Rivers, Floodplains and Wetlands:

Connectivity and Dynamics

Review of the importance of floodplain connectivity and dynamics for riverine biodiversity, including implications for definitions of ecological status under the Water Framework Directive.

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Executive Summary.

The Water Framework Directive (WFD) sets new standards for waters in the European Union. Classification systems will need to take into account the impacts of the physical modifications on riverine biota. Whilst there is a large literature on this subject, rigorous studies are difficult, because of the scale of the area impacted and the long-term nature of adjustment of biological systems. Nonetheless, a comparison of studies using in-stream, between stream, historical record and before-and-after methods show that river modifications, including changes to floodplain condition and connectivity alter the composition and abundance of riverine macrophytes, invertebrates and fish.

River restoration schemes show varying success in restoring biological communities. This may be related to the continuing isolation of rivers from their floodplains, where floodplain habitat diversity plays an important role in maintaining ecological functions. Floodplain wetlands have a role to play in retaining nutrients and improving in-stream water quality. Both forests and grasslands have been shown to be effective in nutrient retention. A comparison of the River Great Ouse in England and the River Biebrza in Poland reveals that the hydro-morphological conditions of the Great Ouse have been dramatically altered in a manner that has demonstrable impacts on fish populations.

Despite problems in identifying reference conditions and comparing complex dynamic systems, it is possible to conclude that the physical modification of rivers and their floodplains results in significant changes to biological communities. Fifteen per cent of lowland UK rivers are considered close to pristine, and a further 15% are considered semi-natural. This suggests that restoration is likely to be required on a large scale, to meet WFD requirements. Restoration should be based on an understanding of the processes that maintain biological communities, rather than focussing exclusively on improvements to channel condition. It should include reinstatement of natural flow regimes and flooding patterns and the creation of floodplain water bodies and side channels. Restoration of this kind, as well as contributing to the achievement of WFD objectives, would assist Government in meeting its obligations to wetland biodiversity, help to manage pollution and mitigate the impacts of flood and droughts, and provide opportunities for people to enjoy, and benefit from, rivers and their environs.

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1 THE WATER FRAMEWORK DIRECTIVE CONTEXT

1.1 Floodplains and the Water Framework Directive

The purpose of the Water Framework Directive is ‘prevent further deterioration and protect and enhance the status of aquatic ecosystems and, with regard to their water needs, terrestrial ecosystems and wetlands directly depending on the aquatic ecosystems’ (Article 1(a). This is to be achieved principally through the aim to prevent deterioration in the states of all rivers, lakes, transitional and coastal and groundwater bodies, and to restore them to good status by 2015. Status for surface water bodies includes biological, hydro-morphological and chemical aspects. In the UK, whilst there has been a reduction in point source water pollution in recent years, eutrophication of waters from point and diffuse sources remains a serious problem. However, the Directive is not only about reducing pollution but also about protecting and enhancing biological communities through the restoration of physical processes, such as flooding frequency and sediment transport. Despite the fact that the word “floodplain” does not feature in the Directive and “wetlands” are mentioned only twice, important characteristics of lowland river floodplains such as flooding, connectivity and physical processes will need to considered in establishing WFD standards for surface water bodies. In addition, wetlands within Protected Areas, and ground-water dependant terrestrial ecosystems (including wetlands) must be protected and enhanced with regard to their water needs. “Water needs” means the right amount of water at the right time of year but, by implication, also means restoring natural patterns, such as flooding frequency and the associated physical work that water carries out, for instance de-silting, sediment redistribution, plant removal or the bringing of fertile sediments and propagules. The physical (seasonal) connections between rivers and floodplain features are crucial to the ecology of aquatic and wetland systems, to allow fish to exploit spawning habitats, for example.

Article 1 of the WFD also states that “The purpose of this Directive is to establish a framework for the protection of inland surface waters, transitional waters, coastal waters and groundwater which: (e) Contributes to mitigating the effects of floods and droughts” (Article 1, Purpose, p. 5 E.U.2000). In the context of the ecological drive of Directive as a whole, mitigating the effects of floods must be best served by restoring the natural function of floodplains, with their ability to hold up floodwaters and reduce the flood pulse. The effects of droughts can similarly be mitigated through the increased retention time of water in functioning floodplain wetlands. The following review aims to establish the relationship between hydro-morphological conditions and biological conditions in rivers, in order to assist in achieving the purposes and ecological objectives of the Water Framework Directive.

1.2 The concept of reference conditions

Ecological status in the Water Framework Directive is determined initially by the establishment of a "reference condition" for a particular "type" of water body. The reference condition is determined using data from sites whose physical and chemical characteristics are considered to have been affected by no, or only minor, anthropogenic disturbance. The biological communities associated with these sites are assumed to be close to "pristine". Where no "real" reference sites exist for a particular type of water body (for example certain kinds of lowland rivers), the Directive allows a virtual reference to be created by from historical data, modelling, or (in the last resort) expert judgement. The "status" of any water body is assessed by measuring its deviation from the reference condition.

1.3 Hydro-morphological Quality Elements in WFD.

The WFD text provides a list of the hydro-morphological elements of rivers most relevant ecological status, and defines their required standard at 'reference condition'. In High Status sites, these aspects of the river must be protected from deterioration. In sites at Good Status and below, they must be protected, restored or enhanced to the degree necessary to achieve the required biological standards (see below). The hydro-morphological requirements of a reference site given by Annex V (page 34) are:

Hydrological regime - The quantity and dynamics of flow, and the resultant connection to groundwater reflect totally, or nearly totally, undisturbed conditions.

River continuity - The continuity of the river is not disturbed by anthropogenic activities and allows undisturbed migration of aquatic organisms and sediment transport.

Morphological conditions - Channel patterns, width and depth variations, flow velocities, substrate conditions and both the structure and condition of the riparian zones correspond totally or nearly totally to undisturbed conditions.

1.3 Biological quality elements and ecological status.

Good ecological status is achieved, in relation to hydro-morphology, when hydro-morphological conditions are consistent with the achievement of good status for the biological quality elements (Annex V, page 43). If the hydro-morphology is impaired such that the biology deviates more than slightly from reference conditions, then the site will fail to achieve good ecological status.

Clearly, it will be critical that biological reference conditions are defined for each river type in sites with pristine, or near-pristine hydro-morphological conditions, in order to be able to detect hydro-morphological impairment using the biota. The biological quality elements to be considered for rivers are invertebrates, fish and macrophytes. Given that, for many river types in the UK, sites at reference conditions do not exist, detailed hindcasting will have to be carried out to work out the biological communities which are likely to have existed, and provide the appropriate biological base-line for WFD classification. This may by done by making comparisons with comparable pristine European rivers or using historical data.

1.4 The concept of the water body

The ecological and physical relationship between a river and its associated floodplain and wetland areas is clarified in the EU Common Implementation Guidance on water bodies (E.U. 2003). A body of surface water is defined as a discrete and significant element of surface water such as a lake, reservoir, stream, river or canal. It is comprised of the quality elements described in the Directive for the classification of ecological status (Annex V 1.1 & Annex V 1.2). In concrete terms this means that, e.g., 'a river water body comprises:

(a) the hydromorphological quality elements, which include the water flow, the bed of the channel, that part of the land adjacent to the channel thats structure and condition is directly relevant to the achievement of the values for the biological quality elements (i.e. the riparian zone), and

(b) the relevant biological elements.'

In relation to floodplain wetlands, this means that those wetlands which are directly influencing the status of a river water body will form part of the riparian zone of that water body.

2 THE THEORETICAL FRAMEWORK

River channels are part of an interconnecting series of biotopes and environmental gradients that constitute lotic floodplain ecosystems. The nature conservation value of floodplain environments is dependant on hydrological and ecological interactions between the river, the riparian zone and the floodplain (connectivity). Connectivity can be defined as: "the ease with which organisms, matter or energy transverse ecotones between adjacent ecological units" (Ward *et al.* 1999 p.129). It is primarily determined by the frequency, extent and duration of flooding and partial connectivity can also be maintained via groundwater pathways. High connectivity is a vital characteristic of a river at reference condition, as it has major implications for biodiversity due to its role in structuring successional patterns in the vegetation on the floodplain – frequently flooded zones tend to be colonised by pioneer aquatic species but as connectivity decreases, terrestialisation of the vegetation occurs.

Much of the biodiversity associated with floodplain river ecosystems is due to heterogeneity at the habitat scale, both within the channel and on the floodplain (Ward 1998). As a river in its natural state migrates across the floodplain, it produces a range of lotic and lentic aquatic environments such as side channels, dead arms connected at one end, abandoned braids, oxbow lakes and ponds. This results in a mosaic of habitat patches, ecotones and successional stages, characterised by different communities and enhanced by disturbance. These are each important for biodiversity in their own right and an understanding of the characteristics and importance of each habitat type is required in order to achieve sustainable catchment management (Williams *et al* in press). In dynamic floodplains, ephemeral water bodies are filled by flooding (Ward *et al.* 1999). Floodplain backwaters can be more important than the channel itself for maintaining biodiversity, (Garcia and Laville 2001) acting as refuges for vegetation and fauna and as spawning grounds for fish.

In Europe many formerly dynamic rivers have become highly managed, single thread channels, isolated from their floodplains (Ward 1998). Indeed, many temperate river-floodplain systems are modified to such an extent that it is difficult to identify variables that once controlled ecological processes and the occurrence of biota (Tockner *et al.* 1998). Flow regulation by dams disrupts the natural disturbance regimes of systems reducing frequency, extent and duration of floodplain inundation (Ward and Stanford 1995). Many biological communities are dependant on these processes to maintain their habitats and therefore regulation causes deviations in ecological status from reference condition.

River regulation in the UK began in the late eighteenth century. By 1936, over 200 reservoirs had been constructed in the Pennines alone. Today, there are 450 large dams in the UK, 80% of which are in upland areas. Dams cause discontinuities of resource gradients by trapping sediment, altering flow regimes and interrupting migration patterns (Tockner *et al.* 1998, Greenwood *et al.* 1999) and cause a major decrease in aquatic floodplain habitats downstream (Ward 1998). Reduction in channel forming flows reduces channel migration. Trapped sediment results in channel degradation below the dam, lowing the water table (Ward and Stanford 1995). The alteration of flow regimes is often claimed to be the most serious threat to the ecological sustainability of rivers and floodplain wetlands (Binn and Arthington 2002).

In addition, various forms of river regulation such as modification of flow dynamics, dredging, channel straightening, bank stabilisation and the construction of levees produce structurally simplified and hydraulically efficient river channels, which permit the rapid clearance of water from the floodplain and disrupt connectivity in many rivers. This lost connectivity prevents the formation of new floodplain water bodies and accelerates the terrestialisation of surviving water bodies. Regional water authorities and Internal Drainage Boards maintain watercourses by extensive dredging, weed control or mowing. River improvement is usually achieved by channelisation, which comprises straightening, widening and deepening of water courses (Hey *et al.* 1994, Gilvear *et al.* 2000). 8 500 km (25%) of main river in the lowlands was severely altered by canalisation, dredging and embankments between 1930 and 1980 and virtually all the remained are managed by lesser practices such as weed cutting (Moss 1998). The River Habitats Survey, carried out between 1994 and 1997 found that only 15% of UK lowland rivers could be classified as “pristine” and only 29.7% as “semi-natural” (Raven 1998). River modifications can seriously impoverish wildlife by disrupting vegetation cover and reducing habitat diversity (Raven 1986). Structural variety of habitats is one of the most important conditions for the existence of well-balanced aquatic communities and river engineering causes key habitats such as riffles and pools, meanders and steep banks to be replaced by uniform sterile and heavily engineered channels (Jingworth *et al.* 1993).

There has been much written about general impacts of river modification on the biota of the channel and its floodplain (e.g. Amoros and Roux 1988, Ward and Stanford 1995). However, this has not always been supported by the results of scientific investigation. Rigorous scientific studies are difficult because of the large scale of the area impacted by river modification and the long-term nature of adjustment of the biological system to modifications. Often there is no data available to describe reference conditions and even when studies were done before modifications were carried out, suitable controls are hard to find as the river is a naturally dynamic system and it is hard to separate anthropogenic change from that which would occur naturally. Replication is also very difficult – different rivers and sections of the river will respond differently due to a whole range of factors. This report aims to use the available scientific data to show how modifications to river hydro-morphology cause deviations from the ecological reference conditions of rivers and floodplains. The studies which have been carried out generally use four main methods:

1) The “within stream” method in which natural and affected areas of the same river are compared.
2) The “between stream” approach in which an affected river is compared to a nearby, unaltered river.
3) The “historical record” approach which involves the use of documented information from prior to the channel alterations.
4) The “before and after” approach where studies are made in the same location before and after channel alterations.

In addition, in recent years there has been interest in the potential for river restoration to ameliorate the effects of channelisation (Biggs *et al.* 1998).

2.1 Definitions of hydrological features

Terms used in this document are defined below:

Channel – a lotic, permanent or semi-permanent water body (river or stream).

Floodplain - all land adjacent to a watercourse over which water flows in the time of flood, or would flow but for the presence of flood defences where they exist. The limits of floodplain are defined by the peak water level of an appropriate return period event on the watercourse or at the coast. On rivers, this will normally be the greater of the 1 in 100 year return period or the highest known water level (based on the definition used by the Environment Agency).

Water body – a discrete area of surface water e.g. a river, stream, lake, pond.

Riparian zone - a transitional zone, bordering the edge of lakes, rivers, streams, ponds and wetlands, which exerts a direct influence on or is influenced by river and stream channels or lake margins, and on the water and aquatic ecosystems contained within them.

Floodplain backwaters – bodies of surface water on the floodplain created by the river system. These include ephemeral and permanent ponds, side channels and oxbow lakes.

3 CHANGES IN RIVER BIODIVERSITY

3.1 Impacts on macrophytes and phytobenthos

Water Framework Directive Status Definitions

High status: The taxonomic composition corresponds totally or nearly totally to undisturbed conditions. There are no detectable changes in the average macrophytic and the average phytobenthic abundance.

Good status: There are slight changes in the composition and abundance of macrophytic and phytobenthic taxa compared to the type-specific communities. Such changes do not indicate any accelerated growth of phytobenthos or higher forms of plant life resulting in undesirable disturbances to the balance of organisms present in the water body or to the physico-chemical quality of the water or sediment. The phytobenthic community is not adversely affected by bacterial tufts and coats present due to anthropogenic activity.

Moderate status: The composition of macrophytic and phytobenthic taxa differs moderately from the type-specific community and is significantly more distorted than at good status. Moderate changes in the average macrophytic and the average phytobenthic abundance are evident. The phytobenthic community may be interfered with and, in some areas, displaced by bacterial tufts and coats present because of anthropogenic activities.

3.1.1 Weed cutting

Submerged macrophytes are important in river ecology. They affect stream characteristics such as water current velocity and sediment composition (Baattrup-Pedersen *et al.* 2002), provide an important refuge from flow for phytoplankton, zooplankton and fish and act as egg laying sites for fish and invertebrates. Macrophytes harbour a distinct and often substantial fauna. Removal of aquatic plants is common practice in many lowland rivers in the UK, primarily to reduce siltation of the channel. In the River Great Ouse, 40% of the channel is weed cut (Garner *et al.* 1996). Baattrup-Pedersen *et al.* (2002) studied the impact of weed cutting on the macrophyte community by comparing weed cut and uncut sections of the River Gels and the River Gryde, Denmark. It was found that the total species coverage was high in both the cut and uncut reaches in both rivers, varying between 65 and 80%. However, species richness and species diversity was lower in the weed-cut compared to the uncut reaches in both rivers with lower numbers of aquatic and semi-aquatic species (no significance level given) (Table 1).

Table 1 – Comparison of species number and dominants on the cut and uncut reaches of the Rivers Gryde and Giel (Baattrup-Pedersen *et al.* 2002).

	Cut		Uncut	
River	Gryde	Giels	Gryde	Giels
No. species per plot	1.02	1.56	1.55	3.28
No. aquatic sp.	5	7	17	17
No. semi-aquatic sp.	8	6	11	15
No. terrestrial sp.	5	17	10	11
Shannon diversity	0.20	0.34	0.36	1.01

Spatial species diversity was also significantly ($P<0.05$) more heterogeneous in the uncut reaches. Overall, it was concluded that weed cutting can have a pronounced negative effect on macrophyte diversity and composition resulting in deviation from reference conditions. Cut reaches were less physically diverse in terms of substratum and depth therefore lower species diversity may be due to less diverse habitat conditions. It was suggested less frequent and intensive cutting could provide opportunities for the establishment of more species rich macrophyte communities.

3.1.2 Flow regulation and channel modification

There have been several studies on the impact of dams and river regulation on aquatic macrophytes. Vanderpooten and Klein (1999) studied the changes in the hydrophyte assemblages from the Alpine Rhine to the middle Rhine and the extent to which river regulation along the entire length has modified zonation from that at reference condition. Previously, when there were rapids upstream, rheophilous mosses with structural adaptations to the strength of flow occurred and downstream the broader alluvial floodplains and substrate movement in the bed opened up areas where less competitive species could establish. The regulation of the river led to a homogenisation of habitat conditions, which greatly influenced the composition of hydrophyte communities. In the High Rhine, hydroelectric dams increased water levels upstream causing the disappearance of all primary habitats reducing niche and bryophyte diversity. Some rare species, characteristic of the area are on the verge of extinction.

Bryophytes can be an important component of stream biota and make a significant contribution to primary production as well increasing habitat for invertebrates and periphyton. Englund *et al.* (1997) studied 52 rapids in central and northern Sweden, comparing the bryophyte flora at unregulated sites, regulated sites with reduced flow and regulated sites with unreduced flow. Regulated sites had larger short-term variations in flow and lower monthly variation. There were a variety of responses to regulation. Some species (e.g. *Fontinalis antipyretica* and *F. dalecarlica*) had reduced abundance in streams with reduced flow (although confidence limits were large). Other species e.g. *Blindia acuta* and *Schistidium agassizii* had higher abundance than expected on sites with regulated but unreduced flow. The results indicated that overall, bryophyte diversity was reduced and species abundance patterns were changed by stream regulation. This may be due to the more constant water level on regulated streams – the highest diversity of bryophytes is usually found in the occasionally submerged zone, which becomes small under regulation.

Rorslett *et al.* (1989) used three case studies to describe the impact of hydropower developments on aquatic macrophytes in Norwegian rivers. At these sites, the impact of regulation on macrophytes was somewhat more dramatic than in the above examples. On the River Otra, hydroelectric development has increased normal winter flows and downstream of the Brokke Reservoir there were large growths of submerged *Juncas bulbosus* in the 1970s and 1980s until up to 55% of the water surface was covered, reducing the amenity value of the river. The River Suldalsågen is one of Norway’s most productive salmon rivers. However, two HEP schemes have caused large hydrological changes - average discharge has been reduced by over 45% compared to pre-regulation level and winter flows are tightly controlled. There are high levels of turbidity. After the initiation of one of the schemes the abundance of submerged bryophytes increased significantly with up to 100% cover. This reduced the in-stream production of salmonids by making the riverbed unfit as a spawning area. On the River Børselva regulation has meant that the river has long dry periods and has been turned into a series of stagnant pools interspersed with wetlands. There has been encroachment by emergent species, particularly the heliophytes *Carex rostrara* and *Equisetim fluviatile*. The remaining open water is choked by dense growths of vascular plants such as *Potomogeton alpinus* and the submerged bryophyte *Fontinalis entipyretica*. They concluded that aquatic macrophytes are important from an ecological point of view but excessive growths can reduce species diversity (Rorslett *et al.* 1989).

Loffler (1990) compared two relatively similar sections of the Danube in Austria – an unaltered section near Hainberg and an altered section near Altenworth where the river is completely disconnected from the alluvial plain. A much greater diversity of macrophytes was found in the river and floodplain backwaters at Hainberg (60 species) compared to Altenworth (20 species) in a survey in 1983. At Hainberg several species found e.g. *Potamogeton acutifolius, Ranunculus raianii* and *Veronica catenata* are rare in Austria. These had disappeared from the disconnected site. Species diversity was higher at Hainberg due to the greater diversity of wetland habitats.

The riparian zone is an integral part of the river system, forming an important water-wetland ecotone. In a pristine, dynamic floodplain, riparian species are alternatively part of the aquatic zone and part of the wetland system as water levels change. The condition of the riparian zone and the condition of the river are intimately linked and in order for the river to be in pristine (reference) state, the land immediately adjoining the river should be in state approaching naturalness (Burt *et al.* 1998). Gumiero and Salmouraghi (1996) in a study of 51 riparian zones of various land uses in the Reno River basin, Italy, found that there was a highly significant ($P < 0.01$) relationship between the naturalness (conservation and biodiversity status) of the riparian zone and benthic invertebrate biodiversity of the river. This was attributed to the fact that riparian zones represent a vital trophic resource for benthic communities. A damaged riparian zone leads to a damaged river community. River regulation by dams causes changes in aquatic communities of the riparian zone. Nilsson *et al.* (1991) studied the effect of regulation on riparian vegetation by comparing the flora of the River Vindel and the River Ume in Northern Sweden. The two rivers are equally long and large, have parallel courses and had similar vegetation before the River Ume was extensively regulated between 1951 and 1967 with hydroelectric development, moving the river margins and altering water level fluctuations. The Vindel, however, retains a pristine water level regime. The riparian zone in this study was defined as the land adjacent to the channel that is periodically flooded, from the summer low water level, equivalent to the lower boundary of the tall *Carex* vegetation to the spring high water level, equivalent to the lower boundary of the continuous *Vacciniuum myrtillus* carpet that covered the valley floor. The total number of species was the same on both rivers but most species were less frequent on the regulated river, occurring at fewer sites. The number of species per area sample was significantly lower on the regulated river (P < 0.002) (Table 2).

Table 2 – Comparison of the number of species found in the riparian zone of the river.

	Vindel	Ume
	Pristine	Regulated
No. species	258	260
No. species present > 20 sites	45	11
No. sp. /$\log_{10}$ area sampled	27.2	19.7

Thirty species occurred at between 10 and 16 fewer sites and 166 species occurred at between 1 and 10 fewer sites. Changes occurred particularly where the regulated river shoreline was located in former terrestrial areas and there were no remnant species available for colonisation. Species that were lost from more than nine sites include some species typical of wetland habitats such as *Parnassia palustris, Mentha arvensis* and *Veronica scutellata.* However, other species that were lost are primarily terrestrial and not of high conservation value such as *Nardus stricta, Agrostis stolenoifera* and *Ranunculus airicamus.* The study shows that river regulation can have an effect on riparian vegetation due to the scarcity of habitat in the regulated river. It was suggested that the mechanism of change was the change in water level fluctuations, which became greater after regulation. Natural river margins have an intermediate level of disturbance leading to maximum species diversity. An increase in disturbance led to a decrease in diversity (Nilsson *et al.* 1991). Similar results were found on a study that compared four free flowing and four regulated rivers in the same area of Northern Sweden (Nilsson and Jonsson 1995). The total number of species per river was found to be similar between the two groups of rivers. However, on a per site basis, there were significantly lower (P<0.01) values of riparian plant species richness in the regulated rivers compared to their unmodified state and many species had become locally extinct due to flow regulation and changes in the variability of water depth.

Channelisation constrains the natural movement of the river within the floodplain. Bravard *et al.* (1986) investigated the impact of regulation of the Upper Rhone on floodplain vegetation using old maps, aerial photographs, interviews and field observations of test plot areas. It is an area of high ecological diversity as there is a coexistence of communities belonging to both lotic and lentic water as well as to marsh and riparian zones. The Upper Rhone was canalised to improve navigation conditions for steamboats between 1880 and 1890. A 100 m channel was established by means of a discontinuous rock fall embankment in sectors where river braiding previously occurred. This was succeeded by hydroelectric projects and the protection of agriculture from flooding. Previously the dynamic braided section ensured the constant regeneration of biotypes and a range of successional stages. However, since regulation, the absence of lateral erosion prevents the initiation of succession and causes the disappearance of early successional stages. Areas have dried out and been colonised by terrestrial species (Bravard *et al.* 1986).

Flow regulation by dams and weirs reduces the regularity of flooding. Bornette and Arens (2002) showed the positive role of flood disturbance in the maintenance of Charophyte species diversity in a study of cut-off channels in the floodplain. Data was collected from the Doubs, Saones, Ain and Rhone rivers from 841 plots along 63 cut off channels. Charophytes are usually considered pioneer species, occurring in disturbed habitats supplied with groundwater. They are expected to occur abundantly in large rivers highly influenced by floods. It was found that *Chara vulgaris* and *Nitella conferuacea* were more frequent in and even limited to channels with high flood disturbance. However, *C. major* and *C. globularis* occurred in channels with little or no flood disturbance indicating that some species can survive under low disturbance conditions. In order to maintain optimal species diversity a range of ages of cut off channels is required, containing different successional stages of vegetation. If rivers are canalised and cut off from side channels, the early successional stages and therefore pioneer species will be lost as all channels gradually reach the climax vegetation.

Williams *et al.* (in press) surveyed water bodies on the floodplain of the River Cole in Oxfordshire for both macrophytes and invertebrates. Eighty sites were sampled in spring and autumn 2000. An area of 75 m^2 was recorded in each water body and some physical characteristics were recorded although there was no measure of connectivity between habitats or human influence on the habitats. The result are shown in Table 3:

Table 3 – Comparison of the contribution to biodiversity of different floodplain water bodies (Williams *et al.* in press)

		Rivers	Ponds	Streams	Ditches
Plants	Mean no. sp per water body	10.7	10.1	7.3	6.1
	Total no. sp.	152	173	124	90
	% Total sp. richness	61%	70%	50%	36%
	Species rarity index*	1.10	1.21	1.09	1.14
Invertebrates	Mean no. sp per water body	45.7	32.6	18.7	12.9
	Total no. sp.	49	67	39	30
	% total sp. richness	56%	76%	44%	34%
	Species rarity index*	1.13	1.10	1.05	1.03

* Derived finding the mean rarity value of the species found in each habitat. Common = 1, local = 2, , nationally scarce = 4, Red Data Book: conservation dependant = 8, Red Data Book: endangered = 16

Rivers were found to have the greatest mean number of species per habitat but were fairly uniform in their species composition. Ponds were very variable and had the greatest overall number of species and the greatest number of unique species. Overall, pond communities differed significantly from the lotic habitats ($P< 0.001$) and contributed the most to overall floodplain biodiversity. The reasons for this may be related to connectivity and to habitat heterogeneity – the ponds have small catchment areas, which are highly distinctive in terms of their land use and geology. A range of floodplain water bodies is important for maintaining a complete range of biodiversity on a floodplain. However, Homes *et al.* (1999) showed that river and floodplain regulation can cause the number of floodplain water bodies to decline. On the Isar floodplain between Mittenwald and Lenggries in Bavaria, Germany the site closest to natural conditions had an abundance of older short-lived ponds close to the stream whilst in sections more strongly influenced by water engineering, the abundance and diversity of ponds declined due to embankments restricting the river to a single channel. This means that new ponds cannot be created and existing ponds are isolated from the river.

Hey *et al.* (1994) looked at the impact of various channelisation schemes throughout England and Wales on aquatic macrophytes. Eighteen flood alleviation schemes were surveyed, including resectioning – dredging widening and regrading to increase the bankful capacity of the river, channel straightening, the creation of diversion channels, adjacent floodbanks and two stage channels. In five schemes non-engineered control sections adjacent to the modified channel were used to ascertain the physical characteristics of the engineered reach and the associated habitats and plants communities in the pre-scheme state, although these are not true controls as they may not exactly represent the channels before the engineering works were carried out. Surveys of both bank and in channel vegetation were carried out. Figure 1 compares the mean species richness within the channel and on the banks with each modification, compared to the semi-natural controls.

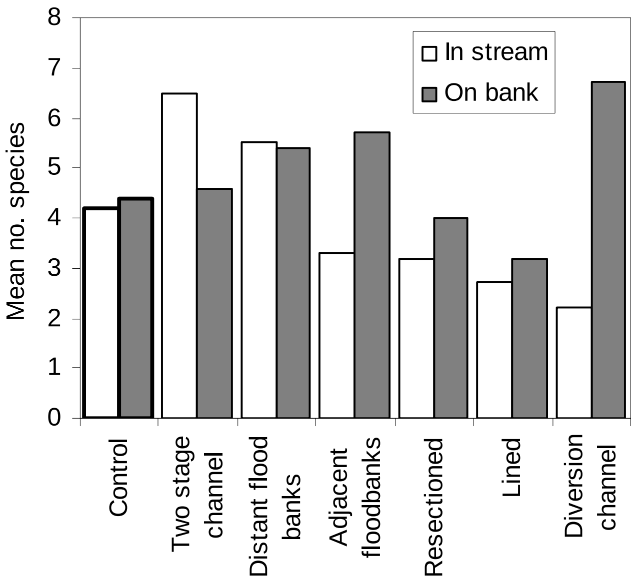

Figure 1 – Results of species richness surveys of channels with various engineering works (data from Hey *et al.* 1994)

Creation of a two-stage channel and distant floodbanks increased in-stream aquatic macrophyte richness above that of the control due to the creation of greater habitat diversity. Creation of a diversion channel did the most damage, followed by lining the channel, resectioning and adjacent flood banks. For riparian plants, again the traditional engineering solutions of resectioning and lining the channel both reduce species diversity below that of the controls as they destroy habitat diversity and limit recolonisation. However almost all engineering methods altered vegetation composition compared to the non-engineered site (representing reference conditions). The increased species diversity in the riparian zone was to some extent the addition of terrestrial habitats and species which does not help to conserve riverine biodiversity. Some species were associated with particular engineering treatments, for example, in the channel *Filligree* and *Juncus articulatus* were associated with distant floodbanks and *Stachys palustris* and *Sarganium ermersium* with two stage channels. The change in species composition post-engineering was due to the change in available habitat types. A reduction in particular habitats accounts for the loss of certain species, whereas increases in particular habitats encouraged colonisation by different species. For example, on the River Cherwell, the channel has been enlarged and the gradient controlled by weirs. This has resulted in decreased slumped banks and decreased shelf, leading to loss of *Rorippa amphibia*, and a decrease in stable banks and a deceased reed fringe leading to loss of *Sagittaria sagittifolia.* On the River Roding, a two-stage channel was created by excavating the floodplain. A decrease in slumped banks and a decrease in shelf margin led to a decrease in *Agrostis stolonifera.* An increase in mud, reed fringe and berms led to an increase in *Vancheria* sp., *Nuphar lutea, Potamogeton percinatus, Spartganium emersim* and *Sagittaria sagittifolia.* All forms of channel engineering lead to a change in available habitat types from those that are naturally present and therefore a divergence from the species community present under reference conditions.

3.1.3 Summary

River modifications have an impact on the community structure and diversity of floodplain macrophytes, generally reducing biodiversity and causing deviation from reference conditions. Weed cutting is a very common management technique and high levels appeared to be incompatible with good status as it was shown by Baattrup-Pedersen *et al.* (2002) to cause significant changes in species diversity compared to uncut river reaches. River regulation has also been shown to cause major changes in community composition and diversity due to a reduction in the heterogeneity of river habitats of both larger macrophytes (Vanderpooten and Klein 1999) and bryophytes (Englund *et al.*1997). Changes in the regularity of flooding combined with rapid changes in water levels downstream of dams causes damage to riparian macrophytes (Nilsson *et al.* 1991). Changes in flow regimes caused by regulation and channelisation has a negative effect on floodplain macrophytes as demonstrated by the community changes experienced by charophytes (Bournette and Arens 2002). Floodplain water bodies are vital for the presence of the full complement of floodplain macrophytes (Williams *et al* in press) but engineering causes the number of ponds to decline as no new ponds are created, connectivity is lost and succession occurs (Homes *et al.* 1999). Reduction in flooding disturbance causes the disappearance of pioneer vegetation as succession occurs and this can lead to a colonisation of previously aquatic zones by terrestrial vegetation (Bravard *et al.* 1986).

3.2 Impact on benthic invertebrate fauna

Water Framework Directive Status Definitions

High status: The taxonomic composition and abundance correspond totally or nearly totally to undisturbed conditions. The ratio of disturbance sensitive taxa to insensitive taxa shows no signs of alteration from undisturbed levels. The level of diversity of invertebrate taxa shows no sign of alteration from undisturbed levels.

Good status: There are slight changes in the composition and abundance of invertebrate taxa from the type-specific communities. The ratio of disturbance-sensitive taxa to insensitive taxa shows slight alteration from type-specific levels. The level of diversity of invertebrate taxa shows slight signs of alteration from type-specific levels.

Moderate status: The composition and abundance of invertebrate taxa differ moderately from the type-specific communities. Major taxonomic groups of the type-specific community are absent. The ratio of disturbance-sensitive taxa to insensitive taxa, and the level of diversity, are substantially lower than the type-specific level and significantly lower than for good status.

Benthic invertebrates play a key role in river ecosystems due to their intermediate position in the food chain linking allochthanous and autochthanous production with higher trophic levels (Munn and Brusen 1991). They are also sensitive to change in their environment and for this reason are often used as indicators of ecological disturbance.

3.2.1 Weed cutting

Macrophytic weed cutting can have an impact on aquatic invertebrates. Macrophytes provide an important refuge for many species of invertebrates. Sampling was done in the middle Great Ouse

(Garner *et al.* 1996). It was found that Cladocera were the dominant large planktonic organisms and were limited to the macrophyte zone whereas rotifers were mainly found in deeper water beyond the cover of plants. Weed cutting removed plant cover beyond 1-2 m from the banks. Densities of rotifers and *Ceriodaphnia quadrangular* increased after weed cutting whereas densities of *Polyphemus pediculus* and Chydarids decreased. There was a distinct shift in the distribution of Cladocera. High densities were found only at points undisturbed by weed cutting within 2 m of the bank. In the longer term, rotifers followed their normal pattern of seasonal decline whereas Cladocera decreased rapidly. The plant cover provides a velocity refuge and food sources for zooplankton. Cutting causes a premature decline in zooplankton, which has knock-on effects on fish.

3.2.2 Flow regulation and channel modification

Scullion *et al.* (1982) compared the invertebrate fauna in the unregulated River Wye and the regulated River Elan in Wales and found a significant decrease in species diversity on regulated rivers compared to their natural state. The River Elan is affected by water quality and temperature changes caused by upstream impoundment. It has high concentrations of iron and manganese from the reservoir and from effluent from a water treatment works and the temperature regime is modified by deep-water release from the impoundment. Lower species biomass and diversity were found on the regulated river (Table 4).

Table 4 – Results of total number of taxa and numbers of individuals of selected invertebrate groups in riffles and pools on the Rivers Elan and Wye (Scullion *et al.* 1982).

	Pool		Sig	Riffle		Sig
	Wye (unregulated)	Elan (regulated)	(P)	Wye (unregulated)	Elan (regulated)	(P)
Mean no. of taxa	23	12	<0.05	25	18	<0.05
No. of individuals	18848	11314	<0.05	32799	15206	<0.05
Ephemeroptera	1056	12	<0.05	48	3946	<0.05
Plecoptera	162	550	<0.05	904	1306	n/s
Diptera:						

(Chironomidae)	13374	9450	n/s	7642	8056	n/s
(Simuliidae)	228	14	<0.05	11474	256	<0.05
Coleoptera	1636	50	<0.05	946	30	<0.05

The River Elan had less Ephemeroptera, Diptera and Simuliidae (Table 4). The reduction of species diversity in the Elan was thought to be related primarily to the deposition of iron and manganese rich materials on the bed. Similar results were found in an apparently separate study also on the effect of impoundment on the invertebrate fauna of the River Elan in comparison to the unmodified River Wye (Inverarity *et al.*1983). Again, it was found that the River Elan downstream of the reservoir was characterised by low numbers of taxa and a lower density of individuals than on the Wye. A total of 77 taxa were collected from the two rivers, 26 found exclusively in the River Wye and 11 species (more than 80% Oligochaeta and Plecoptera) exclusive to the Elan. The Wye fauna is dominated by Ephemeroptera and Plecoptera. Species restricted to the unregulated Wye include *Baetis scambus, Rithrogena semicolorata* and *Edyonurus venosus.* Some species are characteristic to both the Wye and the lower regions of the Elan where metal concentrations are lower such as the Tricopterans *Hydropsyche siltalai* and Seri*costoma personatum*. In the May sampling session, densities were significantly higher (P<0.05) at Wye than the Elan downstream of the reservoir. Metal densities were particularly important in determining invertebrate fauna on the river. The regulation of the River Elan has lead to a severe deviation of the invertebrate community from reference status.

Armitage (1978) looked at the impact of Cow Green reservoir on the invertebrate fauna of the River Tees below the dam. Results of a study of the benthos before the completion of the reservoir were compared to studies downstream of the dam subsequent to impoundment 1972-73. There were large changes in the composition of the invertebrate community (Figure 2), although no indication of significance is given.

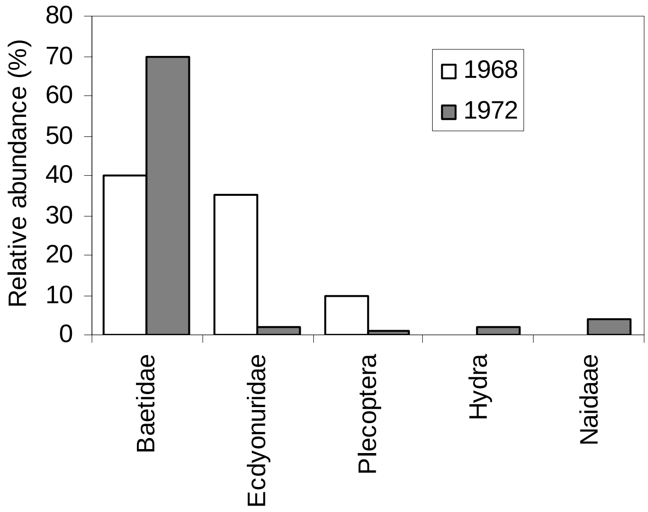

Figure 2 – Changes in composition of invertebrate fauna before and after impoundment on the River Tees (Armitage 1978)

Overall, faunal diversity decreased from a mean of 3.4 in 1968 and 1969 to 1.7 in 1972-73 (no significance level given). These changes were compared to those within Maize Beck, used as a control but here there were no significant changes (P>0.05) since impoundment in any faunal group. In the post impoundment study, the fauna of river sites at varying distances from the dam was compared. It was found that the site immediately below the dam and the next closest site had the least diverse faunas with high numbers of a few species, for example at the site immediately below the dam five species accounted for 90% of individuals.

Although the above studies show a clear decrease in diversity as a result of impoundment, not all authors in agreement with this result. Peeters and Tachet (1989) compared the benthic invertebrates in braided and channelised reaches of the Drôme River, France, a tributary of the Lower Rhone. The invertebrate fauna was dominated by Chironomids (69% of individuals in the braided section and 82% in the embanked section). There were no differences in the number of taxa between the two sections but the mean number of individuals was twice as high in the braided section as in the embanked section. Petts *et al.* (1993) argue that there are changes that occur immediately after impoundment, but then further changes in the invertebrate community may occur over a timescale of 10-100 years as channel characteristics and in-stream hydraulics adjust. They looked at changes in the River Rede, UK below Catcleugh Reservoir comparing sites upstream of and below the reservoir. Both the abundance and number of taxa is greater downstream of the dam than upstream of the impoundment. Greenwood *et al.* (1999) investigated changes in invertebrates on the regulated River Rheidol below Nant-y-Moch Dam Wales. The river was regulated by a dam in 1961 and invertebrate surveys were done in 1981 and 1996, comparing sites upstream of the dam which are unaffected by regulation with a downstream river reach undergoing progressive channel change to new equilibrium conditions under a new flow regime, and an accommodation reach where flows are conveyed by the original channels. The unregulated sites had only 12 taxa, dominated by Chironomidae, with a mean of 399 individuals m^{-2}. In the accommodation reach, 23 invertebrate taxa were found which included 11 new taxa not found at the unregulated site. These were mainly species associated with slow flow habitats. New, more lentic habitats were created where growths of mosses and liverworts created food sources for Plecoptera and Tricoptera. Mean density of individuals was 651 m^{-2}. In the adjusting reach, the number of taxa increased from 19 to 27 between the early 1980s and 1993 with a mean density of around 420 m^{-2}.

Although at these sites a major decrease in biodiversity was not seen, every river modification scheme resulted in a change in the diversity or abundance of invertebrate fauna and therefore a deviation from reference conditions.

Petts and Greenwood (1985) also studied a regulated section of the River Rheidol. Since 1961 runoff from 93% of the catchment has been controlled by the Nant Y Moch reservoir which releases a compensation flow of only 1.6 $m^3 s^{-1}$. This reduction in flow combined with an increase in the rate of sedimentation in the catchment of the Rheidol tributary the Pleithnant has caused a large increase in the rate of sedimentation at the confluence of the two rivers, downstream of the reservoir. Field surveys were done in 1982. Sediment and invertebrate samples were taken at 50 locations in the area of sedimentation, 14 in the stable downstream reach, 5 from the upper Rheidol above the dam and 5 from Upper Pleithnant. It was found that invertebrate density was higher and there was larger number of taxa present in the upstream sites than in the area of sedimentation (no significance levels given). The natural sites were dominated by Chironomidae and Trichoptera. In the sedimented reach however, chironomid density was reduced from 74% to about 25%. Plecoptera and Oligochaetes increased in abundance compared to natural levels at the sedimented site and were associated with flow stabilisation. Trichopterans were twice as abundant in the sedimented region that in the natural or lower Rheidol and Simuliidae also increased due to the formation of pools and riffles. Within the regulated river there was a variety of habitat types such as a gravel bedded, braided reach and a wider, lower velocity region. Each of these has particular compositions of fauna, which means that if they are removed by channel modification, part of the reference fauna is also removed.

Dessaix *et al.* (1995) compared the changes in invertebrate species on the Upper Rhone upstream of the dam. The hydroelectric schemes were completed between 1981 and 1986 and data is available for the pre-regulated state, although this is not entirely natural as the river was embanked and there were upstream dams. Studies began in 1975, sampling both the area upstream of the first dam to be built in 1981, which became a reservoir, and a bypassed section of the river. The upstream area had community changes mainly due to a decrease in flow velocity. Taxa with limnophilic characteristics, which previously only occurred as "accidentals" (sampled less than three times) increased. Their presence was linked to an expansion of macrophytes and the deposition of fine particles. Three rheophilic and lithophilic taxa disappeared (*Theodoxus fluviatilis, Dendrocoelum lacteum* and *Dugesia gonocephala*). In the bypassed section, changes occurred due to a drastic reduction in discharge and increased instability. Potamic species e.g. Hydropsychidae and *Heptagenia suphirea*, which are characteristic of large lotic rivers, became less abundant. Overall, it was found that regulation of the upper Rhone causes significant deviation in the structure and functioning of benthic macrofauna compared to reference conditions. (Dessaix *et al.* 1995)

Loffler (1990) compared two relatively similar sections of the Danube in Austria – an unaltered section near Hainberg and an altered section near Altenworth. A much greater diversity of zooplankton was found at Hainberg compared to Altenworth, although no data are given. There was also greater species diversity within the benthic fauna. At Hainberg, many rare species were found and there were crustaceans typical of the Danube such as *Corodium* sp. and *Limnomysis benedeni*. There was also a unique diversity of molluscs – 35 species offering valuable food resources to fish. Common organisms and only 16 mollusc species inhabited Altenworth.

As with macrophytes, the maintenance of floodplain backwaters is vital for the maintenance of invertebrate communities at reference conditions in the floodplain. In the Garonne River, remnant side arms and floodplain waters are still functional between Toulouse and

Castelsarrasin, although the river is embanked. Four sites were sampled for Chironomid biodiversity –the main channel, a permanently connected side arm, an intermittently connected sidearm and an oxbow lake that is submerged once a year. Samples were taken monthly from each over the course of a year. It was found that there was an increase in Chironomid diversity with lentic status with the oxbow being the most diverse sampling site. 13 and 12 species respectively were exclusive to the temporarily connected and the oxbow sites. Only 7 and 9 species were exclusive to the two lotic sites. The community is dominated by limnophilous species and the floodplain backwaters contain a high proportion of the community diversity and therefore their conservation is vital (Garcia and Laville 2001). In the Danube Restoration project area, within the floodplain there is also a gradual change in Chironomid assemblages from the main side arm to isolated waters. The floodplain is species rich here with 447 benthic invertebrate species compared to just 149 species in a comparable floodplain of an impounded section of the Danube near Altenwörth with less diversity of backwater habitat (Tokner *et. al* 1998).

Homes *et al.* (1999) also demonstrated the importance of a diversity of floodplain backwater habitats for invertebrates. They studied pond occurrence and their invertebrate communities on the Isar Floodplain, Germany. All ponds within six floodplain sites were studied. At site A, the stream is embanked, separated from the floodplain and restricted to a single channel. At sites B-D the stream is managed as a residual flow downstream of a reservoir but the bed is braided and there are various abandoned arms. Site E has large open gravel bars and conditions are close to natural. At site F, the discharge is strongly regulated and peak floods are suppressed. The channel is deeply incised and the gravel bars are stable. Pond density was highest in sites B to E and turnover was also high – more than half the ponds in sites B, D and E were either generated or removed during the study year. Density and turnover were much lower at modified sites A and F. The proportion of young and short-lived pond types is 50% higher in sections B and E. These ponds usually contain stream invertebrates and pioneer species such as *Oreodytes davisi* and *Laccobius alternus.* This is a distinct fauna to the older ponds, which have a greater diversity of Coleoptera and Odonta. Diversity of pond types declines on sites more affected by hydrological engineering and with it the diversity of invertebrate fauna. Many pond species are restricted to lentic habitats and do not occur in the stream itself. When the river is embanked it cannot create new ponds and those that remain are fed from hillslope runoff and may gradually disappear.

The macrozoobenthos community reflects the dynamics of the floodplain. For this reason gastropods were studied on two floodplains of the Upper Rhine – an active floodplain at Au am Rhine and a former, disconnected one at Kastenwort (Orbdlik and Fuchs 1991). The Au am Rhine has natural hydrological dynamics and floods twice a year in June/July and in winter which enables the creation of new aquatic habitats and maintains surface and groundwater connections. The Kastenwort floodplain was cut off from the Rhine by an embankment construction in 1936. Sluice gates are closed in times of flood to protect the land and most of the floodplain water bodies are under-saturated. When floodplains are cut off by floodwall constructions, backwaters suffer from a loss of flow variability and groundwater exchanges between floodplains and abandoned channels are not sufficient to maintain the habitual variability of backwaters. Rapid terrestialisation causes an impoverishment of aquatic biotype diversity. In the Au am Rhine 30 gastropod species were found whereas in the Kastenwort only 19 were present and the community was dominated by the ubiquitous *Bithynia tentaculata* and *Physella acuta*. Within the water bodies the number of gastropod species decreased from small water bodies on the active floodplain, to small water bodies on the former floodplain to the active arms of the Rhine. On the Au am Rhine regular inundation enables the removal of leaf litter and detritus and allows the creation of new biotopes and the exchange of species between these biotopes leading to a higher habitat and species diversity. Gastropods are an immobile group that can only be transported by passive means such as by birds or by floods. Therefore, at Kasenwort the exchange of species is limited. Flood embankments have had serious impacts on the gastropod community at Kasenwort causing deviation from reference conditions.

The study of Van der Brink and Van der Velde (1991) also investigated the diversity of macroinvertebrates in floodplain water bodies and their relation to flood inundation frequency. Up to the middle ages, the Rivers Rhine and Meuse were natural meandering streams with side channels, cut-off channels and oxbow lakes. Today the majority of these rivers is embanked and regulated and as a result, the total floodplain area is fixed and greatly reduced. As a result of former river activity and some human activity, the floodplains harbour numerous water bodies, which have different degrees of connectedness to the main channel. The waterbodies were divided into three groups according to their inundation frequency, (A) those that are located outside the recent floodplain and are no longer flooded but they receive seepage water when the river level is high. The aquatic vegetation is diverse and consists of submerged macrophytes and nymphaeids. The presence of *Cleon simile, Enallagma cynathigerum, Ilyocoris cimicoides, Plea minutissima, Haliplus flauicollis* and *Acentropus nirveus* are associated with the occurrence of aquatic macrophytes. The second group (B) is in the active floodplain and is inundated less than 21 days a year in the winter. Its dominant species *Ischnuria elegans, Noctonecta glauca, Metriocnemus hirticollis* are associated with the broad littoral helophyte zone. The final group (C) are in the active floodplain and are frequently inundated (21-365 days per year) in both winter and summer. The helophyte zone is poorly developed and aquatic macrophytes are sparse or absent. Typical species *Psidium supimum, P. casertanum, P. henslowanum, Sphaerium solidum, Potamopyrgus antipodarum, Lithoglyphus naticoides* and *Oecetis ochracea* are associated with sandy substrates. The entire species composition changes with changing flood frequencies and significant ($P<0.001$-$P<0.05$) differences were found between site groups. In general, the number of mollusc species increases with increased flooding whereas the number of mites and insects (especially Coleoptera and Trichoptera) decreased. Groups A and B are characterised by phytophils and predators and shredders whereas group C is dominated by psammophils who are mainly filter-feeders and collector-gatherers. The conservation of an intact floodplain macroinvertebrate assemblage is dependant on the maintenance of flood frequency creating a diversity of floodplain waterbody habitats. A lack of connectivity will lead to succession in all waterbody types and a loss of pioneer species.

3.2.3 Summary

A whole range of responses of invertebrate fauna to river modification have been seen. Many studies (Loffler, 1990, Scullion *et al.* 1982, Armitage 1978, Inverarity *et al.* 1983) show that regulation causes a decreases in channel species diversity, often with an increase in the abundance of common species such as Baetidae and a decrease in more sensitive species such as Ephemeridae. However Peeters and Tachet (1989) found a decrease in the numbers of individuals but no decrease in the number of taxa and Petts *et al.* (1993) and Greenwood *et al.* (1999) found that the number of taxa present were increased downstream of the dam. It is impossible to generalise on the precise impacts of river modification on channel invertebrate fauna apart from the fact that deviation from reference conditions of biological quality seems inevitable when river hydro-morphology is altered. However, it is clear that a loss of floodplain waterbodies will lead to a dramatic loss in biodiversity as these habitats, and a range of them is vital to a complete community. Chironomid species diversity was shown to be higher in lentic than lotic habitats (Garcia and Laville 2001) and Van der Brink and Van der Velde (1991) demonstrate the distinctly different fauna that is present in waterbodies experiencing various flooding frequencies. Overall, regulation and canalisation results in a reduction of overall floodplain diversity and deviation from reference conditions (Orbdlik and Fuchs 1991; Loffler 1990).

3.3 Impact on fish

Water Framework Directive Status Definitions

High status: Species composition and abundance correspond totally or nearly totally to undisturbed conditions. All the type-specific disturbance-sensitive species are present. The age structures of the fish communities show little sign of anthropogenic disturbance and are not indicative of a failure in the reproduction or development of any particular species.

Good status: There are slight changes in species composition and abundance from the type-specific communities attributable to anthropogenic impacts on physicochemical and hydromorphological quality elements. The age structures of the fish communities show signs of disturbance attributable to anthropogenic impacts on physico-chemical or hydromorphological quality elements, and, in a few instances, are indicative of a failure in the reproduction or development of a particular species, to the extent that some age classes may be missing.

Moderate status: The composition and abundance of fish species differ moderately from the type-specific communities attributable to anthropogenic impacts on physico-chemical or hydromorphological quality elements. The age structure of the fish communities shows major signs of anthropogenic disturbance, to the extent that a moderate proportion of the type specific species are absent or of very low abundance.

Due to their complex habitat requirements, fish communities, especially larval and juvenile fish, are good indicators of habitat structure and ecological integrity of large river systems (Schiemer *et al.* 1991). Channelisation and water regulation for navigation or hydroelectric development can have a particularly severe impact on fish, with recruitment bottlenecks resulting from habitat destruction or changes in flow regime (Copp 1991). The hydrological regime and the geomorphic processes of river systems influence the dynamics of fish reproduction. Anthropogenic alteration of the river regime by channelisation and impoundment would therefore be expected to have effects on fish community structure (Copp 1990). Riverbanks and lentic backwaters (side arms, dead arms, oxbows) represent crucial resource patches for the reproduction of limnophilic fish. However, these are often the first elements to disappear from the fluvial mosaic when a river is canalised. These areas possess environmental conditions essential to the reproduction of some fish species and provide nutritional resources for developing fish, larvae and juveniles. Water retention structures such as weirs, dams and sluices impeded the flow of water required by pelagic and rheophilic spawning fish. A large number of studies have been done on the impacts of river regulation on fish populations.

3.3.1 Weedcuttting

Macrophytes provide an important refuge and egg-laying sites for many species of invertebrates and fish. In the middle Great Ouse, 0+ roach (the dominant species) were found in water less than 1.5 m deep within 8 m of the bank and in the presence of low densities of algae and emergent macrophytes. The roach mainly fed on Cladocera – *Ceriodaphnia quadrangular, C. reticulata, Alona sp.* and *Chydorus sp.* Weed cutting removed plant cover beyond 1-2 m from the banks, particularly beds of *Nuphar lutea.* Densities of rotifers and *C. quadrangular* increased after weed cutting whereas densities of *Polyphemus pediculus* and Chydarids decreased. There was a distinct shift in the distribution of Cladocera. High densities were found only at points undisturbed by weed cutting within 2 m of the bank. The roach made good use of the abundant *C. quadrangular* with high densities being found in the gut. In the longer term, rotifers followed their normal pattern of seasonal decline whereas Cladocera decreased rapidly. With this decline roach switched to aufwachs and this coincided with a decline in growth rates. The loss of *N. lutea* removed the preferred habitat of Cladocera, the main food of young fish (Garner *et al.* 1996).

Swales (1982) also looked at the impacts of weed cutting on fish using an experimental programme on the River Perry, a small lowland tributary of the Severn in Shropshire. Four adjacent 100 m sections were set up; one where the vegetation was completely removed, one with the partial removal of aquatic vegetation leaving riparian vegetation intact and two sections where the vegetation was left undisturbed. Dace and chub were the most abundant fish species with roach, perch and pike being present in low numbers. There were big fluctuations in population density between sampling sessions and between sectors and so the results were inconclusive as it was hard to distinguish natural population fluctuations from the impact of weed cutting. However there was a strong association between fish distribution and weed cover and indicated the important role that aquatic and bank side vegetation have in providing shade and shelter.

3.3.2 River regulation

Continued and extensive regulation of lowland rivers such as the Great Ouse has caused fish populations and species diversity to be reduced. Copp (1990) studied fish recruitment at various sites in the main river, side channels and backwaters along the River Great Ouse between Passenham, Buckinghamshire and Needingworth in Cambridgeshire. 0+ juvenile fish were sampled in late summer. Downstream of Bedford, the river is regulated with weirs, dredging, embankments and navigation locks. Upstream it is only subject to weirs and dredging. The Great Ouse was compared to the unregulated River Biebrza in Poland, which has similar characteristics to the Great Ouse prior to regulation. 2733 fish from 19 species were collected. The generalist species roach (*Rutilus rutilus*) and minnow (*Phoxinus phoxinus*) dominated throughout the system and often made up more than 70% of the fish population. Gudgeon (*Gobio gobio*), three spine stickleback (*Gasterosteus aculeatus*), chub (*Leucisus cephalus*), bullhead (*Coltus gobio*) and silver bream (*Blicca bjoerkna*) occasionally co-dominated. Compared to the River Biebrza the Great Ouse has a relatively poor recruitment of both limnophilic and rheophilic species. Common bream (*Abramis brama*) was only found to have a limited range and abundance. Limnophilic species were generally found at downstream sites only and an extreme rheophile, the barbel (*Barbus barbus*) was only found at two upstream sites emphasising the zonation present in the river, which is usually absent in lowland rivers. In the River Biebrza, limnophilic species such as silver bream, tench (*Tinca tinca*) and rudd (*Scardinus erythrophthalmus*) were found throughout the length of the river, principally in adjacent oxbows and abandoned side channels. The Ouse has few connected floodplain water bodies and therefore the reproduction of limnophilic species is restricted to downstream sites. The absence of the rheophile the burbot (*Lota lota*) was notable as it is a common species of unregulated lowland rivers and their floodplain water bodies and historical records indicate that it was common in the Great Ouse prior to regulation. The general absence of salmonids also corroborates the assumption that regulation has lead to the severe reduction of more sensitive rheophilic fish, of limnophilic fish requiring connected backwaters and a dominance of generalist species (Copp1990). Regulation has caused a significant deviation in the species composition of the river Great Ouse from the reference conditions illustrated by the River Biebrza.

Similar results were found by Loffler (1990) who compared two relatively similar sections of the Danube in Austria – an unaltered section near Hainberg and an altered section near Altenworth where the river is completely disconnected from the alluvial plain. Thirty species of fish were found at Hainberg, whereas at Altenworth there were only four. The reasons for the lack of fish here were postulated as the lack of benthic zones with gravel, the poor development and low species diversity of macrophytes and the separation of the backwaters from the Danube.

Jurajda and Penaz (1994) looked at the impact of river modification on fish species diversity. The River Morava in the Czech Republic was heavily modified between 1968 and 1982. The lower section of the river is completely isolated from its floodplain and former backwaters by dykes. Meanders have been removed and the banks are characterised by stony rip-rap. Fish data collected ten years after completion of channelisation were compared with pre-regulation data. After channelisation, electrofishing found 35 species of fish. Seven species occurred in all sections: roach, chub, gudgeon, barbel, bleak, top mouth gudgeon and bitterling, although of the originally dominant species only bitterling, bleak, chub and roach are still abundant. Anadromous sturgeons have disappeared completely, their migration being blocked by weirs. Of the rheophilous species, only barbel inhabits the lower stretch with nase and zährte almost extinct, again due to blockage of the river by weirs. The low abundance of phytophilous species such as silver bream, tench and rudd could be explained by the absence of suitable spawning sites and nursery areas. The results clearly indicated that the isolation of the backwaters from the main channel and the prevention of flooding have caused a significant deviation from the fish community at reference conditions on the River Morava. Before regulation there was a high abundance of barbel, nase and zährte – these have now declined almost to extinction. Similar results have been found on the Danube due to the interruption of migration patterns by dams. Anadromous sturgeon have disappeared since the construction of dams. Many other rheophilic species have declined due to the reduction in free-flowing sections and the lack of longitudinal exchange possibilities (Scheimer and Spindler 1989).

Cowx *et al.* (1981) compared the fish fauna of two regulated tributaries (the Afon Vyrnwy and Afon Clywedog) and one natural tributary (Afon Dulas) of the River Severn. The Afon Dulas had a flow pattern typical of an upland stream with seasonal fluctuations, drought and flood. In the regulated rivers, flow is reduced to a continuous compensation flow with additional down-draw released for flood mitigation. Above the Afon Clywedog, the Llyn Clywedog has been more intensively used due to increased abstraction and drought conditions in the Severn, which has resulted in a very variable flow regime in the river below. There were also changes in the temperature regimes of the regulated streams with a suppression of temperatures due to the release of cold water from the reservoir. Salmon and trout recruitment was consistent between 1975 and 1978 on the Afon Dulas and Afon Vyrnwy but on the Afon Clywedog, it had declined since 1975. Prior to 1976, there was no significant difference in the growth of trout between the three rivers but after 1976, the growth of trout was significantly faster in Afon Vyrnwy than Afon Clywedog. Regulation has caused the fishery of Afon Clywedog to decline since 1975/76. This is thought to be cause by regulation of flows and particularly the intensely variable flows. Regulation has often led to higher flows and lower temperatures in the period critical to hatching and the development of insects leading to a decline in fauna. Rapid changes in water levels also cause a downstream drift of invertebrates, which due to the dam are not replaced from upstream and this lack of food sources reduces fish growth. In addition, changes in water level cause substrate movement and erosion of gravel beds, which reduce fish spawning areas and therefore recruitment. In the Afon Vyrnwy effects of have been mitigated by the fact that it has many tributaries downstream of the dam allowing easier recolonisation.

3.3.3 Channel modification

Elimination or modification of natural river features such as channel meandering and the pool-riffle pattern together with other changes in river morphology and hydrology can be a particularly damaging to fish populations. Swales (1988) studied the status of the fish community of the River Perry, a small, channelised lowland river in England, in relation to the long-term effects of river management. A major agricultural land drainage scheme was carried out in the upper reaches of the river around the turn of the century and much of the upper reaches are channelised. Since then, regular channel maintenance works have been carried out including dredging, bank regrading and weed cutting to maintain optimal land drainage conditions. These reaches show little recovery from the original channelised conditions and there is a lack of natural features such as pool and riffle patterns and a low level of habitat diversity. Only the lower reaches remain relatively unaffected by major channel alterations with a well established pool-riffle pattern and high level of habitat diversity. The fish community was dominated by running water cyprinids - dace and chub comprised 80% of the fish caught. Atlantic salmon (juveniles) and brown trout were the only salmonid species captured and were restricted to the unmodified site. Overall, fish biomass was consistently lower in the modified reaches than the unmodified (no significance levels given). At the modified upper reaches, dace formed two thirds of the fish community whilst at the unmodified site they accounted for just one third of the population. Density and biomass of chub populations was consistently higher (up to four times) at the unmodified and intermediate sites than at the channelised reaches. Fish size was also affected with a shift towards smaller size classes of dace at the modified reaches, with fewer intermediate or large fish compared to the more natural reaches. The changes in the fish community was thought to be due to the low level of habitat diversity at the channelised sites. Chub are often associated with areas of in-stream cover such as undercut banks and submerged snags, which were lacking at the channelised sites. Reductions in stream cover may favour smaller fish.

On the River Morava, Jurajda (1995) investigated the impact of channelisation on river fish by looking at the nursery preferences of juvenile fish. Stone loach and bitterling showed no preference between stony rip-rap and shallow gravel shorelines. Only chub showed a preference for the rip-rap. Other species such as dace, bleak and roach preferred gravel habitats. Species richness is almost the same now as before modification but the fish assemblage is completely different. Channelisation and the construction of embankments cause a reduction in habitat heterogeneity. At the beginning of the 1980s, studies on the effects of straightening rivers on fish stocks (e.g. Jingwirth 1984) showed that straightening causes a reduction in complex habitats and a subsequent reduction in fish fauna and biomass by up to 90% (Jingwirth *et al.* 1993).

Scheimer *et al.* 1991 also studied the impact of channelisation on fish fry abundance and diversity in different parts of the free-flowing section of the Danube downstream of Vienna, which is influenced by engineering – the river channel has been straightened and strongly embanked, cutting off many lateral river branches. A high number of fish species have been recorded in recent surveys of the area, the largest group being cyprinids (29 out of 50 species). The rheophilic group is the largest, dominated by nase, barbel and Danube gudgeon and has the highest number of endangered species. The fish fry populations found in different habitats were compared. The fry of rheophilic fish is found exclusively in the naturally structured littoral zone of the main channel. Species diversity and rarity of 0+ fish is high in small bays and shallow sloped gravel banks whilst the lowest population density and species diversity is found with the linear rip-rap. In the free flowing zone downstream of Vienna only 18 zones of 1-2 km have been identified as forming potential fish nurseries. Fish require microhabitat diversity. In the larval stage, all fish are bound to the sheltered lentic bays on the shoreline. With increasing age, rheophilic species migrate to the shallow gravel banks. It is also important to have connection to the backwaters – such lentic water bodies are major production zones for plankton, which provides food in early life stages. Only 10% of inshore zones provide high quality habitat for fish. 60% of banks are covered with rip-rap which are practically devoid of fry.

3.3.4 Land drainage

Swales (1982) looked at the impact of land drainage works on fish by comparing the fauna before and after channel alterations were carried out on the River Soar, Leicestershire (1978-79) in an attempt to reduce flooding severity and frequency from once or twice a year to once in every five to ten years. The major change in the fish habitat was a decrease in the amount of stream cover because of bank regarding, the dredging of the river bed and the clearance of trees and bushes along one bank. Prior to drainage, dace, chub and roach were the dominant species with occasional brown trout. One month after the completion of channel work the density decreased by 90% and the standing crop density was reduced by 85% for all species. Three months after completion of the channel works there was an increase in the fish population but they were still well below previous levels. Brown trout had disappeared completely; chub had reduced by 87%, dace by 73.5% and roach by 30.5%. This study provides evidence for the deleterious effects of drainage works on fish stocks. However, the short-term nature of the study means that it is not possible to say whether the effects are caused by disturbance because of channel excavations or a more long lasting effect due to changes in channel structure. This study was continued by Cowx (1986) who compared pre-impact data with fish stocks 5 years later on regraded sections of the river and a 200 m stretch that had been unaffected which has intact pool-riffle topography and abundant macrophytes. Prior to regrading, cyprinid fish density had been low (0.0506 fish m^{-2}) but afterward there were no cyprinid fish in the regarded section for five years, although minnow, sticklebacks and stone loaches remained. After five years, small numbers of roach and dace were caught. Meanwhile in the unmodified section the population increased dramatically. The study demonstrates that the effects of drainage can persist over a long period of time. The loss of fish is likely to be due to the loss of in-stream cover and the removal of the pool-riffle character of the river. The natural area may act as a refuge for displaced fish as well as a spawning and nursery area so recolonisation could eventually occur. If this area were not present, recolonisation may have taken longer.

3.3.5 Loss of connectivity

The high species richness of the Danube between Vienna and the Austrian and Czech border is thought to be due to the habitat diversity of the river-floodplain system (Scheimer and Spindler 1989). There are three types of fish: those strictly bound to the river such as barbel and zährte, riverine species that during certain periods of their lifecycle require zones of low flow such as backwaters with open connection to the river, such as nase, Danube bream and zope. There are also species that are bound to disconnected branches of the river with a strong vegetation development such as moderlieschen, pond loach, crucian carp and tench. Eurytopic species such as roach, pike, bleak and schnieder are found in all habitat types. Connectivity is important for high species diversity. Open branches contain the most diverse and endangered fauna as a range of ecological groups occur, including limnophilic species and riverine species that use the backwater habitat. In disconnected backwaters the fish fry are composed of eurytopic and limnophilic species (Scheimer and Spindler 1989).

When floodplain habitat diversity is lost, this leads to a deviation from reference conditions in terms of the fish community. Scheimer *et al.* (1991) studied the free-flowing section of the Danube downstream of Vienna which, although relatively connected is influenced by engineering – the river channel has been straightened and embanked, cutting off many lateral river branches. The fish fry populations of five different habitats (disconnected backwaters, connected backwaters sheltered bays on river shore, gravel banks on the river shore, and rip rap on the river shore) were compared. It was found that the highest number of individuals and species were found in the connected backwaters where the fish fry is eurytopic. Ward *et al.* (1999) quote data from Scheimer *et al.* (1994) and Spindler (1997) who found that there was a decrease in species richness with decreasing of connectivity from partially connected side channels to isolated water bodies, irrespective of the overall connectivity. For example on a well-connected floodplain, richness decreases from around 27 species to around 12. The proportion of rare species also decreases with decreasing connectivity as most endangered species are rheophilic and occur in the channel whereas disconnected floodplains are inhabited by less specialised species with a low proportion of endangered species. It is also important to have connection to the backwaters because lentic water bodies are major production zones for plankton, which provides food in early life stages.

3.3.6 Summary

River and floodplain modification has been illustrated to have a dramatic effect on fish population. This occurs in several ways. Firstly, canalisation leads to a reduction of habitat diversity within the channel and a loss of spawning and nursery areas for rheophilic fish. Weed cutting causes a removal of shade and shelter both for invertebrate prey and for juvenile phytophile fish. Weirs and dams limit the distribution of anadromous as their migration passages are disrupted. Particularly important in the deviation from the reference condition is the loss of diversity of, and connectivity with, floodplain backwaters. Limnophilic species restricted to these habitats are lost, and decreased connectivity leads to a decline in those species that use connected backwaters as spawning grounds. In general, river modification causes a deviation from the fish community at reference conditions where there is a large diversity of specialist species utilising a large range of lotic and lentic conditions on the floodplain, to a much smaller number of generalist species.

3.4 Restoration of the river channel

In recent years, there has been interest in restoring river channels in order to attempt to recreate the biodiversity of the natural channel. This can be done though active re-meandering of the channel to restore channel morphology or more simply on channelised streams by simply changing channel maintenance, for example a cessation of weed clearance and dredging. Several studies have been carried out looking at the short and long-term impacts of channel restoration. Immediate impacts or channel restoration on the River Brede (Denmark) and the River Cole (UK) were investigated by Briggs *et al.* (1998). Channelisation of the River Brede was carried out in the 1950s. Restoration in 1994 involved remeandering a 3.2 km section and raising bed levels by 0.5 m. At the River Cole a new channel was created and existing channels reshaped over 2 km and bed levels were raised by 1 m. On both channels, wetland vegetation was allowed to recolonise naturally. Vegetation and aquatic macroinvertebrates were surveyed in the restored areas and in upstream control sections. Restoration eliminated most of the river channel and therefore colonisation occurred from scratch. In both rivers, aquatic macroinvertebrate colonisation was rapid and after one year species richness was only slightly below pre-restoration values. However, on the River Cole rare species were significantly less abundant than in the original channel – only 4 out of 13 species had recolonised after one year. Wetland macrophytes also quickly reached pre-restoration levels although aquatic plant species were slower and cover was still well below pre restoration levels after one year (22.7% as compared to 60%) (Briggs *et al.* 1998).

Frieberg *et al.* (1998) carried out a longer-term study of the effects of restoration. In summer 1989 a 1.3 km section of the straightened and canalised River Gelsa in Jutland, Denmark was restored to a 1.85 km meandering course. Channel width was decreased and discharge capacity reduced. This section was compared with an un-restored site where intensive management had been stopped. The sites were sampled in 1989 before restoration when dredging and mechanical weed occurrence occurred at both sites. Subsequent samples were taken after restoration in 1990, 1991, 1993 and 1995. Before restoration in 1989, around 2500 benthic macroinvertebrates were collected in both reaches. In 1990, the number in both reaches increased to 6000. In the restored reach the total peaked at 11000 in 1991 then decreased to the 1990 level in 1993-95. Upstream numbers were constant between 1990 and 1995. Species diversity also increased. In 1989, there were 47 species in the subsequently restored reach and 42 species in the control reach. In 1990, the restored reach had 45 species and the control 55 species, and in 1991, the restored reach had 70 species and the control 56 species (Frieberg *et al.* 1994). In 1991, several species associated with coarse substrates were only found in the restored reach such as Ephemeroptera (*Heptagenia sulphurea*) and Trichoptera (*Hydropsyche siltalai*). Species with poor dispersal abilities such as leaches (Glossiphonia complanata) and a beetle (Limnius volckmari) were absent in the restored reach in 1991 but present in 1993 and 1995. The macroinvertebrate community had stabilised by 1993 and the macroinvertebrate community had improved in terms of biomass and species richness. The upstream reach also recovered well and was still improving in 1995. When macrophytes are not cleared, they can improve the physical heterogeneity of the channel (Frieberg *et al.* 1998).

It has been shown in some studies that restoration of rivers can help to reinstate fish biodiversity back to levels that are closer to natural. For example, a monotonously straightened section of the Melk River in Austria was restructured in 1987. A 1.5 km section was surveyed for fish before restructuring and three times afterwards at annual intervals. Between 1980 and 1987, there were 10 fish species found in this section of the river. One year after restructuring the number of fish species had increased to 16 and by 1990, there were 19 species whilst in the still straightened reference section the number of species remained at 10. Recolonising species include American brook trout, bitterling, dace, carp, nase, roach, rudd, schneider and spined loach. The two species (chub and gudgeon) that were previously dominant were reduced by 50% in the restructured section allowing more balance within the fish community (Jingwirth 1993).

River rehabilitation projects to reform features such as riffles and pools for the benefit of fish populations are widespread in Britain and generally involve restoring in-channel features lost through channelisation such as riffles and pools. The effects on biota, however, are poorly understood, particularly for small scale projects. Pretty *et al.* (2003) looked at two common in-stream methods – artificial riffles and flow deflectors and assessed their benefits for fish in rivers in Eastern and central England. In each of the 13 rivers studied, the rehabilitation scheme was compared with a control reach 100 – 500 m up or down stream that had similar riparian structure and water quality to the manipulated reach. Both riffles and deflectors tended to increase the flow velocity heterogeneity and depth heterogeneity. However, there was no relationship between these changes and the fish community. Where deflectors had been installed, the abundance of stone loach and bullhead tended to be higher than the control but the difference was not significant. Where riffles were added there was no significant difference between the manipulated and control reaches in terms of total fish abundance, species richness or diversity. Overall, there was little evidence that adding artificial riffles and flow deflectors substantially improved the conservation value of the fish assemblage. It is likely that physical conditions were not altered over a large enough area to have a significant effect on the fish. Physical rehabilitation may not necessarily lead to biological rehabilitation. These schemes are aimed at enhancing the current environment rather than restoring it to reference conditions. A reason for their failure may be that these schemes are inappropriate for lowland rivers. In lowland rivers, vegetation plays a more important role in maintaining habitat complexity. In addition, floodplain habitats have a crucial role in the ecological functioning of lowland rivers. Vegetated margins and backwaters connected to the main channels are important habitats for lowland fish, especially juveniles and therefore the creation and conservation of off channel, marginal and floodplain habitats may be more important.

4 IMPORTANCE OF FLOODPLAINS FOR WATER QUALITY

4.1 Introduction

Although the focus of this work is on physical alterations to the channel, and the subsequent loss of connectivity and floodplain backwaters habitats, there are other important connections between the river and its floodplain. Floodplains are also important regulators of the movement of energy and materials through the catchment toward the river and water flowing from surrounding hills and across the floodplain is an important flux.

Diffuse pollution of freshwaters, particularly from urban and agricultural land is an on-going problem. It has been shown that, if floodplains are retained and maintained in a state close to “natural” so that there is a buffer zone between agricultural land and the river, this can reduce inputs of nutrients into the river. Often, however land is ploughed very close to the stream – in Zealand, Denmark 50% of stream banks had a margin of less than 0.25 m (Vought et al.). Floodplains are frequently drained for agriculture, removing the fluctuating water levels which are important if the system is to be sustained, because riparian ecosystems are adapted to water level variation, leading to varying degrees of soil oxidation, which in turn are critical to nutrient retention. River channelisation lowers the stream bed and decreases the zone of saturation, altering floodplain functions. This can permanently lower the water table below the root zone, which is disastrous for nutrient removal, because there is no interaction between the roots and nitrate moving in water flow, reducing root uptake and carbon supply. In an undisturbed condition, the stream and floodplain function as a single unit; anthropogenic disturbance of the riparian zone is therefore in part the cause of reduced stream quality (Fennessey and Cronk 1997). There is a growing interest in restoring floodplain systems to act as a buffer zone between the floodplain and the river.

4.2 Mechanisms of nutrient reduction

Mechanisms of nutrient retention include denitrification, assimilation by vegetation and transformation to ammonium and organic nitrogen followed by retention in the soils. All three mechanisms may be important in different seasons and environments (Correll 1996). The main method of nitrate removal from subsurface flow is usually denitrification, which is dependant on a source of nitrate and organic carbon, together with anaerobic conditions. Natural floodplains can provide ideal conditions for denitrification to occur, due to the occurrence of both saturated and non-saturated soils, a high organic matter concentration and a high water table, lowering oxygen content. Rates of denitrification are a function of nitrate concentration, the amount of organic carbon, the degree of soil saturation, the activity of denitrifying bacteria, temperature and pH of the system. Vegetation is important in maintaining these conditions. Plants must have high primary productivity and enough of the resulting photosynthate must be released below ground to provide enough electrons to drive soil reactions (Correll 1996). Nitrate removal was found to be linear with 20% removed in the first 8 m and 50% in 16 m. Beyond a width of 20-25 m the efficiency of the buffer strip does not seem to increase (Vought *et al.* 1994).

Uptake by plants is a second major path of nitrate removal but this route may not be permanent as it may be leached back from plants, return with plant senescence or animal faeces. Although vegetation uptake is not the most important route for nutrient removal, vegetation has a crucial role to play. Its physical presence promotes sedimentation and prevents erosion, litter provides habitats for microbe colonisation and it improves soil structure and infiltration capacity. Nitrate removal is maximised if water flow is subsurface, especially in winter months and if soils are saturated (Fennessey and Cronk 1997).

More than 70% of phosphorous export from fields is particle bound and therefore enters waterbodies through surface runoff (Vought *et al.*1994). The role of riparian vegetation in trapping sediments and adhering phosphorous has been reported. Vegetation reduces the velocity of the water causing deposition to occur. The fine plant roots and microbial communities near the soil surface assimilate the nutrients. The phosphorous adsorbtion rate depends on amount of clay minerals, aluminium and iron oxides, organic matter and $CaCO_3$. In general phosphate is removed efficiently from surface runoff and the greatest removal is closest to the source with 60% removed within 8 m and 95% within 16 m on an experimental grass buffer strip (Vought *et al.* 1994).

There is uncertainty about the relative roles of different vegetation types. It may be that grass and herbaceous vegetation are more efficient at trapping particulates from storm flow and woody vegetation may be more efficient at removing pollutants from groundwater (Correll 1996).

Grass and beech buffer strips with the same slope angle and a separation of 50 m were compared. It was found the surface runoff extended 16 m on grass strips and only 4 m on beech strips. Riparian buffer strips can be highly efficient at removing nitrate from subsurface water. A buffer strip of 10-20 m will in most cases retain the major part of nitrate and phosphate in surface runoff and most N in subsurface flow. However, retention of phosphate in subsurface flow is very variable and can, under conditions favourable to denitrification be replaced by phosphorus release due to low oxygen conditions (Vought *et al.* 1994).

4.3 Riparian forest case studies

Many lowland floodplains in the UK were originally forested but today there is little riparian forest left, as the land has been cleared for agriculture and urbanisation. Under natural conditions riparian forest ecosystems formed a dynamic yet stable buffering system along most rivers and streams. Riparian forests of mature trees (30-75 years old) are known to effectively reduce non-point source pollution from agricultural fields in certain landscapes. Riparian forest buffer systems can be established to sustain water quality functions over the long term, in a manner similar to natural riparian forests (Lowrance 1996). Lowrance *et al.* (1984) looked at the effectiveness of woody vegetation as a nutrient filter in the watershed of the Little River, Georgia, USA which has 30% riparian forest, 41% arable crops and 13% pasture. The input of nutrients, in fertiliser and lime to the agricultural land is high in this catchment. The riparian zone is almost intact and the dominant tree species in the riparian forest are black gum, tulip tree, sweet bay, red maple and water oak. All watershed nutrients must pass through this zone on their way to the stream channel. Studies showed that 80% of total runoff and 99% of nitrate was moved from the fields as subsurface flow. 68% of nitrogen was retained, 39% of calcium, 30% of phosphorous, 23% of magnesium, 7% of chlorine and 6% of potassium. Gaseous losses of nitrogen were apparently more important than gaseous inputs and losses of nitrate though denitrification were more than twice stream loads. The soils of the riparian zone had ideal conditions for denitrification with high organic matter, seasonal water logging and large inputs of nitrate in subsurface flow. Denitrification alone was enough to remove all nitrate inputs from upland fields to the riparian zone.

Haycock and Pinnay (1993) compared the effectiveness of buffer strips dominated by grass and by poplar forest to determine what effect the type of vegetation has on nitrate removal. It was found that for nitrate, the woody strip was much more efficient removing up to 100%, whilst the grassed strip removed around 84%. This was thought to be because the tree root system produces more organic carbon at greater depths in the soil profile.

4.4 Floodplain meadow case studies

Artificial flooding of meadows has been done since medieval times and results in a large wetland area, which provides favourable sites for waders and waterfowl. Leonardson *et al.* (1994) looked at whether these meadows decreased the nitrate content of streams and rivers in southern Sweden. It was found that outflow water from the meadows had the same total nitrate as that flowing in but nitrate was decreased and ammonium had increased. Overall, nitrate retention during artificial flooding amounted to around 15% of the total. Longer irrigation periods and higher nitrate loading resulted in higher total nitrate retention. In spite of the low retention efficiencies denitrification rates were high (60-95% of the total nitrate load). According to mass balances, mineralisation was intense and supplied denitrification with nitrogen and resulted in ammonium leakage from the area. However, Danish studies show that natural wet meadows retain nitrate more efficiently. Brusch and Nilsson (1990) and Ambus and Hoffman (1990) report retention of 55% and 70-76% of annual nitrate load. These meadows were almost continuously supplied with water which passed the meadows as subsurface flow and created stable conditions and low oxygen concentration in the soil.

Haycock and Burt (1993) monitored the nitrate concentration of groundwater as it entered and left a floodplain on the River Leach, UK to evaluate the floodplain's nitrate absorption capacity Floodplain sections were monitored where the floodplain soils have intensive arable farming. The input of nitrate to the floodplain was principally controlled by subsurface water flow and the principle source was the hill slopes. A transect was done across the floodplain and it was found that between rows six and four the mean concentration of nitrate in groundwater decreased by 82%. They concluded 12-20% was due to dilution by up-valley water and 60-70% due to biological processes. The most probable mechanism was denitrification and this occurred within a very short distance of the hill slope groundwater flowing onto the floodplain. A maximum of 97% of groundwater nitrate load was lost. The results suggest that if hill slope water is allowed to interact with floodplain sediments then denitrification could considerably reduce the nitrate content of the subsurface water and so provide an effective water protection process.

4.5 Conclusion

Studies have shown that floodplains maintained in a relatively “natural” state act as buffer zones between agriculture and urban land and the river can reduce both nitrate and phosphate pollution of surface waters. In order for floodplains to function effectively, water tables must be kept high and vegetation must be productive in order to produce carbon for denitrification reactions. Both forests and grasslands have been shown to be effective in nutrient retention.

5 COMPARISON BETWEEN A UK LOWLAND FLOODPLAIN RIVER AND A RIVER AT REFERENCE CONDITION

Reference condition: River Biebrza (Photograph: W. Walkow)

Not reference condition: River Great Ouse (Photograph: Dorian)

Within the Water Framework Directive, the biological quality of an aquatic ecosystem is assessed by determining the degree to which it deviates from reference conditions. The site chosen to represent reference conditions should have hydromorphic conditions and taxonomic composition and abundance which correspond totally or nearly totally to undisturbed conditions. In the UK there are no lowland floodplain river systems remaining that meet these criteria. However, it has been suggested that the Biebrza Valley in Poland offers such a reference site for a floodplain and fen ecosystems (Wassen *et al.* 2002) and has been considered to have sufficient geographical and geomorphological similarity to be used as a reference system against which to compare the River Great Ouse in the UK (Copp, 1990, 1991). The two rivers are described below and compared as an example of the way the deviation of the River Great Ouse from reference status may be determined.

5.1 Introduction to the River Biebrza, Poland

The Biebrza is 164 km long with a catchment area of 7062 km^2. Almost the whole basin of the river is little affected by human activity and its natural character and hydrological patterns are almost wholly preserved, never having being dammed, diverted, regulated or embanked. It flows through boggy meadows and marshes throughout its course, meandering considerably and forming a large number of old riverbeds and floodplain waterbodies in different stages of succession. During the spring, the narrow river swells to form a vast shallow lake, in places a kilometre wide, and lasting for several months. The long periods of saturation create ideal conditions for mosses that grow and leave their tiny plant skeletons in place year after year. In this stable environment, undisturbed for millennia, the microscopic deposits accumulate to form peat, in places up to 10 m deep, and estimated to be more than 10 000 years old.

Water quality in the Biebrza is very high due to the neutralising and protective action of a broad area of flooded meadows, peatlands and marshes that mean that settlements and fertilized arable land are on the valley margins. Atmospheric deposition is low compared to Western Europe. The pH of the river is 7.15-8.34 as acidity is buffered by high calcium concentrations. Oxygen saturation is very high – 80-93% at the surface.

The vast wetland area created by the abundance of floodplain water bodies and regular flooding has abundant diversity. The Biebrza Valley supports some of the greatest bird species diversity in Europe with a total of 269 species, 186 of them breeding, including black grouse, greater spotted eagle, spotted eagle, white-tailed eagle, eagle owl and white-backed woodpecker. Over 60 plant communities are present in the Biebrza valley, including nearly all the water, marsh and peatland plant communities found in Poland. 872 species of vascular plants have been found in the floodplain, among them many are rare, under extinction, or endangered. There are 67 protected species, including glacial relicts and species of boreal origin. This high diversity of vegetation is a function of the intact vegetation gradient from the river across floodplain to valley margin, caused by the gradient of flood inundation frequency. River floodwater aids the maintenance of species rich, moderately nutrient rich fen and meadow vegetation (Wassen 2002). The whole river except 10 km is within a National Park and protected under the Ramsar convention

5.2 Introduction to the River Great Ouse, UK.

The catchment area of the River Ouse is of a similar magnitude to that of the Biebrza. At 8600 km^2 it is one of the largest catchments in England. In the lower reaches, much of the catchment is below sea level and in its natural state the River Great Ouse would, like the Biebrza, meander through a floodplain of meadows, marshlands and peat bogs. During the early middle ages the river worked its way tortuously to the sea through a complex system of meandering channels, often breaking its banks and extending into wide shallow pools (Pinder *et al.*1997), similar to the Biebrza today. Although some small-scale drainage occurred in the medieval period, the main drainage of the fens was carried out in the seventeenth century. In the middle reaches of the river, between the fens and Bedford, the river has been impounded, regulated and channelised over several centuries for land drainage and flood control, to provide water for the many mills that were once on the river and to facilitate navigation. Regulation attained its maximum development in the 1970s with extensive dredging and channelisation along with the installation of modern locks and automatic sluices associated with the restoration of navigation and improvement in drainage and flood control (Pinder 1997). Upstream of Bedford, the river has weir construction, land drainage (dredging, weed cutting) and the reclamation of abandoned meanders. Downstream regulation for navigation, flood control and water retention has resulted in the loss of many flood meadows, oxbows and floodplain water bodies. A dredged and embanked channel has been created with few natural features. No natural oxbows are known to exist and semi-abandoned channels are rare (Copp 1991). The catchment is now mainly agricultural and contains a large number of urban settlements.

Historical evidence shows that over the long term, there has been slow but progressive change in community structure and biodiversity associated with gradual improvements in drainage and isolation of the river from the floodplain. The most conspicuous casualty is the burbot, which, although it was sufficiently common in medieval times to be valued as a food source, was driven to extinction by the mid twentieth century. In Saxon and medieval times, fenland waterways including the Great Ouse supported a great abundance of fish. This seems to have persisted – historical documents from 1774 indicate good catches of pike, perch and eels. By 1878 there were reports that “In recent years the fish of the fen waters have been greatly reduced in quantity”. This was attributed mainly to drainage of the fens. However, this document still lists 22 species of fish including burbot and sturgeon, which are now extremely rare in English rivers.

Floodplain backwater destruction has reduced the lentic habitat patches for phytophiles. Natural rejuvenation and the creation of biotopes by erosion and aggradation, for the maintenance of habitat diversity is reduced by embankments. This is reflected by low recruitment, as there are few sites for rheophilic species such as barbel, dace, bleak and stone loach. The rejuvenation of backwaters by natural allogenic processes is limited to seasonal floods when lentic side channels and backwaters in direct contact have increased water velocities. There is little transversal heterogeneity in floodplain habitat.

5.3 Comparison of the rivers

The primary difference between the River Great Ouse and its reference comparison is demonstrated by the photographs above. While the Great Ouse is confined to a single channel on a drained agricultural and urban catchment, the Biebrza floodplain is a complex mosaic of semi-abandoned and abandoned (oxbow) channels. The backwater habitat on the Great Ouse is reduced to an occasional natural side channel, but is mainly pleasure boat marinas and remnant water mill works. Flooding on the Great Ouse is regulated and infrequent, whereas on the Biebrza flooding is annual and extensive ensuring connectivity between the channel and floodplains. Historical data shows that pre-industrialisation, these characteristics of the Biebrza were also those of the Great Ouse. Whereas the Biebrza exhibits a vegetation gradient from the river across floodplain to valley margin the Great Ouse has little transversal heterogeneity in floodplain habitat.

Comparison of the flora and fauna of the two rivers should give an indication of what the impact of regulation and human influence has been on the River Great Ouse and the deviation from reference status that has occurred. The most detailed data available is for fish communities. Witkowski (1984) made a detailed study of the fish community of the Biebrza between 1976 and 1980 and this data was used by Copp (1990) as a comparison with their own data collected on the River Ouse. Witkowski (1984) studied adult fish over the whole river basin, divided into four main areas:

(A) Large old riverbeds permanently connected to the river – these are youngest river beds.
(B) The middle and lower reaches of the Biebrza River.
(C) Upper reaches of the river
(D) Small and shallow old river beds only periodically in contact with the Biebrza. These are the oldest riverbeds.

The tables below show the results for each of these sections:

Table 5 – Results of fish surveys on the River Biebrza (Witkowski 1984)

		Individuals ha^{-1}		Biomass (kg ha^{-1})	
		Range	Mean	Range	Mean
Old river beds	A	2400 - 3362	2866	96.77 - 162.45	142.32
	D	420 – 2375	1180	89.50	89.50
Rivers	B	1350 - 2800	1814	77.32 - 156.47	117.22
	C	991 - 5638	2927	10.00 - 13.92	75.62

The fish community is rich in terms of numbers and biomass of individuals. The old river beds are at least as important as the lotic channels as fish habitats. They form a rich array of habitat types being in different stages of succession and are especially in need of protection.

Table 6– Species composition on different sections of the River Biebrza (Witkowski 1984) (Fish types – phyto = phytophile, gen = generalist, litho = lithophile).

	Dominant species	Fish type	% Biomass		Dominant species	Fish type	% Biomass
A	Pike	Phyto	33.17	C	Pike	Phyto	26.84
	Roach	Gen.	20.02		Ide	Litho	21.23
	Rudd	Phyto	13.54		Burbot		17.43
	Tench	Phyto	13.20		Roach	Gen.	16.68
B	Ide		32.36	D	Tench	Phyto	34.39
	Roach	Gen.	18.64		Pike	Phyto	28.99
	Pike	Phyto	16.94		Rudd	Phyto	11.83
	Burbot		12.58		Roach	Gen.	15.34

Phytophiles spawn among dense beds of submerged macrophytes. Their habitats are rare in the Great Ouse and phytophiles are restricted in numbers and distribution. Lithophiles need fast flowing areas with well-oxygenated gravel in which to spawn. Generally, the recruitment of rheophilic species is poor. Phytolithophiles e.g. roach use a whole range of habitats. Other less common species found on the Biebrza include dace, chub, minnow, asp, gudgeon, bleak, white bream, bitterling, crucian carp, stone loach, three spined stickleback, perch and ruff. The old river beds form a rich array of habitat types being in different stages of succession and are especially important and in need of protection. The species composition and fish communities are typical for a lowland river in Poland. They are almost natural and both the numbers and biomass of fish is very high.

Copp (1990) studied fish recruitment at various sites in the main river, side channels and backwaters along the River Great Ouse between Passenham, Buckinghamshire and Needingworth in Cambridgeshire. A total of 2733 fish (compared to 50 000 on the river Biebrza, although these were collected over a much longer time period) from 19 species were collected using electrofishing. The generalist species roach (*Rutilus rutilus*) and minnow (*Phoxinus phoxinus*) dominated throughout the system (Table 7) and generalist species often made up more than 70% of the fish population. Gudgeon (*Gobio gobio*), three spine stickleback (*Gasterosteus aculeatus*), chub (*Leucisus cephalus*), bullhead (*Coltus gobio*) and silver bream (*Blicca bjoerkna*) occasionally co-dominated. Compared to the River Biebrza, the Great Ouse has very different dominant species with relatively poor recruitment of both limnophilic and rheophilic species. Limnophilic species were generally found at downstream sites only and an extreme rheophilic, the barbel (*Barbus barbus*) was only found at two upstream sites emphasising the zonation present in the river which is usually absent in lowland rivers. In the River Biebrza limnophilic species such as silver bream, tench (*Tinca tinca*) and rudd (*Scardinus erythrophthalmus*) were found throughout the length of the river in adjacent oxbows and abandoned side channels. The Ouse has few connected floodplain water bodies and therefore the reproduction of limnophillic species is restricted to downstream sites. The absence of the rheophile the burbot (*Lota lota*) was notable as it is a common species of unregulated lowland rivers and their floodplain water bodies. Historical records indicate that it was common in the Great Ouse prior to regulation. The general absence of salmonids also corroborates the assumption that regulation has lead to the severe reduction of more sensitive rheophilic fish and a dominance of generalist species (Copp1990).

Table 7– Dominant species of the Great Ouse floodplain

	Dominant species	% Biomass		Dominant species	% Biomass
A	Roach	35.92	C	Roach	45.06
	Bream	24.6		Minnow	16.97
	Minnow	12.45		Chub	11

	Chub	9.47		Bream	11.16
B	Minnow	53.93	D	Roach	34.61
	Roach	18.36		Minnow	16.97
	Chub	10.7		Chub	6.9
	Bullhead	6.5		Bullhead	6.54

A = backwaters, B = downstream river, C = upstream river, D = side streams

Information on aquatic vegetation is less detailed. The maximum cover of submerged and floating plants was generally recorded after midsummer in the Great Ouse. The cover was 40% at Godmanchester and Houghton and less at Offord and Huntingdon. Around half the sites recorded as having macrophytes present had low densities of around 20%. The most conspicuous and widespread species were *Nuphar lutea,* also *Sparganium emersum* (Pinder *et al.* 1997). In the Biebrza old river beds have emergent plant cover of around 60% at the water surface and at the younger old channels vegetation cover was up to 90%. Dominant plant species are *Nuphar luteum, Nymphaea alba, Potamogetan natans, P. crispus, P. Leucens, Polyonim amphibium* and *Sagittaria sagittifolia.*

5.4 Conclusions

The hydrological regime, river continuity and morphological conditions of the River Great Ouse have been dramatically altered in comparison to the River Biebrza, representing reference conditions. Within the Water Framework Directive, good hydromorphological status is determined by conditions consistent with the achievement of good status for biological quality elements. The fish community illustrates that such dramatic hydromorphological changes cannot be undertaken without significant changes to the biological communities. In the River Biebrza, around half the fish fauna are found in old channels on the floodplain – on the Great Ouse these channels are almost non-existent. There are large reductions in fish fauna and changes in the composition of species with specialist fish being replaced by generalist species. The changes observed in the river from reference condition are certainly greater than could be described as “slight” and therefore the status of the river could be described as moderate at best, and possibly lower status.

6 CONCLUSIONS AND IMPLICATIONS FOR WFD CLASSIFICATION

6.1 Study summary matrix

Study		Location	River type				Modification							Study type					Confidence in results				
Author	Year		Meandering	Braided	Stream	Floodplain	Channelisation	Impoundment / regulation	Weedcutting	Restoration	Straightening	Drainage	Disconnection	Within river	Between river	Historical document	Before & after	Floodplain waterbodies	Significant deviations	Change in abundance	Change in community	No. repetitions	Control
Macrophytes																							
Baatrup-Pedersen	2002	Denmark							✓					✓	✓				✓		✓	2	✓
Bournette & Arens	2002	France				✓							✓	✓					✓		✓	63	n/a
Bravard et al.	1986	France		✓			✓	✓								✓			×		✓	1	×
Englund	1997	Sweden			✓			✓							✓				×		✓	52	✓
Hey et al.	1994	UK					✓				✓			✓	✓				×		✓	5	✓
Loffler	1990	Austria	✓										✓	✓					×		✓	1	✓
Nillson & Jonsson	1995	Sweden			✓			✓							✓				✓		✓	4	✓
Nillson et al.	1991	Sweden						✓							✓				✓		✓	1	✓
Rorsett et al.	1989	Norway						✓								✓			×	✓	✓	3	×
Vanpooten & Klein	1999	France	✓					✓			✓			✓					×		✓	1	×
Williams *et al.* (in press)		UK				✓							✓	✓									n/a
Invertebrates																							
Armitage	1978	UK		✓				✓							✓		✓		×		✓	1	✓
Dessaix et al.	1995	France						✓									✓		×	✓	✓	1	×
Garcia & Laville	2001	France	✓			✓							✓					✓	×		✓	1	n/a
Garner et al.	1996	UK	✓						✓					✓					×	✓	✓	1	×
Greenwood et al.	1999	UK						✓						✓					×	✓	✓	1	✓
Homes et al.	1999	Germany		✓		✓	✓											✓	×		✓	1	✓
Inverarity et al.	1983	UK						✓							✓				✓		✓	1	✓
Loffler	1990	Austria	✓										✓	✓					×		✓	1	✓
Orbdlik & Fuchs	1991	Germany					✓								✓				×		✓	1	✓
Peeters & Tachet	1989	France		✓			✓							✓					×	✓		1	✓
Petts and Greenwo	1985	UK						✓						✓	✓				×	✓	✓	1	✓
Petts et al.	1993	UK						✓						✓					×		✓	1	✓
Scullion et al.	1982	UK						✓							✓				✓		✓	1	✓
Van der Brink & Va	1991	Netherlands	✓				✓	✓										✓	✓		✓	100	✓
Fish																							
Copp	1990	UK	✓				✓	✓							✓				×		✓	1	✓
Cowx et al.	1981	UK						✓							✓		✓		×	✓		1	✓
Cowx et al.	1986	UK	✓									✓		✓					×	✓	✓	1	✓
Garner et al.	1996	UK	✓						✓					✓					×	✓		1	×
Jarajda	1995	Czech	✓				✓				✓		✓				✓		×		✓	1	n/a
Jarajda & Penaz	1994	Czech	✓				✓				✓		✓				✓		×		✓	1	×
Jingwirth et al.	1993	Austria	✓							✓				✓					×		✓	1	×
Loffler	1990	Austria	✓										✓	✓					×		✓	1	✓
Scheimer et al.	1991	Austria					✓						✓					✓	×		✓	1	n/a
Scheimer & Spindle	1989	Austria					✓				✓						✓		×		✓	1	n/a
Swales	1988	UK	✓				✓							✓					×	✓	✓	1	✓
Swales	1982	UK										✓							×	✓		1	×
Swales	1982	UK							✓					✓					×			1	✓

Within the above grid, ticks represent the river type, study type and river modification studied. A "significant" deviation is ticked if there is a change in the biological community in terms of species number or abundance or individuals calculated by the study authors as statistically significant and assigned a probability referring to the chance of the differences occurring due to chance. A cross means that the authors do not give significance levels. Ticks also represent whether authors describe a change in community or abundance of individuals (significant or otherwise) and the number of replications is given.

6.2 Data availability

A number of studies have been done on the impact of hydromorphological changes on the biota of European rivers, both within the river channels and floodplains. There are a number of inevitable limitations upon this kind of study. River-floodplain ecosystems are extensive and dynamic. Changes to river channels such as canalisation will have impacts over both large spatial and large temporal scales, which makes field study difficult. In addition, even if a comprehensive data set is achieved from a system, a set of data from a single river modification is not enough to draw statistically significant conclusions – replication is needed. Some studies into the impact of river modification on macrophytes do compare several similarly modified and unmodified rivers; e.g Nilsson and Jonsson (1995) compared four regulated and four unregulated rivers in Sweden, and Baatrup-Pedersen *et al.* (2002) compared two weed cut rivers in Denmark. As demonstrated in the matrix above, these studies are exceptional, due to the difficulties in finding sites with suitably similar modifications and fluvial conditions. Even when multiple rivers are compared, every river system is unique and therefore whether they are true replications is questionable, even when the geographical location is similar. In the majority of cases, only one river is investigated and this often means one modified and one unmodified section, although some studies compare multiple sites within the same floodplain.

In general, it is not possible to do experiments on large river systems. Rivers cannot be canalised or regulated simply to determine the impact on the river biota – instead studies must be done as monitoring studies where river management schemes are occurring, or more commonly have occurred. In a few cases, (e.g Armitage 1978, Cowx *et al.* 1981), data is available on the composition of the aquatic community before regulation took place. However, in the majority of cases in Europe, large rivers have been modified for such a long time that there is no record of the pristine biota. Generally, even when before and after type studies are done, the river was rarely at reference status beforehand e.g. Dessaix *et al.* (1995) used data from before hydroelectric regulation on the Rhone to evaluate changes, although the river was already not entirely natural as it was embanked and there were upstream dams. Controls are important even where previous data exists and were used in most studies. Aquatic communities are naturally dynamic and can fluctuate with weather and water levels, among other factors. Therefore, a control is important in order to separate natural change from that which is anthropogenically induced. Where no record of previous community structure is present, it must be approximated from either a section of the river that is deemed to be unmodified or another river. Again, the problem persists as to the reality of the pristine nature of these sections, although they are often referred to as "natural" or "unmodified". In addition, the comparability of the sites also becomes an issue, in terms of how confident we can be that the original flora and fauna of the two sections was the same or similar. These problems come down to the issue of defining reference conditions for these rivers against which to measure change when almost all large European rivers, and certainly most British ones have been modified for an extended period. The comparison between the River Great Ouse in Bedfordshire and the River Biebrza in Poland may not seem an obvious choice but to get a true measure of community change from that at reference conditions this kind of comparison may be required.

Due to the lack of replication and a lack of statistical rigorousness in many of these studies (in common with much environmental monitoring based research), in addition to uncertainly about reference conditions, the evidence presented in these studies must be taken together, rather than individually, in order to draw conclusions. It is possible from the data available to be confident in saying that anthropogenic modification of rivers – canalisation, flow regulation or weed cutting does result in a deviation of biological communities from reference condition. Almost every study shows a change in the range or number of species present and many show a change in the abundance of individuals. Although none of the fish studies show statistical significance or replication (probably due to the fact that it is not easy to divide the river into discrete sections since fish are mobile) studies are in general agreement. There is good evidence that channelisation and rip-rap on the banks lead to loss of spawning ground for riverine fish, dams and weirs prevent migration of anadromous fish and loss of connected backwaters reduce numbers of limnophilic fish. In general, specialist fish are lost and generalist species become dominant.

It is not possible to predict the precise nature of the deviation from reference invertebrate community caused by hydromorphological modification of rivers, due to the diversity of the results presented above, although there is enough evidence to be sure that some change is likely. It appears that different modifications and different rivers result in different invertebrate responses. Again, few results were reported as being statistically significant due to lack of repetition but often community changes were dramatic and convincing e.g. there were twice as many individual invertebrates on the unregulated Wye compared to the regulated Elan (Scullion 1982) and twice as many mollusc species on an unaltered floodplain compared to an altered one (Loffler 1990). Almost all invertebrate studies have been done on rivers which have experienced flow regulation and very few on the effect of canalisation, which is common in British rivers. This is an area where more research is required.

The macrophyte studies range over a large number of types of community and disturbance. There is only one study on weed cutting, though it does include two rivers. Some studies consider in-channel vegetation whereas others consider riparian vegetation. One study is on bryophytes whereas the rest are on macrophytes. There is therefore there is little repetition even when studies are considered together, although fortunately more macrophyte studies involve within–study repetition, increasing confidence in the results. There is only one study on the impact of channelisation on macrophytes in the UK (Hey *et al.* 1994) although this study does include a large number of modified rivers. The main weakness with this study is that the control sections of rivers are probably far from their natural state and therefore there is no real estimate of deviation from reference conditions.

A fact that is clear from these studies is the importance of floodplain backwaters of a range of ages for the presence of a complete reference community. Their importance was shown for charophytes (Bournette and Arens 2002), gastropods (Orbdlik and Fuchs 1991), invertebrates (Holmes 1999) and fish (Scheimer *et al.* 1991) among others. In many cases, it was found that the floodplain backwaters were more important that the channel itself in terms of the number of species for which they provide a habitat.

Many studies have been carried out on modified channels. However, there is a lack of detailed data on the geography, hydromorphical status and biological composition of rivers that could be considered at, or close to reference status. This is necessary in order to assess comparability of potential reference sites and then to make accurate comparisons. The hydrological features determining their biodiversity must be identified to aid restoration of UK rivers back to something close to reference status.

6.3 Discussion of results

If the hydro-morphology of a lowland river system is impaired so that the biological community deviates more than slightly from reference conditions, then the site will fail to achieve good ecological status. In the UK, the vast majority of rivers have been canalised or had their flow regulated. This has occurred to such an extent that a single lotic channel is often perceived to be the "natural" state of rivers. It is only when we begin to look at the floodplains of intact European rivers such as the Biebrza and less heavily modified sections of the Rhine and the Rhone that it is possible to see what a truly pristine system looks like. These do not consist of an isolated channel in a fixed position but of a number of channels that move across the floodplain according to patterns of erosion and deposition. Channels become cut off and eventually abandoned to form lakes. These floodplain waterbodies have a range of flora and fauna from close to that of the river, to a completely different limnophilic community. A natural river will flood regularly, maintaining hydrological connectivity with many of these water bodies. As they become less connected, succession occurs towards a terrestrial habitat but the creation of new waterbodies maintains a complete range of habitat types and a corresponding biological community. In many studies, these floodplain water bodies have been shown to be more important than the channel itself as a habitat, particularly for aquatic macrophytes and juvenile fish. Historical records show that in the past, lowland floodplain rivers in the UK had this dynamic, habitat rich character and that since these times large proportions of the biodiversity has been lost. It is critical that the reference conditions defined for these floodplain rivers are done with reference to perfect or near perfect hydrological conditions, which may require looking outside the UK for suitable sites. There are a number of problems in locating sites that are equivalent to the pristine state of UK rivers. Many European rivers have their origins in high mountainous regions and are strongly affected by snowmelt, which is not the case in the UK. Pristine species compositions of UK and European rivers may not have been identical.

Weed cutting, canalisation and flow regulation have been shown to have a detrimental effect on biological communities and cause a deviation from reference conditions. Only 15% of lowland rivers could be considered pristine and free of any human modification according to the River Habitats Survey 1994-1997 (Raven *et al.* 1998). Only a further 15% could be considered semi-natural with only small modification. To achieve good status within lotic channels these modifications appear to be unacceptable, in almost every study being shown to cause more than slight changes in biota. Major restoration will be required in order to begin to meet the requirements of the Water Framework Directive. Restoration projects have been carried out with varying degrees of success – a relatively large-scale project on the River Cole had some success but re-establishment of rare species was limited. However, it is clear it is unrealistic to hope that many, if not most rivers will attain only a slight deviation from pristine state, as demonstrated by a river such as the Biebrza by 2015. Instead, perhaps the important lesson to learn is the processes that maintain the biological community and attempt to reinstate them in a move towards restoring ecological function. Rather than large scale engineering works, more beneficial to the river may be a reinstatement of natural flow regimes and flooding patterns and the creation of floodplain water bodies and side channels, which will create a greater and more natural variety of habitat conditions.

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