Naiman *et al*. (1992), in discussing general principles of classification and how this relates to river conservation, conclude that river classification is in a formative stage. They attribute this to three facts: that rivers have only relatively recently become recognised as systems in their own right (Vannote *et al*. 1980); that dynamic changes occur over broad spatial and temporal scales; and that classification systems only reflect the current state of knowledge on fluvial function (Frissel *et al*. 1986). Of these three constraints, the latter is perhaps the most relevant to this project. Table 4.1 summarises the main developments in classification for biological conservation and management over the last decades and highlights the shift from early attempts at classifying whole rivers to the hierarchical approaches of today.

**Table 4.1 Evolution of the classification and assessment of rivers (adapted from Naiman *et al*. 1992)**

| Classification type | Examples |
|---|---|
| **Historical concepts** | |
| Whole river schemes | Davis (1890); Shelford (1911); Illies (1961); Bailey (1978) |
| Drainage measures | Horton (1945); Strahler (1957) |
| Biotic zonation – fish | Carpenter (1928); Huet (1954); Karr (1981); Schlosser (1987) |
| Biotic zonation – invertebrates | Macan (1961); Illies and Botosaneanu (1963); BMWP and variants; Vannote *et al*. 1980); Wright *et al*. (1989) |
| Biotic zonation – plants | Harris (1988); Holmes (1983); Holmes and Newbold (1984) |
| Biotic zonation – abiotic factors | Huet (1954); Leopold and Wolman (1957); Hawkes (1975) |
| Landscape ecology | Decamps (1984); Ward and Stanford (1983, 1987, 1995a) |
| **Recent concepts** | |
| Ecoregion concept | Rohm *et al*. (1987); Cupp (1989) |
| Hierarchical classification | Warren (1979); Brussock *et al*. (1985); Rosgen (1985); Frissel *et al*. (1986); Briggs *et al*. (1990); Wadeson (1994); Rountree et al. (2000); van Coller et al. (2000) |
| Linking geomorphology and ecology | Bisson *et al*. (1988); Morin and Naiman (1990); van Coller et al. (1997) |

There are many classification schemes for assessing biological potential, but with consensus on the fundamental attributes of an enduring classification system which relate to (Naiman *et al*., 1992):

- the ability to encompass broad spatial and temporal scales
- the ability to integrate structural and functional characteristics under various disturbance regimes
- the ability to convey information about underlying mechanisms controlling instream features
- the potential for achieving this at low cost and with uniform managerial understanding.

It is worth repeating here that humankind is potentially entering a period of rapid climate change, coupled with the fact that the surface of the planet increasingly bears the signs of human activity. There is a growing acceptance of the need for adaptive tools aimed at sensitive management change under a range of different scenarios. While no single scheme achieves all of this, hierarchical schemes offer the best way forward.

## 4.2 Early schemes

### 4.2.1 Biological Monitoring Working Party (BMWP) Score

CIES (2004) provide an excellent explanation of how the BMWP method originated. The BMWP was set up in 1976 by the Department of the Environment (now the Department for Environment, Food and Rural Affairs or Defra) to recommend a biological classification system for use in national river pollution surveys. The BMWP initially had the following objectives:

- To recommend a biological classification of river water quality for use in river pollution surveys;
- To consider ways and means of implementing classifications;
- To consider relationships, if any, between chemical and biological classifications.

The Working Party initially decided not to investigate correlations of chemical and biological assessments. The interim report of the BMWP was published by the Department of the Environment in 1976. This recommended, by a majority decision, the development of a score system based on benthic macro-invertebrates. Trial use of the recommended methods by the water industry led to major changes in the proposed procedures prior to the presentation of the final report in 1978.

These changes included:

- Reduction in the level of taxonomic identification required;
- Removal of the proposed fauna abundance ratings;
- Reduction of river types to eroding or depositing habitat types only;
- Recognition of the lack of a standardised sampling procedure;
- Allocation of family scores based on their most pollution tolerant species.

In the 1978 report, separate scores were allocated for eroding and depositing zones, which ranged from one for pollution tolerant Oligochaeta (worms) to 100 for the most pollution sensitive families.

The final report was never officially published and, following further trials by the water industry, more changes were made prior to the use of the system for national river surveys. These involved:

- Combining eroding and depositing habitat types into a single type;
- Reducing family scores to a range of one to ten to minimise the final score.

A weakness of the BMWP system, in common with many other score systems, is the effect of sampling effort. Under most circumstances, a prolonged sampling period will produce a higher final score than a sample taken quickly. To overcome this inherent weakness, it became common practice to calculate the average score per taxa (ASPT) by dividing the BMWP Score by the number of taxa. The inclusion of the ASPT in reporting the 1990 National Biological Survey made possible the reappraisal of scores carried out by Walley and Hawkes (1996, 1997). This work showed the significant effects of site type on the score, thus reinforcing the original BMWP's use of different scores for eroding and depositing substrates. The original BMWP Scores and the revised BMWP Scores obtained from this include, for all families where there were sufficient specimens, a habitat-specific score for riffles (less than or equal to 70 per cent boulders and pebbles), pools (less than or equal to 70 per cent sand and silt) and riffle/pools (neither riffle nor pool).

## 4.3 Generic process-based typologies – overseas approaches

### 4.3.1 Indices of biological integrity (IBI)

‘Systems theory’ states that a change in one parameter of a system will lead to a change in most (or all) of the others (Barrow, 1995). In the context of river systems, longitudinal succession and processes such as nutrient spiralling and energy flows mean that impacts made at point sources may affect other elements, functions and processes at different spatial scales (Fisher, 1983). The river system must be seen as a self-maintaining unit, whose parts interact both internally and externally (Slocombe, 1998). Attempting to solve a problem (such as riverine ecosystem sustainability) by looking at only a few of the parts (in other words, only the physical and chemical) can cause additional problems for other interrelated factors (Clayton and Radcliffe, 1996).

For the past few decades, practices and policies of river conservation management have been based upon river water quality data - physical and chemical analyses of the water body. However, these strategies have proved incapable of slowing the degradation of the world’s riverine environment. In recognition of this, heightened public, political and scientific interest in river conservation management has spurred the idea of treating the river as a whole system rather than a mere body of water - a goal of ecological integrity. Francis *et al*. (1993) define ecological integrity as “a river system containing a full complement of native species, processes and structures; and a high quality of water and air". As such, ecological integrity is analogous to 'good ecological status' defined by the WFD.

Attaining both conditions requires the incorporation of biological factors into river system monitoring. A system suffering from a lack of ecological integrity would incur diversity losses, ecosystem function impairment and structural degradation. Karr (1992a) states that ecological integrity is where a system's inherent potential is realised, its condition is stable, its capacity for self-repair is reserved and there is minimal need for external support. Such conditions are at the core of current calls for sustainable development – in other words, an ecologically sustainable state of the river system. Scientists, politicians and the public have called for for a broader and more sustainable approach to protecting biodiversity, given that a "systematic reduction in the capacity of the Earth to support living systems" has been identified (Karr, 1992b). The ecological integrity ethic introduces a broader perspective of the conservation biology movement, recognising that a failure to protect this integrity threatens the health of human society through loss of environmental sustainability (Karr, 1992a).

The past few decades have seen an increasing awareness of humankind's unsustainable impact on nature - our inexorable move towards "global deterioration" (Rees and Wackernagel, 1996). As a result, concern has arisen over the management of the Earth's ecosystems. For riverine ecosystems, there are currently many different conservation strategies underway involving chemical and physical measures of water quality. More recently, however, the river as an ecosystem has become more of a concern and the use of ecological integrity to guide sustainable management is being incorporated into legislation (Karr *et al*., 2000). The current challenge is to promote the use of integrity as opposed to quality in river conservation and management in the UK (see Section 5).

In the US, Canada and France, the concept of environmental integrity has been built into action plans and legislation. The UK, however, continues to focus on water quality as the basis for assessment and action. The underlying assumption is that water bodies are both resilient and resistant to a degree of change (physical or chemical), returning to natural or sustainable states through self-purification and quasi-equilibria of river systems. There are thresholds, however, above which river water and its course may be altered irreversibly without conservation management intervention (Dobbs and Zabel, 1994).

The quality of water is measured through physical, chemical and a limited number of biological factors. As water quality does not address the system as a whole, cumulative effects and anthropogenic impacts upon ecosystem functions and processes may be overlooked (Table 4.2):

**Table 4.2 Advantages and disadvantages of using ecological integrity in river conservation management (from Dionne and Karr, 1991; Keddy *et al*, 1993; Karr, 1992a; Haider and Steadman, 1993; Hilborn *et al*. 1993)**

| Advantages | Disadvantages |
|---|---|
| Quantifiable – can be modelled, which improves theoretical foundations, aids in guidance of future research and strengthens theoretical basis of extrapolation. | Subjective – models are only as good as the foundations upon which they are developed. Depends upon full understanding of dynamics of an ecosystem, and must be geographically generalised. |
| Flexible – availability, type and quality of scientific information is constantly changing; ecological integrity may change with the conditions | The criteria used in IBI are not substantial enough; there is a (costly and time-consuming) need to develop and test a more comprehensive list. |
| Provides a reference point as an ultimate aim for the current system status, and enables identification and correction of factors responsible for degradation. | Rarely find a totally natural river for reference, so a 'sustainable' state is aimed for, but who defines the suitability of this state? (again subjective).<br>There is a large degree of natural variation, which must be taken into account when comparing streams. |
| Combines views of all affected parties (public, scientists and politicians) | Public, scientists and politicians rarely agree |
| Can detect degradation that generic methods (chemical measures) cannot, as it is directly related to protecting environmental processes and ecosystem. | Does not replace chemical, physical and toxicity testing, but adds to it, thereby increasing conservation management costs. |
| Integrates and evaluates the full range of impacts on biotic systems. IBI may assess the degree and type of degradation present, as opposed to chemical, physical and toxicological methods which can only measure above or below some threshold | Methodological bias may be present, depending upon the particular expertise. |
| Due to long life cycles of index organisms of the IBI, a more integrative view of cumulative impacts may be explored. Chemical samples represent one specific place for only a very short period of time. | The 'lifespan' of a politician is around five years; even though long life cycles of indices aid integrative exploration of impacts, policies made may be abolished within short periods. |
| Sensitive, so responds quickly to stresses and, through ambient biological monitoring, provides a direct evaluation of the conditions of the water resource. | |
| Integrates cumulative impacts from point sources, non-point sources and flow alteration. | Impacts have to be researched.<br>Systems are not fully understood, which provides an excuse for not implementing management strategies. |

Under the demands of the WFD, the UK is following the US and Canada in introducing ecological integrity (carrying forward Figure 1.1) into river conservation management in order to create a more sustainable situation for UK's rivers.

The problems with this approach include the inability of physical, chemical and a small number of biological measurements to represent the state of the ecosystem as a whole. Hidden impacts of humans are not adequately assessed via current water quality measurement techniques. Ecological processes and functions may also be overlooked despite being at the core of the riverine ecosystem and in need of conservation management.

### 4.3.2 Instream flow requirements (IFR)

There is growing concern within South Africa that the country's rivers are deteriorating (King and Louw, 1998). The major cause of this deterioration is the escalating demand for water from a rapidly growing population, with the overall result being large-scale direct abstraction from rivers and an extensive programme of dam building. Over the last two decades, the policy of the South African government via the Department of Water Affairs and Forestry (DWAF) has shifted from provision of water in response to demand to one of holistic management of the nation's water resources (King and Louw, 1998). There has been innovative thinking on, and research into, water quality management (DWAF, 1991), water for the environment and the management of low flows to address a number of issues – water quality problems, rural supply and riverine ecosystems (Water Research Commission, 1993). Throughout this period, there has been growing recognition that the riverine environment is not a user of water in competition with other users, but is the source itself. At the same time, there is an urgent obligation to meet the needs of a significant proportion of the population (12 million out of 45 million according King and Louw, 1998) who do not have adequate access to potable water. It is inevitable that the need for more water for human consumption will be in conflict with the desire to maintain or improve the condition of South Africa's rivers.

Initial assessments in the 1990s employed the US Instream Flow Incremental Methodology (IFIM: Stalnaker *et al*., 1994). However, it was soon realised (King and Tharme, 1994) that IFIM could not provide a suitable typing methodology for South African rivers and that the traditional IFIM approach required more data and resources than were available. The consensus was that the IFIM emphasis on target species was inappropriate for a nation where the accent was on management of the *whole* ecosystem (including the riparian system) rather than on fish and other aquatic species being the most important. The IFIM output was also found to be less than the recommended modified flow regime required for whole-river management (King and Louw, 1998). Attention shifted towards developing a local method that could rapidly assess instream flow requirements - this approach is termed the building block methodology (BBM).

### 4.3.3 Building block methodology (BBM)

The basic concept of the BBM is simple: some flows within the total flow regime are more important than others for maintaining the structure and/or function of the river ecosystem. These flows can be described using the following parameters: timing, duration and magnitude. The concept is linked to other schemes under development and testing (such as IHAs (Section 3.1.5): Richter *et al*. 1996, 1998; Taylor *et al*. 2003).

In South Africa, the method is based on best available knowledge and expert opinion, which recommends flow regimes likely to help maintain the river in some pre-determined desired state (King and Louw, 1998).

A number of assumptions underpin the method (King and Louw, 1998):

- Biota within a river can cope with naturally occurring low-flow conditions and may also be reliant on higher-flow conditions at certain times. No matter how extreme, variable or unpredictable they may be, flows that are a normal characteristic of a river are those to which biota are adapted and on which they may be reliant.

- Flows that are not characteristic of the river constitute a disturbance which may fundamentally change the river's character.

- Identifying the most natural components of the flow regime and incorporating them into any managerially modified flow regime will help maintain the natural biota and functioning of the river system.

- Some types of flow influence channel morphology more than others. Again, following the previous recommendation will help maintain the channel structure and diversity of physical biotopes (Section 3.4).

Similar interactions have been highlighted by Bunn and Arthington (2002)– see Section 4.1.5.

Recommended flows should be identified and magnitudes, timing and duration decided upon by expert opinion. Prioritised variables are usually the degree of perrenniality, the magnitude of base flows in dry and wet seasons, magnitude, timing and duration of floods in the wet season and small pulses of higher flows that occur in drier months.

In terms of climate influences, changes in flow regimes almost always result in long-term changes in river ecosystems. The potential for long-term change is recognised in the BBM and dealt with in two ways:

- Channel-flushing flows included in the IFR are designed to help maintain channel form and biotope diversity
- Post-impact monitoring of the river will guide adjustments to the IFR where necessary.

The approach is designed to assess flow requirements at the ecosystem level and can be used where data and time are limited. King and Louw (1998) suggest the following improvements which could be made:

- modelling of hydraulic links at the biotope level
- more substantial identification of habitat-discharge relationships
- more structured methods for assessing the need for flushing flows and flows to maintain riparian biota.

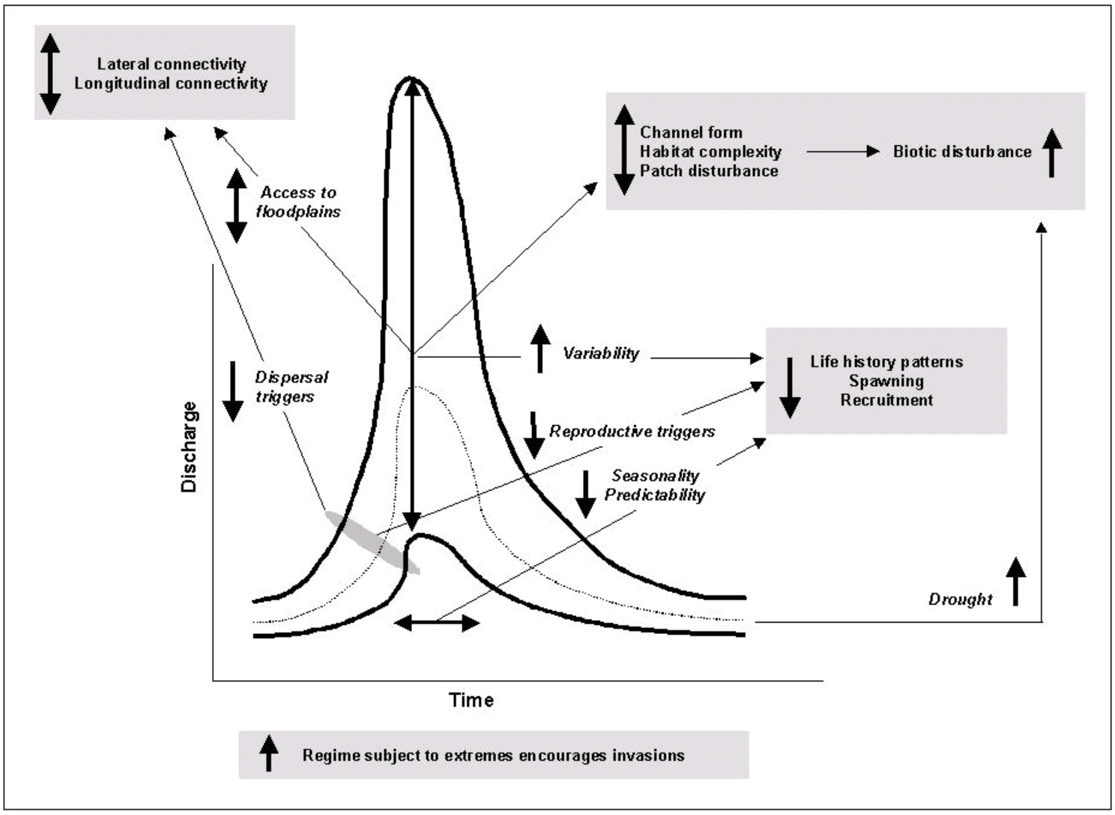


**Figure 4.1 Under UKCIP02 climate change scenarios, river flows will be subject to greater extremes (spates and low flows). The previously interrelated mechanisms encouraging aquatic biodiversity now operate over different spatial and temporal scales. The relationship between channel form, physical structure and biodiversity is adversely affected, while extended low flows limit habitat availability. Disrupted patterns of seasonality, predictability and timing adversely affect life histories, dispersal patterns and system connectivity. (After Bunn & Arthington, 2002)**

### 4.3.4 Water resources classification system for South Africa

The South African water resources classification system provides the framework within which water resources management can achieve equitable, optimum and sustainable use of the water resource. It allows the custodian of the water resource, in consultation with affected parties, to find the appropriate balance between protection and use for different water resources to ensure national uniformity. The system outlines the characteristics of the water resource as well as source-directed controls required to secure these characteristics.

Three resource classes are proposed (Table 4.3):

**Table 4.3 Characteristics of proposed water resources classification system for South Africa**

| CLASS | Characteristics |
|---|---|
| CLASS I | This class focuses on ecological protection. Management would aim to maintain aquatic ecosystems in a natural or near-natural state, which will also secure the quality requirements for basic human needs with minimal treatment (except in those systems where natural water quality is unacceptable). |
| CLASS II | This class aims to balance use and protection of the resource. Management would aim for optimal use of the resource while still ensuring healthy (although slightly modified) aquatic ecosystems. The quality requirements for basic human needs would be met assuming standard treatment, and management would strive to meet the water quality requirements of irrigation and recreational users. |
| CLASS III | This class covers economically beneficial uses of the resource, while still ensuring sustainable use. Management would aim to meet the water quality requirements for all users, although advanced treatment options may be required in some cases. |

Underlying this approach is the concept of the ‘reserve’. The reserve represents that quantity and quality of water that is required to meet basic human needs and to ensure basic functioning of aquatic ecosystems. The basic human needs reserve is the minimum water quantity and quality required for basic sanitation and drinking purposes. It is linked to the population dependent on the resource (in quantity terms) and may increase over time. The ecological reserve is the minimum quantity and quality of water required to ensure aquatic ecosystems do not lose their ability to recover from impacts, and is a property of the particular resource.

### 4.3.5 River health programme

The South African river health programme (RHP) primarily makes use of biological indicators (such as fish communities, riparian vegetation, aquatic invertebrate fauna) to assess the condition of river systems. The rationale for using biological monitoring is that the integrity of biota inhabiting river ecosystems provides a direct, holistic and integrated measure of the health of the river as a whole. The RHP aims to serve as a source of information on the ecological state of river ecosystems in South Africa, to support the management of these resources.

The objectives of the RHP are to:

- Measure, assess and report on the ecological state of aquatic ecosystems;
- Detect and report on spatial and temporal trends in the ecological state of aquatic ecosystems;
- Identify and report on emerging problems regarding aquatic ecosystems;
- Ensure that all reports provide useful information for national aquatic ecosystem management (http://www.csir.co.za/rhp/goal.html)

The RHP advocates a phased approach to monitoring involving a design framework, conceptual development of the programme within the framework and finally, small-scale implementation and 'anchoring' the RHP within management institutions. Formulating a design framework requires the involvement of local resource managers and scientists as well as international benchmarking. This exercise enables programme objectives to be set along with specifications to guide the remaining design phases.

Conceptual development of the programme involves selecting and/or developing technical protocols, such as the selection of monitoring sites, ecological indices and monitoring frequency and the creation of systems to manage data and information. Small-scale implementation tests the programme to ensure it provides a substantial broadening of conventional water quality monitoring along with 'state-of-environment' reporting. The availability of information on ecological reference conditions and the present ecological state of a river helps the process of establishing an ecological reserve for rivers. Finally, there is a need to ensure that the RHP becomes part of the relevant water management institutions in terms of expertise, skills and budgets. The overall goal of the anchoring phase is to guide agencies through the different steps of implementing the programme whilst internalising it in their organisations.

### 4.3.6 Rapid bioassessment protocols

The United States Environmental Protection Agency (USEPA) has developed rapid bioassessment protocols (RBP) that use fish, macroinvertebrates or periphyton to assess stream condition. Metrics representing structural, functional and process elements of the biotic community are calculated for each site and aggregated into an index. This multimetric index represents the biological condition of a site (Barbour *et al*., 1999). Physical and chemical data are also measured at each site and are used to aid the interpretation and calibration of the index as well as defining the reference condition. It is beyond the scope of this document to consider the process of biological metric calculation and calibration. Rather, the focus will be on physical and chemical measurements collected alongside the biota. In particular, the RBP includes a rapid habitat assessment method that uses a scoring system to rate habitat condition (referred to in the literature as HABSCORE).

### 4.3.7 European aquatic monitoring network (EAMN)

Increasing concern in Europe over the human impact on the flora and fauna of the continent's rivers has produced a strong demand for operational tools and assessment frameworks. Research into physical riverine habitat assessment methods in Europe is somewhat fragmented (EAMN, 2004), with overlap and redundancy in some fields and significant gaps in others. The integrated development and management of water resources within Europe mandated by the WFD requires harmonised and comparable monitoring, physical quality assessment protocols, modelling techniques and analysis tools and systems, all oriented from the reach towards the catchment scale.

The European aquatic modelling network (EAMN) established in 2000 aims to help project participants develop new models and methods through a combination of nationally-funded research and international networks. Information is disseminated to end users such as environmental agencies, national institutions and authorities, regulatory bodies and the water and hydropower industry. The main objective is to develop integrated methods and models to assess interactions between aquatic flora and fauna and riverine habitats on the reach scale, transferable to a catchment scale. The first objective has been to define state-of-the-art methods and modelling of riverine habitats and to provide a framework to measure the effect of human influences on aquatic ecosystems. Work is ongoing in this area.

# 5 Integrative approaches

## 5.1 Climate change and water resources

The last two decades have seen a shift from a predominantly protectionist stance to one advocating the sustainable use of resources. This has partly been driven by a growing realisation that climate change represents the greatest long-term threat to the environment and to water resources in particular (see Arnell, 2004). To that end, a number of international conventions now place obligations on signatory governments to manage their resources in a sustainable manner consistent with these principles.

Andreasen *et al.* (2001) recommends a useful definition of sustainability as “maintaining ecosystems and all of their components and processes in a condition such that they continue to provide all of the goods and services that they are capable of providing”. The US, Canada and France have addressed these issues. Environmental integrity is used by the USEPA as a synonym for environmental quality and is seen as a key concept of natural resource management and environmental protection, and as the core of the Clean Water Act. Ecological integrity is now recognised in legislation in thirty-five states of North America, many provinces of Canada (Karr *et al*., 2000), and France in Europe.

Elsewhere, South Africa’s current water resources policy is based on the principles of sustainability, equity and optimal use. As such, it specifically promotes the equitable and economically beneficial use of water. However, policy also recognises that if this goes unchecked, and if the use of water resources extends beyond their ability to recover, sustainable use of the water resource will not be possible. The policy secures sustainability by means of resource-directed measures (RDM) which ensure that resources do not degrade beyond their ability to recover from impacts.

As with many water-stressed nations, South African policies also recognise that some water resources require more protection due to their international, national, provincial and/or local importance. In these cases, RDM must ensure negligible risks or changes to natural functioning of resources. In other cases, the economic benefits to water use warrant a less protectionist stance. However, for the most part water resources need to be maintained in a state which meets the needs of most water users, but which also allows for healthy functioning of aquatic ecosystems and optimal use of the resource (eco-hydrology, Zalewski, 2000, 2002). Policy therefore rests on balancing optimal use of water resources with the needs of the resource.

## 5.2 The need for flexible monitoring and management

The Environment Agency continues to focus upon ‘water quality’ rather than the ‘quality of water resources’, with a host of statutory water quality objectives (www.ukbap.org.uk) addressing nutrient stripping, but with no mention of the ecological integrity essential to sustainable river conservation management. With the advent of WFD in 2005, it is now time for the UK and the rest of Europe to adopt this concept.

innovative approaches are needed to determine how ‘good ecological status’ (as defined in the WFD) may alter with climate change,.  The immediate solution could be to bring in best practice or other schemes from overseas and amalgamate these with flexible and predictive typologies already used in the UK. A broad-based, ecologically-sound, multi-parameter approach is necessary for water resource conservation and management. Biotic integrity should be combined with physical and chemical measurements to identify and manage the degradation of fluvial ecosystems. The method shown to be most effective has been the IBI and there have been a number of attempts to quantify biological integrity through ambient biological monitoring. Karr (1990) defines biological integrity as “the capability of supporting and maintaining a balanced, integrated, adaptive community of organisms having a species composition and functional organisation comparable to that of the natural habitat of the region”.

Under current projections of climate change for the UK, it will be necessary to move away from simplistic approaches to classifying, monitoring and managing the UK’s running water resources towards more predictive ones.  The main focus of attention to date has been on the meso-scale level of analysis. Sommer *et al*. (2004), for example, highlight the deficit of tools for intermediate scale (1-100 km) reaches (Figure 5.1).  While the river continuum concept  (Vannote *et al*., 1980) remains the foundation for much of the current understanding about river and stream structure and function (Sommer *et al*., 2004), there has been insufficient work (Fausch *et al*., 2002) at spatial and temporal scales for the major management decisions needed to address climatically-driven change in river systems.

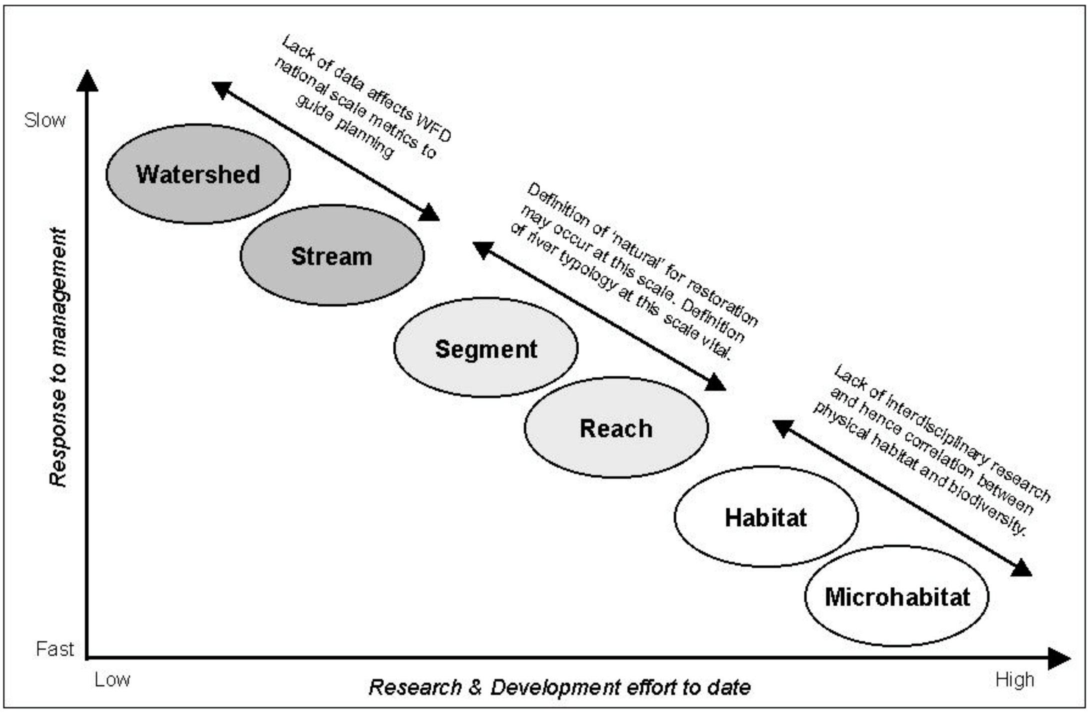


**Figure 5.1 Scale issues regarding existing knowledge of fluvial system sensitivity (from Newson and Large, in press)**

The reach (geomorphologically defined) remains the primary scale at which to identify geomorphological controls on physical habitat quality and quantity (Newson and Newson, 2000). The RHS database may be of considerable value to geomorphologists and should be explored in much greater depth than has been possible to date (Newson *et al*., 1998). Using current knowledge of the flow conditions bespoken by patterns and sequences of biotopes, geomorphologists should implement the following steps to ensure the biotope approach provides an alternative to hydraulic models:

- develop an empirical channel typology or taxonomy to allow rapid characterisation of reaches in the field under a range of climate change scenarios;
- study the changing control exercised by morphological features on biotope patterns at varying flows in a range of channel planforms (the work reported from North East England has been conducted on single-thread, low-sinuosity channels);
- further validate the hydraulic variability prevalent in hydraulically rough channels and in flows controlled by instream vegetation;
- develop a more sophisticated approach for the space/time patterns demonstrated by physical biotopes, for example using the concepts of landscape ecology and the techniques of GIS.

## 5.3 Tools for detecting change

### 5.3.1 Nested systems and linkages

Noss (1990) shows that biodiversity *per se* is currently a relatively minor consideration in environmental policy, having been regarded as too broad and vague a concept to be applied to real-world regulatory and management problems. Yet biodiversity, as it is presently understood, encompasses multiple levels of biological organisation (Figure 5.2 below). Noss (1990) proposed a nested hierarchical approach for environmental monitoring – regional landscape, community/ecosystem, population/species, and genes. Ward *et al*. (1999) further developed the idea for floodplain rivers, emphasising that understanding of the factors that drive diversity patterns of local species requires knowledge of processes that determine species richness at the regional level and the rates of species turnover in a region. Ward *et al.* (1999) suggest that species richness will be at its maximum at some intermediate level of connectivity, a hypothesis consistent with the intermediate disturbance hypothesis of Connell (1978), although data is still needed to support or refute this contention.

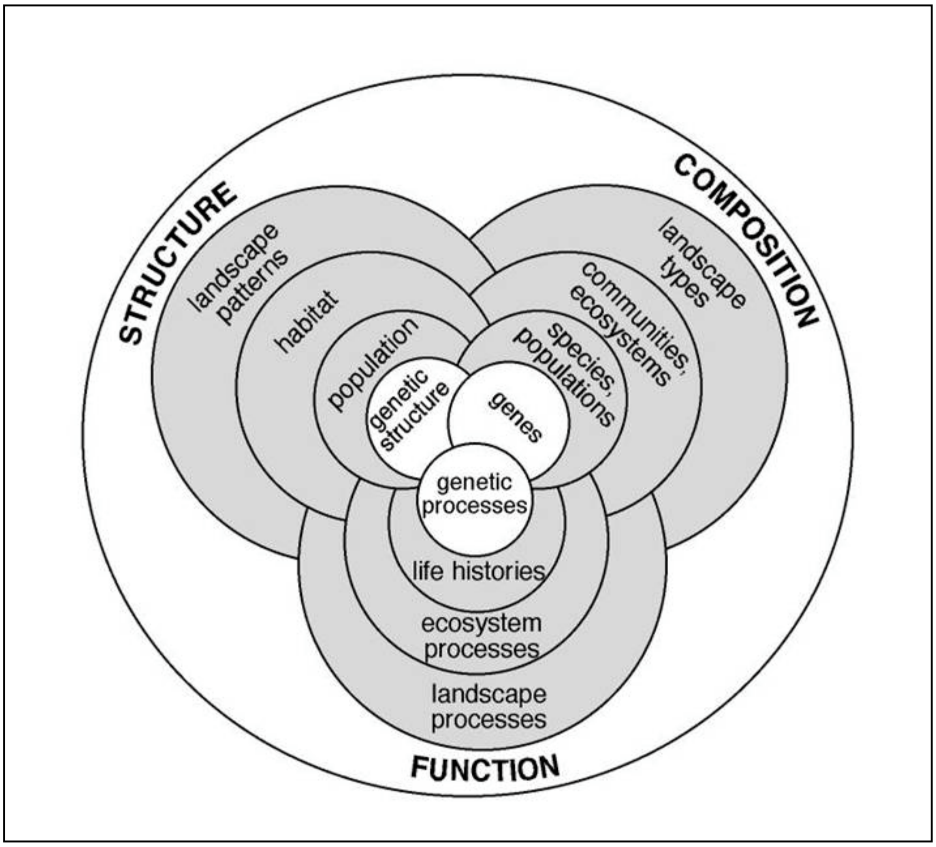


**Figure 5.2 Compositional, structural and functional biodiversity, shown as interconnected spheres, each encompassing multiple levels of organisation (after Noss, 1990)**

This implies that maintaining a diversity of disturbance regimes is of major importance in maximising biodiversity across a floodplain river (in reality, these systems have undergone increasing fragmentation over the last two centuries). Under scenarios of climate change, increasing efforts will have to be made to ensure that fragmentation and isolation do not increase, consistent with the WFD requirement for no deterioration in ecological status. However, the following challenges remain in both variability and scale:

- While the hydrologic regime determines the structure and function of river systems (Junk *et al.*, 1989), detailed descriptions of characteristics (surface area, depth, residence time, velocity) are often lacking at ecologically-relevant spatial and temporal scales (Wiens, 2002)
- Rivers and streams are unique, patchy, discontinuous and strongly hierarchical systems (Hynes, 1975; Poole, 2002)
- Ecological studies measuring concurrent responses of multiple trophic levels to flow in river systems are rare (Sommer *et al.*, 2004)
- Modelling studies of large river-food web dynamics have relied upon generalized hydrologic patterns (Power *et al*. 1995a, 1995b).

### 5.3.2 Potential application in the UK

UK rivers have been shown to adjust to climatic drivers in a number of ways, and have been shown to be quite robust (Section 4.1) except in some cases where they have been heavily modified. However, there is not normally sufficient data to explore complex responses, behaviour and feedbacks or to monitor and manage for change. Research on the scaling of biodiversity from sampling plots and ecosystems to landscapes or regions should go beyond quantifying the diversity of components at different scales to provide ecologically-relevant interpretation. Moreover, there is little point in setting up monitoring schemes if it is not known how to relate observed changes to ecological processes and their drivers. There is a lack of long-term monitoring data and it is thus difficult to differentiate between population fluctuations and real trends. In addition, the usefulness of methods or indicators depends on the considered timescales. While grid data may be useful to monitor species richness over centuries, abundance-based measures should be used for shorter timescales.

Concerns about ecosystem health have previously driven the agenda for river restoration, yet adaptation to climate change may simply be a case of identifying where rivers could be more robust. This offers a best practice definition of good hydromorphological status.

In other countries it has proved easier to establish typologies for river adjustment rather than channel form. An example is the River Styles typology developed in Australia, where 'natural' is defined as a channel that is naturally adjusted, working within a range of variability set by the river style and the catchment context (Brierley and Fryirs, 2000).

In discussing the development of a channel typology to support WFD implementation in Scottish rivers, Grieg (2004) suggests adopting an approach based on the River Styles typology (Brierley and Fryirs, 2000) and the system put forward by Montgomery and Buffington (1993). One drawback of this, in the context of the Environment Agency's responsibilities in England and Wales, is the relative lack of montane reaches (Montgomery and Buffington's work was developed on mountain channels). A similar problem may be encountered when applying Rosgen's scheme, due to the inherent complexity of Britain's fluvial landscape with its variety of controls on channel morphology and response.

### 5.3.3 Trajectories of change

In the UK, does the Environment Agency possess the tools necessary to detect the effects of climate change? What are the gaps in our knowledge base and can they be readily filled? The Environment Agency must adapt to the requirements of the WFD by:

- identifying trajectories of change;
- developing mitigating measures to protect biodiversity and a range of other ecosystem services;
- devising appropriate POMs

Scale issues are central to any assessment of the risks posed by climate change and their effects on achieving WFD objectives. In terms of system sensitivity and response to external drivers, response will be more rapid at the reach scale and below, but there is uncertainty regarding the system components' resilience to change.

Under natural conditions, biotic and abiotic attributes react to system variability and in many cases depend on it. Under climate change scenarios however, it is anticipated that there will be different trajectories of change at different scales in the system (influenced by resilience and natural lags – or resistance – in the system). To assess these trajectories, flexible tools will be needed to:

- parallel change trajectories (Figure 5.3)
- map common links between different catchments, such as surface water-dominated versus groundwater-dominated
- address uncertainties across a range of scales – from grossing up to the scale of the River Basin District to the many uncertainties that remain for system function at the micro-scale (Figure 5.4)

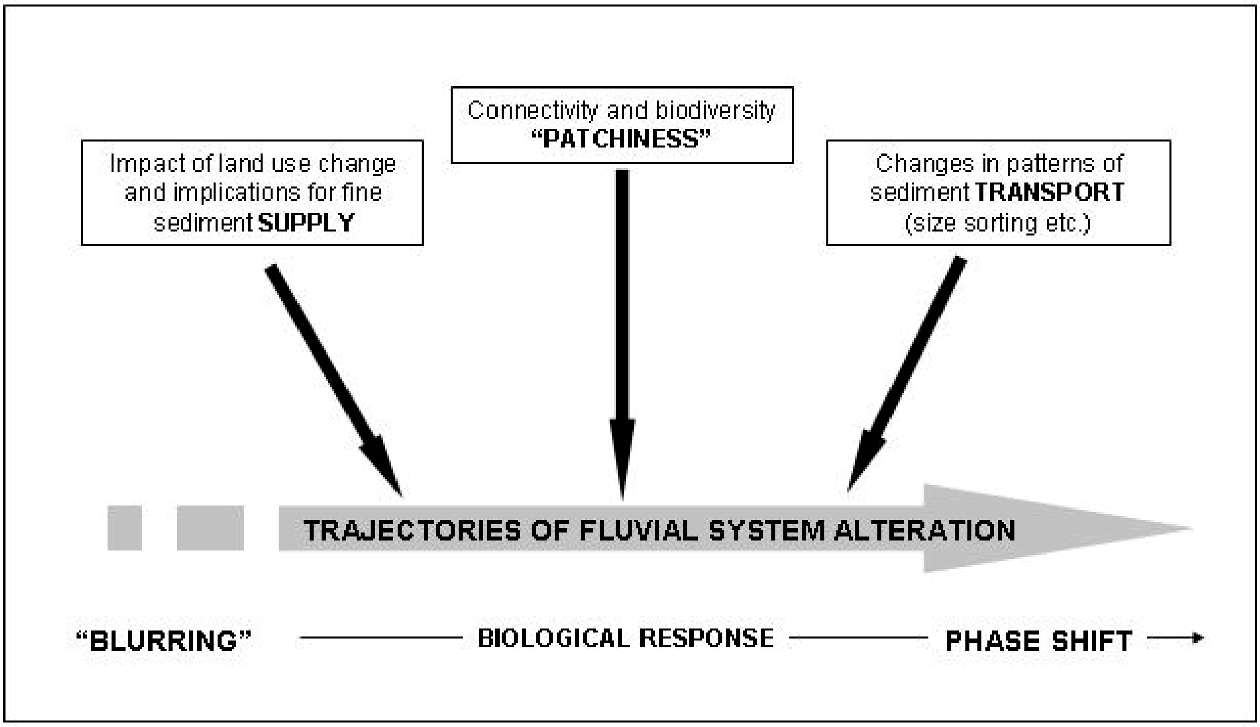


**Figure 5.3 Potential influences of climate change on fluvial system sediment supply and trajectories of change. Sediment transport, not planform change, will be the first impact under climate change**

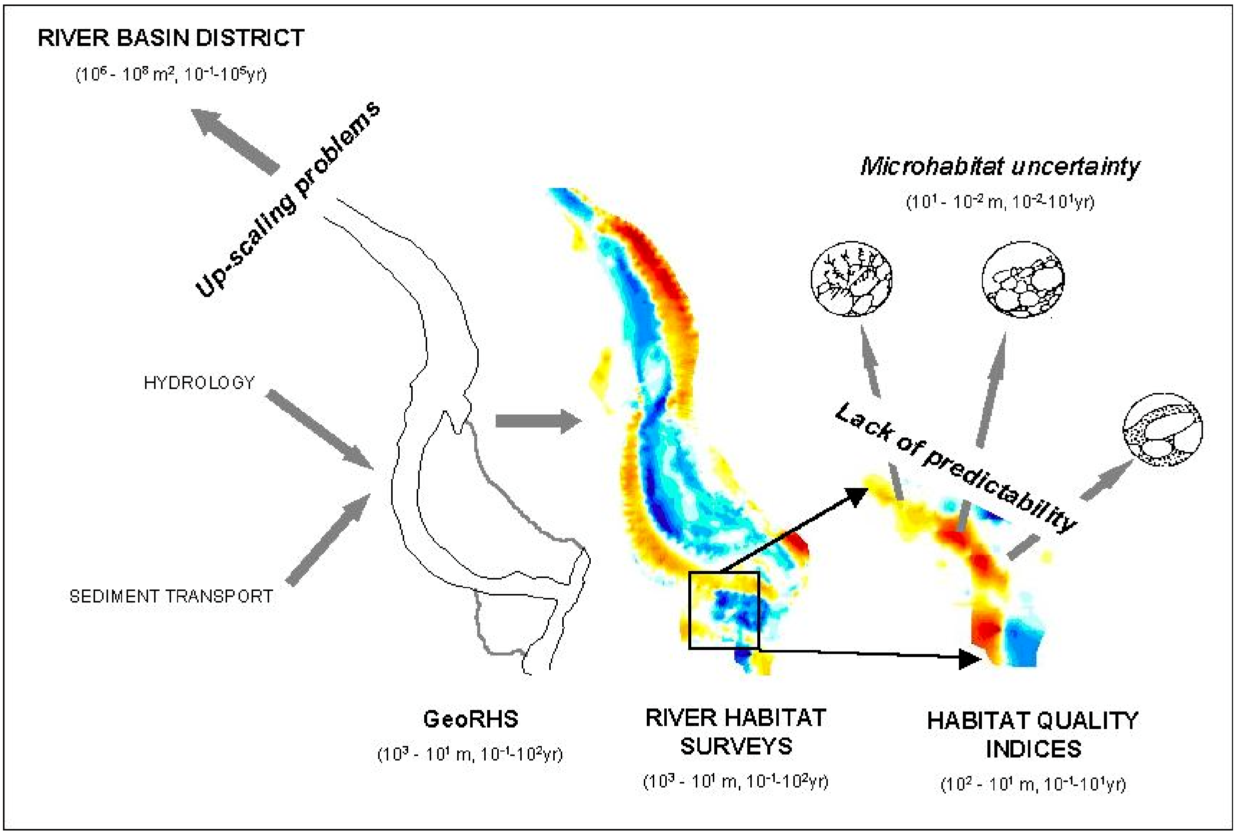


**Figure 5.4 The relationship between system sensitivity and response to external drivers (from Newson and Large, in press)**

Thus far, there has been little application of process-based geomorphology in the UK; most geomorphological studies are topographically biased. "Form is the new process" (Newson, M.D. personal communication, 2005), in that the study of form is replacing studies of process, largely as a result of economies of scale. There is little doubt that in the UK, despite the significant number of geomorphological classifications (Section 4), requirements remain for:

- typologies for system structure
- typologies for sediment transport patterns
- typologies of adjustment ('subtleties' versus phase shifting).

Scientific management in the UK is "ill-adapted" (Everard, 2004) to fluvial environment enhancement and protection. It is also lagging behind the US in the ability to deal with socio-economic and perceptual values (Haider and Steedman, 1993) of ecological integrity in riverine ecosystems. There is a need for the UK to develop an integrated management framework, amalgamating social perspectives of ecosystems with environmental integrity. There is also a need to develop an understanding of the interactions of biophysical and socio-economic environments, if the UK is to achieve water resource sustainability through river conservation management under climate change.

In summary, ecological systems contain a complex mix of physical, chemical and biological components (Andreasen *et al*., 2001). These components are difficult to measure but form an essential basis for management decisions. There are many advantages and disadvantages to using ecological integrity (biological integrity plus physical and chemical monitoring) in river conservation management (Table 3.2). However, assessing ecological integrity and the quality of water resources (as opposed to water quality) introduces subjectivity and value-laden judgements. The aim of using ecological integrity is to produce quantifiable measures of ecosystem status, such as the IBI, as well as social judgements on what is being sought (Francis *et al*., 1993). However opposed scientists may be to using qualitative values in conservation management, the fact remains that highly sought-after solutions to water resource degradation issues will not come from advancing tools of analysis (Karr, 1999). Neither will it come from increased monitoring and controls of chemical activity, but rather from looking at both the elements and functions of a biological system - in other words its ecological integrity.

The UK is ready for an integrated approach that encompasses nested hierarchical scales, ecological relevance and IBI-type judgements. A useful tool already exists in the biotope classificatory tool. Using this technique in conjunction with a geomorphological typology of structure, transport and adjustment may offer a powerful tool to predict possible impacts of future climate change on the UK's river systems.

## 5.4 Indices of biological integrity (IBI)

IBI was first developed for use in small warm water streams (such as those too warm to support salmonids) in central Illinois and Indiana, USA (Karr, 1981). The original version had 12 metrics that reflected fish species richness and composition, number and abundance of indicator species, trophic organization and function, reproductive behaviour, fish abundance, and condition of individual fish (Table 5.1). Each metric received a score of five points if it had a value similar to that expected for a fish community characteristic of a system with little human influence, a score of one point if it had a value that deviated significantly from the reference condition, and a score of three points if it had an intermediate value. Expectations for species richness metrics increased with increasing stream order, and were derived from an empirical relationship between stream size and maximum number of species present, termed the maximum species richness (MSR) line (Fausch *et al*., 1984). The total IBI score was the sum of the 12 metric scores (Table 5.2) and ranged from 60 (best) to 12 (worst). Some authors have reduced the lowest score to zero.

As the IBI became more widely used, different versions were developed for different regions and ecosystems. New versions developed for streams and rivers in France, Canada and the Eastern and Western United States tended to have a very different set of metrics reflecting the substantial differences in fish faunas between these regions and the central United States (USEPA, 2005). Similarly, the metrics used in IBI versions developed for other types of ecosystems, such as estuaries, impoundments and natural lakes, usually bore only a limited resemblance to those of the original version.

**Table 5.1 Original metrics of biological integrity of fish communities (Karr, 1991)**

| Metric | Rating of metric* | | |
|---|---|---|---|
| | | 3 | 1 |
| **Species richness and composition** | | | |
| 1. Total number of fish species* (native fish species)** | | | |
| 2. Number and identity of darter species (benthic species) | Expectation for metrics 1-5 vary with stream size and region | | |
| 3. Number and identity of sunfish species (water-column species | | | |
| 4. Number and identity of sucker species (long-lived species) | | | |
| 5. Number and identity of intolerant species | | | |
| 6. Percentage of individuals as green sunfish (tolerant species) | | | |
| **Trophic composition** | | | |
| 7. Percentage of species as omnivores | <0 | 0-45 | >45 |
| 8. Percentage of species as insectivorous cyprinids (insectivores) | <5 | 5-20 | >20 |
| 9. Percentage of individual as piscivores (top carnivores) | >5 | 5-1 | <1 |
| **Fish abundance and condition** | | | |
| 10. Number of individuals in sample | Expectation for metric vary with stream size and region | | |
| 11. Percentage of individuals as hybrids (exotics etc) | 0 | 0-1 | >1 |
| 12. percentage of individuals with disease, tumours, fin damage and skeletal anomalies | 0-2 | 2-5 | >5 |

* original IBI metrics for Midwest United States
** generalised IBI metrics

**Table 5.2 Index of biological integrity scores (Karr, 1991)**

| **Total IBI score** (sum of 12 metric ratings)* | **Integrity class of site** | **Attributes** |
|---|---|---|
| 58-60 | Excellent | Comparable to the best situations without human disturbance: all regionally expected species for the habitat and stream size, including the most intolerant forma, are present with a full array of age (size) classes; balanced trophic structure. |
| 48-52 | Good | Species richness somewhat below expectation, especially the loss of the most intolerant forms. Some species are present with less than optimal abundances or size distributions. Trophic structure shows some signs of stress. |
| 40-44 | Fair | Signs of additional deterioration include loss of intolerant forms, fewer species, highly skewed trophic structure (such as increasing frequency of omnivorous or other tolerant species); older age classes of top predators may be rare. |
| 28-34 | Poor | Dominated by omnivores, tolerant forms and habitat generalists; few top carnivores; growth rates and condition factors commonly depressed; hybrids and diseased fish often present. |
| 12-22 | Very poor | Few fish present, mostly introduced or tolerant forms; hybrids common; disease, parasites, fin damage and other anomalies regular. |
| ** | No fish | Repeated sampling finds no fish. |

* sites with values between classes assigned to appropriate integrity class following careful consideration of individual criteria/metrics by informed biologists
** no score can be calculated where no fish are present

River management is hampered by a lack of clear standards against which to judge the degree of environmental degradation. Weigel *et al*. (2000) describe the development and characteristics of a fish-based and a macroinvertebrate-based IBI designed to provide such standards. Ten metrics related to assemblage structure, composition, and function comprise the fish IBI: numbers of native, water column and sensitive species; percentages (by number of individuals) of benthic, tolerant, exotic, omnivorous, native livebearing and diseased/deformed individuals; and catch per effort. Seven metrics appear useful in the draft macroinvertebrate IBI: Hilsenhoff Biotic Index; number of species; percentages of Ephemeroptera - Plecoptera – Trichoptera, Chironomidae and sediment-tolerant individuals; percentage gatherer genera; and catch per effort.

IBIs hold promise as indicators of healthy aquatic habitats; being unit less, they allow for comparison between regions. However, this can also be a weakness, in that managers unfamiliar with aquatic systems may end up managing by index rather than by biology. IBIs combine information from structural, compositional, and functional parameters and offer quantitative comparison of different settings in terms of a single metric (Biodiversity Partnership 2005).

The current IBI measures the following metrics:

- total number of fish species
- number of benthic insectivorous species
- number of trout and/or sunfish species
- number of intolerant species
- proportion of individuals as white suckers
- proportion of individuals as generalists
- proportion of individuals as insectivorous cyprinids
- proportion of individuals as trout or proportion of individuals as piscivores (top carnivores)- excluding American eel
- number of individuals in the sample
- proportion of individuals with disease or anomalies (excluding blackspot disease)

(NJDEP, 2005)

Kerans and Karr (1994) used invertebrate data to evaluate the usefulness of 18 characteristics of invertebrate assemblages (attributes) to assess the biological condition of streams. They also developed a comprehensive benthic invertebrate index that reflects important aspects of stream biology and responds to the effects of human society in detectable ways. The following thirteen attributes were found to be valuable in discriminating sites: total taxa richness and taxa richness of intolerant snails and mussels, mayflies, caddisflies and stoneflies; relative abundances of Corbicula, oligochaetes, omnivores, filterers, grazers and predators; dominance; and total abundance.

In Maryland USA, there has been extensive development of these indices through the Maryland Biological Stream Survey (MBSS), a state program that started in the early 1990s. During the MBSS, three key indices were developed (Morgan 1999) – a fish index of biotic integrity (FIBI), a benthic index of biotic integrity (BIBI) and a physical habitat index (PHI). Coupled with the FIBI, a BIBI was also developed using the same general approach.  Indices of habitat quality have lagged behind IBI development. In part, this is because of difficulty in developing accurate, precise and complete methods to quantitatively and qualitatively assess habitat characteristics.

## 5.5 Biotopes

Biotopes provide a standard, descriptive assessment of instream physical structure based on consistent recognition of features over spatial and temporal scales (Table 5.3) (Padmore 1998).  Their basis lies in the development of typologies to underpin the habitat quality index developed as a framework for the protection of rivers (Raven *et al*. 1997). Biotopes provide a means of integrating ecological, geomorphological and water resource variables for management purposes.

According to Newson & Newson (2000), meso-scale research into habitat hydraulics has tended to define spatial units *a priori* and then validate these by combining hydraulic variables. Thus, Wadeson (1994) calls his basic units ‘hydraulic biotopes’ and Padmore (1997) names hers ‘physical biotopes’. Ecologists have also set up *a priori* units, some of them bearing the same names; Pardo and Armitage (1997) define ‘mesohabitats’ as “visually distinct units of habitat within the stream, recognisable from the bank with apparent physical uniformity” (p111).

Wadeson (1994) carried out a substantial review of the terminology available for these units before rationalising them for his study in South African streams; Padmore (1997) produced a similar list for her study in Northern England. The greatest difference between the group of units selected by ecologists and those selected by geomorphologists is that the former (perhaps because they have been established for lowland rivers in the main) contains vegetation and vegetative structures (Table 5.4). The latter is dominated by mineral substrate, morphology and flow, originating as they do in gravel-bed (or coarser) rivers.

**Table 5.3 Descriptions of flow types used to map biotopes in the field (from Newson and Newson, 2000)**

| Flow type | Description | Associated biotopes(s) |
|---|---|---|
| Free fall | Water falls vertically and without obstruction from a distinct feature, generally more than 1m high and often across the full channel width | Water fall |
| Chute | Fast, smooth boundary turbulent flow over boulders or bedrock. Flow is in contact with the substrate and exhibits upstream convergence and downstream divergence | Spill – chute flow over areas of exposed bedrock<br>Cascade – chute flow over individual boulders |
| Broken standing waves | White-water 'tumbling' waves with crest facing in an upstream direction. Associated with surging flow | Cascade – at the downstream side of the boulder, flow diverges or 'breaks'.<br>Rapid |
| Unbroken standing waves | Undular standing waves in which the crest faces upstream without breaking | Riffle |
| Rippled | Surface turbulence does not produce waves, but symmetrical ripples which move in a general downstream direction | Run |
| Upwelling | Secondary flow cells visible at the water surface by vertical 'boils' or circular horizontal eddies | Boil |
| Smooth boundary turbulent | Flow in which relative roughness is sufficiently low that very little surface turbulence occurs. Very small turbulent flow cells are visible, reflections are distorted and surface foam moves in a downstream direction. A stick placed vertically into the flow creates an upstream facing 'V' | Glide |
| Scarcely perceptible flow | Surface foam appears to be stationary and reflections are not distorted. A stick placed on the water's surface will remain still | Pool – occupies the full channel width<br>Marginal deadwater – does not occupy the full channel width |

The common thread between geomorphological/hydraulic studies and those by freshwater ecologists has been that, once the named set of meso-scale units has been confirmed as potentially discriminatory by exploratory study, each type of unit is validated by field measurements. For example, the conversion of Harper *et al*.'s (1995) 'potential habitats' to the final 'functional habitats' occurred through field sampling of invertebrates (Harper and Everard, 1998), followed by multivariate statistical analysis; the same occurred at a larger scale for 'mesohabitats' (Armitage and Pardo, 1995). Jowett (1993), Wadeson (1994) and Padmore (1997) instead used statistical analysis of hydraulic measurements to confirm or refute the *a priori* classification.

**Table 5.4 Biotope patchiness by geomorphological channel type and links to ecosystem theory (after Padmore 1998)**

| Patchiness | Channel morphology | Comments |
|---|---|---|
| High | Unconfined bedrock and boulder channels | Possible 'critical' reaches |
| | Steep headwater boulder dominated channels | Patchiness is greater in channels of low stream order (consistent with the RCC) |
| Intermediate | Mid-gradient, wandering cobble-bed channels | Mid-gradient channels have intermediate numbers of biotopes, and their patchiness is influenced by local channel morphology |
| | Fine gravel, actively meandering channels | Mid-gradient stable cobble-bed channels |
| Low | Regulated mid-gradient, cobble-bed channels | Few biotope sequences are present in regulated channels due to an increase in low flows which drowns out some morphological features |
| | Confined bedrock channels | Possible 'bottleneck' reaches |
| Very low | Low-gradient engineered channels | Low numbers of biotopes dominated by glides as the result of removal of some morphological features (dredging, regrading) |

Questions remain as to the extent to which physical habitats - areas of distinct species assemblages associated with water depth, velocity and substrate combinations - may be mapped onto physical biotopes (riffles, runs, pools, and glides) in rivers. The most notable variable for predicting flow type is the Froude number (Jowett 1993, Wadeson 1994, Padmore 1998). Despite describing flow in the main channel rather than at the bed (Padmore 1998), the Froude number has been shown to correlate with the distribution of benthic invertebrates. Physical biotopes are now important aspects of river inventory and river rehabilitation design, but both empirical (field evidence) and model predictions are lacking, without which it is difficult to establish the connection between physical biotopes and measured aspects of instream species characteristics (Clifford, 2005).

Other issues which arise include:

- the existence of distinct physical habitats and biotopes at the sub-reach scale;
- the changing dynamics of habitat and biotopes - seasonally, event-specific and with more gradual alterations in flow stage;
- dynamic connections between habitat and biotopes over a variety of timescales;
- biotope patchiness by geomorphological channel type and links to ecosystem theory (Table 5.4)

It has been suggested by many freshwater biologists that, whatever the guiding system paradigm for river habitat, a building block approach involving a hierarchy of scales is both appropriate and practicable (for example, Frissel *et al*., 1986). White and Pickett (1985) go so far as to suggest that "the simultaneous occurrence of local dynamics and broad scale equilibria also underscores the central importance of scale hierarchies in the interpretation of natural systems."

Given this central importance it is strange that, despite at least six iterations according to Newson and Newson (2000) referring to the original scale illustration by Frissel *et al*. (1986), little real progress has been made on making hierarchies operational. Maddock and Bird (1996) have at least introduced a version which shows a practical working scale (the PHABSIM calibration scale). A recent study by Newson and Archer (in prep) incorporates flow variability in biotope mapping, where the concept of the biotope has been shown to have a reasonable predictive power in relation to ecological patterns (see Newson and Newson, 2000, and Section 3.5). Their analysis of hydrological variability is based on frequency and duration of pulses above threshold flows, selected as multiples of the median flow. Results show that flow values required to maintain maximum biotope diversity suggest higher environmental flows than are generally used in water resource schemes (that is, $Q_{95}$). This is particularly true for upland channels with more complex hydraulics and geomorphology. Consideration of the number and duration of flood pulses allows analysis of flow regimes rather than single values. Newson and Archer (in prep) list the major changes that occur in physical habitat with flow at seven sites in North East England. This analysis offers a starting point for interdisciplinary assessment of hydromorphological elements of significance to the 'natural' or preferred biota. Subsequently, hydrological analyses could begin to identify changes through time resulting from climate change, land use change or channel/flow management.

# 6 Conclusions and recommendations

## 6.1 Challenges and best practice

Incorporating climate change in river typologies must overcome the following challenges:

- The relationship between climate, land use, catchment hydrology and the fluvial system is one of great complexity, which makes inferences about climate change difficult to make.
- The sensitivity of rivers to change is carried in the links within the system rather than the subcomponents. These links usually lie across disciplinary boundaries, with implications for typology development and new ways of conducting research.
- Rivers and streams are unique, patchy, discontinuous and strongly hierarchical systems.
- While the hydrological regime determines the structure and function of river floodplain systems, detailed descriptions of characteristics are often missing at ecologically-relevant spatial and temporal scales.
- Modelling studies of large river dynamics have relied upon generalized hydrological patterns.
- Ecological studies measuring concurrent responses of multiple trophic levels to flow in river and floodplain habitats are rare.
- Thus far there has been little application of process-based geomorphology in the UK.
- There is an increasing recognition that ecological response is non-linear in nature, while our impacts on systems remain essentially linear (for example, habitat loss). The result may be the crossing of system thresholds with sharp reductions in biodiversity and associated function.
- Land use is an imprecise term in a scientific sense (Newson, 1997), and many gaps exist in our empirical knowledge of land use/cover/management.

For these reasons, the most appropriate response to managing the effects of climate change on rivers may be monitoring and guidance rather than prediction. There is virtually no verifiable climate change data available from contemporary research catchments; most data comes from short-term projects on relatively pristine research sites. As a result, findings tend to be local or regional in scale and cannot be extrapolated with ease. In addition, little is known about the effects of climate change on the physical requirements of instream biota. The Environment Agency is currently leading a consortium of the Countryside Council for Wales (CCW) and English Nature (EN) in the PRINCE project, which is using process-based modelling to determine climate change impacts on selected UK freshwater ecosystems. Conceptual models such as that proposed by Bunn and Arthington (2002) tend only to deal in trends. In some situations only certain species will be affected, while in other cases the requirements of whole communities may alter.

How these adjustments will occur following rapid geomorphological change is unclear. The reality is that, at the catchment scale, the majority of rivers in the UK are likely to be quite robust in nature (from Werritty and Leys, 2001). However, changes in sediment transport characteristics may still significantly affect the future ecological status of water bodies.

## 6.2 Monitoring and guidance versus prediction

Noss (1990) notes that monitoring has not been a “glamorous activity in science”. He also points out that the types of hypotheses posed by a scientist at the start of a project - cause and effect, probabilities, interactions and alternative hypotheses – are not commonly asked by workers carrying out a monitoring programme. In most cases, research and monitoring are uncoordinated and carried out by different agencies. Monitoring is at its most successful when it is designed as scientific research and to test hypotheses that are relevant to policy and management questions. In this context, monitoring is a necessary link in the adaptive management cycle (Figure 6.1) that continuously refines management practices on the basis of data received from monitoring and analysed with an emphasis on predicting impacts (Noss, 1990).

### 6.2.1 Explicit hypothesis testing

The first step is to establish goals and sub-end points (Noss, 1990) of structure or function that the Environment Agency wishes to assess or manage (Figure 6.1). The next stage is to gather and integrate existing data and to establish baseline conditions. From current data it will be possible to determine the distribution and condition of ecosystem subcomponents and their potential stressors. Hotspots of biodiversity, along with geographical areas considered to be at high risk of degradation, can also be identified at this stage from existing datasets.

From these steps, it is possible to formulate specific questions to be answered by monitoring. In terms of changes driven by climate change, information relating to thresholds within fluvial systems will be useful here, as these are the conditions at which adaptive management has to be aimed. As described in Section 5 however, system complexity and limited data have so far prevented detailed analysis of the sensitivity of British rivers to a range of different driving variables. The lack of geomorphological process and rate data (erosion rates, sediment loads) from UK hydrometric networks makes the job of identifying thresholds and threshold behaviour that much more difficult. Remote sensing may be useful here and catchment-scale evaluations of erosion and deposition (known as fluvial audits) are currently being introduced as standard practice in the Environment Agency.

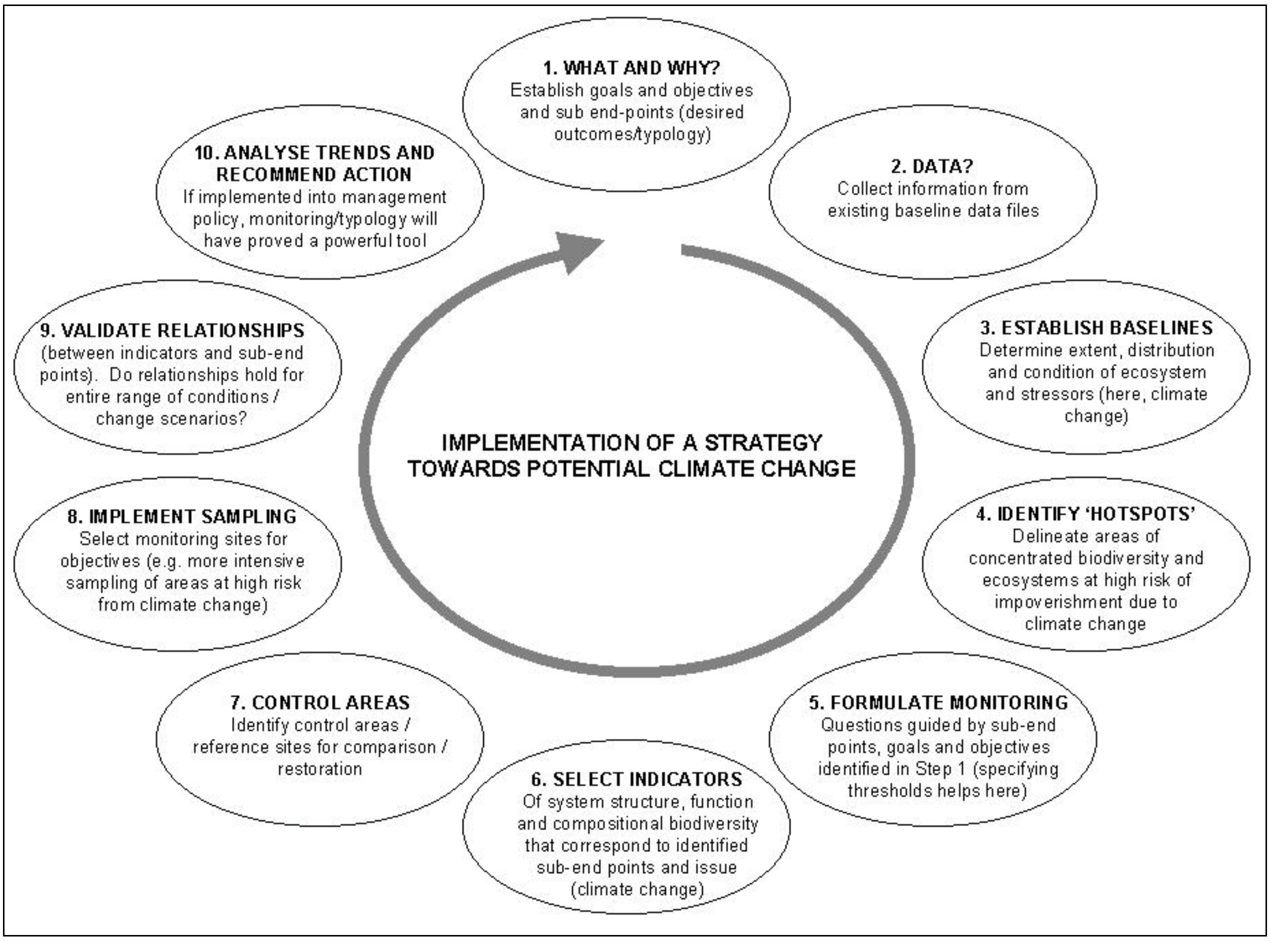


**Figure 6.1 Recommendations for implementing monitoring strategies to assess impacts of climate change on UK rivers (adapted from Noss, 1990)**

It will also be necessary to identify appropriate indicators of structural, functional and compositional diversity at a number of levels that correspond to identified end points and research questions. These levels equate to those identified in Figure 5.2 After identifying control areas and those most susceptible to climate change, sampling and monitoring schemes can be designed at the appropriate scales. While a wide range of compositional, structural and functional methods for typing rivers exist, most benefit will be gained from deriving techniques and typologies that are:

- predictive (rather than simply taxonomic) in nature;
- applicable over a range of scales from the macro- (basin) scale to the level of the meso-scale (or geomorphological unit);
- applicable to the micro-scale (the level at which essential niches and food resources are made available to instream and other biota with the fluvial system).

The potential and applicability of the reviewed techniques approaches and typologies in Sections 2 to 4 of this report are summarised in Table 6.1 under assessment criteria outlined in Section 1.

**Table 6.1 Summary of reviewed techniques, approaches and typologies**

| Method | Process-based | Multi-scale or hierarchical | Sensitive to external drivers | Potential for prediction | Realistic monitoring & data needs | Consistent and quality controllable | Integrates geomorphology hydrology & ecology | Captures system memory | Cross-regional applicability | Informs adaptation or management |
|---|---|---|---|---|---|---|---|---|---|---|
| **Historical flow method** | *** | * | | *** | *** | * | * | *** | * | * |
| **Flow duration curve** | *** | *** | ** | * | *** | * | * | * | ** | ** |
| **Tennant Method** | ** | *** | | * | *** | * | * | * | ** | ** |
| **Discharge method** | | | | | | * | * | * | | |
| **Indices of hydrological hlteration** | ** | *** | ** | *** | ** | | *** | *** | *** | *** |
| **Physical HABitat SIMulation** | *** | | * | *** | ** | * | *** | | *** | *** |
| **Stewart's Method** | *** | *** | | * | *** | | ** | | *** | ** |
| **Low Flows 2000** | * | *** | | *** | *** | ** | ** | ** | *** | *** |
| **Indices of biological integrity** | *** | *** | | ** | ** | * | ** | | ** | |
| **Instream flow requirements** | *** | ** | ** | | *** | * | | | *** | *** |
| **Building block methodology** | ** | *** | ** | *** | | * | ** | | ** | *** |
| **River health programme** | *** | *** | ** | * | *** | * | * | ** | ** | ** |
| **Rapid bioassessment protocol** | *** | | ** | | | | *** | | ** | *** |
| **European aquatic monitoring network** | | *** | ** | | | | | | | |
| **Biotopes** | *** | ** | ** | *** | *** | ** | *** | * | *** | ** |
| **Rosgen** | | * | | | ** | * | * | * | * | * |
| **Montgomery and Buffington** | *** | ** | ** | *** | ** | * | * | ** | * | * |
| **River Styles** | *** | ** | | *** | ** | * | * | ** | ** | *** |
| **Fluvial audit** | *** | *** | *** | *** | *** | ** | ** | *** | *** | *** |
| **River corridor survey** | * | * | * | * | ** | * | * | * | *** | ** |
| **River habitat survey** | * | * | * | * | *** | ** | ** | * | *** | ** |
| **GeoRHS** | ** | *** | * | ** | ** | *** | ** | ** | *** | * |

*** Meets requirements well
** Meets requirements adequately
* Meets requirements to some extent
No star Inadequate

Figure 6.2 attempts to map some of the most-widely used schemes, including those reviewed in this report, along with their usefulness from a hydrological, biological and geomorphological point of view. The most useful methods for monitoring and predicting impacts of (as yet unexperienced) climate-driven change will be those which straddle one or more themes.

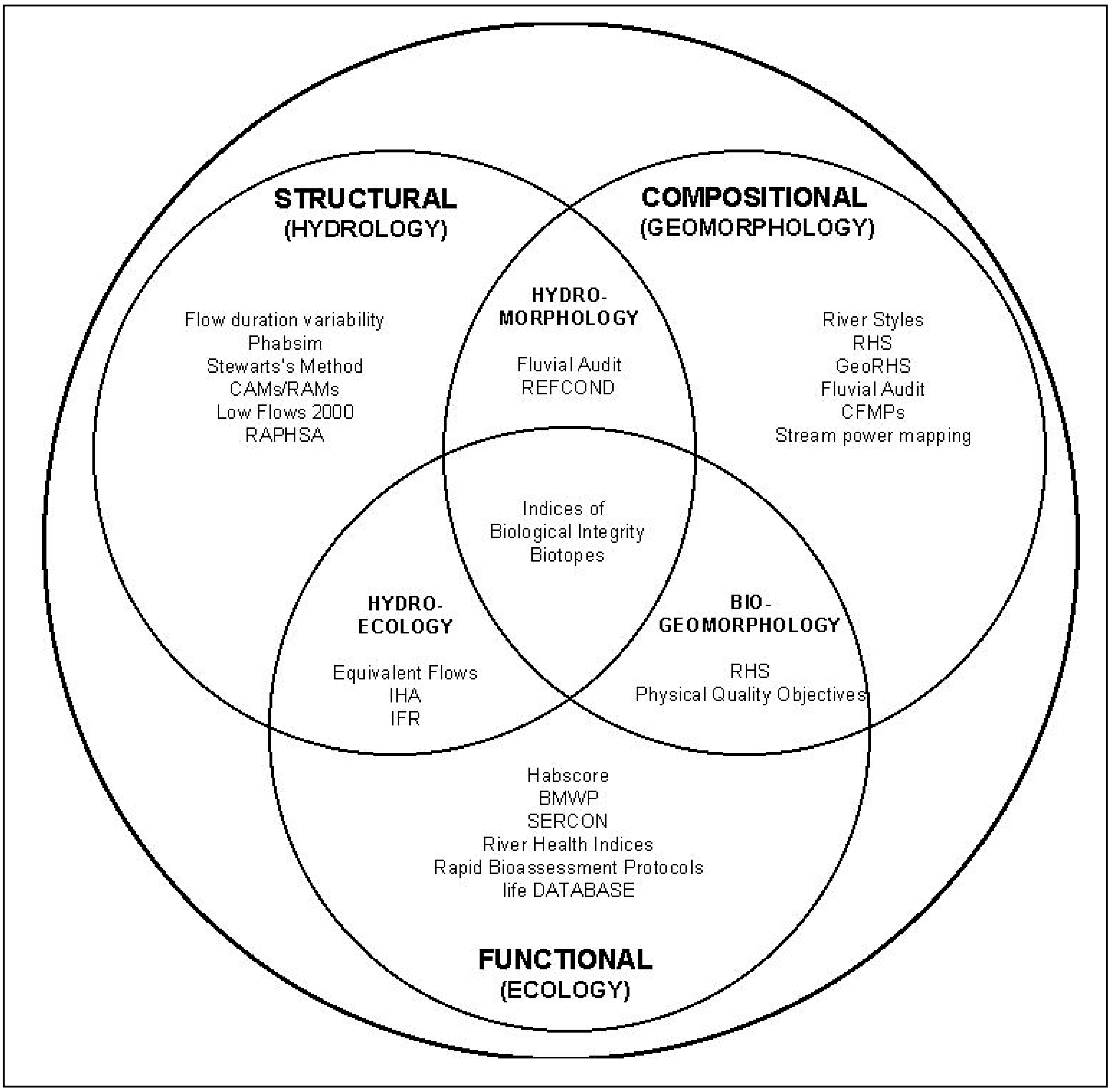


**Figure 6.2 Compositional, structural and functional methods used for typing rivers shown as interconnected spheres, each encompassing multiple levels of organisation. The more predictive techniques are located towards the centre of the diagram**

In the figure, techniques which straddle two spheres deal with hydromorphology, hydroecology and 'bio-geomorphology. Potentially useful here are IBIs and biotopes. Francis *et al*. (1993) define biological or ecological integrity as a system containing a "full complement of native species, processes and structures; and a high quality of water and air". As such, ecological integrity is analogous to the definition of 'good ecological status' in the WFD. Attaining this objective will require the incorporation of biological factors into standard river system monitoring. A system suffering from a lack of ecological integrity would demonstrate diversity losses, impairment of ecosystem function and structural degradation. The biotope approach was developed simultaneously in the UK and South Africa in the late 1990s; as such, the typology is applicable over a range of climate conditions. Being a hierarchical approach, the biotope method is also compatible with other conceptual approaches (such as Frissel *et al*. 1986).

The common thread linking these geomorphological/hydraulic studies with freshwater ecology is that, once the potential set of meso-scale units has been identified by reconnaissance survey, each type of unit is validated by field measurements. Physical biotopes are now key aspects of river inventory and river rehabilitation design. The data gaps identified in Section 5 are thus minimised, although both empirical (field evidence) and model predictions are still lacking, without which it is difficult to establish the connection between these biotopes and measured aspects of instream species characteristics (Clifford, 2005). Specific issues relate to:

- the existence of distinct physical habitats and biotopes at the sub-reach scale;
- the changing dynamics of habitat and biotopes - seasonally, event-specific and with more gradual alterations in flow stage;
- dynamic connections between habitat and biotopes over a variety of timescale;
- biotope patchiness as dictated by channel type geomorphology.

## 6.3 Links to Phase 2 of this project

### 6.3.1 GIS approach

In assessing potential impacts of future climate change on the UK's rivers, it is clear that holistic typologies which integrate the structural, functional and compositional nature of these complex systems are needed. Ultimately the requirement is for a GIS based approach in order to explore available spatial data" operating "at water body/catchment or landscape scale.

Fast Track R&D (such as projects WFD 44, RAPHSA W6-094) is proceeding towards a defensible system of reach typology that will cover relevance to the sensitivity and resilience of both the geomorphological and ecological systems to be protected. Most of the candidate systems are imports to the UK, such as those of Rosgen (1985), Montgomery and Buffington (1998) and the hierarchical schemes of Frissel *et al.* (1896) and Brierley and Fryirs (2000). Scale hierarchical models have an undoubted attraction in a policy field which itself is hierarchical, but it is important to sound warnings about the uncertainties in our knowledge at each level of the hierarchy (Newson and Newson, 2000) and the information penalties to be paid when switching scales (from Figure 5.2). To date there has been insufficient interaction between the principle disciplines included under the term 'hydromorphology' – that is, hydrology, geomorphology and ecology

A pragmatic approach suggests the potential of combining reach-based ecological and physical habitat information with:

- modelling climate change and hydrology at catchment scale;
- modelling (simplified) channel hydraulics at reach or multi-reach scale;
- GIS and land surface/river network modelling.

### 6.3.2 Proposed approach

The second phase of this project will entail collecting spatial data and defining segment and reach typologies using key variables (Table 6.2). These will then be verified with field evidence (fluvial audit and biotope mapping) and other data derived from aerial photography (flow depths, shading and erosion), collated into a catchment-wide set of consistent GIS coverages. Comparison of the physical typology with extensive data on fish populations will explore the ecological relevance of catchment-specific river types.

**Table 6.2 Proposed key variables for defining physical typology**

| Scale | Variables for classification |
|---|---|
| Basin scale | Critical slope classes<br>Stream order<br>Critical basin area<br>Regional geomorphology typology (such as piedmont, lowland) |
| Segment<br>(corresponding to channel type) | Dry valleys (for example, in limestone areas)<br>Presence or absence of floodplain (channel/floodplain width ratio)<br>Important tributary junctions (Strahler increment)<br>Sediment Mobility Index (such as River Calder CFMP) |
| Reach | Eroding (fluvial/stock – as per Eden Rivers Trust)<br>Depositing<br>Eroding and depositing<br>Degree of confinement<br>Planform<br>Modification<br>Scale (such as width/depth ratios)<br>Flow per unit width |
| Ecological unit<br>– meso-scale habitat | Physical biotopes (RHS)<br>Local bed morphological changes and range of relative flow depths (remote sensing/aerial photos)<br>Shading (RHS, aerial photos) |

The third phase will use downscaled climate change data to drive simple hydraulic models through a series of channel types at a variety of catchment locations. This will enable an exploration of potential impacts on local habitat dynamics and the implications for local ecosystems. The existence of a whole catchment, physical, process-based typology will enable extrapolation of potential impacts to the whole catchment.

The method will be applied to the River Eden in Cumbria, North West England in collaboration with the Eden Rivers Trust. The project outputs will potentially guide the development of a river basin management plan for the Eden, identify links with ongoing projects and provide information on adaptive management and specifically how to incorporate climate change in river typologies.

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# List of abbreviations

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|---|---|
| ADF | Average Daily Flow |
| ASPT | Average Score Per Taxa |
| AWB | Artificial Water Body |
| BBM | Building Block Methodology |
| BIBI | Benthic Index of Biotic Integrity |
| BMWP | Biological Monitoring Working Party |
| CAMS | Catchment Abstraction Management Strategy |
| CFMP | Catchment Flood Management Plan |
| CHASM | Catchment Hydrology and Sustainable Management |
| DEM | Digital Elevation Model |
| DWAF | South African Department of Water Affairs and Forestry |
| DWF | Dry Weather Flow Index |
| EAMN | European Aquatic Modelling Network |
| GeoRHS | Geomorphological River Habitat Survey |
| GIS | Geographical Information System |
| HMWB | Heavily Modified Water Body |
| HOST | Hydrology of Soil Types |
| IBI | Indices of Biological Integrity |
| IFIM | Instream Flow Incremental Methodology |
| IFR | Instream Flow Requirements |
| IHA | Indicators of Hydrological Alteration |
| FIBI | Fish Index of Biotic Integrity |
| MAF | Minimum Acceptable Flow |
| MBSS | Maryland Biological Stream Survey |
| MSR | Maximum Species Richness |
| PHI | Physical Habitat Index |
| PHABSIM | Physical Habitat Simulation Model |
| POM | Programme of Measures |
| PSYCHIC | Phosphorus and Sediment Yield Characterisation in Catchments |
| RAM | Resource Assessment and Management Framework |
| RAPHSA | Rapid Assessment of the Physical Habitat Sensitivity to Abstraction |
| RCC | River Continuum Concept |
| RCS | River Corridor Survey |
| RDM | Resource Directed Measures |
| RHP | South African River Health Programme |
| RHS | River Habitat Survey |
| RVA | Range of Variability Approach |
| SERCON | System for Evaluating Rivers for Conservation |
| SHETRAN | Update of SHE (Système Hydrologique Européen) model |
| UKCIP02 | UK Climate Impacts Programme Scenarios |
| USEPA | United States Environmental Protection Agency |
| WFD | European Union Water Framework Directive |

We welcome views from our users, stakeholders and the public, including comments about the content and presentation of this report. If you are happy with our service, please tell us about it. It helps us to identify good practice and rewards our staff. If you are unhappy with our service, please let us know how we can improve it.