SEVENTH FRAMEWORK PROGRAMME

THEME 6: Environment (including Climate Change)

Adaptive strategies to Mitigate the Impacts of Climate Change on European Freshwater Ecosystems

Collaborative Project (large-scale integrating project)
Grant Agreement 244121
Duration: February 1st, 2010 – January 31st, 2014

Deliverable 2.14: Review on processes and effects of droughts and summer floods in rivers and threats due to climate change on current adaptive management strategies

Lead contractor: **NERC (CEH)**
Other contractors involved: **-**

Due date of deliverable: **April 2012**
Actual submission date: **April 2012**

Work package: 2

Contributors: François K. Edwards, Roger Baker, Michael Dunbar and Cedric Laizé

Estimated person months: 2

Project co-funded by the European Commission within the Seventh Framework Programme (2007-2013) Dissemination Level (add X to PU, PP. RE or CO)		
PU	Public	x
PP	Restricted to other programme participants (including the Commission Services)	
RE	Restricted to a group specified by the consortium (including the Commission Services)	
CO	Confidential, only for members of the consortium (including the Commission Services)	

Abstract

Europe is expected to experience a greater frequency of floods and droughts as precipitation and evapotranspiration patterns are modified by climate change in future. Several large scale drought and flooding events have occurred in Europe since 2000. Studies of drought are rare but indicate important impacts on freshwater habitats, water quality, plants and animals, which may have wider consequences for ecosystem functioning. The main factors determining the impacts of droughts and floods are event duration and seasonality of the event. A diverse habitat distribution and the presence of refugia at the reach scale confer the most resilience against droughts and floods. Management measures will also be impacted particularly with regard to riparian zones, channel morphology, flow and floodplain connectivity. However there is a conflict between management actions that target the effects of drought, and those that target floods.

This report reviews information on droughts and aseasonal floods (summer floods) published since 2000 with a principal focus on small lowland rivers. Using several recent (post 2000) reviews on these topics, we describe abiotic and biotic effects of droughts and floods, providing recent European examples where possible. We explain the current status of droughts and summer floods in Europe, and where the main sources of data can be found. We highlight the threats posed by these phenomena to some of the most common current adaptive management strategies in place in the EU. To this end we use measures already described within REFRESH under Deliverables 1.1 and 1.2, and we focused solely on adaptive measures relating to riparian zones, channel morphology, flow and floodplain connectivity.

A review of the processes and effects of droughts and summer floods in rivers and threats due to climate change on current adaptive strategies.

François K. Edwards, Roger Baker, Michael Dunbar and Cedric Laizé

Centre for Ecology and Hydrology, Wallingford UK

April 2012

EU FP7 REFRESH deliverable 2.14

1 Contents

2 Introduction

The aim of the REFRESH project is to identify adaptive strategies that will help mitigate any impacts of climate change on European freshwater systems. Climate change scenarios generally predict a greater frequency and magnitude of seasonal, supraseasonal and aseasonal droughts and floods in the EU in coming decades, though the details can vary between scenarios. River ecosystems are vulnerable to changes in hydrological regime extremes, particularly when those are aseasonal and supraseasonal, and thus fall beyond what natural systems have adapted to cope with. To better predict the ecological impacts of droughts and floods caused by climate change, it is necessary to review the processes and effects of droughts and floods, as well as the management strategies that may be affected by these changes.

This document reviews information on droughts and aseasonal floods (summer floods) since 2000 with a principal focus on small lowland rivers. Using several recent (post 200) reviews on these topics, we describe abiotic and biotic effects of droughts and floods, providing recent European examples where possible. We explain the current status of droughts and summer floods in Europe, and where the main sources of data can be found. We highlight the threats posed by these phenomena to some of the most common current adaptive management strategies in place in the EU. To this end we use measures already described within REFRESH under deliverables 1.1 and 1.2, and we focused solely on adaptive measures relating to riparian zones, channel morphology, flow and floodplain connectivity.

3 Climate change and freshwaters in Europe

3.1 Climate change patterns

There is substantial evidence that global warming is occurring, due to the rising production of greenhouse gases by man since the 19th century. The 100-year trend to 2007 shows a 0.74 °C increase in mean global air temperature, with a higher rate of increase in the last 50 years of 0.65 °C (IPCC, 2007a). The year 2011 was the 35th consecutive year that the yearly global temperature was above the 20th century average. The warmest years on record were 2010 and 2005, which were 0.64°C above average. Including 2011, all eleven years in the 21st century so far rank among the 13 warmest in the 132-year period of record. Only one year during the 20th century, 1998, was warmer than 2011. Future warming is predicted to be even faster with global temperatures expected to rise between 1.1 and 6.4 °C by 2099 depending on levels of greenhouse gas emissions (IPCC, 2007a). Even with an unlikely freeze of greenhouse gas emissions at current levels, past emissions could account for a further 0.5 °C rise in global temperature (Wigley, 2005).

An increase in air temperature is likely to translate directly into warmer water temperatures In line with this, the temperatures of flowing waters have risen in Europe. For example water temperature in the Danube has increased by up to 1.7 °C since 1901 (Webb & Nobilis, 2007), and temperature has increased by 2.6 °C in French rivers between 1979 and 2003 (Daufresne & Boet, 2007), and by 1.4 °C in Welsh streams between 1981 and 2005 (Durance & Ormerod, 2007).

The 2011 globally-averaged precipitation over land was the second wettest year on record, behind 2010, but precipitation varied greatly across the globe in response to natural climatic patterns, such as the El Niño-Southern Oscillation (NOAA, 2011). By the middle of the 21st century, annual average runoff and water availability are projected to increase by 10-40 % at high latitudes, and decrease by 10-30 % over some dry regions at mid-latitudes, some of which are already water-stressed areas (Milly et al., 2005). Drought-affected areas will increase in extent. Heavy precipitation events, which are very likely to increase in frequency, will augment flood risk. Water supplies stored in glaciers and snow cover are projected to decline, reducing water availability in regions supplied by melt water from major mountain ranges (IPCC, 2007b).

There is a already a general trend (1900-2005) for increased precipitation in Northern Europe and a decrease in some parts of Southern Europe (Klein Tank et al., 2002, van Lanen et al., 2007), however no study has yet matched this to patterns in river flows (Lang et al., 2006, Pekarova et al., 2006), with the exception of a Slovak study showing strong decreases in river flows since 1980 in 64 rivers, with greatest decreases in the South of the country and smallest decreases in the North of the country (Majercakova et al., 1997). Linking climatic trends to river flows is difficult, because few rivers have natural flows, thus confounding the impacts of climate change (European Union, 2009). Across Europe, earlier snowmelt and earlier onset of the growing season has occurred since 1989 (Chmielewski & Rotzer, 2002), a pattern also observed in Nordic countries, where this has led to longer summer droughts in 60% of river basins (Hisdal et al., 2006). Though a study of English hydro-meteorological data from 1800 to 2006 could not establish a clear relationship between precipitation patterns and drought frequency (Cole & Marsh, 2006, Marsh et al., 2007), there was a clear trend for wetter winters and drier summers over time (i.e. a more distinct seasonal partitioning of annual rainfall), as predicted by the UK Climate Impact Programme models (Jenkins et al., 2008).

Climate change is expected to affect the hydrology of all European regions in some way. Negative impacts will include increased risk of inland flash floods, and more frequent coastal flooding. Mountainous areas will face glacier retreat and reduced snow cover. In Southern Europe, climate change is projected to worsen conditions (high temperatures and drought) in a region already vulnerable to water shortages. In Central and Eastern Europe, summer precipitation is projected to decrease, causing higher water stress. In Northern Europe, climate change is initially projected to bring mixed effects, including some benefits such as reduced demand for heating, but, as climate change continues, its negative impacts, including more frequent floods are likely to outweigh its benefits.

3.2 Changes in temperature and flow in river ecosystems

The trend for warmer water temperature could have a profound impact on the ecology of European rivers, because warmer temperatures alter fundamental ecological processes and are likely to change species distributions. The growth rates of plants and animals may be stimulated and lengths of lifecycle may change, permanently altering community structure and ecosystem trophic dynamics. Water quality may decrease as microbial activity and decomposition of organic matter increase, adding to reduced oxygen levels associated with higher temperatures. Aquatic species unable to migrate (regionally to cooler climes or within a river to the cooler headwaters) may face local extinctions. Conversely, there is a strong risk that non native invasive species, with broader temperature tolerances, may spread to new territories and establish themselves rapidly, applying further stress to native species. (Poff et al., 2002).

Climatic changes to air and water temperature will cause shifts in the timing and intensity of precipitation and changes in the rates of evapotranspiration. Because these affect the volume and timing of runoff (or snow melt) and modify groundwater recharge, changes to the hydrology of freshwater systems are expected. These include a greater frequency, intensity and duration of extreme events such as storms/floods and droughts, increased peak flows and reduced base flows. These changes mediated by the supply and the quality of water, when combined with other effects of climate change such as higher water temperature, make freshwater ecosystems amongst the most vulnerable to climatic change (Allen & Ingram, 2002).

Hydrological changes to European rivers can be summarised according to run off and stream flow type (Bates et al., 2008):

I. Catchments where stream flow is determined by rainfall and run-off
   Greater flow variability is expected, as well as an increase in the number of drier basins and a decrease in wetter basins

II. Catchments dominated by low elevation snow
    A decrease in snow accumulation is expected, along with an increase in rain precipitation, a shift in spring peak flows to earlier in the season, and lower summer flows
III. Catchments dominated by high elevation snow and glacier melt
    An initial increase in stream flows is expected as glaciers melt, followed by a decrease in stream flow as the volume of glaciers diminish
IV. Catchments dominated by groundwater
    A decline in groundwater recharge is expected, though some local increases may occur depending on land use, geology and groundwater abstraction levels. A greater risk of groundwater and surface water drought is expected in these catchments..

Though temperature changes are well modelled, precipitation and evapotranspiration models have a high degree of uncertainty, limiting the current ability to predict the frequency and intensity of droughts and floods at the river basin scale. However, projections for Europe are in line with the general expectation of increased mean annual precipitation at higher latitudes and decreased mean annual precipitation at mid latitudes, with a corresponding change in run-off and river flow (Kundzewicz, 2009). Some scenarios indicate critical regions in Europe where the return period of 100 year droughts, 100 year floods, or both, will decrease to between 10 and 50 years (Lehner et al., 2006). Models indicate that drought problems could be more severe than flood problems. As a result of climate change, areas of increased water stress will exceed areas of decreased water stress two- to four-fold, though detailed quantitative predictions are model specific (Alcamo et al., 2007). Nonetheless, the intensity of rainfall events is projected to increase even where the mean annual precipitation is likely to decrease, causing a shift towards heavier intensive summertime precipitation over large parts of Europe and increasing the frequency of floods (Christensen & Christensen, 2003). The seasonal distribution of precipitation and run off will also change, increasing in winter and decreasing in summer. The proportion of snow to rain in winter precipitation should decrease causing earlier run off peaks.

Floods and droughts, and their timing, frequency, magnitude and duration exert fundamental effects on the ecology of streams and rivers (Lake, 2007). As both of these hydrological disturbances are expected under climate change scenarios, it is important to understand the impact of floods and droughts on the ecohydrology of flowing waters, so that adequate management strategies can be put in place to protect river systems. It also important to understand the current status of droughts and summer (i.e. aseasonal) floods in Europe and this is summarised in the next section.

4 Current status of droughts and summer floods in Europe

4.1 Droughts

There is an extensive body of literature on drought in Europe, which has been amply reviewed as part of three projects. Detailed reference lists can be found in the reports cited herein, therefore they are not repeated for the sake of conciseness. The main precursor project was the EU Assessment of the Regional Impact of Droughts in Europe (ARIDE) project (IHF/CEH, 2001). This project covered the 1960-1995 period and established the overall methodology used in two subsequent projects aiming at building a catalogue of regional European historical droughts based on gauged data. Chronologically, the first was commissioned by the UK Environment Agency (EA). This catalogue was then revised as part of the Water and Global Change (WATCH) EU project (and expanded to include floods as well; see below). In addition to the drought catalogue, the WATCH project includes several other relevant pieces of work based on global hydrological models (typically 0.5^{o} x 0.5^{o} resolution) to investigate past and future droughts (Europe is therefore included albeit the focus is world-wide).

4.1.1 Historical droughts

In both aforementioned catalogues, large-scale drought across Europe are assessed for 23 ((Lloyd-Hughes et al., 2009); EA project) or 24 ((Parry et al., 2011); WATCH) regions based on the Regional Deficiency Index (RDI), a method allowing intercomparison of streamflow regimes across a range of locations and seasons because it expresses regional drought relative to a particular time of year and location. These studies are based on a European database of gauged flows. The EA catalogue is more detailed but although European in its scope, it dedicates a good part to analysing UK droughts against the European context.

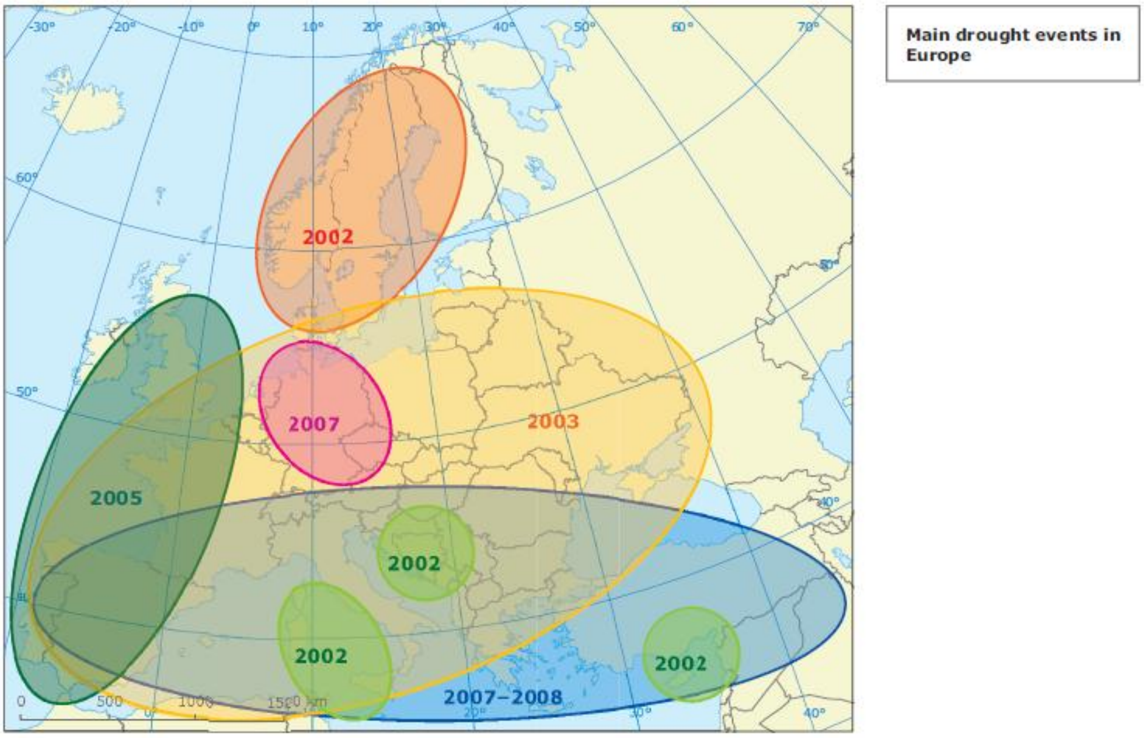
In addition to the catalogues, (Parry et al., 2009) detailed the major drought events of the 1961-2005 period with a particular focus on their spatial coherence. Some of the principal findings are:

- High variability in drought patterns and their drivers (only the 1975-76 event was coherent at European level for a persistent period)
- West-east pattern migration in some major events
- Droughts generally associated with large-scale atmospheric circulation anomalies
- Distinction between short summer droughts and long droughts resulting from combined winter and summer deficiencies (e.g. 1962-64, 1988-92, 1995-97), for which winter conditions are equally important as summer conditions

Tallaksen *et al* (2011) presented a detailed analysis of major historical droughts in Europe for the second half of the 20th century based on observed and on modelled flows (nine-model ensemble). Two extended drought periods were identified: (1) autumn 1975 to late summer 1976; (2) spring and summer 1990. Major differences were found in the drought development of the two events (build-up, consolidation and recovery phase). Main events for 2000-2007 are showed in Figure 1.

4.1.2 Future droughts

Projected future droughts were investigated within the WATCH project. On the one hand, (Williamson et al., 2011) applied the methodology used for the drought catalogue (i.e. to the same European regions) to modelled future flows. They found that in temperate regions of Europe the number of large scale drought events is projected to increase by the end of the 21st century by most models. On the other hand, (Corso-Perez et al., 2011) explored different combinations of seven large-scale models, climate forcing data, and climate change scenarios for the 21st century and they identified an increase in number of drought events in the future.



Source: ETC-LUSI, adapted from Tallaksen, 2007.

Figure 1 Recent extreme drought events in Europe and their approximately area coverage; Tallaksen (2007) in Tallaksen (2011).

4.2 Summer floods

While deaths from flooding appear to have decreased in the 20th century compared to previous times, economic losses have clearly increased dramatically, comprising not just material losses but also disruption costs. the main driver for these losses has been increased exposure and vulnerability rather than simply increased risk (Mitchell, 2003). Conflicting conclusions exist as to flood trends at a European scale.

Flood studies most often focus on flow statistics that do not refer to seasonality (e.g. flow percentiles, annual maxima, peaks over thresholds). Literature consequently refer explicitly to summer flood events when reporting on specific flood events, e.g. (Kundzewicz et al., 1999), or on processes preferentially associated with summer flood events, e.g. (Kundzewicz et al., 2005). Some references, although not specific to summer floods, are cited as they provide insight and/or information relevant to summer events. In addition, some studies focus on high flows, which are not necessarily associated with floods i.e. there can be high flow episodes without flood disasters. As per the previous section on droughts, some of the references cited herein themselves contain reviews of the existing literature, and references are only repeated if necessary.

4.2.3 Historical high flow floods

4.2.3.1 Specific summer flood events

As explained above, the only papers explicitly referring to summer floods tend to be the ones reporting on specific events that occurred during the summer. In this regard, there is a significant body of literature about Central Europe, where this type of floods is quite frequent, notably: Poland in 1997 (Kundzewicz et al., 1999) and Central Europe in 2002 (Ulbrich et al., 2003a, Ulbrich et al., 2003b) . However extremes were these recent events, summer floods are relatively common in Central Europe (Ulbrich et al., 2003a). One also needs to keep in mind that this region of Europe is not exclusively hit by summer floods. (Cyberski et al., 2006) listed the following extreme events for Poland: March 1924, July 1934, March 1947, July 1960, January 1982, July 1997 and July 2001. (Kundzewicz et al., 2005) revisit these extreme events (also including the Vistula flood in 2001) in order to identify their common elements and identifies significance future changes relevant to the flood hazard in the region.

4.2.3.2 Flash floods

Another body of literature that is relevant to summer floods is that of flash floods since many of them happen during that season. Flash floods are generally defined as floods with a short lag between an extreme rainfall event and the flood occurrence (e.g. fewer than six hours). Catalogues of flash flood events were presented by (Gaume et al., 2009) (>550 events in Europe) and (Llasat et al., 2010) (Mediterranean countries so covering Europe as well as North Africa, Turkey, Israel, etc). More detailed analyses of the spatio-temporal patterns of flash floods based on a subset of these are given in (Marchi et al., 2010). In inland continental Europe, flash floods occur during the summer, in Mediterranean Europe, they occur during the autumn. Spatial extent and duration are generally smaller for inland Europe than for Mediterranean Europe. These patterns reflect different climatic forcing in each region. It is worth noting that flash floods are not necessarily summer floods and conversely, summer floods are not necessarily flash floods.

4.2.3.3 High flow and flood cataloguing (not seasonal)

There have been three notable attempts at reviewing high flow or flood events in a comprehensive manner. These studies are specific to the summer period but contain relevant information. First, (Barredo, 2007) presented a map and catalogue of the major flood events for 1950-2005 in the European Union (EU), Bulgaria and Romania in an effort to alleviate the lack of homogeneous and geo-referenced information on flood disasters for large periods in Europe. Second, (Glaser et al., 2010) reconstructed European floods since 1500 and highlighted the complexity of the underlying climatological causes and the high variability of flood events in temporal and spatial dimension. Last, analogue to their aforementioned drought catalogue, (Parry et al., 2011) compiled a high flow catalogue. It uses the same observed flow series and period of records but is based on 23 European regions only (two drought regions are merged, other regions are the same). High flows occurrences are analysed with the Regional High Flow Index, conceptually similar to the RDI used for droughts.

4.2.4 Future floods

As mentioned above, there is no consistent picture regarding the future. (Williamson et al., 2011) applied the methodology used for the high flow catalogue (23 European regions) to modelled future flows: results suggests little change in the number of high flow events, with the exception of north-west Scandinavia (more high flows) and a change in the seasonality in the High Alps (more high flows during Spring). Focusing on flood events, the analysis of long records (up to 150 years) for the Elbe and Oder rivers showed no upwards trends but actually a decrease in winter floods and no change in summer floods (Mudelsee et al., 2003), however, climate model simulations indicate an increase in flood risks globally (Kundzewicz et al., 2010).

5 Processes and effects of droughts

We found few genuine studies on drought emerging from Europe, and none with a rigorous BACI design, reflecting the episodic nature of drought. Much of the expected effects of drought are derived from low flow studies rather than drought studies *per se*. Also much is inferred from studies of naturally intermittent systems, which are adapted to drought and are nor representative of streams that would not normally experience annual droughts. We rely on several extensive reviews published post 2000 (Lake, 2003, Dewson et al., 2007, Lake, 2011b), adding European examples published since then.

5.1 Defining drought

Numerous definitions for drought exist, most based on different anthropocentric viewpoints: there is a clear temporal progression from lack of rainfall (meteorological drought) to progressive soil moisture deficits through to low levels of rivers and standing water bodies (hydrological drought), and to impacts upon water-dependent ecosystems (ecological drought). Surface water drought may progress to groundwater drought.

Sector-specific definitions such as agricultural drought also exist: drought is one factor influencing water scarcity (socio-economic drought) which may be defined as the imbalance between water availability and demand. More details of the various definitions can be found in (Tate & Gustard, 2000), (Mishra & Singh, 2010) and (Lake, 2011e). Overall, at any particular place and time, whether a drought situation exists is largely based on expert judgement, weighting up the importance of the above factors. (Lake, 2011e) makes a distinction between “normal” seasonal dry periods and supra-seasonal drought associated with a longer term lack of wet-season rain.

Drought clearly affects society, there are now documented cases of drought destroying historical civilisations (Medina-Elizalde & Rohling, 2012). As with floods, droughts interact with human behaviours, human demand for water can often increase during drought (e.g.

for agricultural irrigation). In the modern world drought is a clear reminder of the ecosystem services that the hydrological cycle provides, affecting water supply for people and industry, power generation, food and fibre production, a recent estimate is that European droughts have cost €100 billion over the past 30 years with average annual costs of €5.3 billion (CEC, 2007) .

5.2 Processes

The temporal sequence of definitions highlighted above indicates the main hydrological processes associated with drought. Droughts, as with floods, are controlled primarily by meteorology. Clearly, they are the opposite of floods in that their defining characteristic is an absence of water, however they also differ in other ways. For example floods generally happen rapidly (although large river flooding can clearly take some time to develop), whereas droughts happen gradually and may persist for extended periods, this leads to difficulties in defining ongoing drought status. Relative to their slow onset, the physical characteristics of drought tend to diminish more quickly when rain returns, providing that rainfall leads to the elimination of soil moisture deficits. Droughts tend to be spatially extensive, whereas floods tend to be restricted to individual river basins, with the exception of major regional winter floods caused by sudden snowmelt and / or ice breakup. There tends to be a correlation between drought severity and spatial extent. Soil moisture deficits will be exacerbated by increased evapotranspiration caused by warm temperatures, which are associated with particular weather systems.

Conventional wisdom is that the natural environment plays a major role in mitigating the effects of droughts, for example the perceived benefits from wetlands in buffering water flow. However this is not always the case (Robinson, 1990, Bullock & Acreman, 2003). What is clear is that human activities, particularly the abstraction of water for human uses, can exacerbate drought. Although such activities largely do no destroy water, they lead to its removal from river and groundwater systems and bypassing of surface waters via constructed water distribution and removal networks. Unless checked through voluntary or compulsory restrictions, human usage of water can increase during drought associated with warm temperatures, further exacerbating drought impacts. This is particularly an issue in the use of water by agriculture, much of which is lost through evapotranspiration. There seems to be the general opinion that other human activities in catchments (i.e. not related to direct water use) can exacerbate the effects of drought (Lake, 2011d), however hard evidence is lacking. For example, (Lake, 2011d) suggest that increasing catchment tree cover as a drought mitigation measure. However there is differing opinion as to the effects of tree cover on hydrological conditions, trees may transpire more water, exacerbating low flows (Querner et al., 1997), however such models may not include all relevant processes (Roberts, 1983)

Data on droughts are continually increasing and improving, with satellite imagery now providing a new dimension. Seasonal forecasting, essential for droughts and desirable for floods, is less developed, and historical analyses suggest that it is hard or impossible to define a “typical” European drought, every past drought is different (Parry et al., 2010, Stahl et al., 2010, Hannaford et al., 2011). Studies of teleconnections between large scale meteorological and oceanic phenomena are much more common for the Americas (Mishra & Singh, 2010) than for Europe (e.g. (Svensson & Prudhomme, 2005)).

5.3 Effects

5.3.5 Abiotic effects

Drought impacts both terrestrial and aquatic environments. At the catchment scale, plant cover is reduced, plant litter accumulates, and nitrates may build up in soils. The input of water, sediment, ions, nutrients and organic carbon from the catchment into the riparian zone of streams and rivers is reduced (Dewson et al., 2007). In agricultural landscapes, inputs of N and P nutrients in stream water decrease, as diffuse nutrient run off is reduced (Fritz & Dodds, 2004). In forested landscapes, decreases in particulate organic matter (POM), dissolved organic carbon (DOC), dissolved inorganic nitrogen (DIN) and soluble reactive phosphorous (SRP) in stream waters have all been observed during drought, as links are severed between the stream and the catchment (Golladay & Battle, 2002). On the other hand as flow decreases and dilution is reduced, the concentrations of solutes increases in stream water (Zielinski et al., 2009), which can be important when sewage effluents form part of the river inputs. Generally changes to water chemistry represent a lowering of the ratio of organic to inorganic nutrients, which has the potential to shift stream metabolism from heterotrophy to autotrophy as there is less DOC and algae take up nitrogen (Dahm et al., 2003). The amount of suspended particles and turbidity decrease with decreasing flow, and fine sediment will be deposited in and on the stream bed (McKenzie-Smith et al., 2006). With lack of flow, particulate organic matter (POM) can accumulate in the stream channel, which can lead to water quality problems as it decomposes, particularly when combined with higher temperatures and low oxygen (McMaster & Bond, 2008).

As the riparian zone dries, and water levels drop below root depth, the streamside vegetation will die if the drought persists. Leaf senescence will eventually occur, which in turn alters the input of organic matter into streams, but also reduces shading and therefore refugia from warmer water temperatures. Decomposition of organic matter slows down and the rate at which nutrients are processed slows down. During drought, more herbivores may use the riparian zone to access water and streamside vegetation, this further depletes riparian vegetation, but also increases soil erosion and the input of nutrients via excretion (Jacobs et al., 2007).

As flow decreases, air temperature and solar radiation have a greater influence on stream water temperature. A model by Meier *et al* (2003) predicted a water temperature increase due to low flow in a stream in the Swiss Alps, but there is little empirical evidence on the relative contribution of lower flow vs. elevated air temperature. In the case of small streams with high groundwater inputs, temperature may drop with lower surface flows (Dewson et al., 2007), but in most cases temperature would be expected to increase. Warmer water temperature causes heat stress to the fauna, in particular fish, but also increases decomposition rates and lowers oxygen concentrations (reduced solubility). This has the potential to cause hypoxic conditions, particularly at night time when the oxygen demand is higher. Water conductivity may increase as a result of decreased dilution and increased evaporation (Zielinski et al., 2009), and this can be followed by changes in pH (Sprague, 2005). Woodward *et al* (2002) observed a rise in pH in response to reduced run off during a low rainfall period in a naturally acidic UK stream. The potential for dilution of wastewater and sewage is reduced with lower flows and the concentrations of dissolved solids and ions such as chloride and sulphate may increase (van Vliet & Zwolsman, 2008, Wilbers et al., 2009). This is partly compensated for by the reduced run-off which lowers the input of nutrients from the catchment to the stream (Caruso, 2002), and also the greater contribution of low nutrient groundwater to the total stream discharge (Dahm et al., 2003). However, once flow drops past a certain threshold, concentrations of nitrogen and phosphates in the water will increase, and combined with low flow and high temperature, can create algal and cyanobacteria blooms, leading to eutrophication and an associated drop in oxygen saturation (Suren et al., 2003b). Increase sedimentation could also occur with lower flows, as more sediment will drop out of suspension onto the stream bed, the stream margins and interstitial spaces. By contrast, Bond (2004) observed decreased sedimentation with low flows because there was less suspended sediment in the water.

As droughts persist, decreased discharge causes a gradual drop in water velocity and depth. This reduces the amount of wetted habitat and changes the stream hydrodynamics, although there is a lot of variability within and between streams dependant on channel morphology (in particular width to depth ratios). Aquatic linkages are severed as drought sets in (Lake, 2011f), first laterally between the channel and the riparian zone as the stream withdraws from the littoral edge; then longitudinally as riffles, runs and glides are replaced by a mosaic of remnant pools and dry areas (Boulton & Lake, 2008). The stream continuum thus becomes a linear series of heterogeneous patches (Boulton, 2003), disconnected from terrestrial inputs of allochtonous litter and invertebrates (Baxter et al., 2005). Organic matter will accumulate and be deposited in remnant pools and dry areas and detritus processing will slow (Pinna & Basset, 2004). In remnant pools, biotic interactions intensify and species with lentic traits and predators will be initially favoured, and the assemblages in each pool may diverge according to local conditions (Stanley et al., 2004). However the water quality in pools will gradually deteriorate as the concentrations of solutes increase and organic matter breakdown depletes oxygen levels (Suren et al., 2003a). Higher temperatures may also cause conductivity to increase, and bring about thermal stratification in larger pools (Caruso, 2002). These pools may persist but will completely disappear in longer, more intense, droughts. Usually some subsurface water will persist in the hyporheic zone, providing the last benthic refuge for fauna, and decomposition and nutrient transformation may also continue in the interstitial water, but, nonetheless, the principal pathways of material transfer in the stream ecosystem are severed.

Once dried, benthic soft sediment patches may crack and develop fissures and important changes in microbiology, chemistry and mineralogy can occur. The anoxic layer of the stream bed retreats deeper, reducing denitrification and increasing phosphate retention, microbial biomass is reduced and sulphur can get re-oxidised to sulphates, potentially causing acidification problems when re-wetted (Baldwin & Mitchell, 2000, Lamontagne et al., 2006)

The increase in bare ground reduces the water retention capacity of the whole catchment when surface run off returns after the drought, which increases the entrainment of sediment, organic matter and nutrients into water bodies. If drought breaks with a heavy downpour rather than a prolonged spell of steady precipitation, erosion may be severe and large amounts of sediment and nutrients may be exported out of the catchment (Lake, 2011f). The re-wetted riparian zones and floodplains release DOC from soil and litter (Worrall et al., 2006), which once flushed into the stream, can increase microbial activity and reduce water quality (O'Connell et al., 2000, Howitt et al., 2007). Soils and wetlands in catchments also accumulate nitrates and re-wetting produces a pulse of these into streams, causing a dramatic increase in nitrate concentrations and nitrogen loading. Re-wetting wetlands and floodplains may also release sulphates as sulphuric acid, which is washed into streams causing an acid pulse, and a drop in pH, which may also be associated with a mobilisation of metals, which can reach toxic concentrations (Tipping et al., 2003, Eimers et al., 2008). Rewetting bed sediments can release nitrogen as nitrates, and this may stimulate high levels of microbial activity, but may also increase algal activity and sometimes cause a burst of macrophyte growth (Baldwin et al., 2005). In streams where POM and leaf litter accumulate in the dry bed during drought; there is a pulse of DOC and nutrients that may be readily incorporated into food webs, contributing to ecosystem recovery. However if this is also associated with pulses of acidity, sulphates or metals, recovery may be dampened (Romani et al., 2006).

5.3.6 Algal biofilms

At the onset of droughts, low flows, water clarity and higher nutrient concentrations can initially promote algal growth (Wade et al., 2002, Wright et al., 2002a), creating chlorophyll-rich biofilms and increasing stream primary production (Dahm et al., 2003). During low flows, there is a change from assemblages dominated by diatoms to assemblages dominated by filamentous green algae, in response to increased temperature, reduced water velocity and in some cases higher nutrient concentrations (Suren et al., 2003b). There is a corresponding shift from low to high biomass, though the benthic algal biomass will decrease substantially as the drought progresses past a flow threshold (Kinzie et al., 2006). As the amount of wetted habitat is gradually reduced, desiccation of attached algae, cyanobacteria and biofilms will occur, and rapid drying may kill algal cells (Ledger et al., 2008) and primary production will drop. The algal biofilm can resist drought by using cysts and extracellular mucilage layers (Stanley et al., 2004), and in these final drying stages, the assemblage often shifts from dominance by green algae back to dominance by diatoms (Caramujo et al., 2008). The rate of drying is a crucial factor, as rapid drying will impair any desiccation resistance strategy (Robson & Matthews, 2004).

Re-wetting often sees a rapid recovery of the biofilm, favouring filamentous algae, coccoid green algae and diatoms (Ledger & Hildrew, 2001), which may peak before herbivores have returned. This occurs due to the reactivation of surviving cells and from propagules in the drift, if upstream reaches have remained wet (Stanley et al., 2004). Remnant pools may also provide a refuge for algae and act as recolonisation foci (Robson & Matthews, 2004).

5.3.7 Higher plants

Drought can substantially alter macrophyte communities by eliminating some species and creating gaps for opportunistic species to establish. Plant communities shift from aquatic to terrestrial as the stream bed dries out during drought, which combined with the deposition of sediment and litter may change the streambed morphology (Lake, 2011c). As the stream ecosystem dries, remaining plants can act as an important refuge for small invertebrates. However if drying continues, all aquatic macrophytes will eventually die. The dead plant material is deposited on the stream bed and riparian zone and can act as an important source of moist habitat, and will become a source of organic matter when flows return.

Higher plants have a wide range of strategies to tolerate drought, such as changes in osmotic physiology, cell turgor, stomata reduction and leaf orientation, but also have drought avoidance strategies such as reliance on propagules and seeds (Touchette et al., 2007, Romanello et al., 2008). Seeds may remain viable for a long time and be stored in sediments providing an important drought refuge as a seed bank (Brock et al., 2003). They may not necessarily germinate on re-wetting, providing a long term refuge capable of surviving multiple drought years. The recovery of higher plants is important for ecosystem recovery as they provide a trophic resource as well as habitat for invertebrates and fish. It is usually marked by a succession of different plant assemblages, which can take years to return to pre-drought condition, depending on the strength and duration of the drought, and the extent to which the community has become terrestrialised (Westwood et al., 2006). Recovery of plant assemblages can be strongly limited by siltation which shades seedlings (Wright et al., 2002b) and abstraction which maintains low flows beyond the drought (Franklin et al., 2008).

5.3.8 Invertebrates

Reduced flows and wetted area during drought can initially lead to higher invertebrate densities (Dewson et al., 2003) as they become more concentrated in the available wet habitat (McIntosh et al., 2002), but generally this is followed by reduced invertebrate densities, as food quantity and quality changes and competitive and predatory interactions become more intense (Wood et al., 2000). The responses of food resources, in particular algae, to drought conditions can have a strong influence on the density response of invertebrates (Smakhtin, 2001).

In perennial streams, reduced flows can lead to decreases in biodiversity (species richness), which can be attributed to the loss of types of habitats such as riffles and cascades. If habitat diversity persists through reduced flow periods, then richness is little affected, unless there are associated stressors such as increased sedimentation or a rise in water temperature. Hence channel morphology has a strong influence on the invertebrate community response to flow, with more heterogeneous reaches being the most likely to lose habitat types and species during low flow events. The extent of the drought and flow reduction is also a strong determinant of habitat persistence and species survival (Dewson et al., 2007). Because of these changes in richness and abundance, there is usually a change in invertebrate community composition, as habitat suitability increases for some species and decreases for others (Gore et al., 2001). Often total abundance and species richness is similar pre and post drought, but the constituent species differ (Beche et al., 2009). In a small English stream, Wood *et al* (2000) observed that taxa with preferences for low water velocity and fine sediment characterised the sites in drought years, and taxa with preferences for high water velocity and gravel substrate characterised the sites during non drought years. Similarly, trait composition differed between unregulated and regulated sections of a river in Portugal (Cortes et al., 2002). The invertebrate assemblage of the regulated section was more tolerant of reduced oxygen saturation, was better adapted to low flows and had shorter life cycles. Differences in invertebrate community structure are also related to the rate and extent of drying. First order streams have a more stable community as the rapid drying produces a robust set of surviving species. In higher order streams, drying takes longer and may be less extensive, producing a less stable and less specific community (Beche & Resh, 2007).

In perennial streams, drought has the potential to have strong impacts on the abundance and species richness of invertebrate assemblages, but the effects are a function of the duration and severity of the drought, as well as local geomorphology which determines which habitats and stream sections dry the most. Generally, the invertebrate fauna is reduced by drought but this is somewhat taxon specific, as some are not very affected e.g. nematodes, turbellarians, oligochaete worms, mites, beetles and some dipteran larvae, while others are strongly affected (e.g. EPT taxa, leeches, amphipods, molluscs). Rheophylic taxa, such as filter feeding species, will be rapidly eliminated. As flow drops, species specialised in shallow habitats such as riffles risk the risk of being stranded, especially if they have limited motility, such as mussels (Gagnon et al., 2004). Due to increasing concentration of nutrients and other chemicals in the remaining water (in particular remnant pools), and physical processes such as the deposition of fine sediments and litter with dropping flows, taxa that are well adapted to low water quality and degraded stream conditions are also well adapted to tolerate drought (Boulton & Lake, 2008). As long as flow persists to some extent during the drought, the recovery is rapid, but if the channel dries completely and significant losses in population densities and species richness occur, the recovery will be slow and incomplete (Ruegg & Robinson, 2004). If several droughts occur close together, or in several successive years, then the effects are severe. For supraseasonal droughts, the unpredictability in timing, duration and severity create a selective force which few adaptations can withstand. Furthermore, supraseasonal droughts are sometimes also characterised by the absence of (naturally occurring) scouring floods earlier in the year, which increases the disturbance effect (Power et al., 2008).

In general fauna have a low resistance to drought (though some bury and some have resistant eggs) but resilience is strong and substantially mediated by the use of refugia. These principally consist of remnant pools (Covich et al., 2003), moist habitats (e.g. under logs and stones), the hyporheic zone (Fenoglio et al., 2006) or migration to another permanent reach or stream. Drift enables invertebrates to avoid unfavourable conditions such as drought. Though passive drift reduces during low flow, active drift tends to increase, but recent studies are rare (Kinzie et al., 2006, Dewson et al., 2007). Though drifting may increase exposure to predators such as fish and may strand the drifting organism in even less suitable habitat, it is often the first and strongest community response to flow reduction. However, once a flow threshold is reached, drifting will not be possible so that invertebrates will have to use benthic refugia. For example, insects primarily use migratory refugia when capable, but other invertebrates such as crustaceans and molluscs have to use sedentary in-stream refugia, if these are available (Fonnesu et al., 2005). Densities of hyporheic invertebrates can therefore increase during drought, but this is limited by siltation, which can also increase during drought (Stubbington et al., 2009, Wood et al., 2010).

The biota of intermittent waters such as Mediterranean streams has evolved to contend with seasonal droughts, which provides an insight on the adaptations that increase resilience and resistance to atypical and supraseasonal droughts. For temporary stream fauna, drying is a normal environmental event, but supraseasonal drought can also exert significant effects. Seasonal drought in intermittent Mediterranean streams is predictable, with high winter flows and low or no summer/autumn flows. Typically complex, species rich invertebrate assemblages dominate in spring and early summer, succeeded by a simpler poorer assemblage in mid to late summer as streams become trickles or series of separate pools (Bonada et al., 2006). In intermittent streams, traits associated with adaptation to natural annual droughts include laying eggs in vegetation, free isolated eggs, parthenogenesis, diapause, spiracular respiration, tegument respiration, flying adult stage, passive aquatic dispersal, cocoons, endobenthic habitat, surface swimming, microorganisms and fine detritus as food (Bonada et al., 2007).

Recolonisation by invertebrates can be very variable and depends on the intensity of the drought as well as the nature of the system (i.e. perennial or naturally intermittent) (Wood & Armitage, 2004). Recolonisation occurs from several pools of colonists. Recolonisation by some insects after drought occurs via highly mobile flying adult stages from remnant pools, wetted reaches or other water bodies. Drought favours good colonisers such as chironomids (Acuna et al., 2005) which are multivoltine, but for many species of insects, recolonisation may not take place until the next generation of adults, the following year (Hynes, 1958, Hynes, 1961, Morrison, 1990, Ruegg & Robinson, 2004). Thus, the timing of the drought in relation to life cycles is crucial in determining the speed and strength of recovery. Recolonisation can also occur from drought resistant eggs (Boulton, 2003) or from wetted benthic refugia. Surviving individuals in remnant pools, if any, contribute to the recolonisation by dispersing out of pools when flows return either actively, or passively via drift (Fritz & Dodds, 2004). Recovery follows a similar trajectory in intermittent and perennial streams, with the first colonists having short life cycles (e.g. chironomids, ceratopogonids, simuliids, etc) and species richness increasing as longer-lived taxa re-appear (collector/gatherers, grazers), then eventually levelling off once larger predators return (Lake, 2007). Griswold (2008) suggested that important traits in early recovery are small body size, sclerotisation, tubular body shape and ability to drift. As recovery progresses these traits give way to soft bodies, poor resistance to desiccation, rare in drift, crawl or cling.

Sometimes the post drought community can be very different from the pre drought community because key species such as grazers, shredders and detritivores are lost and the trophic structure is altered. This is particularly so for supraseasonal droughts, where recovery is subject to a time lag and much less predictable because there has been some degree of species turnover and microhabitat distribution has changed. Power (2008) described how the lack of scouring floods in drought years allowed cased caddisfly larvae to proliferate and limit algae through grazing, in contrast with non-drought years with few caddisfly larvae and extensive algal growth. In a replicated experiment simulating drought in artificial streams, secondary production of invertebrates was halved, and functional feeding groups were affected differently , with shredders and predators most affected by the drought (Ledger et al., 2011). When flow returns, the heavy influx of nutrients, DOM, POM from the catchment and the dried bed, give rise to a heterotrophic stream metabolism, with detritus as a major source of food for invertebrates, as autotrophic metabolism and primary production may lag behind as the re-growth of algal biofilms may take some time, thus detritivores may recover quicker (Artigas et al., 2009).

5.3.9 Fish

As drought sets in and flow diminishes, fish change behaviour and habitat, for example dominance hierarchy and territoriality disappear. Initially, fish may redistribute themselves at the reach scale as they seek patches of cooler and deeper water, such as shaded pools (Elliott, 2006), where they also run less risk of being stranded (Dekar & Magoulick, 2007). Other species may be able to exploit the early stages of drought if they have a higher tolerance to higher temperatures and low oxygen levels. Elliott (2006) observed that during drought, wetted habitat and water quality were impaired, brown trout (*Salmo trutta* L.) density was reduced, and fish migrated to cooler pools. However bullhead (*Cottus gobio* L.) densities increased, as they are more temperature tolerant, and they were able to exploit benthic resources in shallow areas created by drought and abandoned by trout. Once water temperature reaches a species specific critical limit, fish kills may start to occur. Below lethal limits, effects on condition may be apparent due to physiological stress and also because the invertebrate food source is usually depleted (Hakala & Hartman, 2004).

In remnant pools, fish species richness and abundance increase with pool size, as habitat quantity and variety is greater (Pires et al., 2010), but the presence of predatory species may drive any drought survivors to extinction (Labbe & Fausch, 2000). Predation by terrestrial mammals, birds and reptiles may also increase (Magalhaes et al., 2002, Antolos et al., 2005). Outbreaks of parasitism such as white spot disease may occur in pools due to stress and high density (Maceda-Veiga et al., 2009). Pools can thus act as an environmental filter, shaping the composition of the remnant fauna and the post drought fish assemblage. Salmonid populations, for example, are typically density dependent, but drought causes a shift to population regulation by density independent factors (Nicola et al., 2009).

As drought persists and the stream bed starts to dry, deteriorating water quality in pools means fish are most likely to migrate to a perennial reach, or run the risk of being trapped in a pool that may eventually completely dry out (Matthews & Marsh-Matthews, 2003). Fish production thus drops in drought impacted reaches (McMaster & Bond, 2008). This migration strategy does carry some risk as drying may create barriers to migration, and downstream perennial reaches have greater exposure to large fish predators.

As long as wetted areas are available and accessible during drought, recovery of fish populations is usually rapid, much faster than other biota, as they are highly mobile (Magoulick & Kobza, 2003). The timing of the drought and fish life cycles is important because juvenile fish are at much greater risk of predation in remnant pools or during migration to a perennial reach, and eggs may die if fine sediments are deposited on the stream bed (Magalhaes et al., 2003, Hakala & Hartman, 2004). Nonetheless, recruitment is often strongest following a drought year, though it is unclear whether this is driven by environmental conditions, habitat effects or biological interactions (Keaton et al., 2005). Recovery from a supraseasonal drought is less predictable for fish, and ranges from rapid to slow and incomplete (Lake, 2007).

5.3.10 Ecosystem function

There is little data on how droughts and floods might impact ecosystem function, other than extrapolating from effects to the physical habitats and community structure described in previous sections. It is clear there is the potential for large effects on functionning. One study of supraseasonal drought in German lowland streams (Schlief & Mutz, 2011) does however clearly demonstrate such impacts. They found that microbial activity, shredder colonization and leaf decay rates were low during the onset of drought, presumably caused by drought-related environmental conditions. Microbial activity and temperature-corrected decay rates increased after the flow resumption but not leaf mass loss and shredder colonization. During both periods, exposed leaves appeared physically unaffected suggesting strongly reduced shredder-mediated leaf decay despite shredder presence. Thus their results indicate that hydrological drought can affect organisms and processes in temperate lowland streams even after flow resumption. Several major reviews on drought or low flows (Poff et al., 2002, Lake, 2003, Dewson et al., 2007, Lake, 2007, Poff & Zimmerman, 2010, Lake, 2011a) stress the need for a clearer understanding of the impacts on ecosystem functionning, which may highlight as yet unrecognised long-term impacts of drought.

5.3.11 Human activities and drought:

Humans through their activities exert a range of pressures on natural systems, particularly freshwater ecosystems. However a number of these activities have the potential to increase the frequency, intensity and duration of drought, and exacerbate its effects on stream and river ecosystems:

- Changes in land use and land cover

  Deforestation, or the replacement of deep rooted vegetation with shallow rooted crops, can influence regional climates by reducing evapotranspiration, precipitation and surface water availability, and in turn reduce run off and groundwater recharge (Bala et al., 2007). The clearing and grazing of land leads to greater soil compaction so that soils retain less moisture and water flushes through the catchment (Lawrence & Chase, 2009). Grazing also leads to greater soil erosion and greater inputs of fine sediments into streams and rivers, particularly when riparian zones are affected. The proliferation of impervious surfaces, especially in urban areas, greatly reduces the storage of water in soils and groundwater at the catchment scale (Sophocleous, 2002).

- Groundwater abstraction/recharge

  Excessive abstraction may alter the dynamics of surface waters and influence the severity of drought and the ecological recovery. Other activities that reduce recharge include building impervious surfaces, regulating river flow and reducing flooding in floodplains (Burgess, 2002). All these activities which reduce the groundwater levels serve to lower the water table below the level of the stream bed, which has important consequences for systems where there is an active connection between surface and ground water, for example gaining streams may become losing streams (Shepley et al., 2009).
- Dams and levees

  These have the potential to exacerbate drought because they reduce river flow, prevent floods and reduce the connectivity of the groundwater/river/floodplain system longitudinally (dams), laterally (levees) and vertically (bed sedimentation due to reduced flow downstream of dams).
- Channel clearing and resectioning

  This reduces habitat diversity and the occurrence of drought refugia. For example snag habitats maintain moist patches which provide refugia for some bivalve molluscs (Gagnon et al., 2004, Golladay et al., 2004). Resectioning also further disconnects the river from its floodplain.
- Invasive species

  These are often spread through human activity (Hänfling et al., 2011) and are usually good colonisers so drought may facilitate their establishment by reducing competition from native species. However invasive species can also have a negative effect on native species during droughts as they increase pressures on resources and space.
- Climate change

  This is discussed in section 1, and covers a wide array of human activities and ecological effects; however man-made climate change is expected to increase the frequency, severity and duration of droughts in Europe via changes in precipitation patterns.

Reactive measures that may help mitigate the impacts of these human activities on drought include emergency releases from dams, groundwater pumping to maintain streams and the maintenance of instream drought refugia (Lake, 2011d). However restoration measures that increase the resistance and resilience to drought may be more effective on the long term, e.g. restoration of riparian zones and river channels, reinstatement of floodplain connectivity, establishment of natural flow regimes in regulated rivers, management of land use and land cover (Poff et al., 2002). These adaptive management and restoration efforts are themselves at risk from increasing frequency and magnitude of drought, which may limit their feasibility, durability and effectiveness. These aspects are discussed in section 7.

6 Processes and effects of summer floods

6.1 Defining summer floods

Floods may simply be defined as the inundation of land not normally covered with water. They are controlled primarily by meteorological conditions, but also by soil/geology. Further controls include geometry both of the river network (including overall catchment size) and the river channel. The latter and to some extent the former act as an overall natural control which can be extensively altered by human activities. Feedback processes operate, hence floods create channel forms which in turn can alter some flood characteristics. Human interventions to reduce flood risk in one area (for example straightening channels and elevating flood banks) may increase it in another downstream area. Flood risk management is increasingly recognised as strongly interdisciplinary, integrating physical and social sciences. Land cover (described by vegetation structure, notably urbanisation and deforestation (Tollan, 2002)), in-channel and riparian vegetation and land use (e.g. under-drainage (Robinson, 1990), soil compaction, presence of embankments) interact to control flood regimes in complex ways, especially at larger spatial scales. Presence of natural and artificial lakes also affects flood regimes (Reed & Robson, 1999), extreme flows also have important consequences for dam safety.

Four main flood generating mechanisms are recognised: convective rain, long-lasting rain, snowmelt and ice-breakup. Clearly only the first two apply in the context of summer floods, with convective rainfall being a particularly important driver of floods in summer compared to winter. Distinction can be made between surface water flooding (pluvial flooding), river flooding (i.e. rivers breaking their banks), groundwater flooding. Hence antecedent conditions, in terms of soil moisture, river and groundwater levels are a strong controlling factor. Coastal storm surges, which often occur jointly with other types of flooding have been the single greatest flood-related cause of loss of life in Europe (Mitchell, 2003).

Conventional wisdom is that the natural environment plays a major role in mitigating the effects of floods, for example the perceived benefits from wetlands and land use change in flooding. However this is not always the case: wetlands can ameliorate and exacerbate flooding dependent on the particular situation (Bullock & Acreman, 2003, Mudelsee et al., 2003, Glaser et al., 2010). Data on floods are continually increasing and improving, with satellite imagery now providing a new dimension. Near-time forecasting of flooding is now routine, although there have been notable forecasting failures. These have not only been linked with predicted rainfall uncertainty but also with assumptions as to flood wave velocity: presence of summer vegetation will retard water velocity, systems may be calibrated for more "typical" winter conditions. Seasonal forecasting is less well developed (Svensson & Prudhomme, 2005).

6.2 Processes

In his recent review of droughts and temporary waters, (Lake, 2011c) points out that greater work has been undertaken to understand the effects of floods on ecology than the effects of drought. Typical floods may be considered to be part of the natural hydrological variability, integral to the hydrological regime and ecosystem productivity (Naiman & Decamps, 1997). However this review considers the specifics relating to unpredictable summer, or aseasonal, floods and as such the disturbance and subsequent ecological responses may differ from those often reported.

The characteristics of the disturbance of summer flooding are dependent upon the nature of the receiving waterbody, but also upon the subsequent response of the ecological community therein. The disturbance resulting from floods tend to be characterized as pulses, i.e. short term and delineated, in contrast to presses (arising sharply and increasing steadily then reaching a maintained) or ramps (increase steadily without an endpoint, such as some droughts) (after (Bender et al., 1984, Lake, 2000). It should be noted that while floods are a form of disturbance to the aquatic environment, they can act as vital resetting events whereby they alter the abiotic environment with subsequent effects upon the biological communities (Junk et al., 1989, Poff, 1992). The effect of such a disturbance on the biota will have spatial and temporal scale-dependence. For example a component of the community that measure generation times in years will respond differently to one with multivoltine traits (Lake, 2000).

This review considers constrained streams/rivers that are not floodplain rivers, in which water frequently moves out of the channel inundating the floodplain and creating a patchwork of connective channels and pools. In floodplain rivers, floods are considered to be powerful and necessary regenerative events for the ecological processes both within the channel but also to terrestrial elements of the floodplain (Lake, 2007).

The depletion of stream biota in response to floods has been long understood, with the potential damaging effects of unpredictable floods being observed (Moffett, 1936, Hoopes, 1974). The rapid recovery that is usually seen, for example in desert streams, led to the theory that the communities were stochastic and that such disturbances are the dominant organising factors in stream ecology (Fisher et al., 1982, Lake & Barmuta, 1986, Resh et al., 1988).

Within a typical stream or river system, floods cause disturbance by altering habitats, washing out fauna, algae and higher plants, removing biofilms, non-fatal and fatal damage to organisms through sediment mobilisation, scour and deposition, removal of physical structures such as dams, sediment deposits, and alteration of the riparian habitat (Resh et al., 1988, Lake, 2000, Lake, 2007). However flood events may also have beneficial effects, especially where they are predictable in their timing and may be exploited; for example in the migration of fish (David & Closs, 2002).

The overall effect of a flood may be characterized by the event's duration, magnitude, frequency and predictability combined with consideration of elements of the stream's substrate composition, stability, refugia, and natural hydrograph (Poff, 1997, Lake, 2000, Lake, 2007). Due to these variable components the effects of flooding may be seen to vary from small spates or freshets causing little geomorphological change, to extended powerful high discharge events which can alter the structure of the stream channel entirely (Costa & O'Connor, 1995). Aseasonal floods are acknowledged to be more damaging than those that occur during typical wet seasons (Lytle, 2003, Giller, 2005).

The effects of a flood on stream biota can be lessened through the use of refugia; defined by (Sedell et al., 1990) as 'habitats or environmental factors that convey spatial and temporal resistance and/or resilience to biotic communities impacted by biophysical disturbances', While refugia may not enable resistance of the community as a whole, they may improve resilience through the resistance of patches from which recolonisation can occur (Lancaster & Belyea, 1997, Lake, 2007). (Lake, 2007) cites the example of the use of refugia via habitat change being the movement of surficial invertebrates into the hyporheos during floods (DoleOlivier et al., 1997) or into pools (Fausch et al., 2002); while refugia use within a habitat is demonstrated by organisms persisting on stable substrates (Matthaei et al., 2004).

While resistance to flood disturbance is generally low, the biotic resilience is generally high (Lake, 2000). The extent to which this holds true is dependent upon the intensity of the event, its spatial extent, the availability of a source of recolonisers and the composition of the community itself. It is acknowledged that exceptions to this rapid recovery may include instances where the event is particularly severe (Scrimgeour et al., 1988), or where the flood occurs out of season (Giller et al., 1991).

6.3 Effects

6.3.12 Abiotic effects

Floods, comprising the movement of large volumes of rapid water, have the potential to cause geomorphic changes to a river channel and its adjacent alluvial valley (Schmidt et al., 2001). The effects on the abiotic features of a river channel are dependent upon many characteristics of the flood and of the channel itself. For example, the extent of sediment transport will depend upon the carrying capacity of the water and the nature of the sediment (Biedenharn et al., 2005). During the rising limb of the flood hydrograph there is the potential for materials from the exposed soils within the catchment to be mobilised and transported into the river by overland flow, Coarse particles will be transported over short distances while fines may be carried over much greater distances (Lane, 2007). It should be noted that such ‘muddy floods’, where fine sediments are mobilised in this way, are less likely when the land is vegetated, as is more likely in aseasonal summer floods.

It is well recognized (Reice et al., 1990) that sediments of different particle size will be mobilized by different water velocities. (Middleton & Southard, 1977) developed a model to relate the Reynolds number to current velocity, particle diameter, fluid density and dynamic viscosity. In general the velocities required to mobilize sand are lower than for other particles due to the increases in critical velocities required for larger particles, and the adhesive properties of finer particles (Reice et al., 1990).

While the degree of sediment transport from surrounding land is less likely in summer floods than in winter due to the presence of vegetation, erosion of the channel bed and banks will still occur (Thorne, 1990, Julien, 1998). Inevitably a flood will cause areas of scour and removal of fine sediment but with subsequent deposition. Such processes will often result in a net transfer of particles from the channel bed to the banks and to areas of recirculation (Hazel et al., 1999, Schmidt et al., 2001). However following the controlled aseasonal flood below the Glen Canyon Dam on the Colorado River, the bed topography of some reaches of channel appeared to be rearranged rather than experience net change. Areas of thick (>1m) sand deposition were recorded (Schmidt et al., 2001). The controlled flood at Glen Canyon Dam lasted 7d, with a further 3 d at a lower discharge. During this time the size of fine sediment deposited was seen to coarsen. Fine particles of silt and clay were the first to be mobilised and the percentage of sand in deposits increased as the duration of the flood increased (Schmidt et al., 2001). The greatest effect of this flood was to redistribute fine sediments from low to higher depositional sites along the channel's margin. The extent of erosion and deposition were spatially variable, for example low elevation sandbars were extensively eroded while shoreline deposition was common as the waters receded (Schmidt et al., 2001). Back water channels were excavated by the changes in flow patterns which, combined with some areas of deposition on existing bars, led to the creation of areas of backwater habitat. These flood induced changes were only seen to last a few years with backwater areas disappearing by 1999.

As a flood recedes the sediment transport capacity decreases and the mobilised sediments are progressively deposited. Where the banks are overtopped, much of this sediment is deposited on the floodplain, however the rest is deposited in-channel in point, mid-channel and side bars (Steiger et al., 2001). The deposition of new sediments over pre-existing vegetation on bars provided an increase in available nutrients stimulating vegetation regrowth. This soil nutrient availability was enhanced for at least two years post flood on the below the Glen Canyon Dam (Parnell et al., 1999, Schmidt et al., 2001). (Shannon et al., 2001), describe how the flood waters in the Colorado River were seen to clear suspended particles in the water column and as a result the water clarity increased. This was thought to have aided the recovery of the benthic biota. Of the other water quality parameters, including temperature, conductivity, dissolved oxygen and pH, recorded during this experiment, none were seen to vary outside of the typical patterns recorded prior to the flood event.

Debris, in the form of both organic (large woody material) and 'trash' (anthropogenically derived) may also accumulate in the system following a flood event. Such material can cause localised flooding and interrupt sediment transfer (Wallerstein & Thorne, 1998). However such debris can also increase morphological diversity and have beneficial effects on the river habitat.

6.3.13 Algal biofilms

Floods can potentially disrupt metabolic process where primary production is reduced and ecosystem respiration may fall. However after this occurred in a stream in Arizona, (Fisher et al., 1982) recorded a rapid recovery with the system once again becoming autotrophic in the short-term.

In the controlled flood experiment on the Colorado River in 1996, (Shannon et al., 2001) reported an increase in the biomass of the cyanobacterium *Oscillatoria spp*. following the flood pulse, however this was attributed in part to the area of the channel that was being sampled at the time. A shift from the filamentous algae *Cladophora glomerata* to miscellaneous algae and bryophytes at certain sites was also observed. Scour and entrainment of primary producers was observed throughout the system, however the phytobenthos were seen to recover on hard substrates within just one month.

Slower, but still relatively rapid, recovery was also recorded by (Blinn et al., 1995) and by (Benenati et al., 1998). While much of the phytobenthos was scoured, the coarser substrates remained upon which the rhizoidal holdfasts of *C. glomerata* remained. The high water clarity was thought to have contributed to the rapid recruitment of the community.

The primary impact that high discharge events have on periphyton communities is through their dislodgment due to sheer stress from flow and abrasion by sediments (Steinman & McIntire, 1990). The potential for damage of organisms by abrasion is related to the amount and type of sediment in the system. Furthermore, abrasion is more likely to influence those organisms with a habitat preference for smaller lighter substrates due to the likelihood of them being displaced during a flood (Douglas, 1958). The growth form and certain environmental conditions are known to influence the susceptibility of the periphyton, with filamentous forms being susceptible above velocities of 50cms^{-1} (Horner & Welch, 1981). Methods of attachment also have a bearing on retention with *Rhizoclonium sp.*, a filamentous chlorophyte with basal attachment cells, being much more resistant than *Spirogyra sp.,* which has less robust terminal hapteroid outgrowths (Power & Stewart, 1987). In their study into the structure and dynamics of the ciliate population of the potamoplankton of the River Rhine, (Scherwass & Arndt, 2005) found that sessile Peritrichs were particularly susceptible to being washed out due to high discharge events. Typical spring and autumn flood events had the effects of causing a short term succession from peritrichs to pelagic oligotrichs.

The characteristics of the habitat have been shown to influence greatly the rate of recovery (Steinman & McIntire, 1990) with rapid recovery of susceptible filamentous chlorophytes in their typical habitat of high nutrients and light conditions. Recovery of periphyton following a flood disturbance will be dependent upon the abundance of propagules and their composition as well as the local environmental conditions (Steinman & McIntire, 1990).

6.3.14 Higher Plants

High flows can have both damaging and beneficial effects on macrophyte communities. Macrophytes can be entirely stripped, or receive mechanical damage by the flood waters and transported sediments, however these same flows can facilitate colonisation through the redistribution of plant fragments or seeds. Sediment deposition can cause localised disturbances but also create new areas of habitat for plant colonisation. For the duration of the flood event, increased water depth and turbidity can reduce light penetration and the ability of the plants to photosynthesise.

Study of winter and summer flood disturbance on the Rhone showed that macrophyte community recovery times were similar to that of other elements of the aquatic ecosystem (Barrat-Segretain & Amoros, 1995). This resilience was attributed to their high potential foe dispersal, colonisation and growth.

The relationship between floods and hydrochory (waterborne propagule dispersal) has received recent attention with high flows playing a major role in the transport and dispersal of seeds along river communities (Gurnell et al., 2008). As such the timing of such flood events can be crucial for hydrochorus plants.

(Combroux et al., 2001) found floods to be highly important in influencing the introduction of species to areas of deposition. (Tickner et al., 2001) have found that the distribution of many invasive species, such as *Fallopia japonica, Heracleum mantegazzianum* and *Impatiens glandulifera*, relies on their own or their seeds', transportation during floods.

At the time of their study on the recovery of flood affected macrophyte communities on the Rhone, (Barrat-Segretain & Amoros, 1995) noted that very few studies addressed the effects of flood disturbance on aquatic vegetation. In their experiments, recovery from summer disturbance was seen to be within a few weeks with species such as *Elodea* appearing very rapidly. Some species did not reappear, while others, capable of exploiting open patches of river bed, appeared at the study site post-flood despite not being recorded previously. Recovery from a winter disturbance was also rapid but with a different order of succession, possibly due to the different abilities of the species to disperse, and establish at different times of year. The effects of the flood were considered greater during the summer event, since this occurred when the development of the plants was at its maximum. In winter the disturbance occurs when most species lack their vegetative development.

The response of a number of species including *Potamogeton pectinatus, Cladophora sp.* and *Myriophyllum spicatum* to an increase in flow was studied at the Rabodanges dam (Bernez et al., 2007)). After an initial decrease in cover all the plants were observed to recover to previous levels.

6.3.15 Invertebrates

Macroinvertebrate communities may be composed of many species with differing ecological requirements and thus their responses to flood disturbances are considered complex. The three key variables considered to influence community composition are flow, temperature and substrate stability (Cortes et al., 2002, Fleituch, 2003, Lytle & Poff, 2004b). A flood may have an effect of benthic invertebrate community structure and function as a result of substratum stability and availability, water velocity, water quality among other variables (Shannon et al., 2001). (Junk et al., 1989) established the flood pulse concept which describes the disturbance of floods and responses of the biota in river floodplain systems.

(Wood et al., 2000) found that hydrological conditions played a dominant role in influencing community composition on the base flow dominated Little Stour. However it is clear that resistance to higher discharges is greater for those species within the community exhibiting suitable adaptations, thus altering community composition in their favour (Cortes et al., 2002, Fleituch, 2003). (Ward, 1976) suggested such alterations would include increasing the abundance of the Plecoptera, while members of the Diptera, Trichoptera and Ephemeroptera may be increased or reduced. Meanwhile, comparing a site downstream of an impoundment to an unimpounded one, (Armitage, 1978) noted more Oligochaeta, Chironomidae, Diptera and microcrustacea below the impoundment. These same sites were studied over 30 years and it was found that many of the species at the impounded site declined in abundance due to its regulated nature (Armitage, 2006). The impounded site had fewer than half the number of high flow events and numbers of *Hydra sp.*, *Ancylus fluviatilis,* Naididae, Heptageniidae, Leuctridae and *Brachycercus subnubilis* had all declined in abundance by a factor of five or more.

The effect of floods on the invertebrate community is sometimes limited due to their generally high mobility and ability to exploit refugia during high flow events (Mackay, 1992, Lake, 2000, Lake, 2007). Recolonisation is then rapid from these patches of resistant organisms. Other major routes for recolonisation are from aerial dispersal, surface movement, active swimming and passive drift (Gray & Fisher, 1981, Mackay, 1992). Filter feeders often dominate early communities with these organisms being able to exploit the high levels of suspended material (Downes & Lake, 1991).

Community composition is often predictably reset by the occurrence of seasonal floods. The disturbance of winter floods typically created space for Simuliids to dominate in a Californian stream, however in the absence of such floods the dominant taxa were hydropsychid caddis larvae (Hemphill & Cooper, 1983). When one species is better adapted to cope with the high discharges than another, flooding can disrupt predator-prey relationships (Lake, 2007); (Thomson, 2002) found that the prey species were typically more resilient to flooding than their predators. Such natural disturbances may act as environmental filters which alter the dynamics of the community and create new opportunities for some organisms (Lake, 2007).

Immediately following a high flow event the invertebrate population is typically reduced in both diversity and abundance. (Stubbington et al., 2009) studied sites of varying permanence on the River Lathkilldale and found invertebrate abundance fell at perennial sites when subjected to extreme summer floods. The test flood episode at the Glen Canyon Dam on the Colorado River was found to significantly alter the aquatic community composition in the short term, however recovery of the system was very rapid. Those invertebrate species that associate with the fine sediments were most susceptible to disturbance. When compared to organisms able to exploit interstitial spaces or adapted to inhabit the higher flows on the benthos (Shannon et al., 2001). It was estimated that within 24h of the start of the flood >90% of the benthos was removed and peak drift mass was recorded during the initial 2d.

Macroinvertebrate recovery in the Colorado River followed similar patterns to the phytobenthos with a return to ‘typical’ biomass within two months of the flood ending. Indeed primary consumer biomass was the greatest for six years prior to the flood with record diversity (Shannon et al., 2001). In this experimental system, with an upstream impoundment regulating flow in the period after the flood, it is likely that these rapid recoveries in the biota were accelerated due to this stable environment enhanced by improved water clarity and denuded substrate.

Rapid recoveries of macroinvertebrate numbers are widely reported, even in systems that are rarely exposed to flood disturbance (Yount & Niemi, 1990). In a spring-fed stream responding to a rare flood event, (Thorup, 1970) recorded macroinvertebrate species richness returning to pre-flood levels within two months. The exceptions to this rule were two slow growing species that lived on stones at the time, and would have been affected by rolling substrate. Both these species *Wormaldia occipitalis* and *Ancylus fluviatilis* had returned within two years. (Robinson et al., 2004) reported that the recovery time of macroinvertebrates can be less than the generation time of most of the species represented suggesting high resilience.

The movement of bed load associated with a flood event is considered to have either a beneficial or deleterious effect on invertebrates. For a burrowing organism a movement of substrate may have devastating effects whereas for a filter feeding organism on stable substrate the increase in food in suspension can be of benefit (Reice et al., 1990).

In many systems, the biota present will have evolved strategies for responding to flood events, with many having life cycles adapted to rely on such periods. However, where these events happen unexpectedly the communities may not be in a state to apply these adaptations or responses with deleterious consequences (Yount & Niemi, 1990).

As part of a study into the diversity, distribution and seasonality of blackfly larvae in a glacial stream system in the Swiss Alps, (Burgherr et al., 2001) noted negative effects on the community densities and species richness caused by high summer discharge conditions.

Benefits to lentic species of invertebrate have been recorded as a result of floodplain inundation due to summer floods. (Minar et al., 2001) reported a marked increase in population density of mosquitoes following a flood of the Morava River and subsequent inundation of the floodplain in 1997. In addition the effects of flooding on those organisms in the terrestrial environment will be highly affected, in particular those with low mobility such as the molluscs. (Ilg et al., 2009) recorded the particularly damaging effects of the summer floods on the River Elbe in 2002, happening as they did in a period of high physiological activity.

6.3.16 Fish

The distribution and use of habitat by fish is strongly influenced by flow regime, flow requirements and flow tolerances not just between species, but also between life stages of the same species. Direct effects of floods on fish populations include the facilitation of migration, washout or stranding while indirect effects may also be caused through the alteration of food availability, changes in habitat and enabling access to new habitat such as floodplains.

Many fish species common in upland reaches have life history adaptations to improve recruitment success and increase survival. Salmonids are ecologically well adapted to survive in rapidly flowing water, for example, excavating deep egg nests to minimise flood scour (DeVries, 1997). Spawning is timed such that fry emerge when flows are likely to improve survival (Fausch et al., 2001). Spawning triggers are largely a combination of photoperiod and temperature, however a rise in water velocity is also thought to trigger the behavioural response to begin migration and congregate at spawning grounds (Ladle, 2002). (Franssen et al., 2006) suggested that elevated flows increase the longitudinal connectivity and allow access for species to upstream areas for nursery and spawning grounds.

During floods, fish survival and the subsequent potential for recolonisation is strongly influenced by the existence of refugia (Townsend, 1989). Under flood conditions fish are attracted to areas of low flow to save energy, and to avoid displacement, injury or mortality (White & Harvey, 2001). (Jurajda et al., 2004) found that, in the long term, floods actually increased the abundance of phytophilous and phytolithophilous species of fish due to the flooded vegetation providing food and shelter.

While some beneficial effects do result from flooding, floods can have very damaging effects on fish populations, particularly aseasonal floods impacting on early life stages. Extreme high flows may scour eggs from spawning sites (Carline & McCullough, 2003, Cowx & de Jong, 2004), fine sediments transported in flood waters may be deposited and prevent alevin emergence (Phillips et al., 1975), or starve eggs of oxygen (Meyer, 2003). Adults are less susceptible than larvae, but extreme floods may redistribute individuals, cause mortality or damage through bed instability or drifting debris (Erman et al., 1988, Lusk et al., 1998, Weng et al., 2001). Fish may also experience a reduction in feeding efficiency due to turbidity (Arndt et al 2002) or a food shortage due to the washout of their prey (Jensen & Johnsen, 1999).

The controlled flood experiment on the Colorado River in 1996 had little effect on the distribution, abundance and movement of the native fish species, however some short term effects were seen in the densities of non-native species (Valdez et al., 2001). Radiotracked individuals were not observed to differ in their movements or habitat choice during the flood event. Analysis of diet suggested that opportunistic feeding occurred as composition changed but biomass did not. Of the species present only the young non-native rainbow trout (*Oncorhynchus mykiss*) were displaced downstream, with densities recovering to pre-flood levels within eight months.

Substratum movements as a result of flooding can have beneficial effects on the salmonid fish community through the scour of gravel spawning grounds and subsequent removal of fine sediments. In contrast, for species that spawn on plants the removal of vegetation may reduce recruitment (Reice et al., 1990). While adult fish may possess the mobility to maintain their position during increased river discharge, or at the least return to habitat if displaced, juvenile stage are unlikely to do so and will be washed out of the system reducing subsequent recruitment.

(Jurajda et al., 2006) found relatively little difference in fish population structure at sampling points on the River Morava (Danube catchment) following an extensive summer flood. The dominant species before the flood (chub *Leuciscus cephalus* and gudgeon *Gobio gobio*) remained dominant following the event, however the age structure did change in favour of the one-plus individuals of gudgeon. Declines in abundance were observed in the bleak (*Alburnus alburnus*), roach (*Rutilus rutilus*) and barbel (*Barbus barbus*), no change recorded in chub (*Leuciscus cephalus*) and burbot (*Lota lota*) but increases in perch (*Perca fluviatilis*).

(Lojkasek et al., 2005) found no significant differences in average fish population species diversity, abundance and biomass from one month before to two months after a summer flood in the River Oder in 1997. However differences in biomass and abundance were observed at the reach scale. Catches of *Salmo trutta* and *Thymallus thymallus* increased in the year of the flood, although T. Thymallus catches declined in subsequent years. Catch data for *Barbus barbus* and *Chondrostoma nasus* in the period after the flood showed no obvious trends.

As part of a study into the possible effects of climate change on brown trout populations in Switzerland, (Burkhardt-Holm, 2009) considers the possible increases in flood events (albeit in winter) to include an increase in mobilization of fine sediments which affect the health of juvenile fish and subsequent overall reproductive success. Arguably flood events timed later in the year are likely to have less effect on reproductive success as they come at a time when the juvenile fish are older, stronger and better adapted to survive such stress.

The importance of terrestrially derived invertebrate prey for fish populations in the Kol River (Russia) is considered by (Eberle & Stanford, 2010). Species specific variation was considerable, with an average of 68% of the diet of Coho salmon compared to 13% for Dolly Varden being derived from a terrestrial source. Much of this prey was provided in spring floods and the timing of this may be crucial for the predator. Summer floods will similarly provide considerable increases in terrestrially derived food sources however the predators may not be adapted to exploit this potential food source when it is present out of season.

The differential responses of species to flood disturbance can also affect the invasions of invasive species. An example of this relationship is the reduced success of the invasive mosquito fish (*Gambusia affinis*) in replacing the Sonoran topminnow (*Poeciliopsis accidentalis*) in systems where flash floods are frequent occurrences (Meffe, 1984).

(Lusk et al., 2004) studied the extreme flood events that occurred at the confluence of the rivers Morava and Dyje (1997) and Luznice River in 2002. They observed considerable migration of species both laterally, into the floodplains, but also upstream, as well as some downstream washout. In the later stages of the floods, low levels of dissolved oxygen did cause some localised fish mortality. Mass spawning events of species able to display portional or delayed spawning was recorded and this may have contributed to a tenfold increase in abundance and fivefold increase in biomass in the study sites following the floods on the Luznice River. Species were also recorded that were believed to have washed down from the headwaters of the river while no actually detrimental effects on populations were recorded in upstream reaches. The effects of elevated discharge on young-of-year cyprinid fish was also considered by (Reichard & Jurajda, 2004) who found that it affected ”neither drift abundance nor size, age, or taxonomic composition of drifting fishes”. The elevated discharge studied occurred only during daylight and was not associated with an increase in turbidity. It was concluded that water turbidity, rather than current velocity is the causative factor for drift behaviour in such structurally complex habitats.

Experimental manipulations of rivers that have been historically manipulated by impoundment are often carried out to clear out accumulations of fine sediments. One such experimental flood on the River Spol (Switzerland) was found to noticeably improve the spawning grounds of the brown trout population (Ortlepp & Murle, 2003). Despite some localised fatality (<2% overall) of individuals during the flood event due to physical damage or stranding, the condition of adults was unchanged and the number of redds increased three-fold.

The timing of an aseasonal flood may have implications on the levels of agriculturally derived contaminants entering the watercourses. (Polard et al., 2011) recorded different levels of genetoxicity in basal flow, winter flood and spring flood hydrological conditions. Exposure to spring flood conditions resulted in the greatest damage to the model fish (Crucian carp, *Carassius carassius*). Chemical analysis showed these water samples to have the highest contamination level, mainly explained by a peak of metolachlor. The mercury levels of the organs and tissues of 10 bream caught before and 10 caught after the 1997 flood in the Vistula River were compared to test the hypothesis that floods disturb the bottom deposits and mobilise contaminants therein (Zarski et al., 2002). A 2.5 fold increase in the mercury concentrations was recorded after the river had flooded.

(Bischoff & Wolter, 2001) studied the effects of the summer floods of 1997 on the River Oder on the population of 0+ fish. The floods were seen to have affected local community composition and alter the physical habitat structure. After an initial loss of fish to washout, particularly in the poorly structured meso-habitats, the 0+ fish were able to re-colonize the habitat. Diversity, particularly of the rheophylic species, increased in this post flood period as individuals benefitted from the increased habitat heterogeneity that had been created during the disturbance.

7 Threats to adaptive strategies

We reviewed the adaptive measures already described in REFRESH deliverables 1.1 and 1.2 (Nõges et al., 2010b, Nõges et al., 2010a) that were relevant to rivers, and determined these could be grouped into five main areas of adaptive management: riparian zones, channel morphology, hydrological regime, water quality and floodplain connectivity (we did not include estuarine measures). We discuss how these (excluding 'water quality' as this was too intermeshed with the other 4 groups) may be impacted by the threat of droughts and summer floods induced by climate change in rivers. Measures bear the same reference number as per Nõges et al (2010b) and are listed in Annex 1.

7.1 Riparian zone management

Relevant measures: 65, 150, 298, 351, 370, 380, 381, 382, 383, 387, 388, 395, 396, 397, 408, 453

By far, the most common climate change adaptation strategy for rivers is the establishment and management of a natural riparian zone and vegetation. The main premise is that shading by vegetation is one of the main drivers of stream water temperature and thus it will mitigate the effects of rising air temperature on stream systems (Davies-Colley & Rutherford, 2005). This is particularly important for organisms with low critical thermal limits, such as salmonid fish (Hendry et al., 2003). However many more benefits can be derived from the riparian zone, such as buffering the flux of nutrients, sediments and organic matter from the catchment/floodplain into the stream/river (Parkyn et al., 2005), controlling algal growth (Hutchins et al., 2010) or increasing stream bank heterogeneity with roots. Thus, the management of riparian vegetation is seen as a 'win-win' adaptive measure which promotes the ecology of the stream/river system as well as provides resilience in the face of climate change (Nixon, 2008).

7.1.1 Effects of drought

The principal risk from an increasing frequency and magnitude of seasonal and supraseasonal drought concerns the loss of vegetation, or at least foliage, from the riparian zone. This would entail a reduction in shading and an increase in stream water temperatures. Measures potentially susceptible to this include M65, M150, M351, M380 – 383, M395 – 397 (Nõges et al., 2010b). The riparian zone would lose a lot of its buffering ability, affecting the flows of nutrients and sediment into the stream. Once soils crack, the soil chemistry in riparian zones may change. If tree roots eventually die, the stability of the stream banks may be weakened, thus measures which seek to stabilise the banks, such as M453, M408 (Nõges et al., 2010b) may provide extra resistance and resilience to drought. On rewetting, large quantities of nutrients may be flushed from the catchment along streams and rivers. Other impacts include loss of plant biodiversity at the catchment scale and loss of habitat for fauna associated with the riparian zone.

From a socio-economic perspective, this could mean that riparian zones cannot become self sustaining and would require frequent and intensive management actions to replace the lost vegetation. This is at odds with the principle that riparian management is a low maintenance strategy once riparian zones are established (SEPA, 2009).

7.1.2 Effects of summer floods

A well established riparian flora will reduce erosion in the event of floods, maintaining the channel’s integrity and limiting the mobilisation of sediments. The instability of bed sediments can cause damage to the river fauna and its eventual deposition smothers the substrate. In addition the riparian vegetation, once inundated, provides refuge for fish and invertebrates against the high velocity water improving their resistance to displacement. The timing of summer floods would guarantee that the riparian zone is well vegetated and at its most effective in holding back flood water, providing refugia and stabilising the river banks. In contrast, if the floods occur in the winter months more of the riverbanks and flood plain are likely to be unvegetated and exposed to erosion. Measures which stabilise banks thus seem to be important in managing the threat of more frequent and severe floods due to climate change (e.g. M351, M382, M453)

The increase in riverbank stability may also have the effect of reducing the amount of flood plain intrusion by holding back flood waters for longer. The additional debris that may be brought into the river channel from the riparian zone, including fallen trees or trash, will have the potential to cause dams and create additional flood risks, particularly at existing pinch points such as bridges or weirs, and this constitutes the main threat posed by more frequent and stronger summer floods. Through careful management of the riparian zone, such debris may be minimised and damage may be reduced, but as with droughts, this implies a socio-economic cost.

7.2 Channel morphology

Relevant measures: 36,46,56,148,172,229,352,353,354,365,369,385,386,389,407,452

Human activities, such as urbanisation and agriculture, have led to a widespread modification of stream and river channels, particularly in lowlands. Such actions include the straightening and dredging of river channels, and the clearing of boulders and woody debris (Feld, 2004, Tavzes et al., 2006). Humans also change the natural drivers of river channel morphology and its dynamics, for example through the regulation of the river flows which would normally shape these channels via erosion and deposition (Norris et al., 2007). Because channel morphology, in part, determines the diversity of habitats and species, it is fundamental to the functionning of the river ecosystem, and thus, it is generally accepted that most stressors are aggravated by hydromorphological degradation, for example low flows, siltation etc (Elosegi et al., 2010).

7.2.1 Effects of drought

Many man-made channel modifications are often aimed at moving water out of the system more rapidly, rendering stream ecosystems more vulnerable to low flows and droughts. The re-establishment of natural stream morphology is a ‘win-win’ suite of adaptive measure, which promote biodiversity, ecosystem functionning and resilience to man-made disturbances, including climate change. Restoration measures include bank profiling, instream habitat devices (boulders, logs), pool/riffle sequencing, substrate reinstatement, and re-meandering. These types of measure tend to increase residence times at the reach scale, reducing the impacts of drought. These measures also have the potential to create habitat mosaics which confer some resilience to the effects of drought for the fauna, for example deeper pools can promote the survival of fish during drought (Elliott, 2006), logs and boulders provide damp zones if the stream bed dries (Gagnon et al., 2004).

A greater frequency and intensity of droughts does however pose some threats to channel management strategies. Under low flow conditions, organic matter may accumulate in reaches, initially increasing the amount of wetted habitat but ultimately leading to a decrease in water quality due to the decomposition of organic matter (Larned, 2000). In severe drought conditions, when flow disappears and the bed dries, terrestrialisation of the river channel may occur, with non-aquatic plants taking over the river channel. This in turn modifies the physical habitat so that width, depth and flow habitats will defer post-drought from their pre-drought state (Lake, 2011c). Measures addressing channel morphology must be flexible enough to provide refuge habitat in case of sustained very low flows as may be expected with climate change. Thus channel morphology measures in abstracted and regulated rivers should be designed in unison with environmental flow setting to promote the greatest level of resilience to droughts. Measures that aim to slow water down and retain it as long as possible in reaches are hence advantageous e.g. M352; however they are at odds with adaptive strategies designed to manage floods e.g. M172, M229, M354, and M386. The report by Nõges et al (2010b), only identifies one adaptive measure that addresses both low flows and floods (M407).

7.2.2 Effects of floods

Changes are made to river channel morphology for a number of reasons, not least for flood defence purposes in order to facilitate the rapid conveyance of water downstream and away from urban areas. The straightening and dredging of river channels dramatically reduces the habitat diversity of the river and is often accompanied by further structures and bed lining to reduce substrate erosion. Dunbar *et al* (2009) have shown a clear relationship between channel morphological characteristics and the quality of the macroinvertebrate community in their work on the DRIED-UP models in rivers in the UK. These studies have shown that there can be considerable improvements made in the ecology of a river through the restoration of river channels, even when taken in the absence of increased river flows.

Where channels are heavily modified, the presence of refugia from periods of high flows is likely to be reduced and the potential for organisms to be redistributed is much higher. Where species, or certain life stages, are immobile, these will be highly susceptible to washout in highly modified systems. The resilience of the communities, and speed of recolonisation following a flood, will be much greater where habitat refuges have not been removed by straightening or dredging. However, most current management strategies dealing with floods are entirely centred on the free passage of the water in the channel e.g. M229, M354, M386 (Nõges et al., 2010b). As indicated above, the principal challenge facing managers is how to deal with both increased floods and droughts under predicted climate change scenarios. New opportunities may arise from the field of integrated water resources management, where the concept of storing floodwaters and releasing them during drought/low flow episodes is rapidly emerging as an environmentally sensible option, as well as an economically viable one (Michael Norton, personal communication).

7.3 Flow

Relevant measures: 62,181,201,250,278,313,314,346,347,390,402,403,404,405,406

A range of human activities affect stream flows so that they are increased, decreased or their fluctuation is modified. These activities either affect flow generation processes as well as the stream flow directly (Nõges et al., 2010b):

- Abstraction of groundwater and surface water
- Land use changes (floodplain drainage, vegetation clearing/planting, urbanisation)
- Water resources management (import/export of water between catchments)
- Effluent/return flows from industry, agriculture and homes
- River flow regulation by dams and impoundments

Stream flows vary naturally on a seasonal and inter-annual basis, with low flows occurring in summer (low precipitation, high evaporation) and to a lesser extent in winter (soil frost). During these natural periods of low flows, river systems are particularly vulnerable to anthropogenic impacts and climate change. Thus, an important facet of river management has been environmental flow setting, so that there is an adequate water supply to sustain key species, ecological processes and associated goods and services (Arthington et al., 2006). In recent years, the field on environmental flow setting has moved from arbitrary minimum flows, to more natural flow regimes incorporating seasonal variability in flow (Lytle & Poff, 2004a). These man-made flow regimes principally arise from management of water abstraction, effluent inputs and of compensation flows downstream of dams. Though land use management options do affect stream flow, the relationship is so complex and context specific that the impacts of climate change on this relationship are impossible to predict at a scale useful to water managers.

7.3.1 Effects of drought

The main threat to environmental flow setting from a greater frequency and magnitude of droughts concerns situations where a generic operating rule is applied, with no flexibility. Baseline flow conditions against which the flow regimes are set will need to be updated periodically to reflect the potential reduction in river flows due to drought. Measures potentially subject to these restrictions include M62, M250, M278, M346, M347, M402 and M403, though the description of these measures are too general to conclude anything about the flow setting (Nõges et al., 2010b). Management of dams will need to consider the need for more freshet releases, particularly in warmer dryer summers, and compensation flows from dams will also need to be flexible enough to mitigate the impacts of drought in the regulated zone of rivers e.g. M313 and M404. Recent approaches to compensation flow setting have centred upon achieving a more natural hydrograph with natural seasonal variations, e.g. M405 and M406, however this may be harder to achieve if climate changes causes aseasonal changes to hydrological patterns. Because drawdown rates in reservoirs are likely to increase during times of drought, there may be a conflict between reducing drawdown rates and ensuring compensation flows downstream. Thus dam management will need to be periodically reviewed to take account of climate change. Considering the potential for an increased frequency of both droughts and floods in some European areas, it would seem any measure that captures high flows for later release during times of reduced flow would provide the best resistance to climate change impacts, e.g. M181, M201, M314 (Nõges et al., 2010b).

7.3.2 Effects of floods

Direct management of flow, particularly in regulated and abstracted rivers, usually concerns issues with environmental/residual flows and thus threats from floods are not really relevant, especially when compared to the threat from droughts. However, this study is particularly concerned with summer flood events which might be considered aseasonal. As such, even where a ‘typical’ hydrological regime might include flood events, it may be that a summer flood will be experienced by species at a life stage less suited to such an event. This is particular recognised with regard to fish populations, in particular salmonids, which require high flows at some time of year, and lower flows at other times of year to breed successfully (DeVries, 1997). Thus when floods occur aseasonally, efforts should be made to maintain the hydrograph in regulated rivers to something typical for that season.

7.4 Floodplain connectivity

Relevant measures: 26,151,363,375,384

The historical association between rivers and human settlements means that many unregulated rivers are managed to reduce the risk of over-the-bank floods, for example using levees and barriers, deeper and straighter river channels so flood waters are rapidly moved downstream. Therefore, there has been a loss of lateral connectivity between the river and its floodplain. In regulated rivers, these measures aren’t always necessary because they are no or few flooding episodes, nonetheless lower flows and less variable flows have also entailed disconnection of the river and its floodplain. As a consequence of this loss of connectivity and of inundation periods, a wide range of floodplain habitats such as wetlands are threatened or have disappeared, impairing important ecosystem services such as carbon or nutrient sequestration. Also, a range of plants and animals have lost their main means of dispersal, and many birds and fish have lost crucial breeding/nursery habitat. At a catchment scale, water that would be normally ‘stored’ in wetlands and soils of the floodplain is lost from the catchment, affecting water budgets.

There has been a paradigm shift in water management away from ‘fighting’ flood water, towards more integrated approaches to flood risk, i.e. ‘making room for flood water’. An increasingly appealing measure is the reconnection of rivers and their floodplains, which may potentially accomplish three primary objectives: flood-risk reduction, an increase in floodplain goods and services, and resiliency to potential climate-change impacts. This approach is constrained by the location of human settlements and agricultural land, so needs careful management. Nonetheless because of increasing water scarcity, there is a lot of value in being able to store floodwaters for later use, and a reconnected floodplain is a way to achieve this.

7.4.1 Effects of drought

Despite providing some resilience to climate change, activities concerning the reconnection of floodplains are themselves threatened by an increasing frequency and intensity of droughts. Reduced stream flows and groundwater recharge means that the water budget of a reconnected floodplain must be carefully managed. For example, under severe drought conditions there may not be enough water available to maintain both the stream flow and wetland habitats in floodplains. If stream flows are reduced, the floodplain may not see any water for some time, during which the vegetation may have shifted from that desired. The main consequence is that money and effort will be wasted if floodplain reconnection plans do not take account of the predicted impacts of climate change on precipitation, stream flows and groundwater levels. Measures M26 (widen floodplain), M375 (use floodplain for tertiary treatment) and M384 (mini paddy fields) for example, are all vulnerable to increased droughts caused by climate change (Nõges et al., 2010b).

7.4.2 Effects of floods

By definition floodplains are expected to receive, store and release flood waters so are far more resilient to an increased frequency of floods that to an increased frequency of drought. The main risk posed by floods arises from damage to the physical habitat, as may be associated with very strong floods with high scouring flows. These can have very high socioeconomic costs, particularly when much time and effort has been invested into floodplain reconnection, and management of the floodplain habitat. Climate change induced floods pose the additional threat that they are likely to be aseasonal. This can have important consequences for the floodplain fauna or flora. Ground nesting wetland birds for examples would be particularly vulnerable during the breeding and rearing season, when major floods are not normally expected. The same argument could be made for a range of animals and plants. Little can be done to manage this risk unless an alternative route for the water can be found. Floodplain connectivity could be managed so that flood waters are routed to the floodplain during the normal flood season, and either conveyed in the main river channel or stored safely for future release at other times.

8 Conclusions

- Climate change in coming decades is likely to cause an inc4rease in river water temperature and affect the hydrological cycle so that both droughts and floods in Europe are likely to occur more often, occur seasonally and be of a greater magnitude.
- All river types are likely to be affected, but some important characteristics of rivers will determine how much they are affected by climate change, in particular morphology, and the proportion of groundwater input, as these two factors provide most of the resilience to drought
- Several major drought and summer flood events have occurred in Europe since 2000, some pan-European, some affecting only some countries.
- Several projects have already compiled information on the occurrence of droughts and floods at the European continent level.
- Both droughts and floods have the potential to affect river ecosystems profoundly, and all types of flora and fauna are likely to be impacted: algae, macrophytes, invertebrates and fish. Also terrestrial organisms associated with rivers will also be impacted, e.g. birds, mammals, amphibians.
- Though it is clear that these ecological effects will translate into impacts on ecosystem functionning and the delivery of ecosystem services, little consideration has so far been given to this subject area.
- Taxa adapted to poor water quality and habitat degradation are also better adapted to drought. From a biomonitoring perspective, ecological quality metrics for drought impacted systems are likely to be low.
- As droughts and floods are episodic in their nature, there is virtually no well designed empirical studies looking at their effects, i.e. with before after or treatment/impact experimental designs.

- Most of the information concerning the impacts of drought come from Australia. There is very little European information, and this is usually biased towards Mediterranean streams, which are naturally adapted to seasonal droughts.
- It was difficult to find recent (post 2000) information on the ecological impacts of floods on in-stream processes and biota, particular from Europe.
- It is clear that a number of human activities exacerbate the impacts of drought, but this isn't the case for floods.
- In stream habitat diversity, in particular the presence of refugia such as pools, wetted areas and the hyporheic substrate, are essential for ecological recovery from drought. The same could be said for floods, in this case in stream refugia might include low flow zones, large boulders, emergent macrophytes etc
- Adaptive management strategies concerning riparian zones, channel morphology flow and floodplain connectivity are all likely to be affected by droughts and floods, but droughts are likely to have the biggest impact, particularly when supraseasonal.
- Few adaptive strategies are resilient to both floods and droughts
- Adaptive strategies to mitigate for the impacts of droughts and of floods are sometimes incompatible.
- A strategy to capture and store flood and high flow waters, for use during droughts and low flows, is desirable.

9 References

Acuna, V., Munoz, I., Giorgi, A., Omella, M., Sabater, F. & Sabater, S. (2005) Drought and postdrought recovery cycles in an intermittent Mediterranean stream: structural and functional aspects. *Journal of the North American Benthological Society,* **24,** 919-933.

Alcamo, J., Florke, M. & Marker, M. (2007) Future long-term changes in global water resources driven by socio-economic and climatic changes. *Hydrological Sciences Journal-Journal Des Sciences Hydrologiques,* **52,** 247-275.

Allen, M. R. & Ingram, W. J. (2002) Constraints on future changes in climate and the hydrologic cycle. *Nature,* **419,** 224-232.

Antolos, M., Roby, D. D., Lyons, D. E., Collis, K., Evans, A. E., Hawbecker, M. & Ryan, B. A. (2005) Caspian tern predation on juvenile salmonids in the mid-Columbia River. *Transactions of the American Fisheries Society,* **134,** 466-480.

Armitage, P. D. (1978) DOWNSTREAM CHANGES IN COMPOSITION, NUMBERS AND BIOMASS OF BOTTOM FAUNA IN TEES BELOW COW GREEN RESERVOIR AND IN AN UNREGULATED TRIBUTARY MAIZE BECK, IN 1ST 5 YEARS AFTER IMPOUNDMENT. *Hydrobiologia,* **58,** 145-156.

Armitage, P. D. (2006) Long-term faunal changes in a regulated and an unregulated stream - Cow green thirty years on. *River Research and Applications,* **22,** 947-966.

Arthington, A. H., Bunn, S. E., Poff, N. L. & Naiman, R. J. (2006) The challenge of providing environmental flow rules to sustain river ecosystems. *Ecological Applications,* **16,** 1311-1318.

Artigas, J., Romani, A. M., Gaudes, A., Munoz, I. & Sabater, S. (2009) Organic matter availability structures microbial biomass and activity in a Mediterranean stream. *Freshwater Biology,* **54,** 2025-2036.

Bala, G., Caldeira, K., Wickett, M., Phillips, T. J., Lobell, D. B., Delire, C. & Mirin, A. (2007) Combined climate and carbon-cycle effects of large-scale deforestation. *Proceedings of the National Academy of Sciences of the United States of America,* **104,** 6550-6555.

Baldwin, D. S. & Mitchell, A. M. (2000) The effects of drying and re-flooding on the sediment and soil nutrient dynamics of lowland river-floodplain systems: A synthesis. *Regulated Rivers-Research & Management,* **16,** 457-467.

Baldwin, D. S., Rees, G. N., Mitchell, A. M. & Watson, G. (2005) Spatial and temporal variability of nitrogen dynamics in an upland stream before and after a drought. *Marine And Freshwater Research,* **56,** 457-464.

Barrat-Segretain, M. H. & Amoros, C. (1995) Influence of flood timing on the recovery of macrophytes in a former river channel. *Hydrobiologia,* **316,** 91-101.

Barredo, J. (2007) Major flood disasters in Europe: 1950–2005. *Natural Hazards,* **42,** 125-148.

Bates, B. C., Kundzewicz, Z. W., Wu, S., Palutikof, J. P. & (Eds) (2008) Climate change and water. Technical paper of the intergovernmental panel on climate change, IPCC secretariat, Geneva. 214 pp

Baxter, C. V., Fausch, K. D. & Saunders, W. C. (2005) Tangled webs: reciprocal flows of invertebrate prey link streams and riparian zones. *Freshwater Biology,* **50,** 201-220.

Beche, L. A., Connors, P. G., Resh, V. H. & Merenlender, A. M. (2009) Resilience of fishes and invertebrates to prolonged drought in two California streams. *Ecography,* **32,** 778-788.

Beche, L. A. & Resh, V. H. (2007) Short-term climatic trends affect the temporal variability of macroinvertebrates in California 'Mediterranean' streams. *Freshwater Biology,* **52,** 2317-2339.

Bender, E. A., Case, T. J. & Gilpin, M. E. (1984) PERTURBATION EXPERIMENTS IN COMMUNITY ECOLOGY - THEORY AND PRACTICE. *Ecology,* **65,** 1-13.

Benenati, P. L., Shannon, J. P. & Blinn, D. W. (1998) Desiccation and recolonization of phytobenthos in a regulated desert river: Colorado River at Lees Ferry, Arizona, USA. *Regulated Rivers-Research & Management,* **14,** 519-532.

Bernez, I., Chicouene, D. & Haury, J. (2007) Changes of Potamogeton pectinatus clumps under variable, artificially flooded river water regimes. *Belgian Journal of Botany,* **140,** 51-59.

Biedenharn, D. S., Thorne, C. R. & Watson, C. C. (2005) Wash load/bed material load concept in regional sediment management. *Proceedings of the 8th Federal Interagency Sedimentation Conference*. Reno, Nevada.

Bischoff, A. & Wolter, C. (2001) The flood of the century on the River Oder: Effects on the 0+ fish community and implications for floodplain restoration. *Regulated Rivers-Research & Management,* **17,** 171-190.

Blinn, D. W., Shannon, J. P., Stevens, L. E. & Carder, J. P. (1995) CONSEQUENCES OF FLUCTUATING DISCHARGE FOR LOTIC COMMUNITIES. *Journal Of The North American Benthological Society,* **14,** 233-248.

Bonada, N., Rieradevall, M. & Prat, N. (2007) Macroinvertebrate community structure and biological traits related to flow permanence in a Mediterranean river network. *Hydrobiologia,* **589,** 91-106.

Bonada, N., Rieradevall, M., Prat, N. & Resh, V. H. (2006) Benthic macroinvertebrate assemblages and macrohabitat connectivity in Mediterranean-climate streams of northern California. *Journal Of The North American Benthological Society,* **25,** 32-43.

Bond, N. R. (2004) Spatial variation in fine sediment transport in small upland streams: The effects of flow regulation and catchment geology. *River Research and Applications,* **20,** 705-717.

Boulton, A. J. (2003) Parallels and contrasts in the effects of drought on stream macroinvertebrate assemblages. *Freshwater Biology,* **48,** 1173-1185.

Boulton, A. J. & Lake, P. S. (2008) Effects of drought on stream insects and its ecological consequences. In: *Aquatic Insects. Challenges to Populations* (eds J. Lancaster & R. A. Briers). CAB, Wallingford, UK. pp 81-102

Brock, M. A., Nielsen, D. L., Shiel, R. J., Green, J. D. & Langley, J. D. (2003) Drought and aquatic community resilience: the role of eggs and seeds in sediments of temporary wetlands. *Freshwater Biology,* **48,** 1207-1218.

Bullock, A. & Acreman, M. (2003) The role of wetlands in the hydrological cycle. *Hydrology and Earth System Sciences,* **7,** 358-389.

Burgess, D. B. (2002) Groundwater resource management in eastern England: A quest for environmentally sustainable development. In: *Sustainable Groundwater Development* (eds K. M. Hiscock, M. O. Rivett & R. M. Davison). pp 53-62

Burgherr, P., Ward, J. V. & Glatthaar, R. (2001) Diversity, distribution and seasonality of the Simuliidae fauna in a glacial stream system in the Swiss Alps. *Archiv Fur Hydrobiologie,* **152,** 19-37.

Burkhardt-Holm, P. (2009) Climate change and decline in abundance of brown trout - is there a link? Results from Switzerland. *Umweltwissenschaften und Schadstoff-Forschung,* **21,** 177-185.

Caramujo, M. J., Mendes, C. R. B., Cartaxana, P., Brotas, V. & Boavida, M. J. (2008) Influence of drought on algal biofilms and meiofaunal assemblages of temperate reservoirs and rivers. *Hydrobiologia,* **598,** 77-94.

Carline, R. E. & McCullough, B. J. (2003) Effects of floods on brook trout populations in the Monongahela National Forest, West Virginia. *Transactions of the American Fisheries Society,* **132,** 1014-1020.

Caruso, B. S. (2002) Temporal and spatial patterns of extreme low flows and effects on stream ecosystems in Otago, New Zealand. *Journal of Hydrology,* **257,** 115-133.

CEC. (2007) *Addressing the challenge of water scarcity and droughts in the European Union. Communication from the Commission to the European Parliament and the Council. Commission of the European Communities. Brussels, 18 July 2007, COM(2007) 414 final.*

Chmielewski, F. M. & Rotzer, T. (2002) Annual and spatial variability of the beginning of growing season in Europe in relation to air temperature changes. *Climate Research,* **19,** 257-264.

Christensen, J. H. & Christensen, O. B. (2003) Climate modelling: Severe summertime flooding in Europe. *Nature,* **421,** 805-806.

Cole, G. A. & Marsh, T. J. (2006) An historical analysis of drought in England and Wales. In: *Climate Variability and Change - Hydrological Impacts* (eds S. Demuth, A. Gustard, E. Planos, F. Scatena & E. Servat). pp 483-489

Combroux, I., Bornette, G., Willby, N. J. & Amoros, C. (2001) Regenerative strategies of aquatic plants in disturbed habitats: the role of the propagule bank. *Archiv Fur Hydrobiologie,* **152,** 215-235.

Corso-Perez, G., Lanen, H. v., Bertrand, N., Chen, C., Clark, D., Folwell, S., Gosling, S., Hanasaki, N., Heinke, J. & Voss, F. (2011) Drought at the global scale in the 21st Century. Technical Report No 43. WATCH deliverables D 4.3.1. pp

Cortes, R. M. V., Ferreira, M. T., Oliveira, S. V. & Oliveira, D. (2002) Macroinvertebrate community structure in a regulated river segment with different flow conditions. *River Research and Applications,* **18,** 367-382.

Costa, J. E. & O'Connor, J. E. (1995) Geomorphologically effective floods. In: *Natural and anthropogenic influences in fluvial geomorphology* (eds J. E. Costa, A. J. Miller, K. W. Potter & P. R. Wilcock). American Geophysical Union, Washington, DC. pp

Covich, A. P., Crowl, T. A. & Scatena, F. N. (2003) Effects of extreme low flows on freshwater shrimps in a perennial tropical stream. *Freshwater Biology,* **48,** 1199-1206.

Cowx, I. G. & de Jong, M. V. (2004) Rehabilitation of freshwater fisheries: tales of the unexpected? *Fisheries Management and Ecology,* **11,** 243-249.

Cyberski, J., GrzeŚ, M., Gutry-Korycka, M., Nachlik, E. & Kundzewicz, Z. W. (2006) History of floods on the River Vistula. *Hydrological Sciences Journal,* **51,** 799-817.

Dahm, C. N., Baker, M. A., Moore, D. I. & Thibault, J. R. (2003) Coupled biogeochemical and hydrological responses of streams and rivers to drought. *Freshwater Biology,* **48,** 1219-1231.

Daufresne, M. & Boet, P. (2007) Climate change impacts on structure and diversity of fish communities in rivers. *Global Change Biology,* **13,** 2467-2478.

David, B. O. & Closs, G. P. (2002) Behavior of a stream-dwelling fish before, during, and after high-discharge events. *Transactions of the American Fisheries Society,* **131,** 762-771.

Davies-Colley, R. J. & Rutherford, J. C. (2005) Some approaches for measuring and modelling riparian shade. *Ecological Engineering,* **24,** 525-530.

Dekar, M. P. & Magoulick, D. D. (2007) Factors affecting fish assemblage structure during seasonal stream drying. *Ecology of Freshwater Fish,* **16,** 335-342.

DeVries, P. (1997) Riverine salmonid egg burial depths: review of published data and implications for scour studies. *Canadian Journal Of Fisheries And Aquatic Sciences,* **54,** 1685-1698.

Dewson, Z. S., Death, R. G. & James, A. B. W. (2003) The effect of water abstractions on invertebrate communities in four small North Island streams. *New Zealand Natural Sciences,* **28,** 51-65.

Dewson, Z. S., James, A. B. W. & Death, R. G. (2007) A review of the consequences of decreased flow for instream habitat and macroinvertebrates. *Journal of the North American Benthological Society,* **26,** 401-415.

DoleOlivier, M. J., Marmonier, P. & Beffy, J. L. (1997) Response of invertebrates to lotic disturbance: Is the hyporheic zone a patchy refugium? *Freshwater Biology,* **37,** 257-276.

Douglas, B. (1958) The ecology of the attached diatoms and other algae in small stony streams. *Journal of Animal Ecology,* **46,** 295-322.

Downes, B. J. & Lake, P. S. (1991) DIFFERENT COLONIZATION PATTERNS OF 2 CLOSELY RELATED STREAM INSECTS (AUSTROSIMULIUM-SPP) FOLLOWING DISTURBANCE. *Freshwater Biology,* **26,** 295-306.

Dunbar, M. J., Scarlett, P., Mould, D. J. & Laize, C. (2009) *Distinguishing the Relative Importance of Environmental Data Underpinning flow Pressure Assessment: DRIED-UP 3*. Report to Environment Agency (Water Resources).

Durance, I. & Ormerod, S. J. (2007) Climate change effects on upland stream macroinvertebrates over a 25-year period. *Global Change Biology,* **13,** 942-957.

Eberle, L. C. & Stanford, J. A. (2010) IMPORTANCE AND SEASONAL AVAILABILITY OF TERRESTRIAL INVERTEBRATES AS PREY FOR JUVENILE SALMONIDS IN FLOODPLAIN SPRING BROOKS OF THE KOL RIVER (KAMCHATKA, RUSSIAN FEDERATION). *River Research and Applications,* **26,** 682-694.

Eimers, M. C., Watmough, S. A., Buttle, J. M. & Dillon, P. J. (2008) Examination of the potential relationship between droughts, sulphate and dissolved organic carbon at a wetland-draining stream. *Global Change Biology,* **14,** 938-948.

Elliott, J. M. (2006) Periodic habitat loss alters the competitive coexistence between brown trout and bullheads in a small stream over 34 years. *Journal of Animal Ecology,* **75,** 54-63.

Elosegi, A., Díez, J. & Mutz, M. (2010) Effects of hydromorphological integrity on biodiversity and functioning of river ecosystems. *Hydrobiologia,* **657,** 199-215.

Erman, D. C., Andrews, E. D. & Yoderwilliams, M. (1988) EFFECTS OF WINTER FLOODS ON FISHES IN THE SIERRA-NEVADA. *Canadian Journal Of Fisheries And Aquatic Sciences,* **45,** 2195-2200.

European Union (2009) Common implementation strategy for the water framework directive (2000/60/EC). Guidance document No.24. River basin management in a changing climate. Luxembourg: Publications office of the European Union. 132 pp

Fausch, K. D., Taniguchi, Y., Nakano, S., Grossman, G. D. & Townsend, C. R. (2001) Flood disturbance regimes influence rainbow trout invasion success among five holarctic regions. *Ecological Applications,* **11,** 1438-1455.

Fausch, K. D., Torgersen, C. E., Baxter, C. V. & Li, H. W. (2002) Landscapes to riverscapes: Bridging the gap between research and conservation of stream fishes. *BioScience,* **52,** 483-498.

Feld, C. (2004) Identification and measure of hydromorphological degradation in Central European lowland streams. *Hydrobiologia,* **516,** 69-90.

Fenoglio, S., Bo, T. & Bost, G. (2006) Deep interstitial habitat as a refuge for Agabus paludosus (Fabricius) (Coleoptera : Dytiscidae) during summer droughts. *Coleopterists Bulletin,* **60,** 37-41.

Fisher, S. G., Gray, L. J., Grimm, N. B. & Busch, D. E. (1982) TEMPORAL SUCCESSION IN A DESERT STREAM ECOSYSTEM FOLLOWING FLASH FLOODING. *Ecological Monographs,* **52,** 93-110.

Fleituch, T. (2003) Structure and functional organization of benthic invertebrates in a regulated stream. *International Review of Hydrobiology,* **88,** 332-344.

Fonnesu, A., Sabetta, L. & Basset, A. (2005) Factors affecting macroinvertebrate distribution in a Mediterranean intermittent stream. *Journal of Freshwater Ecology,* **20,** 641-647.

Franklin, P., Dunbar, M. & Whitehead, P. (2008) Flow controls on lowland river macrophytes: A review. *Science of the Total Environment,* **400,** 369-378.

Franssen, N. R., Gido, K. B., Guy, C. S., Tripe, J. A., Shrank, S. J., Strakosh, T. R., Bertrand, K. N., Franssen, C. M., Pitts, K. L. & Paukert, C. P. (2006) Effects of floods on fish assemblages in an intermittent prairie stream. *Freshwater Biology,* **51,** 2072-2086.

Fritz, K. M. & Dodds, W. K. (2004) Resistance and resilience of macroinvertebrate assemblages to drying and flood in a tallgrass prairie stream system. *Hydrobiologia,* **527,** 99-112.

Gagnon, P. M., Golladay, S. W., Michener, W. K. & Freeman, M. C. (2004) Drought responses of freshwater mussels (Unionidae) in coastal plain tributaries of the Flint River basin, Georgia. *Journal of Freshwater Ecology,* **19,** 667-679.

Gaume, E., Bain, V., Bernardara, P., Newinger, O., Barbuc, M., Bateman, A., Blaskovicová, L., Blöschl, G., Borga, M. & Dumitrescu, A. (2009) A compilation of data on European flash floods. *Journal of Hydrology,* **367,** 70-78.

Giller, P. S. (2005) River restoration: seeking ecological standards. Editor's introduction. *Journal of Applied Ecology,* **42,** 201-207.

Giller, P. S., Sangpradub, N. & Twomey, H. (1991) *CATASTROPHIC FLOODING AND MACROINVERTEBRATE COMMUNITY STRUCTURE*.

Glaser, R., Riemann, D., Schönbein, J., Barriendos, M., Brázdil, R., Bertolin, C., Camuffo, D., Deutsch, M., Dobrovolný, P., van Engelen, A., Enzi, S., Halíčková, M., Koenig, S., Kotyza, O., Limanówka, D., Macková, J., Sghedoni, M., Martin, B. & Himmelsbach, I. (2010) The variability of European floods since AD 1500. *Climatic Change,* **101,** 235-256.

Golladay, S. W. & Battle, J. (2002) Effects of flooding and drought on water quality in gulf coastal plain streams in Georgia. *Journal of Environmental Quality,* **31,** 1266-1272.

Golladay, S. W., Gagnon, P., Kearns, M., Battle, J. M. & Hicks, D. W. (2004) Response of freshwater mussel assemblages (Bivalvia : Unionidae) to a record drought in the Gulf Coastal Plain of southwestern Georgia. *Journal Of The North American Benthological Society,* **23,** 494-506.

Gore, J. A., Layzer, J. B. & Mead, J. (2001) Macroinvertebrate instream flow studies after 20 years: A role in stream management and restoration. *Regulated Rivers- Research & Management,* **17,** 527-542.

Gray, L. J. & Fisher, S. G. (1981) POST-FLOOD RECOLONIZATION PATHWAYS OF MACROINVERTEBRATES IN A LOWLAND SONORAN DESERT STREAM. *American Midland Naturalist,* **106,** 249-257.

Griswold, M. W., Berzinis, R. W., Crisman, T. L. & Golladay, S. W. (2008) Impacts of climatic stability on the structural and functional aspects of macroinvertebrate communities after severe drought. *Freshwater Biology,* **53,** 2465-2483.

Gurnell, A., Thompson, K., Goodson, J. & Moggridge, H. (2008) Propagule deposition along river margins: linking hydrology and ecology. *Journal of Ecology,* **96,** 553-565.

Hakala, J. P. & Hartman, K. J. (2004) Drought effect on stream morphology and brook trout (Salvelinus fontinalis) populations in forested headwater streams. *Hydrobiologia,* **515,** 203-213.

Hänfling, B., Edwards, F. & Gherardi, F. (2011) Invasive alien Crustacea: dispersal, establishment, impact and control. *BioControl,* **56,** 573-595.

Hannaford, J., Lloyd-Hughes, B., Keef, C., Parry, S. & Prudhomme, C. (2011) Examining the large-scale spatial coherence of European drought using regional indicators of precipitation and streamflow deficit. *Hydrological Processes,* **25,** 1146-1162.

Hazel, J. E., Kaplinski, M., Parnell, R. & Manone, M. (1999) Topographic and bathymetric changes at thirty-three long-term study sites. In: *The controlled flood in Grand Canyon* (eds R. H. Webb, J. C. Schmidt, G. R. Marzolf & R. A. Valdez). American Geophysical Union, Washington, D.C. pp

Hemphill, N. & Cooper, S. D. (1983) THE EFFECT OF PHYSICAL DISTURBANCE ON THE RELATIVE ABUNDANCES OF 2 FILTER-FEEDING INSECTS IN A SMALL STREAM. *Oecologia,* **58,** 378-382.

Hendry, K., Cragg-Hine, D., O’Grady, M., Sambrook, H. & Stephen, A. (2003) Management of habitat for rehabilitation and enhancement of salmonid stocks. *Fisheries Research,* **62,** 171-192.

Hisdal, H., Roald, L. A. & Beldring, S. (2006) Past and future changes in flood and drought in the Nordic countries. In: *Climate Variability and Change - Hydrological Impacts* (eds S. Demuth, A. Gustard, E. Planos, F. Scatena & E. Servat). pp 502-507

Hoopes, R. L. (1974) FLOODING, AS RESULT OF HURRICANE AGNES, AND ITS EFFECT ON A MACROBENTHIC COMMUNITY IN AN INFERTILE HEADWATER STREAM IN CENTRAL PENNSYLVANIA. *Limnology And Oceanography,* **19,** 853-857.

Horner, R. R. & Welch, E. B. (1981) STREAM PERIPHYTON DEVELOPMENT IN RELATION TO CURRENT VELOCITY AND NUTRIENTS. *Canadian Journal Of Fisheries And Aquatic Sciences,* **38,** 449-457.

Howitt, J. A., Baldwin, D. S., Rees, G. N. & Williams, J. L. (2007) Modelling blackwater: Predicting water quality during flooding of lowland river forests. *Ecological Modelling,* **203,** 229-242.

Hutchins, M. G., Johnson, A. C., Deflandre-Vlandas, A., Comber, S., Posen, P. & Boorman, D. (2010) Which offers more scope to suppress river phytoplankton blooms: Reducing nutrient pollution or riparian shading? *Science of the Total Environment,* **408,** 5065-5077.

Hynes, H. B. N. (1958) The effect of drought on the fauna of a small mountain stream in Wales. *Verh. Internat. Verein. Limnol.,* **13,** 826-833.

Hynes, H. B. N. (1961) The invertbrate fauna of a Welsh mountain stream. *Archiv für Hydrobiologie,* **57,** 344-388.

IHF/CEH (2001) Assessment of the Regional Impact of Droughts in Europe. Final Report to the European Union ENV-CT97-0553 Institute of Hydrology, University of Freiburg Germany. pp

Ilg, C., Foeckler, F., Deichner, O. & Henle, K. (2009) Extreme flood events favour floodplain mollusc diversity. *Hydrobiologia,* **621,** 63-73.

IPCC. (2007a) *Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge University Press, Cambridge and New York.

IPCC (2007b) Summary for policy makers. In: Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press Cambridge and New York. 7-22 pp

Jacobs, S. M., Bechtold, J. S., Biggs, H. C., Grimm, N. B., Lorentz, S., McClain, M. E., Naiman, R. J., Perakis, S. S., Pinay, G. & Scholes, M. C. (2007) Nutrient vectors and riparian processing: A review with special reference to African semiarid Savanna ecosystems. *Ecosystems,* **10,** 1231-1249.

Jenkins, G. J., Perry, M. C. & Prior, M. J. (2008) The Climate of the United Kingdom and Recent Trends Met Office Hadley Centre Exeter, UK. 120 pp

Jensen, A. J. & Johnsen, B. O. (1999) The functional relationship between peak spring floods and survival and growth of juvenile Atlantic Salmon (Salmo salar) and Brown Trout (Salmo trutta). *Functional Ecology,* **13,** 778-785.

Julien, P. Y. (1998) *Erosion and Sedimentation*. Cambridge University Press, Cambridge.

Junk, W. J., Bayley, P. B. & Sparks, R. E. (1989) THE FLOOD PULSE CONCEPT IN RIVER-FLOODPLAIN SYSTEMS. *Canadian Special Publication of Fisheries and Aquatic Sciences,* **106,** 110-127.

Jurajda, P., Ondrackova, M. & Reichard, M. (2004) Managed flooding as a tool for supporting natural fish reproduction in man-made lentic water bodies. *Fisheries Management and Ecology,* **11,** 237-242.

Jurajda, P., Reichard, M. & Smith, C. (2006) Immediate impact of an extensive summer flood on the adult fish assemblage of a channelized lowland river. *Journal of Freshwater Ecology,* **21,** 493-501.

Keaton, M., Haney, D. & Andersen, C. B. (2005) Impact of drought upon fish assemblage structure in two South Carolina Piedmont streams. *Hydrobiologia,* **545,** 209-223.

Kinzie, R. A., Chong, C., Devrell, J., Lindstrom, D. & Wolff, R. (2006) Effects of water removal on a Hawaiian stream ecosystem. *Pacific Science,* **60,** 1-47.

Klein Tank, A., Wijngaard, J. & Engelen, A. (2002) Climate of Europe - The assessment of observed daily temperature and precipitation extremes. European Climate Assessment (ECA), KNMI, De Bilt, Netherlands 36 pp

Kundzewicz, Z. W. (2009) Adaptation to floods and droughts in the Baltic Sea basin under climate change. *Boreal Environment Research,* **14,** 193-203.

Kundzewicz, Z. W., Hirabayashi, Y. & Kanae, S. (2010) River floods in the changing climate—observations and projections. *Water Resources Management,* **24,** 2633-2646.

Kundzewicz, Z. W., Szamalek, K. & Kowalczak, P. (1999) The Great Flood of 1997 in Poland. *Hydrological Sciences Journal,* **44,** 855-870.

Kundzewicz, Z. W., Ulbrich, U., brücher, T., Graczyk, D., Krüger, A., Leckebusch, G. C., Menzel, L., Pińskwar, I., Radziejewski, M. & Szwed, M. (2005) Summer Floods in Central Europe – Climate Change Track? *Natural Hazards,* **36,** 165-189.

Labbe, T. R. & Fausch, K. D. (2000) Dynamics of intermittent stream habitat regulate persistence of a threatened fish at multiple scales. *Ecological Applications,* **10,** 1774-1791.

Ladle, M. (2002) Review of Flow Needs for Fish and Fisheries.R & D Technical Report W159. pp

Lake, P. S. (2000) Disturbance, patchiness, and diversity in streams. *Journal Of The North American Benthological Society,* **19,** 573-592.

Lake, P. S. (2003) Ecological effects of perturbation by drought in flowing waters. *Freshwater Biology,* **48,** 1161-1172.

Lake, P. S. (2007) Flow-generated disturbances and ecological responses: floods and droughts. In: *Hydroecology and ecohydrology: past, present and future* (eds P. J. Wood, D. M. Hannah & J. P. Sadler). John Wiley & Sons, Chichester. pp 75-92

Lake, P. S. (2011a) Conclusions. In: *Drought and Aquatic Ecosystems: Effects and Responses*. John Wiley & Sons, Ltd. pp 290-299

Lake, P. S. (2011b) *Drought and aquatic ecosystems: Effects and responses*. John Wiley & Sons, Chichester.

Lake, P. S. (2011c) Drought and Temporary Waters. In: *Drought and Aquatic Ecosystems: Effects and Responses*. John Wiley & Sons, Ltd. pp 100-133

Lake, P. S. (2011d) Human-Induced Exacerbation of Drought Effects on Aquatic Ecosystems. In: *Drought and Aquatic Ecosystems: Effects and Responses*. John Wiley & Sons, Ltd. pp 265-289

Lake, P. S. (2011e) Types of drought and their assessment. In: *Drought and Aquatic Ecosystems: Effects and Responses*. John Wiley & Sons, Ltd. pp 20-34

Lake, P. S. (2011f) Water Bodies, Catchments and the Abiotic Effects of Drought. In: *Drought and Aquatic Ecosystems: Effects and Responses*. John Wiley & Sons, Ltd. pp 68-99

Lake, P. S. & Barmuta, L. (1986) Stream benthic communities: persistent presumptions and current speculations. *Limnology in Australia* (eds P. De Deckker & W. D. Williams). CSIRO, Melbourne.

Lamontagne, S., Hicks, W. S., Fitzpatrick, R. W. & Rogers, S. (2006) Sulfidic materials in dryland river wetlands. *Marine And Freshwater Research,* **57,** 775-788.

Lancaster, J. & Belyea, L. R. (1997) Nested hierarchies and scale-dependence of mechanisms of flow refugium use. *Journal Of The North American Benthological Society,* **16,** 221-238.

Lane, S. N. (2007) Other floods and their drivers. In: *Future Flooding and Coastal Erosion Risks* (eds C. R. Thorne, E. P. Evans & E. C. Penning-Rowsell). Thomas Telford, London. pp

Lang, C., Gille, E., Francois, D. & Auer, J. C. (2006) PRESAGES: A collection of tools for predicting low flows. In: *Climate Variability and Change - Hydrological Impacts* (eds S. Demuth, A. Gustard, E. Planos, F. Scatena & E. Servat). pp 145-150

Larned, S. T. (2000) Dynamics of coarse riparian detritus in a Hawaiian stream ecosystem: a comparison of drought and post-drought conditions. *Journal of the North American Benthological Society,* **19,** 215-234.

Lawrence, P. J. & Chase, T. N. (2009) Climate Impacts of Making Evapotranspiration in the Community Land Model (CLM3) Consistent with the Simple Biosphere Model (SiB). *Journal of Hydrometeorology,* **10,** 374-394.

Ledger, M. E., Edwards, F. K., Brown, L. E., Milner, A. M. & Woodward, G. U. Y. (2011) Impact of simulated drought on ecosystem biomass production: an experimental test in stream mesocosms. *Global Change Biology,* **17,** 2288-2297.

Ledger, M. E., Harris, R. M. L., Armitage, P. D. & Milner, A. M. (2008) Disturbance frequency influences patch dynamics in stream benthic algal communities. *Oecologia,* **155,** 809-819.

Ledger, M. E. & Hildrew, A. G. (2001) Recolonization by the benthos of an acid stream following a drought. *Archiv Fur Hydrobiologie,* **152,** 1-17.

Lehner, B., Doll, P., Alcamo, J., Henrichs, T. & Kaspar, F. (2006) Estimating the impact of global change on flood and drought risks in europe: A continental, integrated analysis. *Climatic Change,* **75,** 273-299.

Llasat, M., Llasat-Botija, M., Prat, M., Porcú, F., Price, C., Mugnai, A., Lagouvardos, K., Kotroni, V., Katsanos, D. & Michaelides, S. (2010) High-impact floods and flash floods in Mediterranean countries: the FLASH preliminary database. *Advances in Geosciences,* **23,** 47-55.

Lloyd-Hughes, B., Prudhomme, C., Hannaford, J., Parry, S., Keef, C. & Rees, G. (2009) The Spatial Coherence of European Droughts-UK and European Drought catalogues. Science Report SC070079/SR1. Environment Agency UK. pp

Lojkasek, B., Lusk, S., Halacka, K., Luskova, W. & Drozd, P. (2005) The impact of the extreme floods in July 1997 on the ichthyocenosis of the Oder Catchment area (Czech Republic). *Hydrobiologia,* **548,** 11-22.

Lusk, S., Halacka, K. & Luskova, V. (1998) The effect of an extreme flood on the fish communities in the upper reaches of the Ticha Orlice river (the Labe drainage area). *Czech Journal of Animal Science,* **43,** 531-536.

Lusk, S., Hartvich, P., Halacka, K., Luskova, V. & Holub, M. (2004) Impact of extreme floods on fishes in rivers and their floodplains. *Ecohydrology & Hydrobiology,* **4,** 173-181.

Lytle, D. A. (2003) Reconstructing long-term flood regimes with rainfall data: Effects of flood timing on caddisfly populations. *Southwestern Naturalist,* **48,** 36-42.

Lytle, D. A. & Poff, N. L. (2004a) Adaptation to natural flow regimes. *Trends in Ecology and Evolution,* **19,** 94-100.

Lytle, D. A. & Poff, N. L. (2004b) Adaptation to natural flow regimes. *Trends in Ecology & Evolution,* **19,** 94-100.

Maceda-Veiga, A., Salvado, H., Vinyoles, D. & De Sostoa, A. (2009) Outbreaks of Ichthyophthirius multifiliis in Redtail Barbs Barbus haasi in a Mediterranean Stream during Drought. *Journal of Aquatic Animal Health,* **21,** 189-194.

Mackay, R. J. (1992) COLONIZATION BY LOTIC MACROINVERTEBRATES - A REVIEW OF PROCESSES AND PATTERNS. *Canadian Journal Of Fisheries And Aquatic Sciences,* **49,** 617-628.

Magalhaes, M. F., Beja, P., Canas, C. & Collares-Pereira, M. J. (2002) Functional heterogeneity of dry-season fish refugia across a Mediterranean catchment: the role of habitat and predation. *Freshwater Biology,* **47,** 1919-1934.

Magalhaes, M. F., Schlosser, I. J. & Collares-Pereira, M. J. (2003) The role of life history in the relationship between population dynamics and environmental variability in two Mediterranean stream fishes. *Journal of Fish Biology,* **63,** 300-317.

Magoulick, D. D. & Kobza, R. M. (2003) The role of refugia for fishes during drought: a review and synthesis. *Freshwater Biology,* **48,** 1186-1198.

Majercakova, O., Fendekova, M. & Leskova, D. (1997) The variability of hydrological series due to extreme climate conditions and the possible change of the hydrological characteristics with respect to potential climate change. In: *Friend'97-Regional Hydrology: Concepts and Models for Sustainable Water Resource Management* (eds A. Gustard, S. Blazkova, M. Brilly, S. Demuth, J. Dixon, H. VanLanen, C. Llasat, S. Mkhandi & E. Servat). pp 59-66

Marchi, L., Borga, M., Preciso, E. & Gaume, E. (2010) Characterisation of selected extreme flash floods in Europe and implications for flood risk management. *Journal of Hydrology,* **394,** 118-133.

Marsh, T., Cole, G. A. & Wilby, R. (2007) Major droughts in England and Wales, 1800 - 2006. *Weather,* **62,** 87-93.

Matthaei, C. D., Townsend, C. R., Arbuckle, C. J., Peacock, K., Guggelberger, C., Kuster, C. E. & Huber, H. (2004) Disturbance, assembly rules, and benthic communities in running water: A review and some implications for restoration projects. In: *Assembly rules and restoration ecology. Bridging the gap between theory and practice* (eds V. M. Temperton, R. J. Hobbs, T. Nuttle & S. Halle). Island Press, Washington, USA. pp

Matthews, W. J. & Marsh-Matthews, E. (2003) Effects of drought on fish across axes of space, time and ecological complexity. *Freshwater Biology,* **48,** 1232-1253.

McIntosh, M. D., Benbow, M. E. & Burky, A. J. (2002) Effects of stream diversion on riffle macroinvertebrate communities in a Maui, Hawaii, Stream. *River Research and Applications,* **18,** 569-581.

McKenzie-Smith, F. J., Bunn, S. E. & House, A. P. N. (2006) Habitat dynamics in the bed sediments of an intermittent upland stream. *Aquatic Sciences,* **68,** 86-99.

McMaster, D. & Bond, N. (2008) A field and experimental study on the tolerances of fish to Eucalyptus camaldulensis leachate and low dissolved oxygen concentrations. *Marine And Freshwater Research,* **59,** 177-185.

Medina-Elizalde, M. & Rohling, E. J. (2012) Collapse of Classic Maya Civilization Related to Modest Reduction in Precipitation. *Science,* **335,** 956-959.

Meffe, G. K. (1984) EFFECTS OF ABIOTIC DISTURBANCE ON COEXISTENCE OF PREDATOR-PREY FISH SPECIES. *Ecology,* **65,** 1525-1534.

Meier, W., Bonjour, C., Wuest, A. & Reichert, P. (2003) Modeling the effect of water diversion on the temperature of mountain streams. *Journal of Environmental Engineering-Asce,* **129,** 755-764.

Meyer, C. B. (2003) The importance of measuring biotic and abiotic factors in the lower egg pocket to predict coho salmon egg survival. *Journal of Fish Biology,* **62,** 534-548.

Middleton, G. V. & Southard, J. B. (1977) *Mechanics of sediment movement. Short courses No.3*. Society of Economic Paleontologists and Mineralogists, Binghampton.

Milly, P. C. D., Dunne, K. A. & Vecchia, A. V. (2005) Global pattern of trends in streamflow and water availability in a changing climate. *Nature,* **438,** 347-350.

Minar, J., Gelbic, I. & Olejnicek, J. (2001) The effect of floods on the development of mosquito populations in the middle and lower river Morava Regions. *Acta Universitatis Carolinae-Biologica,* **45,** 139-146.

Mishra, A. K. & Singh, V. P. (2010) A review of drought concepts. *Journal of Hydrology,* **391,** 202-216.

Mitchell, J. K. (2003) European River Floods in a Changing World. *Risk Analysis,* **23,** 567-574.

Moffett, J. W. (1936) A quantitative study of the bottom fauna in some Utah streams of different size from the same drainage basin. *American Naturalist,* **125,** 16-38.

Morrison, B. R. S. (1990) Recolonisation of 4 small streams in central Scotland following drought conditions in 1984 *Hydrobiologia,* **208,** 261-267.

Mudelsee, M., Börngen, M., Tetzlaff, G. & Grünewald, U. (2003) No upward trends in the occurrence of extreme floods in central Europe. *Nature,* **425,** 166-169.

Naiman, R. J. & Decamps, H. (1997) The ecology of interfaces: Riparian zones. *Annual Review Of Ecology And Systematics,* **28,** 621-658.

Nicola, G. G., Almodovar, A. & Elvira, B. (2009) Influence of hydrologic attributes on brown trout recruitment in low-latitude range margins. *Oecologia,* **160,** 515-524.

Nixon, S. (2008) Summary of information received from member states on best practices and approaches for a climate check of the first programmes of measures. . WRc report to the strategic steering group on climate change and water. Version no. 2, 21 August 2008 41 pp

NOAA (2011) NOAA National Climatic Data Center, State of the Climate: Global Analysis for Annual 2011, published online December 2011, retrieved on January 20, 2012 from http://www.ncdc.noaa.gov/sotc/global/. pp

Nõges, T., Nõges, P. & Cardoso, A. C. (2010a) Climate change adaptation and mitigation strategies already in practice based on the first river basin management plans of the EU member states.EUR 24663 EN. Luxembourg: Publications Office of the European Union. 127 pp

Nõges, T., Nõges, P. & Cardoso, A. C. (2010b) Review of published climate change adaptation and mitigation measures related with water.EUR 24682 EN. Luxembourg: Publications Office of the European Union. 127 pp

Norris, R. H., Linke, S., Prosser, I., Young, W. J., Liston, P., Bauer, N., Sloane, N., Dyer, F. & Thoms, M. (2007) Very-broad-scale assessment of human impacts on river condition. *Freshwater Biology,* **52,** 959-976.

O'Connell, M., Baldwin, D. S., Robertson, A. I. & Rees, G. (2000) Release and bioavailability of dissolved organic matter from floodplain litter: influence of origin and oxygen levels. *Freshwater Biology,* **45,** 333-342.

Ortlepp, J. & Murle, U. (2003) Effects of experimental flooding on brown trout (Salmo trutta fario L.): The River Spol, Swiss National Park. *Aquatic Sciences,* **65,** 232-238.

Parkyn, S. M., Davies-Colley, R. J., Cooper, A. B. & Stroud, M. J. (2005) Predictions of stream nutrient and sediment yield changes following restoration of forested riparian buffers. *Ecological Engineering,* **24,** 551-558.

Parnell, R. A., Bennett, J. & Stevens, L. (1999) Mineralization of riparian vegetation buried by the 1996 controlled flood. In: *The controlled flood in Grand Canyon* (eds R. H. Webb, J. S. Schmidt, G. R. Marzolf & R. A. Valdez). American Geophysical Union, Washington, D.C. pp

Parry, S., Hannaford, J., Prudhomme, C., Lloyd-Hughes, B. & Williamson, J. (2011) Objective drought and high flow catalogues for Europe. Technical Report No 33. WATCH deliverable 4.1.6a and 4.1.6b. Centre for Ecology & Hydrology UK. pp

Parry, S., Lloyd-Hughes, B., Hannaford, J., Prudhomme, C. & Keef, C. (2009) The Spatial Coherence of European Droughts-Summaries of Major Historical Droughts. Science Report SC070079/SR4. Environment Agency UK. pp

Parry, S., Prudhomme, C., Hannaford, J. & Lloyd-Hughes, B. (2010) Examining the spatio-temporal evolution and characteristics of large-scale European droughts. *BHS Third International*

Symposium, Managing Consequences of a Changing Global Environment, pp. 135-142. British Hydrological Society, Newcastle.

Pekarova, P., Miklanek, P. & Pekar, J. (2006) Long-term trends and runoff fluctuations of European rivers. In: *Climate Variability and Change - Hydrological Impacts* (eds S. Demuth, A. Gustard, E. Planos, F. Scatena & E. Servat). pp 520-525

Phillips, R. W., Lantz, R. L., Claire, E. W. & Moring, J. R. (1975) SOME EFFECTS OF GRAVEL MIXTURES ON EMERGENCE OF COHO SALMON AND STEELHEAD TROUT FRY. *Transactions of the American Fisheries Society,* **104,** 461-466.

Pinna, M. & Basset, A. (2004) Summer drought disturbance on plant detritus decomposition processes in three River Tirso (Sardinia, Italy) sub-basins. *Hydrobiologia,* **522,** 311-319.

Pires, D. F., Pires, A. M., Collares-Pereira, M. J. & Magalhaes, M. F. (2010) Variation in fish assemblages across dry-season pools in a Mediterranean stream: effects of pool morphology, physicochemical factors and spatial context. *Ecology of Freshwater Fish,* **19,** 74-86.

Poff, N. L. (1992) WHY DISTURBANCES CAN BE PREDICTABLE - A PERSPECTIVE ON THE DEFINITION OF DISTURBANCE IN STREAMS. *Journal Of The North American Benthological Society,* **11,** 86-92.

Poff, N. L. (1997) Landscape filters and species traits: Towards mechanistic understanding and prediction in stream ecology. *Journal Of The North American Benthological Society,* **16,** 391-409.

Poff, N. L., Brinson, M. M. & Day Jr, J. W. (2002) Aquatic ecosysems and global climate change: Potential impacts on inland freshwater and coastal wetland ecosystems in the United States. Pew Center on Global Climate Change, Arlington, VA. 45 pp

Poff, N. L. & Zimmerman, J. K. H. (2010) Ecological responses to altered flow regimes: a literature review to inform the science and management of environmental flows. *Freshwater Biology,* **55,** 194-205.

Polard, T., Jean, S., Gauthier, L., Laplanche, C., Merlina, G., Sanchez-Perez, J. M. & Pinelli, E. (2011) Mutagenic impact on fish of runoff events in agricultural areas in south-west France. *Aquatic Toxicology,* **101,** 126-134.

Power, M. E., Parker, M. S. & Dietrich, W. E. (2008) Seasonal reassembly of a river food web: Floods, droughts, and impacts of fish. *Ecological Monographs,* **78,** 263-282.

Power, M. E. & Stewart, A. J. (1987) DISTURBANCE AND RECOVERY OF AN ALGAL ASSEMBLAGE FOLLOWING FLOODING IN AN OKLAHOMA STREAM. *American Midland Naturalist,* **117,** 333-345.

Querner, E., Tallaksen, L., Kasparek, L. & Van Lanen, H. (1997) Impact of land-use, climate change and groundwater abstraction on streamflow droughts using physically-based models. In: *Regional hydrology: concepts and models for sustainable water resource management; proceedings of the Third International Conference on FRIEND held at Postojna, Slovenia, from 30 September to 4 October 1997* (eds A. Gustard, S. Blazkova, M. Brilly, S. Demuth, J. Dixon, H. Van Lanen, C. Llasat, S. Mkhandi & E. Servat). IAHS Press, Wallingford, UK. pp 171-180

Reed, D. & Robson, A. (1999) *Volume 3: Statistical procedures for flood freequency estimation*. Centre for Ecology and Hydrology.

Reice, S. R., Wissmar, R. C. & Naiman, R. J. (1990) DISTURBANCE REGIMES, RESILIENCE, AND RECOVERY OF ANIMAL COMMUNITIES AND HABITATS IN LOTIC ECOSYSTEMS. *Environmental Management,* **14,** 647-659.

Reichard, M. & Jurajda, P. (2004) The effects of elevated river discharge on the downstream drift of young-of-the-year cyprinid fishes. *Journal of Freshwater Ecology,* **19,** 465-471.

Resh, V. H., Brown, A. V., Covich, A. P., Gurtz, M. E., Li, H. W., Minshall, G. W., Reice, S. R., Sheldon, A. L., Wallace, J. B. & Wissmar, R. C. (1988) THE ROLE OF DISTURBANCE IN STREAM ECOLOGY. *Journal Of The North American Benthological Society,* **7,** 433-455.

Roberts, J. (1983) Forest transpiration: A conservative hydrological process? *Journal of Hydrology,* **66,** 133-141.

Robinson, C. T., Aebischer, S. & Uehlinger, U. (2004) Immediate and habitat-specific responses of macroinvertebrates to sequential, experimental floods. *Journal Of The North American Benthological Society,* **23,** 853-867.

Robinson, M. (1990) Impact of improved land drainage on river flows. Institute of Hydrology Wallingford, UK. 233 pp

Robson, B. J. & Matthews, T. G. (2004) Drought refuges affect algal recolonization in intermittent streams. *River Research and Applications,* **20,** 753-763.

Romanello, G. A., Chuchra-Zbytniuk, K. L., Vandermer, J. L. & Touchette, B. W. (2008) Morphological adjustments promote drought avoidance in the wetland plant Acorus americanus. *Aquatic Botany,* **89,** 390-396.

Romani, A. M., Vazquez, E. & Butturini, A. (2006) Microbial availability and size fractionation of dissolved organic carbon after drought in an intermittent stream: Biogeochemical link across the stream-riparian interface. *Microbial Ecology,* **52,** 501-512.

Ruegg, J. & Robinson, C. T. (2004) Comparison of macroinvertebrate assemblages of permanent and temporary streams in an Alpine flood plain, Switzerland. *Archiv Fur Hydrobiologie,* **161,** 489-510.

Scherwass, A. & Arndt, H. (2005) Structure, dynamics and control of the ciliate fauna in the potamoplankton of the River Rhine. *Archiv Fur Hydrobiologie,* **164,** 287-307.

Schlief, J. & Mutz, M. (2011) Leaf Decay Processes during and after a Supra-Seasonal Hydrological Drought in a Temperate Lowland Stream. *International Review of Hydrobiology,* **96,** 633-655.

Schmidt, J. C., Parnell, R. A., Grams, P. E., Hazel, J. E., Kaplinski, M. A., Stevens, L. E. & Hoffnagle, T. L. (2001) The 1996 controlled flood in Grand Canyon: Flow, sediment transport, and geomorphic change. *Ecological Applications,* **11,** 657-671.

Scrimgeour, G. J., Davidson, R. J. & Davidson, J. M. (1988) RECOVERY OF BENTHIC MACROINVERTEBRATE AND EPILITHIC COMMUNITIES FOLLOWING A LARGE FLOOD, IN AN UNSTABLE, BRAIDED, NEW-ZEALAND RIVER. *New Zealand Journal of Marine and Freshwater Research,* **22,** 337-344.

Sedell, J. R., Reeves, G. H., Hauer, F. R., Stanford, J. A. & Hawkins, C. P. (1990) ROLE OF REFUGIA IN RECOVERY FROM DISTURBANCES - MODERN FRAGMENTED AND DISCONNECTED RIVER SYSTEMS. *Environmental Management,* **14,** 711-724.

SEPA (2009) Engineering in the environment good practice guide:riparian vegetation management.WAT-SG-44. Scottish Environmental Protection Agency, Stirling. 47 pp

Shannon, J. P., Blinn, D. W., McKinney, T., Benenati, E. P., Wilson, K. P. & O'Brien, C. (2001) Aquatic food base response to the 1996 test flood below Glen Canyon Dam, Colorado River, Arizona. *Ecological Applications,* **11,** 672-685.

Shepley, M. G., Streetly, M., Voyce, K. & Bamford, F. (2009) Management of stream compensation for a large conjunctive use scheme, Shropshire, UK. *Water and Environment Journal,* **23,** 263-271.

Smakhtin, V. U. (2001) Low flow hydrology: a review. *Journal of Hydrology,* **240,** 147-186.

Sophocleous, M. (2002) Interactions between groundwater and surface water: the state of the science. *Hydrogeology Journal,* **10,** 52-67.

Sprague, L. A. (2005) Drought effects on water quality in the South Platte River Basin, Colorado. *Journal of the American Water Resources Association,* **41,** 11-24.

Stahl, K., Hisdal, H., Hannaford, J., Tallaksen, L. M., van Lanen, H. A. J., Sauquet, E., Demuth, S., Fendekova, M. & Jódar, J. (2010) Streamflow trends in Europe: evidence from a dataset of near-natural catchments. *Hydrology and Earth System Sciences,* **14,** 2367-2382.

Stanley, E. H., Fisher, S. G. & Jones, J. B. (2004) Effects of water loss on primary production: A landscape-scale model. *Aquatic Sciences,* **66,** 130-138.

Steiger, J., Gurnell, A. M. & Petts, G. E. (2001) Sediment deposition along the channel margins of a reach of the middle River Severn, UK. *Regulated Rivers-Research & Management,* **17,** 443-460.

Steinman, A. D. & McIntire, C. D. (1990) RECOVERY OF LOTIC PERIPHYTON COMMUNITIES AFTER DISTURBANCE. *Environmental Management,* **14,** 589-604.

Stubbington, R., Greenwood, A. M., Wood, P. J., Armitage, P. D., Gunn, J. & Robertson, A. L. (2009) The response of perennial and temporary headwater stream invertebrate communities to hydrological extremes. *Hydrobiologia,* **630,** 299-312.

Suren, A. M., Biggs, B. J. F., Duncan, M. J., Bergey, L. & Lambert, P. (2003a) Benthic community dynamics during summer low-flows in two rivers of contrasting enrichment 2. Invertebrates. *New Zealand Journal of Marine and Freshwater Research,* **37,** 71-83.

Suren, A. M., Biggs, B. J. F., Kilroy, C. & Bergey, L. (2003b) Benthic community dynamics during summer low-flows in two rivers of contrasting enrichment 1. Periphyton. *New Zealand Journal of Marine and Freshwater Research,* **37,** 53-70.

Svensson, C. & Prudhomme, C. (2005) Prediction of British summer river flows using winter predictors. *Theoretical and Applied Climatology,* **82,** 1-15.

Tallaksen, L. M., Stahl, K. & Wong, G. (2011) Space-time characteristics of large-scale droughts in Europe derived from streamflow observations and WATCH multi-model simulations. Technical Report 48. WATCH deliverable 4.2.3. University of Oslo Norway. pp

Tate, E. L. & Gustard, A. (2000) Drought definition: a hydrological perspective. In: *Drought and Drought Mitigation in Europe* (eds J. V. Vogt & F. Somma). Kluwer, Dordrecht. pp 23-48

Tavzes, B., Urbanic, G. & Toman, M. J. (2006) Biological and hydromorphological integrity of the small urban stream. *Physics and Chemistry of the Earth,* **31,** 1062-1074.

Thomson, J. R. (2002) The effects of hydrological disturbance on the densities of macroinvertebrate predators and their prey in a coastal stream. *Freshwater Biology,* **47,** 1333-1351.

Thorne, C. R. (1990) Effects of vegetation on riverbank erosion and stability. In: *Vegetation and Erosion* (ed J. B. Thornes). John Wiley & Sons, Chichester. pp

Thorup, J. (1970) INFLUENCE OF A SHORT-TERMED FLOOD ON A SPRINGBROOK COMMUNITY. *Archiv Fur Hydrobiologie,* **66,** 447-&.

Tickner, D. P., Angold, P. G., Gurnell, A. M. & Mountford, J. O. (2001) Riparian plant invasions: hydrogeomorphological control and ecological impacts. *Progress in Physical Geography,* **25,** 22-52.

Tipping, E., Smith, E. J., Lawlor, A. J., Hughes, S. & Stevens, P. A. (2003) Predicting the release of metals from ombrotrophic peat due to drought-induced acidification. *Environmental Pollution,* **123,** 239-253.

Tollan, A. (2002) Land-use change and floods: what do we need most, research or management? *Water Science & Technology,* **45,** 183-190.

Touchette, B. W., Iannacone, L. R., Turner, G. E. & Frank, A. R. (2007) Drought tolerance versus drought avoidance: A comparison of plant-water relations in herbaceous wetland plants subjected to water withdrawal and repletion. *Wetlands,* **27,** 656-667.

Townsend, C. R. (1989) THE PATCH DYNAMICS CONCEPT OF STREAM COMMUNITY ECOLOGY. *Journal Of The North American Benthological Society,* **8,** 36-50.

Ulbrich, U., Brücher, T., Fink, A. H., Leckebusch, G. C., Krüger, A. & Pinto, J. G. (2003a) The central European floods of August 2002: Part 1–Rainfall periods and flood development. *Weather,* **58,** 371-377.

Ulbrich, U., Brücher, T., Fink, A. H., Leckebusch, G. C., Krüger, A. & Pinto, J. G. (2003b) The central European floods of August 2002: Part 2–Synoptic causes and considerations with respect to climatic change. *Weather,* **58,** 434-442.

Valdez, R. A., Hoffnagle, T. L., McIvor, C. C., McKinney, T. & Leibfried, W. C. (2001) Effects of a test flood on fishes of the Colorado River in Grand Canyon, Arizona. *Ecological Applications,* **11,** 686-700.

van Lanen, H. A. J., Tallaksen, L. M. & Rees, G. (2007) Droughts and climate change. Annex II in: Comission staff working document impact assessment (SEC(2007)(993)). Water policy in the EU: communication on water security and drought. Brussels. 13 pp van Vliet, M. T. H. & Zwolsman, J. J. G. (2008) Impact of summer droughts on the water quality of the Meuse river. *Journal of Hydrology,* **353,** 1-17.

Wade, A. J., Whitehead, P. G., Hornberger, G. M. & Snook, D. L. (2002) On modelling the flow controls on macrophyte and epiphyte dynamics in a lowland permeable catchment: the River Kennet, southern England. *Science of the Total Environment,* **282,** 375-393.

Wallerstein, N. P. & Thorne, C. R. (1998) *Computer model for prediction of scour at bridges affected by large woody debris*.

Ward, J. V. (1976) Effects fo flow patterns below large dams on stream benthos; a review. *Instream flow needs symposium viol 2* pp. 235-253. American Fish Society.

Webb, B. W. & Nobilis, F. (2007) Long-term changes in river temperature and the influence of climatic and hydrological factors. *Hydrological Sciences Journal-Journal Des Sciences Hydrologiques,* **52,** 74-85.

Weng, Z. Y., Mookerji, N. & Mazumder, A. (2001) Nutrient-dependent recovery of Atlantic salmon streams from a catastrophic flood. *Canadian Journal Of Fisheries And Aquatic Sciences,* **58,** 1672-1682.

Westwood, C. G., Teeuw, R. M., Wade, P. M. & Holmes, N. T. H. (2006) Prediction of macrophyte communities in drought-affected groundwater-fed headwater streams. *Hydrological Processes,* **20,** 127-145.

White, J. L. & Harvey, B. C. (2001) Effects of an introduced piscivorous fish on native benthic fishes in a coastal river. *Freshwater Biology,* **46,** 987-995.

Wigley, T. M. L. (2005) The Climate Change Commitment. *Science,* **307,** 1766-1769.

Wilbers, G.-J., Zwolsman, G., Klaver, G. & Hendriks, A. J. (2009) Effects of a drought period on physico-chemical surface water quality in a regional catchment area. *Journal of Environmental Monitoring,* **11,** 1298-1302.

Williamson, J., Parry, S., Goodsell, G., Prudhomme, C. & Hannaford, J. (2011) Large-scale hydrological extremes in Europe: past and future simulations. Technical Report No 29. WATCH deliverable 4.3.1a and 4.3.1b. Centre for Ecology & Hydrology UK. pp

Wood, P. J., Agnew, M. D. & Petts, G. E. (2000) Flow variations and macroinvertebrate community responses in a small groundwater-dominated stream in south-east England. *Hydrological Processes,* **14,** 3133-3147.

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

Wood, P. J., Boulton, A. J., Little, S. & Stubbington, R. (2010) Is the hyporheic zone a refugium for aquatic macroinvertebrates during severe low flow conditions? *Fundamental and Applied Limnology,* **176,** 377-390.

Woodward, G., Jones, J. I. & Hildrew, A. G. (2002) Community persistence in Broadstone Stream (UK) over three decades. *Freshwater Biology,* **47,** 1419-1435.

Worrall, F., Burt, T. P. & Adamson, J. K. (2006) Trends in drought frequency - The fate of DOC export from British peatlands. *Climatic Change,* **76,** 339-359.

Wright, J. F., Gunn, R. J. M., Winder, J. M., Wiggers, R., Kneebone, N. T. & Clarke, R. T. (2002a) The impact of drought events in 1976 and 1997 on the macroinvertebrate fauna of a chalk stream. In: *International Association of Theoretical and Applied Limnology, Vol 28, Pt 2, Proceedings* (ed R. G. Wetzel). pp 948-952

Wright, J. F., Gunn, R. J. M., Winder, J. M., Wiggers, R., Vowles, K., Clarke, R. T. & Harris, I. (2002b) A comparison of the macrophyte cover and macroinvertebrate fauna at three sites on the River Kennet in the mid 1970s and late 1990s. *Science of the Total Environment,* **282,** 121-142.

Yount, J. D. & Niemi, G. J. (1990) Recovery of Lotic Communities and Ecosystems from Disturbance - a Narrative Review of Case-Studies. *Environmental Management,* **14,** 547.

Zarski, T. P., Rokicki, E., Rieder-Zarska, J. & Zarska, H. (2002) Effect of flood stage on mercury tissue concentration in fish caught in the middle course of the Vistula River. *Medycyna Weterynaryjna,* **58,** 703-704.

Zielinski, P., Gorniak, A. & Piekarski, M. K. (2009) The effect of hydrological drought on chemical quality of water and dissolved organic carbon concentrations in lowland rivers. *Polish Journal of Ecology,* **57,** 217-227.

Annex 1: Adaptive measures relevant to rivers and listed in REFRESH deliverables 1.1 and 1.2, organised by type of measure (Nõges et al., 2010b)

Type	Number	Desciption
Riparian zones	**M065**	Planting of natural riparian tree vegetation along water courses.
	M150	Riparian stand establishment and rehabilitation
	M298	Create buffers around water bodies
	M351	improve bank and shore vegetation
	M370	Creat buffer strips, which reduce nutrient loads of lakes and rivers and also offer shading of water;
	M380	Creating riparian shading at small and moderate-size watercourses to control excessive algal growth during summer periods
	M381	Planting large shade trees on the dikes
	M382	Restoration of streambank vegetation for temperature control
	M383	Retaining forested buffers along stream banks in logging areas
	M387	management of the riparian zone to prevent overshading
	M388	Fencing and the protection of riparian vegetation
	M395	Enhance habitat for warmwater fish by (1) maintaining buffer strips; (2) removing some trees to allow more sunlight to reach the water surface; (3) allowing a few mature trees to die in place; and (4) prohibiting livestock from entering the buffer strip.
	M396	Treatments for enhancing cold-water fish habitat by (1) maintaining buffer strips with at least 70% relative density; (2) allowing a few mature trees to die in place; and (3) creating small penings less than 0.1 ha in size.
	M397	Measures of intensive protection for riparian areas by (1) maintaining buffer strips with at least 70% relative density; (2) allowing a few mature trees to die in place; and (3) creating small openings less than 0.1 ha in size.
	M408	Manipulating fluvial erosion by incorporating structures such as live trees, dead trunks and brushwood into the riparian zone.
	M453	Place bundles of live willow withies behind the trunks at stream shores where they will develop roots and shoots, binding bank material together and trapping silts which act as a growth medium for colonising emergent macrophytes.
Channel morphology	**M036**	Establish natural course of rivers
	M046	Initiating self dynamic processes of river morphology development in densely populated regions.
	M056	Improvement of river structures
	M148	Revitalization of watercourses and inappropriate drainage, improvement of watercourse passability
	M172	Implementation of precautionary flood protection measures on minor watercourses and in their basins and erosion protection measures on forest land, repairs of bank scours, erosion scours and damming, stabilization of ravines on land designed to play the role of a forest.
	M229	Maintenance of river beds to ensure the passage of the flood peak.
	M352	Re-engineer more natural bed and bank features (eg recreate meanders in straightened rivers)
	M353	Modify, reduce or cease maintenance works (eg dredging)
	M354	Move embankments further away from banks and shores; reduce pressures from hard engineering structures on beds, banks and shores (eg improve design, use softer engineering techniques, remove)

	M365	Stregthen and raise sea and river dikes
	M369	Improve the connectivity of river systems;
	M385	In-stream salmonid habitat restoration techniques with turbulence generated when water flows over weirs into plunge pools, or merely surface agitation from flow constriction
	M386	trash dam removal to facilitate either fish passage and/or land drainage
	M389	Channel modification using riparian structures
	M407	Designing a two-stage channel to reinstate favourable velocity and depth conditions below a dam resevoir. The low flow channel will carry compensation flows and normal regulation releases, and a higher level channel over shallow side berms will carry high flows.
	M452	Stream channels in riparian restoration projects should be designed narrow enough to overflow into the floodplain on a 1.5 to 2 year timescale
flow	**M062**	Develop environmental flow in regulated rivers
	M181	Prospecting for, survey and assessment of the possibilities of controlled groundwater recharge by surface water (artificial groundwater recharge) from watercourses or reservoirs
	M201	additional measures to first retain water for later slow release
	M250	Prohibition on issuing permits for water for bodies of groundwater-dependent Intermittent rivers
	M278	Establishment of rules for the minimum residual flows at hydropower plants
	M313	Measures to maintain compensation flows at reduced available water resources in summer to contribute to fish migration within systems particularly around or across barriers such as weirs.
	M314	Water capturing during peak flows of extreme rainfall events, and off-channel storage to reduce flood hazard and increase storage infrastructure.
	M346	Provide improved river flows by integrated operation of scheme; changing pattern of abstraction
	M347	Provide improved river flows by reducing net abstraction
	M390	Increasing river flow to control phytoplankton standing stock in rivers
	M402	Flexible abstraction rules with a higher prescribed flow (PF) at times and in locations where fish are migrating, and a relaxation at other times of year when no migration
	M403	Abstraction operating rules allowing only a proportion,typically 50%, of the flow above the prescribed flow (PF) to be taken
	M404	Maintain compensation flows at dam reservoirs
	M405	Releases from dam reservoirs to stimulate upstream migration of adult salmon and sea trout or downstream migration of smolts, to provide flushing flows to clean gravels prior to the spawning season
	M406	Adjust release patterns from hydroelectric power plants to simulate a natural spate hydrograph for the river
floodplain connectivity	**M026**	More space for rivers (enlarging floodplains)
	M151	Grassing of arable land, in particular along watercourses
	M363	provide flood areas for excess riverwater from the Maas and Rijn in the Krammer-Volkerak, the Zoommeer, the Grevelingen and possibly the Oosterschelde
	M375	Using natural wetlands for tertiary treatment
	M384	grass filters and creation of mini-paddy fields in combination with forest buffer

water quality	**M134**	Adaptation measures at waste disposal and storage sites
	M135	Adaptation measures at sewage collection and treatment sites
	M155	Technical and biological measures to reduce euthropication of surface water
	M160	Reduction of surface water and groundwater pollution from agricultural sources
	M299	Connection of unsewered wastewater discharges to municipal system in selected areas where assimilative capacity is available during low flow
	M303	Make provision for pre-treatment requirements for industrial wastewater entering the collection systems and treatment plants considering the potentially reduced assimilative capacity in rivers in summer
	M311	Addressed the decrease in assimilative capacity of water bodies receiving pollutant loads from point and non-point sources.