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Geography & Environment

Investigating the effects of large wood and forest management on flood risk and flood hydrology

by

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Abstract

The changes to catchment scale flood risk following river restoration works, including the addition of large wood logjams to the channel, are poorly quantified in the literature. Key concerns following river restoration for river managers and other stakeholders are changes to flood hydrology at the reach and catchment scale and changes in the mobility of large wood pieces. The effects of accumulations of large wood (logjams) on local flood hydrology have been documented in the literature, showing logjams slow flood wave travel time and increase the duration and extent of local overbank inundation. Modelling studies conducted at a reach scale have shown that these local effects can slow flood wave travel time through a reach and delay the timing of flood peak discharge at the reach outflow. How these local and reach scale effects translate to the catchment scale remains to be illustrated in the literature.

In this thesis a combination of field and modelling studies are used to; elucidate the link between logjam form and function, to quantify the mobility of pieces of large wood relative to their physical characteristics, to predict the changes in floodplain forest restoration over time and to provide predictions of changes to catchment scale flood hydrology following river restoration at a range of scales and locations.

It is shown that logjams inducing a step in the water profile are most effective at creating diverse geomorphology and habitats. Logjams were found to account for 65% of flow resistance in forested river channels, rising to 75-98% of flow resistance where the logjam was inducing a step in the water profile.

Large wood in small forested river channels was found to be highly mobile with 75% of pieces moving, with the longest transport length of 5.6km. Large wood mobility is governed primarily by the length of a piece of wood with wood in excess of 1.5x channel width a threshold for a lower probability of movement.

Hydrological modelling using OVERFLOW shows that reach scale river restoration can lead to modest changes in catchment scale flood hydrology. It is concluded that flood risk management can incorporate river restoration, but that results are likely to be unpredictable if engineered logjams are used alone. Substantial benefits in reducing catchment outflow peak discharge (up to 5% reduction) are modelled for floodplain forest restoration at the sub-catchment scale (10-15% of catchment area), rising to up to 10% reductions as the forest matures and becomes more complex.

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- CD containing modelling codes for running OVERFLOW model using MATLAB.

Academic Thesis: Declaration Of Authorship

I, Simon James Dixon declare that this thesis and the work presented in it are my own and has been generated by me as the result of my own original research.

Investigating the effects of large wood and forest management on flood risk and flood hydrology

………
………

I confirm that:

1. This work was done wholly or mainly while in candidature for a research degree at this University;
2. Where any part of this thesis has previously been submitted for a degree or any other qualification at this University or any other institution, this has been clearly stated;
3. Where I have consulted the published work of others, this is always clearly attributed;
4. Where I have quoted from the work of others, the source is always given. With the exception of such quotations, this thesis is entirely my own work;
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Acknowledgements

I want to thank my wife Gemma without whom this thesis would not have been possible, she braved New Forest winters, became one of only about four people in the world who know how to use an ancient Geodimeter total station and read every single word.

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Finally I'd like to dedicate the thesis to my daughter Emmeline who arrived half-way through the PhD making me a lot more efficient and reminding me there are more important things in life.

The angry waters rose and seethed around Achilles; they beat down on his shield and overwhelmed him. Unable to maintain his stance, he laid hold of a full-grown elm. But the tree came out by the roots, brought the whole bank away, and crashed into the middle of the river where it bridged from side to side, clogging the stream with a tangle of branches.

Homer, The Iliad, Book XXI

1. Introduction

1.1. Framing the research question

In-stream wood is a natural part of a forested river ecosystem. Historically, in many rivers, the volumes of in-stream wood were higher than that currently observed (Collins et al., 2002). However, wood has been progressively removed to aid navigation, for recreation, to aid fish passage and to mitigate local flood risk (Brooks et al., 2004). In recent years, in some environments, wood is being placed back into rivers as part of river restoration programmes (Brooks et al., 2004; Krejčí and Máčka, 2012). The relationship between in-stream wood, floodplain forests and flood hydrology is a complex one involving feedback loops. Forested floodplains act as a source for large pieces of dead wood to a river (Bendix and Cowell, 2010; Cline et al., 1980), and these pieces can become organised into logjams (Braudrick et al., 1997; Gurnell et al., 2002). During flood events these logjams can redirect flow towards the banks and also force water onto the floodplain earlier in the event and for a longer duration (Jeffries et al., 2003; Sear et al., 2010). Flow directed towards the banks can cause erosion and the toppling of riparian trees into the channel, overbank flows can float deadwood from the floodplain into the river channel (Gurnell et al., 2002; Millington and Sear, 2007; Pettit and Naiman, 2005), thus further increasing the volumes of wood in the river.

There remain many unknowns in the interactions between floodplain forests, in-stream wood and flood hydrology. Information on the effects of land cover changes (e.g. forest planting) on flood hydrology is sparse in catchments larger than 10km^2 (Archer et al., 2010) and the effects of in-stream wood and riparian forests on flood hydrology have typically only been studied along short river reaches (e.g. Sholtes and Doyle, 2011; Thomas and Nisbet, 2012). Therefore there is a need to study the effects of changing land cover and large wood loads at a catchment scale. Although the transport mechanisms of large wood in a river have been documented (e.g. Bocchiola et al., 2006a; Braudrick et al., 1997) there is a need for studies reporting variables such as the potential transport length of mobile wood and the stability of wood relative to the size of the river channel over multiple years, which remain rare in the literature (Bertoldi et al., 2013; MacVicar and Piégay, 2012; Wohl et al., 2010). Addressing these gaps in the literature will also help river management bodies such as the Environment Agency in the UK understand how land cover and in-stream wood affects flood risk and flood hydrology. In addition information on wood mobility will allow assessment of the risk of in-stream wood moving during flood events and causing potential infrastructure damage. Information on flood risk factors will allow management bodies to make informed choices about the potential for using floodplain forest and engineered logjams as part of flood risk mitigation programmes.

1.2. Importance of flooding

The gaps identified in the scientific literature in section 1.1 set out the scientific justification for the research conducted within this thesis. However, it is also important to consider the societal and philosophical imperatives for conducting a given piece of research. Ideally research should be not only scientifically novel, but also socially useful.

1.2.1. Economic importance of flooding

Flooding is the most damaging natural disaster globally, both in terms of loss of life (accounting for half of all natural disaster related deaths) and in terms of economic losses resulting from damage and disruption (Berz, 2000). The cumulative value of losses due to all flood events in Europe in the later part of the 20th century was estimated at $2bn per annum (Robinson et al., 2003). Several recent large magnitude flood events globally have highlighted the catastrophic economic cost of such disasters; 2007 floods in England estimated to have cost £3bn and caused 13 deaths (Marsh, 2008), 2012 flooding in Thailand costs an estimated £29bn (Ziegler, 2012) and 2013 flooding on the Danube initially estimated to have caused “several billion US dollars” of damage (Blöschl et al., 2013). The frequency and magnitude of large floods is expected to increase with climate change (Kleinen and Petschel-Held, 2007; Wilby et al., 2008) and flooding is expected to be the most serious political impact of climate change (McCarthy et al., 2001). There is a need to critically examine methods of flood mitigation to limit exposure to loss and damage resulting from floods as part of adapting to climate change.

1.2.2. The philosophical imperative of flood research

Mankind has long recognised the risks to life and livelihood posed by flood events and has sought to understand flooding in order to attempt to mitigate its impact. Floods have been a central element in human consciousness across many regions for well over four thousand years; some of the very earliest writing in human history from ancient Sumerian literature (Lambert et al., 1999) has a central narrative of a flood myth in the epics of Gilgamesh, Ziusudra and Atrahasis (Dalley, 1989). Similar flood myths are found in religious texts from the Abrahamic religions (George, 2003) and in the Hindu scripture Shatapatha Brahmana (Klostermaier, 2007). In these texts, floods are seen as devastating events capable of wiping out entire communities and destroying infrastructure, and although the link between heavy and prolonged rainfall is understood, floods are seen as unavoidable ‘acts of God’. Flooding appears numerous times in Homer’s Iliad and Odyssey (Homer, 1961; Homer, 2008); floods raised by wrathful gods seeking to inflict death or damage on humans. The attribution of divine control to floods in the ancient world reflects the complex processes of flood generation which no doubt confounded ancient scholars; indeed the link between rainfall and river discharge, and with it the birth of the modern science of hydrology was not first made until the later part of the 17th Century (Freeze, 1974). The impacts of land use change on flood behaviour were noted in Pliny the Elder’s Natural History in the first century AD (Andréassian, 2004) where he observed disastrous torrents on newly clear felled areas of mountain forests (XXXI, 30). Despite subsequent advances, there remains considerable uncertainty around the process dynamics and organising principles that underlie rainfall runoff processes (McDonnell et al., 2007) and with them flood generation. Many of the mechanisms of flood behaviour remain to be elucidated.

1.3. Adapting to flooding and mitigating flood risk

Historically the preferred method of responding to flood risk was through the construction of engineered flood defences, such as levees and embankments (Howe and White, 2001). Climate change is likely to increase the frequency and magnitude of flood events (Kleinen and Petschel-Held, 2007; Wilby et al., 2008); this coupled with increasing encroachment of housing development onto the floodplain (Howe and White, 2001; Werritty, 2006) puts an increasing number of homes at risk of flooding. Currently in the UK an estimated 5.2 million homes are believed to be at risk of flooding (EA, 2009a). In light of increasing exposure to flood risk it is recognised that increasing the height and extent of engineered flood defences is unsustainable (Broadmeadow and Nisbet, 2009; Johnson et al., 2007; Johnson and Priest, 2008), and that alternative approaches are needed (Johnson and Priest, 2008; Werritty, 2006).

1.3.1. Flood mitigation through land use

Alternative approaches have received attention in recent UK policy and legislative documents (e.g. Defra, 2005a; Defra, 2005b; Defra, 2007; Pitt Review, 2008; EA, 2009a; EA, 2009b), which include altering land cover to attenuate runoff from hillslopes and using rural land on the floodplain as a temporary water store to keep flood waters away from vulnerable urban locations (Pitt Review, 2008). River restoration programmes have potential for integration into flood risk management (Acreman et al., 2003); however the impacts of river restoration on flood hydrology remain poorly understood (Wharton and Gilvear, 2006). Studies are needed at the catchment scale in order to demonstrate the potential of river restoration to reduce flood risk (Parrott et al., 2009; Robert, 2011).

1.3.2. Role of wood in rivers

Wood within a river channel induces changes in local hydraulics, which acts to increase flood wave travel time. Changes to local hydraulics also lead to local variations in shear stress, which in turn leads to local patterns of sediment erosion and deposition. The influence of in-stream wood on channel processes is dependent on the setting, but in general wood leads to reduced sediment transport, pool formation (Curran and Wohl, 2003; Gurnell et al., 2002; Jeffries et al., 2003; Montgomery et al., 1995; Montgomery et al., 2003) and the formation of sedimentary structures on the floodplain (Sear et al., 2010) and in the channel (Abbe and Montgomery, 1996; Montgomery et al., 2003), such as bars and riffles (Gurnell et al., 2002). Wood plays a number of important ecological roles within a stream ecosystem; geomorphological features provide important habitats for a variety of biota (Benke and Wallace, 2003; Dolloff and Warren, 2003; Millington and Sear, 2007; Montgomery et al., 2003), whilst logjam structures provide shelter for fish during high flow and pollution events (Wheaton et al., 2004), logjams act to trap and retain course particulate matter which is an important allochthonous source of carbon for macroinvertebrates (Gurnell et al., 2002; Piégay and Gurnell, 1997).

Despite numerous studies linking the presence of wood in rivers with general geomorphological diversity and improved biotic conditions, a link between logjam characteristics, such as size, and geomorphological features remains to be illustrated in the literature. Investigating such a relationship is important from a river management perspective with a view to developing river restoration guidelines on specific logjam types which are likely to produce desired geomorphological features.

1.4. Research Objectives

The overall aim of this thesis is to understand how river restoration programmes can affect flood hydrology and impact on flood risk. Within this overarching aim there are a number of specific objectives;

- Understand the changes in flood hydrology at a catchment scale following river restoration programmes using both engineered logjams and restoration of forests. This will be addressed using a numerical modelling approach and is detailed in Chapters 9 and 10.
- Conduct a logjam survey to determine which independent catchment and reach variables control the distribution of logjams in a forest river, in order to predict

how logjam abundance may change if these variables are altered in the future. This study is reported in Chapter 5.

- Using data from a forest river analyse which features of logjams lead to a greater likelihood of the structure creating geomorphological features and/or enhancing habitat diversity. Any relationships found can be used by river managers seeking to use logjams in river restoration to restore specific geomorphological features or habitat for specific species. This objective is addressed using data from the logjam survey reported in Chapter 5.
- To establish transport lengths of mobile wood in a forest river, factors influencing wood stability and relationships between discharge sequences and log mobility by tracking the position of individual pieces of wood in a forest river over multiple years. This objective is addressed through a field study reported in Chapter 6, and will enhance understanding of the risk of wood in rivers moving during flood events and causing damage to infrastructure.
- Quantify the hydraulic resistance values of logjams within a forest river during high flow events. This will assist in understanding how logjams influence flood hydrology and will help to parameterise hydrological models of logjams. This objective is addressed in Chapter 7.
- Develop a conceptual model of riparian forest succession and associated input of wood to the river channel. This will inform the development of modelling scenarios following forest restoration and is reported in Chapter 8.

2. Thesis Strategy

The thesis is presented as a series of analysis chapters which individually can stand alone, but also which form increments of the thesis and contribute towards answering the central question; which is how river restoration and associated floodplain forest management affects flood hydrology and flood risk. The effects of river restoration on flood risk and flood hydrology are ultimately directly addressed in a final hydrological modelling study. However each individual study provides valuable information towards the parameterisation of this modelling approach. The thesis begins with a review of literature related to flooding and large wood, and each analysis chapter also includes a short review of literature relevant to the individual study.

A review of literature on flooding is presented in Chapter 3. A history of recent large magnitude flood events both globally and in the UK is laid out. The mechanisms of flood generation related to precipitation, runoff and hydraulic connectivity are explained, along with the spatial and temporal effects on runoff during prolonged rainfall events. The potential changes to flood hydrology through climate change and changing patterns of land use are explored. River and forest restoration are identified as having potentially significant impacts on flood risk and the effects of different types of restoration are examined. Changing patterns of in-stream large wood are identified as an important hydraulic resistance component which can be altered as part of river restoration. Gaps in the literature are identified relating to the impacts of river restoration and land cover on flood hydrology which will be investigated in this thesis.

Drivers for changing in-stream large wood loads have been identified in Chapter 3 along with the importance of large wood for flood hydrology. In chapter 4 the wider effects of large wood on the fluvial environment are explored in a review of the large wood and logjam literature. The importance of large wood on ecology, hydrology and geomorphology are explored. The mechanisms by which large wood becomes organised into logjams are detailed. Gaps in the literature on wood mobility and logjam effects on geomorphology are identified.

The first of the analysis chapters (Chapter 5) is "Controls on the distribution and geomorphological performance of large wood accumulations". This chapter summarises the effects of key structural pieces on forming logjams, as well as the importance of logjam characteristics on the formation of geomorphological and habitat diversity.

The second analysis chapter (Chapter 6) "The influence of geomorphology on large wood dynamics in a low gradient headwater stream" the mobility of in-stream wood is examined through an experimental study. This study is important in the context of flood risk to quantify the mobility of large pieces of wood which can cause damage if mobilised during floods.

In the third experimental analysis chapter (Chapter 7) "Hydraulic resistance properties of logjams during flood flows" the flow resistance properties of a range of logjams are quantified and used to parameterise later hydrological modelling.

The fourth analysis chapter is a modelling study; "Developing a conceptual model of riparian forest succession following restoration". In this chapter (Chapter 8) a riparian forest growth model, along with literature values, are used to develop a conceptual model describing how a restored riparian forest changes over time. This model is then used to design hydrological modelling scenarios in later chapters.

The fifth and sixth analysis chapters describe a hydrological modelling approach using the model OVERFLOW (Odoni and Lane, 2010). The first (Chapter 9) comprises a description of the model used and its calibration and parameterisation using data gathering during the experiments described in Chapters 5 and 7. The second of these chapters (Chapter 10) analyses the output of hydrological modelling for scenarios of engineered logjam insertion as part of river restoration, as well as for the restoration of floodplain forests over 100 years. Scenarios are designed using data collected in Chapter 5 and the conceptual model from Chapter 8. Values for hydraulic resistance are parameterised with field data from Chapter 7 and values from the literature.

The thesis concludes with a summary of findings, the limitations of the work and suggestions for future avenues of research.

3. Flood hydrology and river restoration

3.1. Flood risk

Flooding is the most common natural catastrophe, accounting for around one third of natural disasters and responsible for a half of all natural catastrophe related deaths (Berz, 2000) and it impacts upon human civilisation at scales from local (Golding et al., 2005) to continental (Blöschl et al., 2013). Floods cause; loss of life, economic loss, damage to infrastructure, illness from polluted water and vector borne diseases, soil and land erosion, loss of agricultural crops, temporary loss of aquatic habitats and reductions in aquatic species populations (Burrel et al., 2007). The annual cost of flooding related damage and economic losses in Europe between 1995-1999 was estimated at €2bn (£1.7bn) (Robinson et al., 2003), with recent large magnitude events estimated to cause several billion US dollars in losses individually, such as the summer 2013 floods of the Danube (Blöschl et al., 2013), and 2012 monsoon flooding in Thailand estimated to have cost a total of $45bn (£29bn) in damage and losses (Ziegler, 2012). Despite such devastating effects upon human lives and infrastructure, floods also provide many benefits such as nutrient and organic matter transfer between the aquatic and terrestrial environments (Junk et al., 1989) which enriches floodplain soils. Indeed the annual floods in the Nile Delta have been the foundation of agriculture in Egypt for over five thousand years (Bell, 1970). Floods are not in and of themselves damaging, however problems arise where human infrastructures are exposed to flood risk.

3.2. Flooding in the United Kingdom

In recent years the UK has experienced a number of large magnitude flood events causing loss of life and substantial damage to homes and infrastructure. In 2000 the UK Met Office reported the wettest twelve month period since records began in 1766, leading to widespread flooding affecting 10,000 homes in 700 separate locations (Met Office, 2012). In 2004 a large flood event caused extensive, but localised damage to the coastal village of Boscastle (Golding et al., 2005). In Summer 2007 prolonged heavy rain lead to localised flash flooding in which 55,000 homes were flooded at an estimated insurance cost of £3 billion with 13 deaths making the event the costliest flood in the world during 2007 in economic loss terms (Marsh, 2008; Pitt Review, 2008). In 2009 a rainfall rate of 316mm in 24 hours in Cumbria (Miller et al., 2012) lead to severe flooding in the towns of Cockermouth and Workington with an estimated peak discharge on the River Derwent of $700m^3s^{-1}$ (Miller et al., 2013), leading to 1500 homes flooded, the destruction of two bridges over the River Derwent and one death (Sibley, 2010).

In total the Environment Agency estimates some 5.2 million homes in England are at risk of flooding (EA, 2009a). The extent and cost of recent flooding, notably the 2007 events, prompted a governmental review into flood defence policy, the Pitt Review, which was published in 2008.

3.3. Flood generation

In order to understand factors which may mitigate or exacerbate flood risk it is necessary to understand the mechanisms of streamflow generation. Figure 3.1 shows a simplified diagram of the processes involved in streamflow generation where input of water into the system is from precipitation and outputs are through evapotranspiration and stream flow. Within the system there are numerous pathways and stores for water; vegetation within the system can both intercept a portion of precipitation as well as draw up water from the unsaturated soil moisture zone, this water can remain stored in the vegetation, or released from the system as evapotranspiration. Water reaching the ground surface will either infiltrate into the soil or move as overland runoff depending on factors such as soil saturation and precipitation rate. Where water moves as overland runoff it will eventually flow into the channel network as channel flow. Water infiltrating into the soil may either move as sub-surface runoff, eventually reaching the channel network as exfiltration or seepage, or will move through the unsaturated soil moisture zone and recharge groundwater in the saturated groundwater zone where it can remain as storage. Water also moves out of the saturated groundwater zone as groundwater discharge where it can move into the channel flow as exfiltration.

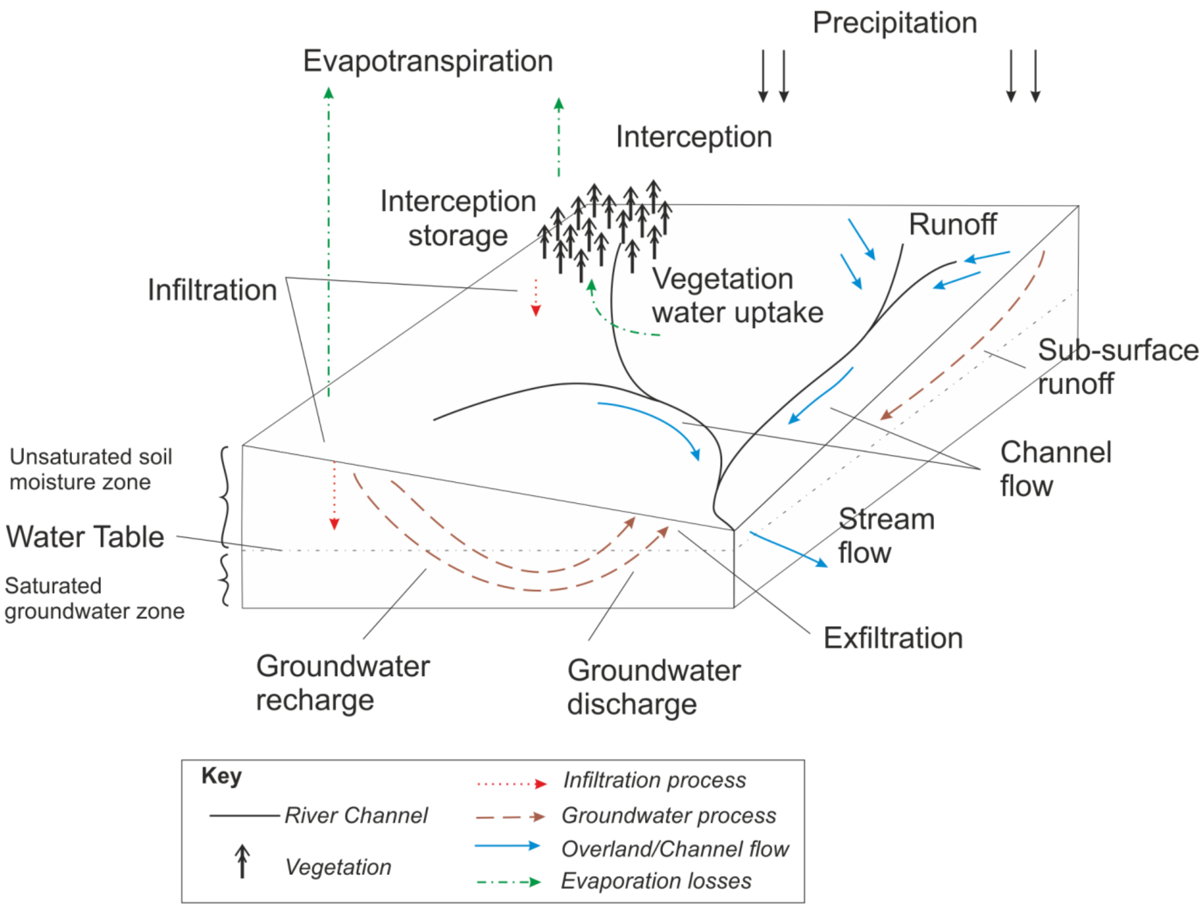


Figure 3.1– a conceptual model of rainfall runoff processes. (after Freeze, 1974)

A key variable in flood hydrology is the speed at which water from precipitation reaches the channel network. The speed of transfer for precipitation to river channel is a function of connectivity, which will depend on the level of vegetation interception and the capacity of the soil for infiltration; although these factors will be governed to a certain extent by catchment characteristics they are neither spatially nor temporally constant within a river catchment (Bergkamp et al., 1996; Bracken and Croke, 2007).

Hydrological connectivity is a measure of the passage of water from one part of the catchment to another and during rainfall events will typically include some proportion of runoff response (Bracken and Croke, 2007), exactly what proportion of precipitation that moves as runoff depends on a number of factors which exhibit complex spatial and temporal patterns even at small scales (Bracken and Croke, 2007; Cammeraat and Imeson, 1999; Fitzjohn et al., 1998; Ludwig et al., 1999a; Ludwig et al., 1999b; Ludwig et al., 2000; Morgan, 1988).

Flood generation is a mass transfer problem where channel flow is a function of the rainfall rate and catchment area, which govern the input rate of water to the system, and the capacity for the catchment to absorb this water into temporary storage and the transfer rates of water in excess of storage capacity. High channel discharges are produced by a combination of high storm intensity and optimum basin morphometry (Costa, 1987). Equation 3.1 shows a simplified relationship between rainfall and water available for transfer from where it falls as precipitation to another part of the catchment.

$$\Delta V = \int_0^t PA - S - F \tag{3.1}$$

Where ΔV is change in available water for transfer via flowpaths, P is precipitation rate (depth per time unit), A is catchment area, S is temporary storage (e.g. within vegetation and topographic depressions) and F is losses (e.g. evapotranspiration). A key variable in flood generation is the relative speed of transfer of water from hillslopes to the channel via different flow paths. Runoff is generated when water cannot infiltrate into the soil and thus moves over the surface, the relative importance of runoff is therefore related the soil's capacity for infiltration. Infiltration can depend on crusting and surface roughness of soil (Auzet et al., 1993; Helming et al., 1998; Singer and Le Bissonnais, 1998), the soil surface can also be sealed by the impacts of rainsplash dislodging particles and filling soil pores (Bradford et al., 1987a; Bradford et al., 1987b; De Ploey, 1984; Govers, 1991). In humid climatic zones runoff is largely via saturated overland flow and in arid/semi-arid zones where the precipitation rate exceeds that infiltration rate water will transfer downslope via Hortonian overland flow; both these types of runoff are governed by slope and microtopography and can be very rapid (Bracken and Croke, 2007). In many environments sub-surface stormflow is considered to the most important mechanism of rapid water transfer to channels during rainfall events (McDonnell, 2003; Noguchi et al., 1999; Tromp-van Meerveld and McDonnell, 2006; Weiler et al., 2006). Infiltration and subsequent transfer of water via sub-surface flow is enhanced by the presence of soil macropores (McDonnell, 2003; Noguchi et al., 1999) where living and dead vegetation root systems contribute preferential flow pathways (Noguchi et al., 1999). Where sub-surface stormflow is the dominant lateral transfer process for water the routing process can be via bedrock topography, rather than soil topography with transient water tables developing at the soil-bedrock interfaces (Freer et al., 2002; McDonnell, 2003). The transfer of water in hillslopes via sub-surface stormflow is highly threshold dependent (McDonnell, 2003; Tromp-van Meerveld and McDonnell, 2006) and once activated by sufficient precipitation transfer will be by rapid lateral flow via soil pipes within the transient saturated zone, or via discontinuities at the soil-bedrock interface (Uchida et al., 2001).

3.4. Spatial effects on flood generation

Land use, particularly vegetation type and density, impacts runoff at all scales (Bergkamp et al., 1996; Bracken and Croke, 2007; Bull et al., 2000; Imeson and Verstraten, 1988; Lasanta et al., 2000). Vegetation has a large impact on the amount of precipitation which is intercepted by the leaf canopy (Robinson et al., 2003), but also on soil characteristics within the catchment, generally having positive effect on infiltration rates by increasing organic matter and bulk density (Boix-Fayos et al., 1998; Nicolau et al., 1996). Different types of vegetation have varying water uptake from soil (Robinson et al., 2003), species which have high water usage such as conifers (Robinson et al., 2003) may result in lower moisture levels in the unsaturated soil zone and thus lead to the soil having a higher capacity for infiltration during a storm (Bracken and Croke, 2007). Furthermore the type and density of vegetation impacts on surface runoff rates (Bergkamp et al., 1996; Imeson and Verstraten, 1988) through the complexity of the ground surface slowing runoff rates in the case of mature old growth forests (Broadmeadow and Nisbet, 2009), or speeding runoff through drainage systems as in the case of commercial tree plantations (Robinson, 1986). Runoff generated on relatively impermeable bare slopes can infiltrate when it runs through vegetated areas (Boer and Puigdefábregas, 2005; Dunkerley, 1999; Puigdefábregas, 2005; Sanchez and Puigdefabregas, 1994; Valentin et al., 1999), illustrating the complex spatial patterns that exist in natural catchments, although much of the work in this area has been done in arid and semi-arid areas.

3.5. Temporal effects on flood hydrology

Antecedent soil moisture conditions are important in determining the capacity of the soil for infiltration (Bracken and Croke, 2007) and will depend on both inter-storm and intra-storm factors. A dry catchment which has not experienced recent rainfall is likely to have higher infiltration rates than the same catchment under humid conditions (Bracken and Croke, 2007), similarly a rainfall event which starts as drizzle can wet a catchment and reduce the capacity of the soil to absorb moisture and make the catchment susceptible to a later high intensity pulse later in the storm (Bracken and Croke, 2007). Generally during a storm infiltration capacity will be reduced (Bracken and Croke, 2007), as soil moisture increases connectivity between runoff sources increases and the channel system expands headwards (Figure 3.2) (Freeze, 1974) with surface topography controlling the time evolution of saturated areas (Weill et al., 2013); however the response is dynamic and will vary depending on catchment and antecedent conditions and rainfall duration and intensity (Bracken and Croke, 2007). The importance of the variable source area process will vary between catchments and in many situations the expanding wedge of saturated soil may only hold in and around the riparian zone (McDonnell, 2003), where the presence of a capillary fringe, a quickly saturated zone with low storage, can also result in groundwater discharge to the channel as well as runoff depending on antecedent conditions (Abdul and Gillham, 1989). On hillslopes water transfer will be controlled by lateral matrix and pipeflow behaviour (McDonnell, 2003) and where slopes are steep with conductive soils subsurface stormflow may be the main process of storm runoff (Weiler et al., 2006). Increased connectivity lowers the proportion of runoff which subsequently infiltrates into soil and increases the travel length of runoff pathways (Fitzjohn et al., 1998) leading to run-on which is runoff flowing from source to channel without infiltrating (Bracken and Croke, 2007).

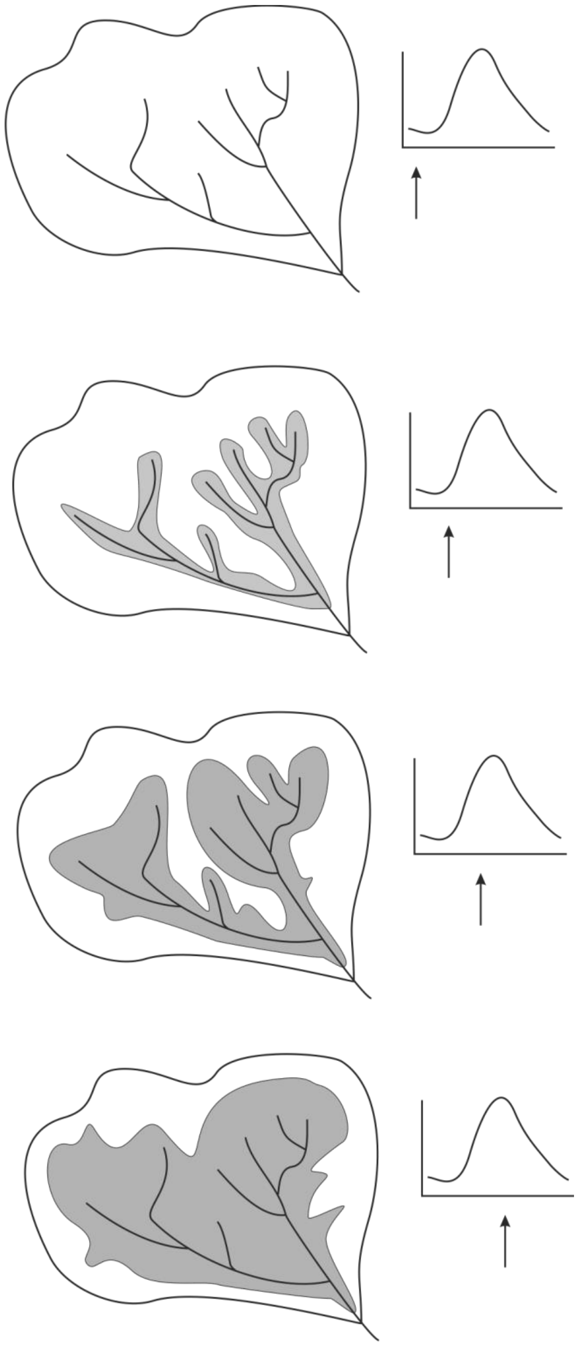


Figure 3.2 – The expansion of source area and the channel network during a storm event under the variable source area concept. The black lines are the channel network and the grey area is that which is providing direct runoff to the channel network. During a storm event the areas contribute direct runoff to the channel expand headwards, increasing connectivity and extending the channel network. (after Freeze 1974, his figure 6)

Despite catchment characteristics exhibiting important controls on flood hydrology the intensity and duration of rainfall is of prime importance for flood risk in all catchments

(Pitlick, 1994; Schick, 1987) and the primary control on downstream flood risk is peak flow magnitudes in contributing sub-catchments (Pattison et al., 2008), therefore changes in precipitation patterns under climate change are expected to have a large effect on future flood risk (Wilby et al., 2008).

3.6. Climate Change impact on flooding

Flooding is expected to become the most widespread and serious political impact of climate change (McCarthy et al., 2001). There is already an increased public perception and awareness of flood risk (Brown and Damery, 2002; Evans et al., 2006b; Posthumus et al., 2008; Thomas and Nisbet, 2012), and the detection of a climate change signal in global precipitation trends is well established (Zhang et al., 2007) although there remains a debate as to whether trends in flooding are already reflecting a changing climate or are due to climatic variations (Lane, 2003; Lane, 2008; Robson, 2002). Robson (2002) was unable to find any significant trends in flooding at a UK level, in part this may be due to the relatively short duration of many flow records (<50 years) making it hard to identify trends (Lane, 2003). Furthermore any observed changes in flood frequency at a local level are hard to attribute to changes in precipitation given the complex relationship between flood generation, land use change and changing river conveyance which alters the flood-discharge relationship for a given river (Lane, 2003). Climate change will lead to more extreme weather events (Arnell, 2003), with lower mean summer rainfall but a global increase in extreme rainfall events (Christensen and Christensen, 2003; Wilby et al., 2008). As a result of changing weather patterns there will be an increasing frequency of large magnitude floods (Wilby et al., 2008) with flood return periods becoming shorter; a current 50 year return period could be shortened to a 25 year return period over the next hundred years (Kleinen and Petschel-Held, 2007). Despite broad regional patterns, and an indication that Europe is experiencing a relatively "flood rich" period (Figure 3.3), the impacts of climate change on local weather patterns (Lambert et al., 2004) and flood risk remain uncertain (Lane, 2003; Prudhomme et al., 2003).

Estimates of future flood risk are subject to many areas of uncertainty (Lane, 2003) not least as the climate change signal is small compared to large inter-annual variability in precipitation (Lane, 2003), and there is a complex interplay between downscaled climate change scenarios and regional variations in catchment characteristics (Wilby et al., 2008).

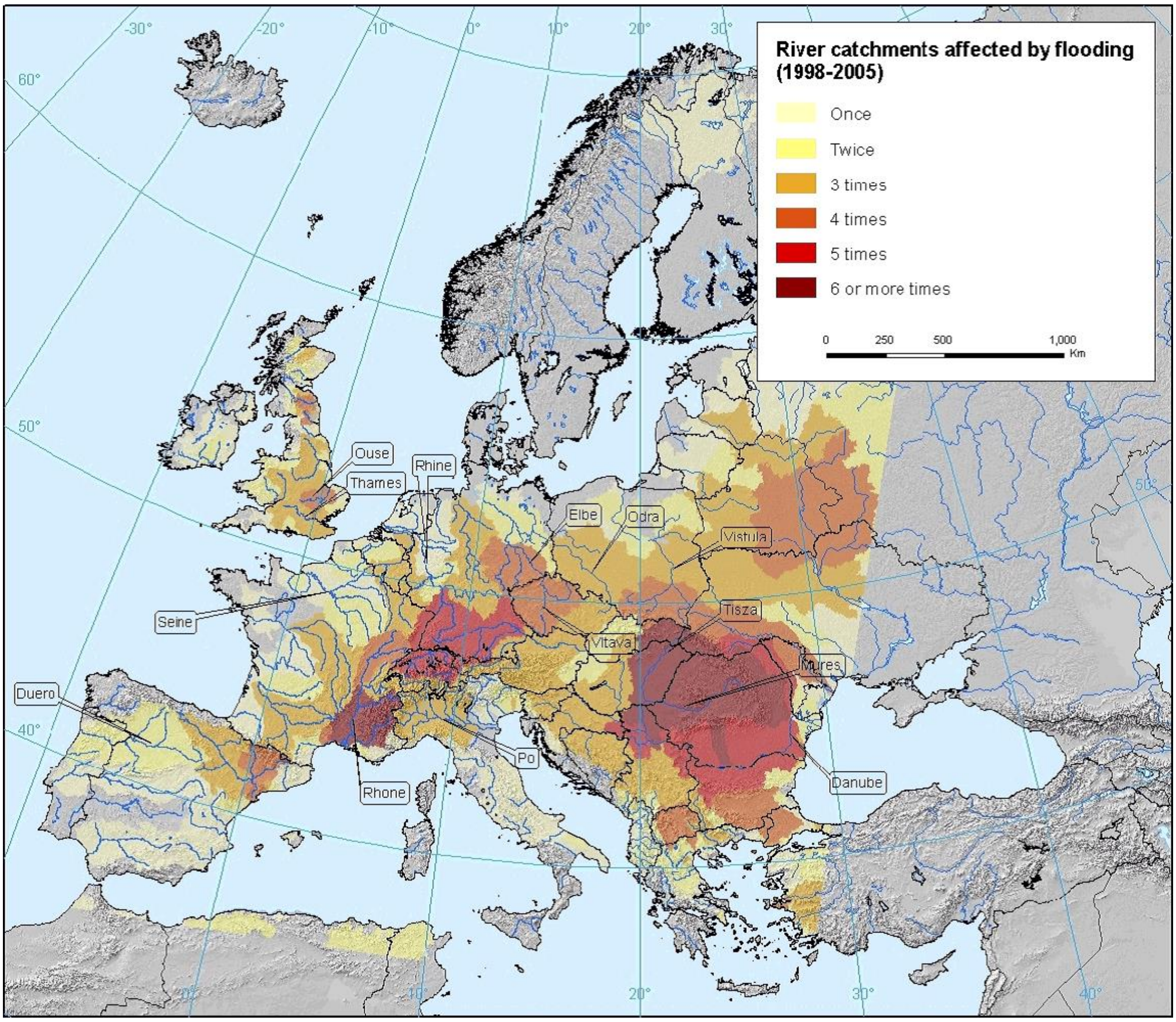


Figure 3.3– recurrence of major flood events in Europe between 1998 and 2005, demonstrating Europe is currently experiencing a relatively "flood rich" period. From Wilby et al, 2008

3.7. Flood control

Flood management encompasses actions to prevent floods, reduce the probability of floods or lessen the damage and disruption caused by unavoidable flood events (Burrel et al., 2007). The dominant paradigm in flood defence up until the later part of the last century was one of controlling flood waters through construction of levees, dams

(Howe and White, 2001); so called “hard defences” are especially important, and through maximising capacity of river channels in urban areas which are often trapezoidal & concrete lined (Werritty, 2006).

As early as 1945 White proposed an alternative approach of human adjustment to flood risk (White, 1945 in Burrell et al, 2007), however this idea did not receive widespread acceptance in the UK until the past ten years (e.g. Defra, 2005a; Pitt Report, 2008; EA, 2009a; EA, 2009b). Adjusting to flood risk can involve a combination of awareness of potential risk, adaptations to minimise the magnitude of flood events and building resilience into structures such that water is kept out or the amount that enters is minimised when flooding does occur (Pitt Review, 2008). Awareness of flood risk will be through a combination of reliable warning and forecast systems for floods, along with improved preparedness of people living in zones of flood risk which can be enhanced through public participation in flood mitigation plans (EA, 2009a; Schelfaut et al., 2011), furthermore developmental control on floodplains can play a part in minimising exposure to flood risk (EA, 2009a; Kelman, 2001; Pitt Review, 2008). Increasing the extent of permeable rather than impermeable surfaces can help to minimise the magnitude of flood events, for example limiting paving over of gardens and an increasing use of permeable pavements (Vis et al., 2003)(Pitt Review, 2008); water can also be kept away from vulnerable urban areas by allowing flooding of rural land as temporary floodplain storage (Pitt Review, 2008). Structural flood resilience is increasingly receiving attention and is expected to be incorporated into future UK building regulations (Escarameia et al., 2007), such resilience can take the form of dry proofing, which is to keep water out of a building, or wet proofing which can accelerate recovery from flooding (Lamond and Proverbs, 2009). Wet proofing and dry proofing is especially important in residential buildings due to indoor dampness and mould growth leading to an increase in respiratory and dermal diseases among people living in flood affected homes (Azuma et al., 2013).

With steady encroachment of homes and infrastructure onto floodplains raising exposure to risk from flood events (Howe and White, 2001; Werritty, 2006) and the increasing perception that climate change will lead to increased flood risk (Beven, 1993;

Wilby et al., 2008) the unsustainability of increasing the height and extent of structural defences has been recognised (Broadmeadow and Nisbet, 2009; Johnson et al., 2007; Johnson and Priest, 2008). As well as compelling arguments of unsustainability of structural defences there are also environmental concerns such as the Water Framework Directive (WFD) which requires all EU member states to achieve "good ecological status" for all watercourses. The WFD has encouraged states to look for ecological gain wherever possible when designing flood mitigation measures and to try and avoid installing structures which may damage the environment unless there is no other option (Werritty, 2006). Several authors make the argument that although structural engineering solutions to flooding can reduce flood risk they tend to put responsibility onto the state and make individuals exposed to flood risk complacent (e.g. Kelman, 2001; Lamond and Proverbs, 2009; Ziegler, 2012). As a result of the above mentioned drivers there is now a paradigm shift away from a sole reliance on structural defences to a balance between structural and non-structural mitigation (Johnson and Priest, 2008; Werritty, 2006).

The UK Department of Food and Rural Affairs (DEFRA) published a landmark document in 2005 "Making Space for Water" setting out the government's long term strategy for flood defence, this states; "we need to consider how we adapt to climate change, incorporating allowances into our consideration of flooding and erosion risk, ensuring our measures are reversible and adaptable, and that we review our approach on a regular basis using the foundation of best available science" (Defra, 2005a). Making Space for Water was followed by the Floodwater Management Act of 2010 which takes a wider view of flood risk management than just addressing 'at-a-point' defences (Ball, 2008). As a result of these legislative and cultural changes flood mitigation is now assessed in terms of feasibility, sustainability and environmental impact at the catchment as well as the local scale (Burrel et al., 2007). With an increasing environmentally sustainable focus to flood mitigation, attention has begun to turn towards the potential of using river restoration as part of flood control (Acreman et al., 2003) although the potential of river restoration to reduce downstream flood risk remains to be demonstrated (Wharton and Gilvear, 2006). Two areas in which river restoration may impact the balance of water transfer in the conceptual model shown in Figure 3.1 are in increased loadings of large wood to river channels which slow channel flow (Thomas and Nisbet, 2012) and in changing land use with forests specifically identified as reducing runoff and reducing flood risk (EA, 2009b).

It is important to understand the potential effects of river restoration upon flood hydrology and flood risk; river restoration can represent a current and future change to land cover which may impact upon flood hydrology but which currently largely operates independently of flood risk programmes and is poorly understood (Wharton and Gilvear, 2006), river restoration is also a potential tool as part of future flood mitigation schemes, but studies are needed to demonstrate this potential at reach and catchment scales (Naiman and Décamps, 1997; Parrott et al., 2009; Robert, 2011).

3.8. River restoration

The context for restoration of rivers is a background of progressive degradation throughout human history. Human modifications to European river systems are not a recent phenomena; local deforestation occurred as early as the Neolithic (6500BP), with increasing riparian forest clearance for farming and pasture in the Iron Age (2000BP) (Higler, 1993; Montgomery et al., 2003). The Roman Empire conducted the first projects that could be considered river engineering (Oleson et al., 2004; Surian and Rinaldi, 2003), with circumstantial evidence of rivers in Germany converted from anabranching to single-thread to improve navigation (Herget, 2000). Techniques to improve river navigation such as meander cut-off to straighten channels appear during Medieval times (Herget, 2000) and the practice of de-snagging to remove in-stream large wood became common place up to and including the last century (Brooks et al., 2004; Piégay and Gurnell, 1997). Modification to river systems has progressively led to the hydrological processes of the channel, floodplain, riparian and hyporheic zones of many rivers becoming disconnected from each other (Kondolf et al., 2006).

As a result of this legacy of river modification the majority of UK rivers have been affected in some way; a survey showed that only 23% of sampled river reaches could be classed as geomorphologically natural (Sear et al., 1998). In the USA more than a third of rivers are impaired or polluted (Bernhardt et al., 2005) and extinction of freshwater fauna is five times that of terrestrial fauna, with many more species imperilled and a projected extinction rate matching that of tropical forests (Ricciardi and Rasmussen, 1999).

In the past thirty years, in response to cultural demands and increasing legislation (e.g. EU, 2000) the degraded nature of river systems has begun to be addressed and ways of halting or reversing further damage have received increased attention and funding (Kondolf et al., 2006) Initially most river restoration projects focused on improving water quality and removing or mitigating pollution sources (Harper et al., 1999), in the past 15 years there has been an increase in projects attempting to improve biodiversity or the abundance of target species through habitat restoration (Clarke et al., 2003) and in the past ten years focus has increasingly turned towards the benefits of integrated catchment management (Beechie et al., 2010).

3.9. Types of restoration

The techniques and methods employed in river restoration projects are informed by the goals of the project as well as by physical and socioeconomic constraints (Caruso and Downs, 2007; Sear and Arnell, 2006; Wohl et al., 2005). These techniques can be grouped into a hierarchy based on the degree of intervention; feature, form and process based restoration. Feature based restoration projects are designed to be stand alone and are not typically linked into wider restoration of the system, for example the placement of pieces of large wood in the channel in order to provide discrete habitat zones (Larson et al., 2001). Form led restoration typically involves restoring the form of a reach towards a reference state, but without significant interventions occurring outside of the target reach (Clarke et al., 2003); for example re-meandering and raising the bed in a reach of channelised river (Nagayama et al., 2008). Process based restoration involves attempting to restore the eco-hydromorphic processes that drive a dynamic river system (Beechie et al., 2010), this can be thought of as more of a passive restoration where the river with restored processes will do the work to create geomorphic forms (Brierley and Fryirs, 2009; Wheaton et al., 2008). Process based restoration is often accompanied by a degree of form restoration where the river is especially degraded but the focus is on long-term sustainability rather than short-term solutions (Beechie et al., 2010). Process based restoration will often need to be at the watershed scale to successfully address root causes of initial process degradation (Clarke et al., 2003; Harper et al., 1999), even if some of the physical works are conducted on a reach scale nested within catchment scale objectives (Wheaton et al., 2008).

3.9.1. Feature based restoration

Feature based restoration is the most common form of river restoration (Bernhardt et al., 2005) despite doubts over the effectiveness of the approach (Whiteway et al., 2010). The most common type of feature restoration is the introduction of in-stream objects, such as large wood or boulders to create habitat for target species (Bernhardt et al., 2005). Large wood added as part of river restoration impacts on flood hydrology by adding geomorphological heterogeneity and increasing the hydraulic roughness of the channel , acting to dissipate energy and increase flood wave travel time (e.g. Gregory et al., 1985; Sholtes and Doyle, 2011; Thomas and Nisbet, 2012); a full analysis of the effects of in-stream large wood on flood hydrology is covered in section 3.11.

The perceived benefits of restoring a particular feature of a river system which has been degraded, or is perceived to be missing, are that it is usually cheap (Bernhardt et al., 2005), the results are immediate and are often visual and easily understood by all stakeholders (Tunstall et al., 2000). However the results, although immediate, are often short-term (Kondolf, 1998) and there are few reciprocal benefits to other parts of the system (Larson et al., 2001). This type of approach is more accurately called rehabilitation, as there is no attempt to restore to a previous condition, only to mimic a feature and/or habitat from a less degraded reference example (Van Diggelen et al., 2001).

Although use of large wood structures in river restoration remains widespread there are numerous studies indicating such an approach has only limited, short-term benefits. After installation of large wood in a channel physical habitat diversity is commonly improved and macroinvertebrate richness can be increased (Miller et al., 2010), however salmonid populations generally fail to show increases post-restoration and there are not commensurate improvements to habitat quality beyond increases in physical heterogeneity (Larson et al., 2001; Whiteway et al., 2010). The permanence of installed large wood both in restored and natural rivers also remains an unknown quantity (Powell et al., 2009). There are no detectable improvements in biological conditions associated with the wood and although there have been shown to be modest improvements in physical habitat these are only on scales of 2-10 years (Larson et al., 2001; Sweka et al., 2010). Larson et al (2001) examined several log placement projects and concluded they produced only limited success at controlling downstream sedimentation where this was an objective. Two recent meta-analyses showed that macroinvertebrate biodiversity can be enhanced by habitat restoration, but there are not corresponding increases in density (Miller et al., 2010), and that introduction of large wood structures on their own were not effective at increasing salmonid populations (Whiteway et al., 2010). The failure of large wood structures to increase target species populations may be due to inappropriate usage of these structures in systems where salmonid populations are not limited by lack of habitat (Rosenfeld and Hatfield, 2006; Sweka et al., 2010)

Imposing a feature within the fluvial landscape can also cause problems which necessitate continuing management, where a feature, such a spawning gravels introduced for salmon may need continuing maintenance in order to prevent it being washed away (Newson et al., 2002).

3.9.2. Form based restoration

The approach of restoring a river reach to a pre-historic, or reference form is often driven by ecological goals; for example increasing the numbers of a target species, or the biodiversity of a target group (Harper et al., 1999). Form based restoration is usually undertaken on a reach scale, and rarely are catchment wide processes included in project designs (Harper et al., 1999). The ecological objectives of a project are typically intended to be met after a short adjustment period post completion (Clarke et al., 2003).

Form based restoration can include; reconnection of the channel to the floodplain through bed level raising and re-meandering, enhancing the potential for overbank flow and for flood waves to be attenuated as water flows over the complex floodplain surface (Anderson et al., 2006), the planting of riparian woodland or buffer strips (Collins et al., 2012), which adds further hydraulic roughness to the floodplain and further attenuates flood waves (Archer, 1989; Thomas and Nisbet, 2007).

A form based approach to river restoration has received numerous critical analyses in the geomorphological and ecological literature as the lack of process restoration often results in failure to meet long-term objectives for ecosystem regeneration (Beechie et al., 2010; Clarke et al., 2003; Gilvear, 1999; Harper et al., 1999; Kemp et al., 1999; Kondolf, 1998; Sear, 1994; Sear et al., 1995), furthermore an imposed form may have unintended ecological consequences outside of the target reaches (Sear, 1994; Wheaton et al., 2008).

3.9.3. Process based restoration

Restoring the eco-hydromorphic processes within an entire catchment as part of integrated catchment management is an approach that has received increasing attention in the past two decades (Beechie et al., 2010; Downs et al., 1991). This technique is based on identifying the root causes of degradation within the catchment and then attempting to reverse or address them (Brookes and Sear, 1996; Kondolf, 1998; Newson et al., 2002), so that the system recovers towards a dynamic equilibrium position over time (Sear, 1994). It involves detailed analysis of past landscape change and hydrological history as well as quantifying processes such as the sediment transport regime (Beechie et al., 2010; Brookes and Sear, 1996; Clarke et al., 2003; Ward et al., 2001). The final project plan is designed to normalise the eco-hydromorphic processes in the catchment and deliver a sustainable river system (Beechie et al., 2010).

The restored river system is designed to occupy a dynamic equilibrium and so can respond to natural disturbance events and adjust to natural variations in climate (Hughes et al., 2005; Newson et al., 2002); at any given time the system will occupy a variable state within a range of acceptable, predicted limits (McDonald et al., 2004). The planning of process based restoration and predicting the outcomes of a project is dependent on quantifying and understanding the eco-hydromorphic processes within the system as well as the historical landscape forms and disturbance regime, referred to as the historical range of variability (Agee, 2003). The first papers describing a process based approach to river restoration did not appear until the mid-1990's and so fully monitored examples of this approach remain rare (Beechie et al., 2010). One example from the literature is a project to restore the Rhone in Switzerland from a channelized river to an uncontained braided form has progressively increased the mosaic of habitats over 13 years and increased the density of wading birds (Arlettaz et al., 2011).

Process based river restoration can involve spatially extensive changes to land use both on hillslopes and in the riparian zone, leading to a high potential for attenuating runoff (Bracken and Croke, 2007) and slowing down flood waves (Anderson et al., 2006; Archer, 1989). Due to the greater spatial extent of process based restoration compared to form and feature based approaches there is potential for catchment scale flood hydrology to be altered; for example changing the relative timings of sub-catchment flood waves which have been shown to be important in flood magnitude (Pattison et al., 2008). The influence of land use on flood hydrology is explored in more depth in section 3.10.

3.9.4. Forest Restoration

As well as programmes to restore rivers there are widespread programmes to encourage the restoration of forests on hillslopes and floodplains with the restoration of riparian forests a goal of many river restoration projects (Nislow et al., 2002) and biophysical landscape development and riparian forest planting recommended as part of holistic river restoration (Collins et al., 2012). European forest cover is expanding through; commercial forestry plantations, environmental protection and for recreation and amenities (Robinson et al., 2003). Policy and practise in North West Europe encourages the restoration of natural forests (Nislow and Lowe, 2006) and early successional forests are protected across New England, USA and millions of dollars are spent annually to create and maintain early successional forest habitats across the USA (King et al., 2011). Where forests are newly planted there are likely to be changes to aquatic ecosystems (Nislow and Lowe, 2006) and runoff (Harr, 1986) as forest succession proceeds; despite these twin effects forest restoration is driven by ecological goals and there are currently no examples in the literature of flood control as a driver of forest regeneration. Although paired catchment experiments have been reported (e.g. Archer et al., 2010; Krishnaswamy et al., 2012; Robinson et al., 1998), these are either; afforestation experiments, deforestation experiments, regrowth experiments or forest conservation experiments (Brown et al., 2005), rather than flood control experiments.

3.10. Land cover and flood hydrology

Land cover has a large effect on flow paths and storage within the rainfall runoff process. The key mechanisms by which land cover can mitigate flood risk are through; increasing interception (Robinson et al., 2003), increasing infiltration (Bracken and Croke, 2007), increasing storage (Ghavasieh et al., 2006), attenuating runoff (Broadmeadow and Nisbet, 2009) and slowing conveyance (Lane et al., 2007). Table 3.1 summarises the effects of specific land cover changes upon aspects of rainfall runoff and flood generation processes. Changing land use and increasing the extent of forests has been identified in the UK as having potential to reduce downstream flood risk as part of DEFRA's "Making space for water" strategy (Defra, 2005a; Defra, 2005b; Defra, 2007; EA, 2009b). Serious flooding along the Rhine and Meuse in the early 1990s also highlighted the possibility that changing land use and deforestation had affected river flow regimes (Mendel, 1996 in Robinson et al, 2003).

There is not currently a consensus in the scientific literature regarding the effects of forest land cover on flood hydrology (Andréassian, 2004; Archer et al., 2010; Robinson et al., 1998), with Van Dijk et al (2009, p110) claiming "the effects of [forest cover] on flooding is a hotly debated issue" and that evidence does not suggest a strong role of trees in floods, however Van Dijk et al (2009) focused on increased interception and infiltration from trees, whilst neglecting runoff attenuation, potentially underestimating effects of forests on flood hydrology. Other studies show that as forests mature effects on attenuating runoff increase (Harr, 1986). Flood attenuation effects however are less, or non-existent, for extreme rainfall/flood events (Grant et al., 2008; McCulloch and Robinson, 1993; Robinson et al., 2003) and less where only parts of the basin are forested (Swanson et al., 2010). Part of the uncertainty in the effects of land cover on flooding is that variability in hydrological behaviour due to climatic variations is of the same order of magnitude as any effects of land cover changes

(Fritsch, 1990 in Andréassian, 2004). In a review of the literature Andréassian (2004) concludes that despite much sound hydrology work it is still the case that the flood hydrology response to reforestation is "highly variable and for the most part, unpredictable" (Hibbert, 1967 in Andréassian, 2004) and that overall process and watershed scale research only agrees on the direction of change and not the magnitude (Andréassian, 2004), however much of the uncertainty on flood hydrology responses to changing land use is attributable to the lack of evidence in larger catchments over $10km^2$ (Archer et al., 2010). Despite uncertainty in the effects of land cover on flood hydrology DEFRA's 'Making Space for Water' report (Defra, 2005a) indicated the potential for land use and increased forest cover to reduce flood risk (EA, 2009b).

Forests intercept rainfall and reduce throughfall; with density of trees, uniformity of the canopy and the nature of bark and leaf types important variables in the proportion of interception (Crockford and Richardson, 2000). Conifers have greater aerodynamic roughness and thus enhance interception losses and storage (Robinson et al., 2003). Forests increase total water usage (Robinson et al., 2003), with certain tree species such as conifers (Robinson et al., 2003) and eucalyptus (King, 2013) having particularly high water usage, this increased water usage also enhances total annual evaporation loss (Robinson et al., 2003).

Land management and land cover type affects runoff generation (Bull et al., 2000; Lasanta et al., 2000) and connectivity at all spatial scales (Bracken and Croke, 2007), with vegetation reducing runoff (López-Moreno et al., 2006). Intense afforestation reduces peak runoff and total runoff volume (Hundecha and Bárdossy, 2004), and degraded forests and forest plantations are dominated by quickflow compared to remnant old growth forests (Krishnaswamy et al., 2012). The most important element in slowing or reducing runoff is the spatial configuration of land cover, rather than the absolute area (Cammeraat and Imeson, 1999; Fitzjohn et al., 1998; Ludwig et al., 2005). Agricultural land use tends to increase connectivity and thus runoff (Archer et al., 2010; Ludwig et al., 1995); cereal crop cultivation increases overland flow and soil erosion (Boardman, 1995; Sullivan et al., 2004), plough furrows concentrate runoff and channel flow (Govers et al., 2000; Kirkby et al., 2002) and drainage within the riparian zone results in rapid transfer of storm water to channels (Burt, 1997; Burt and Haycock, 1996; Burt and Pinay, 2005; Pinay et al., 1998).

Within the riparian zone, extensive vegetation may act to limit runoff connectivity (Bracken and Croke, 2007) and reduces runoff volume (Broadmeadow and Nisbet, 2009). Where riparian forest restoration has taken place in-channel effects of the vegetation will increase as the forest matures (Fisher et al., 2010)

Vegetation greatly increases infiltration (Bracken and Croke, 2007) due to increased organic matter and bulk density of soil (Boix-Fayos et al., 1998). Runoff generated on bare hillslopes can infiltrate when it runs through areas of vegetation (Boer and Puigdefábregas, 2005; Dunkerley, 1999; Puigdefábregas, 2005; Sanchez and Puigdefabregas, 1994; Valentin et al., 1999), illustrating the potential for forest buffer strips or vegetation strips to attenuate runoff. Forests have been identified as contributing to increased infiltration (Broadmeadow and Nisbet, 2009). Forests have high water usage compared to other vegetation types (Robinson et al., 2003) and thus in the presence of climatic variability with periods of hydrological surplus a forest can proportionally dry the soil out more than other vegetation types increasing potential infiltration rates and capacity during rainfall events (Andréassian, 2004). In addition to bulk infiltration volumes old forests have been shown to have deep subsurface stormflow and groundwater pathways which lag storm water delivery to streams (Krishnaswamy et al., 2012).

Local storage can slow flood wave propagation (Thomas and Nisbet, 2007) and retain flood waters in non-critical areas reducing flooding downstream (Liu et al., 2004). Re-connecting rivers with their floodplains can attenuate flood magnitude (Acreman et al., 2003) and delay flood peaks (Ghavasieh et al., 2006), with riparian vegetation slowing overbank flow (Archer, 1989). Heavily vegetated riparian zones typically have a very rough soil surface and an intact litter layer which increases hydraulic resistance and thus slows conveyance (Bracken and Croke, 2007; Broadmeadow and Nisbet, 2009). Anderson et al (2006) found reductions in peak discharge of 12% with tall (3m canopy height) compared to non-tall (0.5m canopy height) riparian vegetation and Thomas and Nisbet (2007) found riparian woodland showed the greatest effect on attenuating floods. Retarding the passage of flood waves is considered to be the greatest potential flood mitigation effect of forested floodplains (Broadmeadow and Nisbet, 2009).

In the case of partly forested catchments, the effects on flood height and timing will depend greatly on the timings and synchronisation of peak flows from sub-catchments (McCulloch and Robinson, 1993). Although field and modelling studies have shown promising results in attenuating flooding upscaling the effects of land cover change remain problematic (Pattison and Lane, 2012b) and there is little evidence from catchments greater than 10km^2 (Archer et al., 2010). It is difficult to demonstrate a link between land cover and flooding at catchment scales (Parrott et al., 2009); there are problems of extrapolating small scale studies to the catchment scale as the relative importance of different processes changes as scale changes (Archer, 2003; Blöschl et al., 2007; López-Moreno et al., 2006). At the plot scale storage, interception and infiltration dominate, whereas at catchment scales channel processes are dominant (Archer, 2003). Figure 3.4 shows a conceptual relationship of climate and land cover importance at increasing scale, this figure suggests that land use changes have the largest impact in small catchments and as catchment size increases land use changes are likely to have a smaller impact than the range of climate variability. Field studies of land cover change are problematic as changes to land use occur as a patchwork and effects of one change maybe counteracted by changes elsewhere in the catchment (Archer, 2003) and changes may proceed slowly over time as vegetation grows (Archer, 2003). Furthermore the effects on catchment hydrology of land cover change will be of the same order of magnitude as climatic variations making trends difficult to discern (Archer, 2003; Fritsch, 1990 in Andréassian, 2004). The effects of vegetation will also depend on context with lithology, catchment size and location of vegetated areas within the catchment all important in determining flood response (López-Moreno et al., 2006). Although paired catchment approaches have provided useful data on land cover changes and flood response these are only circumstantial (Brown et al., 2005) and further research is needed at the catchment and representative hillslope scale to show the effects of riparian systems on upland hydrologic inputs (Naiman and Décamps, 1997).

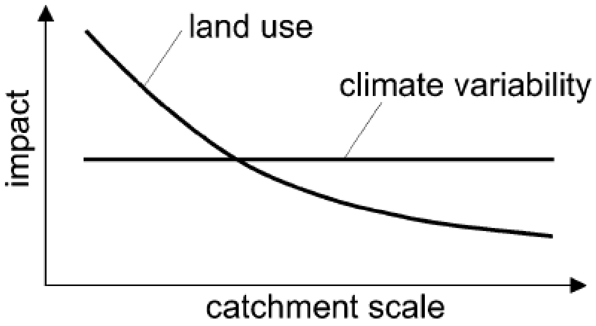


Figure 3.4 - Hypothesized impact of land use and climate variability on hydrological response as a function of scale (from Blöschl et al., 2007, their Figure 1)

Land cover impacts on flood hydrology are complicated by different magnitudes and directions of effects at different points in the growing cycle (Archer, 2003). In young forests losses due to interception and evapotranspiration are low and variable and response time of the catchment to rainfall is short; evapotranspiration and interception losses increase each year as a forest ages and approaches canopy closure (Robinson et al., 1998). Within newly established forest plantations artificial drainage channels will dominate and increase runoff, as the forest matures tree growth results in increased interception, infiltration and evapotranspiration which counteract the drainage effects (Archer, 2003; Robinson et al., 1998). As a forest grows there will be feedback mechanisms operating (Blöschl et al., 2007; López-Moreno et al., 2006); reductions in peak flows and floodplain inundation aids the establishment of riparian vegetation and the stabilisation of the alluvial plain, which in turn can reduce peak flows (López-Moreno et al., 2006). Feedback loops operating in catchments are poorly understood and are likely to exist at a range of scales, feedback between vegetation, soil moisture and local climate are likely to be especially important in flood hydrology and need to be elucidated (Blöschl et al., 2007).

A key variable in the effects of land cover on flood hydrology is event magnitude. During moderate, less intense rainfall events a reduction in peak flows with vegetation cover will occur due to increased interception and infiltration (Caissie et al., 2002; Gallart and Llorens, 2003; Lewis et al., 2001; López-Moreno et al., 2006). In large rainfall events catchments are dominated by vertical flowpaths (Salemi et al., 2013) as interception and infiltration capacities are quickly saturated (López-Moreno et al., 2006). The effects of vegetation cover and land use on runoff is uncertain during intense rainfall events (Andréassian, 2004; Gallart and Llorens, 2003; Lane et al., 2005; Niehoff et al., 2002); it is well established that the magnitude of events with a long return period (greater than 100 years) are not significantly affected by afforestation or deforestation (Andréassian, 2004), although moderate floods show clear evidence of change with afforestation (Archer et al., 2010; López-Moreno et al., 2006).

Overall the effects of vegetation growth, mature forests and event magnitude on flood hydrology remain uncertain (Andréassian, 2004; Archer et al., 2010; Robinson et al., 1998).

3.11. Large wood and logjams

Historically wood has been removed from watercourses by man (Brooks et al., 2004; Krejčí and Máčka, 2012), but is now routinely inserted into channels as Engineered Logjams (ELJs) as part of river restoration (Brooks et al., 2004; Collins et al., 2012; Reich et al., 2003; Shields Jr et al., 2006). ELJ installation remains a popular method of river restoration (Abbe and Brooks, 2011) despite benefits such as changes to habitat heterogeneity being short-lived. Although not recommended as a holistic method of river restoration (Beechie et al., 2010) the installation of ELJs is seen as an important initial step in programmes of landscape restoration (Abbe et al., 2003; Collins and Montgomery, 2002) where newly planted forests may take many years to mature sufficiently to be a source of large wood to the river channel (Benda and Sias, 2003; Bragg, 2000).

Large wood and logjams increase hydraulic roughness (Curran and Wohl, 2003; Kitts, 2011; Sear et al., 2006; Shields Jr and Gippel, 1995) which reduces flow velocity (Shields Jr and Gippel, 1995), increases flood wave travel time (Gregory et al., 1985; Thomas and Nisbet, 2012) as well as providing temporary storage (Shields Jr and Gippel, 1995). Logjams also increase connectivity with the floodplain and force water out onto the floodplain surface at lower discharges and earlier in flood events (Sear et al., 2010).

Process/Store	Example Activity		Potential system impacts
	Increase	Decrease	
Canopy layer interception storage	Forest restoration	Land cover type, Crop or forest harvesting, urbanisation, fire, disease	Evapotranspiration, travel time (precipitation to reach soil level)
Litter layer storage	Forest restoration	Fire control, urbanisation, forest management/harvesting	Evapotranspiration, infiltration, runoff volume
Root zone storage	Conservation, Forest restoration	Agriculture (tillage, stocking density, soil compaction, modify soil organic content)	Evapotranspiration, infiltration, runoff volume
Infiltration-excess overland flow	Agriculture (tillage, stocking density, soil compaction, modify soil organic content), urbanisation	Crop cultivation, Forest restoration, Riparian buffer strips	Runoff volume and timing
Saturation-excess of overland flow		Land drainage, forest plantations, irrigation	Antecedent soil moisture, runoff volume and timing, ground water recharge
Subsurface interflow	Land drainage		Antecedent soil moisture, runoff volume and timing, ground water recharge

Ground water recharge		Urbanisation, irrigation, ground water abstraction.	Antecedent soil moisture, runoff volume and timing
Channel storage	Channel remeandering, riparian vegetation restoration, in-stream woody debris, dredging	Urban sewer networks, channel engineering, siltation	Hydraulic roughness, overbank flow, runoff/channel flow travel time
Floodplain storage	Riparian forest restoration, wetland conservation	Flood defence structures, urbanisation, hedgerow removal	Hydraulic roughness, runoff/channel flow travel time, ground water recharge/discharge

Table 3.1 – showing potential land cover effects on aspects of rainfall runoff/flood generation processes and storages, with examples of land cover types/land usage which may increase or decrease process flows and storage. Adapted from Wilby et al, 2008

Changes to hydraulic roughness occur as a result of skin friction, eddy losses and form resistance as flow distorts around the large wood structures (Einstein and Barbarossa, 1952; MacVicar, 2013), this results in increased drag (Buffington and Montgomery, 1999; David et al., 2011). The largest contribution to wood related hydraulic resistance is where spill occurs (Curran and Wohl, 2003; Kitts, 2011) in smaller channel logjams can behave in a similar way to broadcrest weirs (Dust and Wohl, 2012).

ELJs have been shown to alleviate downstream flooding in modelling studies (e.g. Sholtes and Doyle, 2011; Thomas and Nisbet, 2012), although results are mixed. ELJs slow the passage of flood waves, although they do not reduce peak magnitude, this effect was attributed to water moving onto the floodplain and flowing back into the channel a short distance downstream of the ELJ (Thomas and Nisbet, 2012). Sholtes and Doyle (2011) modelled small scale changes in channel characteristics representing river restoration and found these provided minimal quantifiable impact due to the small scale of restoration compared to the size of catchments and suggest scales as much as 5-10km of restoration would be needed to see catchment scale effects. Liu et al (2004) modelled increases in channel hydraulic resistance and sinuosity to represent restoration and found an average reduction in peak flow magnitude of 14%. Curran and Wohl (2003) conclude from field measures of logjam roughness that distribution and function of large wood and not just abundance is a critical variable in hydraulic resistance and resulting changes to flood hydrology.

3.12. Conclusion

Flood generation is a complex processes operating at a range of scales. With increased future flood risk under climate change there is an imperative for a new way of thinking and undertaking flood mitigation strategies given that continuing to increase the height and extent of at-a-point engineered flood defences is unsustainable. One possible adaptation is to incorporate catchment wide land cover management plans into flood mitigation, where hillslope runoff can be reduced or attenuated and complex river channel and floodplain morphology and vegetation can slow flood wave travel time and attenuate flood peaks.

There remains a debate in the scientific literature as to the effects of both land cover and engineered logjams on flood hydrology. There is a great need for more studies investigating the catchment scale flood effects of spatially distributed land cover change, for which numerical modelling can play a valuable role. Numerical models of flood hydrology can be particularly useful in relation to forest restoration where effects on flood hydrology will be temporally invariant as the forest stand matures and changes in structural composition as it ages. Catchment wide flood hydrology studies will help to answer questions of land cover and flood hydrology interactions and also help inform governmental policy on forest restoration and catchment wide flood risk management plans.

In this thesis using field studies, literature reviews and numerical modelling I will address gaps in the scientific literature by:

- Quantifying the hydraulic resistance of logjams and using a numerical hydrology model to study the effects on catchment scale flood hydrology of inserting engineered logjams within a channel as part of river restoration.
- Developing a conceptual model of riparian forest growth based on a forest growth model and literature values and then using a numerical hydrological model to study the effects on catchment scale flood hydrology of spatially distributed forest restoration over 100 years of forest growth.

4. Effects of large wood and logjams on hydrology and geomorphology

The presence of pieces of large wood within a river channel and on the surrounding floodplain is a natural element of a forested river system. The amount of wood in the channel, its behaviour when in the river system, and its longevity are all functions of both the riparian and aquatic environments, and the connectivity between them.

Historically wood has been removed from river channels (Brooks et al., 2004) in order to aid navigation, minimise local flood risk and for aesthetic and recreational reasons. The trend of decreasing large wood loads in river channels is beginning to be reversed in recent years through river restoration (Brooks et al., 2004; Collins et al., 2012; Reich et al., 2003; Shields Jr et al., 2006) and riparian forest restoration (Nislow and Lowe, 2006; Nislow et al., 2002) (see Chapter 3 for review of river restoration effects on large wood).

Wood will become organised into accumulations, or logjams, where there is a balance between mobile and stable pieces of large wood and where there are sufficiently high loading rates of wood to the river channel. In addition to the effects of isolated pieces of large wood, logjams have been shown to have an important impact on eco-hydromorphic processes (Collins et al., 2002).

In the scientific literature large wood is commonly defined as a piece of living or dead wood exceeding 10mm diameter and 1m in length, lying within the channel or riparian zone, however minimum size criteria vary greatly between studies (Wohl et al., 2010) and criteria for including or excluding pieces of large wood in a study are rarely linked to channel dimensions (Wohl et al., 2010). A lack of standardisation in large wood criteria or the use of dimensionless units, such as pieces length/channel width, makes comparisons between studies of large wood problematic. In-stream wood is variously referred to as 'Large wood', 'Large woody Debris' (LWD) and 'Coarse Woody Debris' (CWD) in the scientific literature. Figure 4.1 shows scientific papers published by year using different descriptors for in-stream wood, this shows both the explosion in interest in the subject in the late 1990's and the increasing use of 'large wood' in the literature. Coarse woody debris remains a popular descriptive term and is found mainly in ecological journals, particularly those based in the USA. Both large woody debris and large wood have been used more often in European based journals and in recent years the popularity of the latter has grown due to perceived perjoritive connotations of using "debris". In recent years large wood has gained preference in the geomorphological literature, due to the possible pejorative connotations of using "debris" for an ecosystem component with numerous documented benefits and thus 'Large wood' will be used throughout this thesis.

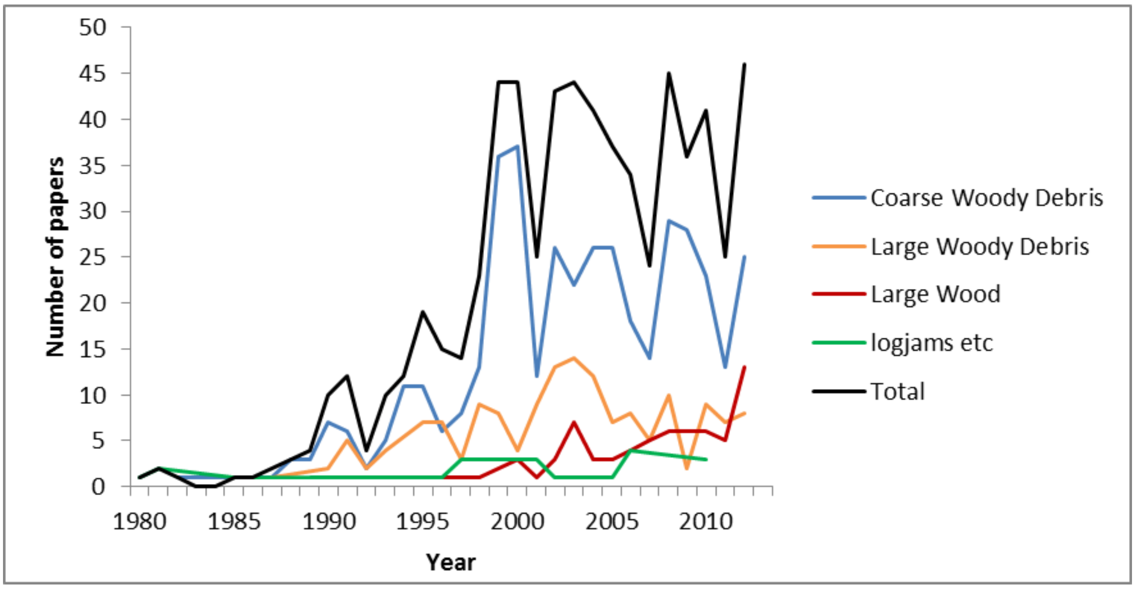


Figure 4.1 – papers published by year using different terms to describe in-stream wood in the paper title. This shows the steady growth in studies from the early 1980's and the explosion in interest during the late 1990's. Source: Web of Science searches by publishing year for in-stream wood descriptive terms in paper title, 15/05/2013, papers not related to in-stream wood were manually excluded from figures.

4.1. Eco-Hydromorphic Functions of large wood

Large wood and accumulations of large wood influence the hydrology, geomorphology and ecology of a river (Brooks et al., 2004; Gurnell et al., 2002;

Millington and Sear, 2007; Piégay and Gurnell, 1997). The effects of individual pieces of large wood and logjams are usually local and are highly site specific (Montgomery et al., 2003). Despite the local scale of effects and a high degree of spatial variability the influence of large wood can be observed on hydrology, sediment transport and ecology at the catchment scale (Montgomery et al., 2003; Sear et al., 2006; Sear et al., 2010).

4.1.1. Effects on geomorphology

In-stream large wood and logjams exert an influence on local hydraulics and cause local variations in shear stress, these local variations in shear stress in turn lead to heterogeneity in local patterns of sediment erosion and deposition. Large wood and logjams increase the hydraulic resistance of the channel (Gregory et al., 1985; Sholtes and Doyle, 2011). Increased hydraulic resistance dissipates energy, reduces erosive power (Gregory et al., 1985; Montgomery, 2003) and reduces the transport competence of the channel at reach and catchment scales (Buffington et al., 2004). Hydraulic resistance occurs as a result of large wood and logjam obstacles due to skin friction between obstacles and flow (Einstein and Barbarossa, 1952), drag (Buffington and Montgomery, 1999; David et al., 2011; Shields Jr and Gippel, 1995), turbulence due to eddy losses (Einstein and Barbarossa, 1952), spill over obstacles (Dust and Wohl, 2012; Yochum et al., 2012) and flow expansion and distortion around obstacles (Einstein and Barbarossa, 1952; MacVicar, 2013).

Channel characteristics are important in determining the potential influence of in-stream large wood and logjams on geomorphology. Kitts (2011) showed there is a continuum across channel types with in-stream wood where hydraulic resistance increases with slope. Studies in high gradient step-pool channels have shown spill with resulting sharp changes in velocity (Curran and Wohl, 2003) associated with wood is the dominant source of hydraulic resistance in these environments (David et al., 2011; Dust and Wohl, 2012; Yochum et al., 2012). At low to medium discharges increased hydraulic resistance associated with large wood slows stream velocity (Gregory et al., 1985; Gurnell et al., 2002; Millington and Sear, 2007) and reduces the competence of the channel to transport sediment (Montgomery, 2003) although large flood events can drown-out hydraulic resistance effects of in-stream wood (Gregory et al., 1985).

At local scales the type and orientation of in-stream large wood can be important for patterns of erosion and deposition. Orientation of individual pieces of large wood is important for drag forces (Shields Jr and Alonso, 2012), as is the complexity of large wood shape with branching logs causing higher drag than single trunks (Hygelund and Manga, 2003). The orientation of large wood with an attached rootwad affects scour patterns with upstream orientated rootwads causing primarily flow deflection and minimal scour, whereas downstream orientated rootwads lead to local scour erosion around the rootwad and subsequent bar deposition downstream (Svoboda and Russell, 2011).

4.1.1.1. Channel form

In-stream wood is the primary influence on channel form in small rivers (Collins et al., 2002), although there are fewer studies in large rivers (Collins et al., 2012; Collins et al., 2002). Wood has been shown to be important in braided river systems where it reduces overall sediment transport (Montgomery et al., 2003; Piégay and Gurnell, 1997), stabilises emergent bars and mediates island formation (Bertoldi et al., 2013; Gurnell et al., 2005).

Recent research has suggested the development of meandering river styles coincided with the evolution of trees. Sedimentary alluvial deposits from the Cambrian period (541-485 million years ago), before the evolution of terrestrial plants, are broad unconsolidated blankets of coarse sediment (Davies and Gibling, 2011) reflecting river systems which were highly unstable and characterised by wide sandy beds and abundant channel migration (Davies and Gibling, 2012). The appearance of terrestrial plants during the Palaeozoic period (541-252 Ma) has been described as one of the most significant changes to the Earth system (Davies and Gibling, 2011). During the Late Silurian (427-419 Ma) and Devonian (419-358 Ma) periods the appearance of plants with root systems coincides with the development of channelized sand bed rivers (Gibling and Davies, 2012), and by the end of the Devonian period, stable floodplains and a diverse array of meandering and braided channel styles (Davies and Gibling, 2011; Montgomery et al., 2003). As plants evolved the development of complex and diverse root systems stabilised river banks and floodplain surfaces (Davies and Gibling, 2011; Davies et al., 2011). Fossilised trees begin to appear in alluvial sediments during the Carboniferous period (358-299 Ma) along with development of large mainstream meandering channels (Davies et al., 2011). As tree like plants evolved and expanded in geographical range during the Early Pennsylvanian period (323-307 Ma) narrow fixed channels appear, including the development of anastomosing and island-braided styles (Gibling and Davies, 2012). The development of dense floodplain forests would have led to stable floodplain surfaces due to root strengthening (Gibling and Davies, 2012); the input of abundant large wood from floodplain forests to river channels would have acted as an avulsion trigger leading to development of anastomosing rivers (Davies and Gibling, 2011). During the Carboniferous period floodplains and river morphology became increasingly controlled by vegetation as tree habitats expanded (Davies and Gibling, 2011), with fluvial styles also influencing plant development (Gibling and Davies, 2012).

4.1.1.2. Pool formation

Large wood promotes geomorphological heterogeneity and change through altering local erosional patterns (Gurnell et al., 2002; Montgomery et al., 2003; Piégay and Gurnell, 1997). Numerous studies in the scientific literature report the effects of large wood on pool formation; plunge pools can be scoured at the downstream end of step-forming logjams (Curran and Wohl, 2003; Richmond and Fauseh, 1995; Wohl et al., 1997), in low gradient streams back-water pools can form upstream of channel spanning logjams (Jeffries et al., 2003) and local scour associated with flow diversion around individual pieces of large wood and non-channel-spanning logjams can form pools (Braudrick et al., 1997; Millington and Sear, 2007; Piégay and Gurnell, 1997; Svoboda and Russell, 2011). The packing of logjam interstitial space with fine organic material such as leaf litter is a key determinant in logjam porosity (Bilby and Ward, 1991); logjams of lower porosity are more likely to cause a backwater effect and a step in the water profile (Bilby and Ward, 1991), thus leading to pool formation. Packing of logjams with fine organic material is largely restricted to rivers with deciduous floodplain forests and is thus highly seasonal (Jeffries et al., 2003). Forested rivers have more pools than non-forested rivers (Gurnell and Sweet, 1998; Montgomery et al., 1995), and most pools within forested rivers are in association with large wood (Collins et al., 2002; Montgomery et al., 2003). Various studies have reported the association of pools and in-stream large wood; Gurnell et al (2002) reported 87% of in channel wood had a pool associated with it in the New Forest, UK, Collins et al (2002) that 61% of pools in the Nisqually River, Washington, USA were associated with large wood and Richmond and Fauseh (1995) that 78% of pools were formed by large wood in Rocky Mountain streams in Northern Colorado, USA. Pools found in association with large wood are deeper than non-wood formed pools, but also display a greater variability in depth (Abbe and Montgomery, 1996; Montgomery et al., 2003).

4.1.1.3. Reach and Catchment scales

Large wood and logjams cause local variations in shear stress which affects sediment transport and deposition (Keller and Tally, 1982) acting to sort sediment as well as scour pools (Assani and Petit, 1995; Beschta, 1979; Kasprak et al., 2012; Thompson, 1995). Reductions in overall erosive power and bed shear stress reduces the competence of the channel to mobilise its bed at low to medium discharges (Montgomery et al., 2003), leading to increased retention of sediment both in the channel and on the floodplain (Gregory et al., 1985; Gurnell and Sweet, 1998; Sear et al., 2010; Wallerstein and Thorne, 2004), as well as reducing the average surface grain size in the bed (Buffington et al., 2004; Montgomery et al., 1996a; Montgomery et al., 2003). Scour associated with logjams acts to clean and sort bed materials suitable for salmonid spawning by exposing and flushing fines from gravels (Lewis, 2010). Logjams are efficient at trapping and retaining sediment (Bilby and Likens, 1980) compared to other in-stream obstructions (Fisher et al., 2010); Fisher et al (2010) found reaches with large wood to have sediment accumulation rates of 0.7g cm^{-2} d^{-1} and residence times of over 100 days, compared to accumulation rates of 0.2g cm^{-2} d^{-1} and residence times of less than 100 days in reaches with sparse large wood. As well as bulk storage of sediment there is also an impact of large wood on the formation of sedimentary structures both on the floodplain (Sear et al., 2010) and in the channel (Abbe and Montgomery, 1996; Montgomery et al., 2003), with increases in riffles and bars over non-forested river channels (Gurnell et al., 2002). In high gradient streams sediment will accumulate upstream of wood obstructions and in low gradient streams deposition will be downstream, often in mid channel bars (Daniels, 2006).

The effects of in-stream wood and forested floodplains on bank erosion and channel planform are a complex balance between forces. Root structures of trees can stabilise floodplain surfaces and bank materials and provide resistance to erosion (Beechie et al., 2006; Shields Jr and Gray, 1992), in addition logjams and individual pieces of wood in the channel can act as armouring for banks. Conversely logjams divert flow towards banks and cause local variations in shear stress which increase local erosive power (Davies-Colley, 1997; Montgomery et al., 2003) and alter local sediment transport (Sear et al., 2010). Logjams divert flow towards banks and cause local bank erosion leading to avulsions, side channels and ephemeral floodplain channels (Sear et al., 2010), in some cases logjams can maintain a multiple channel planform (Collins and Montgomery, 2002; Collins et al., 2002; Montgomery et al., 2003). A channel which has large wood accumulations and associated sedimentary deposition can incise or aggrade in the absence of climate or tectonic changes in its boundary conditions (Montgomery et al., 2003).

Riparian trees capable of rooting from fallen trees or broken branches, such as willow (*salix spp*.) or black poplar (*Populus nigra*) can act to stabilise emergent bars or islands. In braided rivers, such as the Tagliamento River in Italy, bankfull flows can topple and disperse significant numbers of large riparian trees (Bertoldi et al., 2009), which are then deposited on bars, usually bar heads, (Bertoldi et al., 2013; Gurnell, 2014), often only a very short distance downstream from the recruitment site (Bertoldi et al., 2013). Deposited large wood, including whole trees regenerate freely and quickly establish saplings with root systems in the first year after deposition (Bertoldi et al., 2009; Gurnell, 2014), these newly vegetated patches on bars then increase local hydraulic resistance leading to local patterns of scour and sediment deposition (Bertoldi et al., 2011; Gurnell, 2014), as well as trapping sites for other mobile large wood (Gurnell, 2014). Overall riparian tree cover induces sediment deposition, with root networks also stabilising deposited sediment (Bertoldi et al., 2011), leading to aggradation and the formation of pioneer islands (Bertoldi et al., 2009; Gurnell, 2014), once established these islands then aggrade to floodplain level and are thus less susceptible to inundation and erosion (Bertoldi et al., 2009; Gurnell, 2014).

4.1.2. Ecology

As well as altering the hydrology and morphology of the channel, large wood accumulations also provide an important ecological function by providing and maintaining diverse habitats and sources of food for aquatic biota. Pools, bars, backwaters, exposed riverine sediments and coarse gravel riffles are all habitats which are created in association with large wood and are important for macroinvertebrates and fish, particularly juvenile fish (Chin et al., 2008; Dolloff and Warren, 2003; Gurnell et al., 2002; Millington and Sear, 2007; Mossop and Bradford, 2004). The scour pools associated with large wood and dams provide habitats for juvenile salmonids, both for rearing as well as refuges in the lee of structures during periods of high flow or pollution incidents (Braudrick et al., 1997; Gurnell et al., 2002; Wheaton et al., 2004). Bars deposited in association with large wood accumulations can also form important spawning areas of fine gravel deposits, especially in systems where the general average grain size is too large for salmonid spawning (Montgomery et al., 2003).

Logjams also have a great retentive capacity for organic material such as leaf litter, which will be retained for a sufficient length of time to allow it to break down. This leads to enhanced temporal and spatial availability of particulate organic matter which is an important allochthonous source of carbon for macroinvertebrates (Gurnell et al., 2002; Piégay and Gurnell, 1997).

The increased form drag and associated hydraulic roughness of log-jams, coupled with reduced channel capacity leads to an increase in the frequency and duration of local overbank flooding (Gregory et al., 1985; Jeffries et al., 2003). These events deposit a large volume of fine sediment onto the floodplain instead of such sediment flowing out to estuarine environments (Collins et al., 2002). This creates a complex and diverse mosaic of habitats on the floodplain as erosional and depositional floodplain features form around vegetation and topography (Piégay and Gurnell, 1997; Sear et al., 2010). Such floodplain environments can have high biodiversity, temporary pools for example provide habitat for rare macroinvertebrates (Davis et al., 2007). The frequent overbank events are also important in maintaining the crucial niche habitat of wet woodland, which has unique ecological characteristics found in neither upland or fluvial environments (Braccia and Batzer, 2008).

4.1.3. Research gaps

Although many features associated with large wood and logjams have been documented in the literature the precise characteristics of individual logjams which lead to the creation of additional habitats or geomorphological complexity remain to be illustrated. In general wood remains an incompletely quantified component of river systems (MacVicar and Piégay, 2012). This thesis aims to address this gap by conducting an experimental study of logjam characteristics in association with geomorphological features to attempt to elucidate whether size, location or hydraulics of logjams are more important in creating habitat and geomorphological complexity.

4.2. Life Cycle of Large wood

The amount of large wood in a river system is a function of its supply rate into the channel, called the *loading rate*, and the *depletion rate* at which it is removed from the system (Millington and Sear, 2007). The balance between these two rates can be calculated to give the *wood budget* of the river system, which defines the annual rates of input and removal and the amount of wood remaining in storage within the system. Wood enters the river channel through falling trees, windthrow, fluvial transport from upstream, bank erosion (Fetherston et al., 1995; Gurnell et al., 2002; Swanson and Lienkaemper, 1978) or is removed anthropogenically. Residence or storage times of large wood in a reach vary from minutes to decades. Wood then leaves the system through decay or transport out of the reach (Gurnell et al., 2002) or is removed by anthropogenic management.

Several studies (e.g. Gurnell et al., 2002) have estimated the in-channel wood budget for given river systems or reaches, using a general mass-balance equation. Equation 4.1 shows a modified version of an in-channel wood budget model.

$$\Delta S_c = \left(\frac{(L_i + Q_i + W + A_i) - (L_0 + Q_0 + A_0 + D)}{x}\right)\Delta t \qquad 4.1$$

Where ΔS_c is change in storage in reach length x over time Δt. L_i is lateral recruitment of wood to the channel by bank erosion and transport off the floodplain and L_0 is deposition of wood onto the floodplain during a flood event. Q_i and Q_0 are fluvial transport of wood into and out of the reach. A_i and A_0 are the input or removal of wood by anthropogenic management. W is wood recruitment to the channel through wind-throw, and D is within reach decay of wood (Benda and Sias, 2003).

Measuring input and removal of wood via all pathways in Equation 4.1 is difficult in the field, so typically in-stream wood loads are measured in a reach over time, along with estimates of inputs of new wood to the channel. From these two measurements it is possible to estimate depletion rate. Calculating wood budgets is useful for illustrating how wood storage in the system varies over time, as well as estimating the rate of turnover of pieces of large wood; however the approach is sensitive to yearly fluctuations in rates when measured in the field. One of the key challenges in estimating wood budgets is in obtaining quantitative evidence of wood supply rates, however, recently new techniques have been developed such as using aerial photography (Gurnell et al., 2002; Johnson et al., 2000; Piégay et al., 1999).

There will be a high degree of temporal and spatial variability in loading and depletion rates, as they are dependent on a highly complex web of environmental variables such as; forest stand age, the growth of woodland, death and fall of trees, transport of logs into the river channel, mechanical breakdown, biological decay and transport out of the river system (Bilby, 1984; Fetherston et al., 1995; Gurnell et al., 2002; Lienkaemper and Swanson, 1987; Millington and Sear, 2007; Murphy and Koski, 1989). Of these, tree fall and transport into and out of the river system tend to be most sensitive to extreme environmental events such as wind storms, floods and wild fires (Bendix and Cowell, 2010). Rates of tree growth, tree death and biological breakdown are governed by prevailing environmental conditions and are likely to show less variation over time (Benda and Sias, 2003).

Wood budgets and large wood loadings show a great deal of variability between forest types, with conifer dominated forests in the USA having large wood loading an order of magnitude greater than loadings in deciduous forests (Harmon et al., 1986). Table 4.1 shows a summary of large wood loadings for a range of settings, these figures reflect that conifer forests can have much higher large wood loadings, but also show a great deal of variability within similar forest types and within broad geographical regions, for example grouping all conifer data together this shows a range of 2.5-1700 m^3/ha within USA forests. All values reported in Table 4.1 are by volume as there are limitations in reporting biomass; many studies catalogue piece number and size for large wood but assume a uniform density for all species of wood. Due to wide differences in species density (Zanne et al., 2009) this can lead to large overestimates of load where conifers dominate (>20-40%) and underestimates of load where hardwoods dominate by the same amount (Hedman et al., 1996).

Location	**Dominant tree species/Forest type**	**In-stream wood loading (m^3/ha)**	**Reference**
Worldwide Average	Conifer plantation	240	(Gurnell et al., 2002)
New Hampshire, USA	Conifer, unmanaged	30-80	(Harmon et al., 1986)
North-Western USA	Mixed, unmanaged	812	(Wohl and Jaeger, 2009)
California, USA	Giant Redwood, unmanaged	240-4500	(Harmon et al., 1986; Keller et al., 1995 in Harmon et al., 1986)
USA	Fir, unmanaged	50-216	(Harmon et al., 1986; Lambert et al., 1980)
Tennessee, USA	Norway Spruce, unmanaged	140-220	(Harmon et al., 1986)
Idaho, USA	White Spruce, unmanaged	50-88	(Harmon et al., 1986)
Alaska, USA	Sitka Spruce, unmanaged	55-300	(Swanson et al., 1984 in Harmon et al., 1986)
British Columbia, Canada	Sitka Spruce, unmanaged	320-1700	(Hogan, 1987 in Harmon et al., 1986; Toews and Moore, 1982 in Harmon et al., 1986)
USA	Pine, unmanaged	30-82	(Harmon et al., 1986; Sacket, 1979 in Harmon et al., 1986)**
Idaho, USA	Pine, unmanaged	2.5-120	(Harmon et al., 1986)
Washington, USA	Douglas Fir, unmanaged	308-1421	(Franklin et al., 1981 in Harmon et al., 1986; Harmon et al., 1986; Huff, 1984 in Harmon et al., 1986)**
Oregon, USA	Douglas Fir, unmanaged	45-1200	(Harmon et al., 1986)
California, USA	Douglas Fir, unmanaged	10-1200	(Harmon et al., 1986)

California, USA	Giant Sequoia, unmanaged	555-1000	(Harmon et al., 1986)
USA	Birch, unmanaged	82	(Harmon et al., 1986)
USA	Yellow Poplar, unmanaged	51	(Harmon et al., 1986)
USA	Mixed Oak, unmanaged	46-94	(Harmon et al., 1986)
USA	Chestnut Oak, unmanaged	132	(Harmon et al., 1986)
Tennessee, USA	Mixed hardwood, unmanaged	40-300	(Harmon et al., 1986)
Nisqually River, Washington, USA	Mixed, unmanaged (>80% conifer)	633	(Collins et al., 2002)
Snohomish River, Washington, USA	Immature riparian forest, leveed river	52	(Collins et al., 2002)
Stillaguamish River, Washington, USA	Immature riparian forest, leveed river	24	(Collins et al., 2002)

Table 4.1 – showing in-stream large wood loading for forests of varying types and with varying dominant species across a range of locations. These data show in-stream large wood loadings are highly variable even within the same type of forest and same geographical region.

4.2.1. Conceptual Model

In order to understand the variability of large wood within different river systems it is necessary to examine the processes that affect the loading and depletion rates. Following a review of the literature a conceptual model (Figure 4.2) has been developed to illustrate the important areas in large wood dynamics which may be

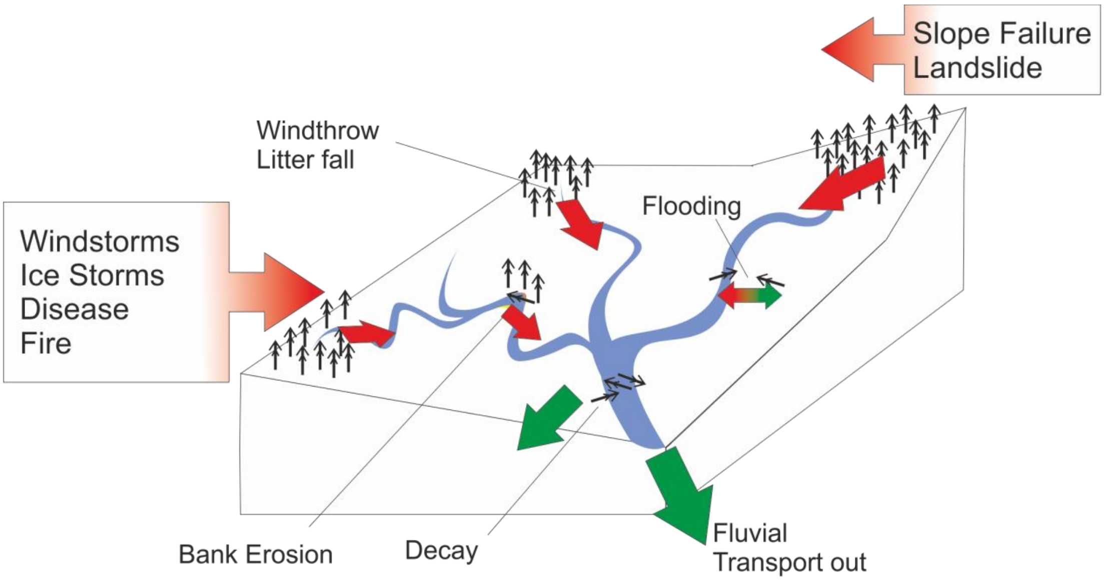


Figure 4.2 Conceptual model showing main factors influencing large wood loads within forested rivers. Arrows highlighted in red are inputs of large wood to the channel and green arrows are wood leaving the channel. Processes in boxed arrows are episodic in nature and may not affect all catchments, unboxed processes are chronic and will occur to a greater or lesser degree in all river systems.

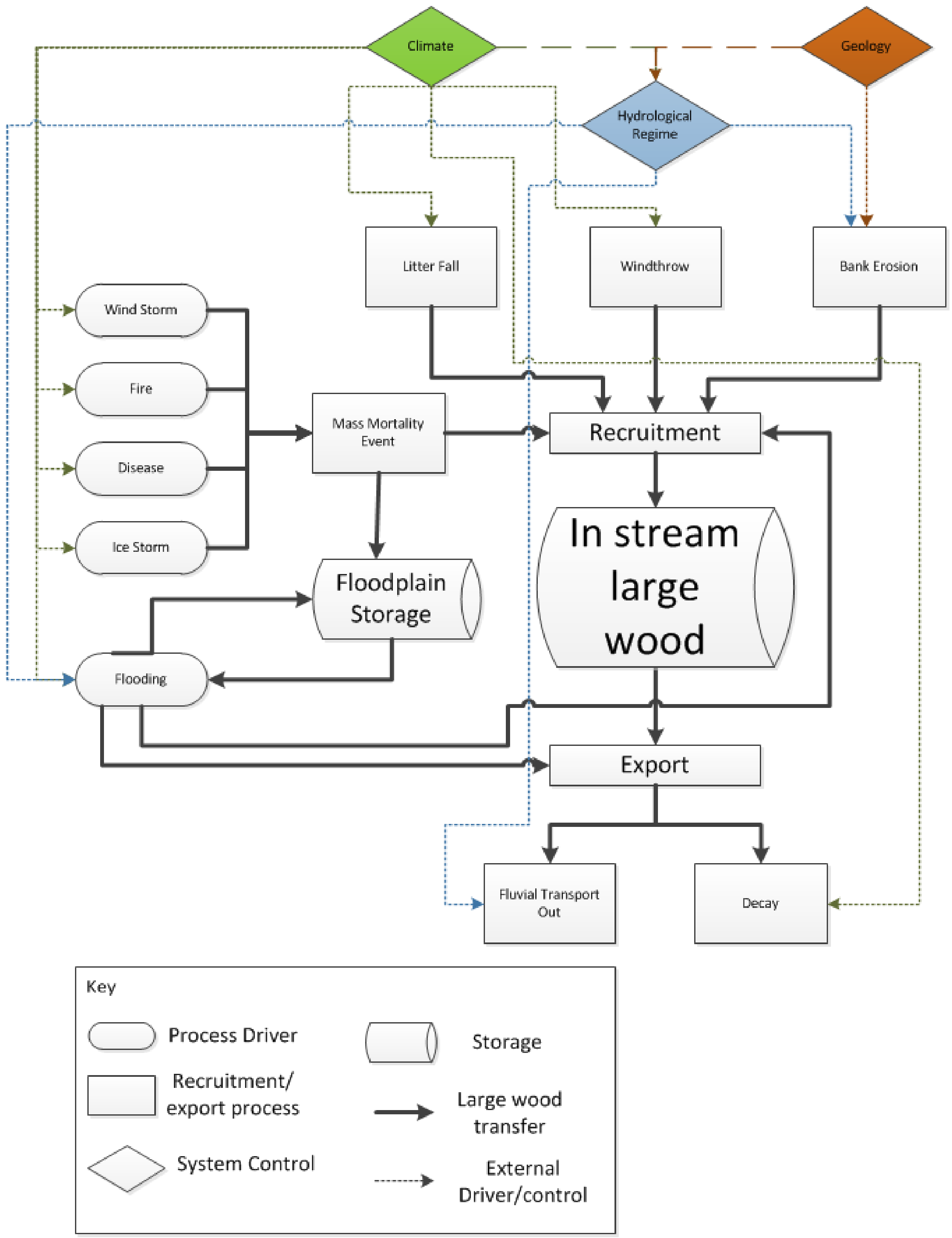


Figure 4.3 – Process flow diagram showing the external controls on large wood recruitment and export within a river systems and the interactions between processes determining large wood storage within the channel and on the floodplain.

affected by river restoration schemes, and in turn may affect flood behaviour in a river catchment. The conceptual model in Figure 4.2 has been extended to a systems and process diagram (Figure 4.3) to illustrate how climate and geology are the controls on the relative importance of each of the input and output (recruitment and export) processes for large wood in the system.

4.2.2. Floodplain Forests

In the absence of management initiatives such as river restoration projects using engineered logjams, a prerequisite for in-channel large wood is the presence of a forested floodplain to act as a source for material.

The residence time and in channel effects of large wood will vary depending on size and species (Gurnell et al., 2002). Factors governing species composition, forest succession and growth of individual trees within a floodplain forest will control the impact of large wood within the river system. The size of floodplain trees controls the maximum size of large wood available for recruitment to the river and thus limits the potential geomorphological effectiveness of large wood in the channel network (Montgomery et al., 2003). Maximum size of trees varies with genus and species and also shows considerable regional variability as does growth rate which is partly controlled by climate (Montgomery, 2003).

Where forests are managed, rotation time of trees can be reduced from centuries to decades compared with natural forest succession, in turn managing forests reduces the maximum size of individual trees in the forest reducing the available size of large wood (Lofroth, 1998). In plantations trees will often be an even-aged cohort, which can reduce the diversity of large wood sizes; furthermore, management thinning in plantations has been shown to reduce the self-thinning in a stand and results in less large wood (Dahlström et al., 2005).

4.2.3. Creation of Dead Wood

The processes which create dead wood are fundamental for large wood to be present in the aquatic environment. The *dead wood cycle* is described as the process of tree death, fall and decay within the forest ecosystem (Lofroth, 1998). This process begins with the germination of a live tree and ends with the incorporation of the material of a dead tree into the soil organic horizon, or the aquatic environment (Montgomery et al., 2003). Although these processes are biotic, they are also affected by biogeomorphic factors and factors outside of the river and floodplain systems (Benda and Sias, 2003; Bendix and Cowell, 2010). The characteristics of large wood in the channel and on the floodplain are dependent on the nature of the forest supplying it as well as the processes supplying the wood to the system (Bendix and Cowell, 2010; Cline et al., 1980; Lofroth, 1998).

Tree death and tree fall are separate processes. Given an absence of environmental forces needed to topple a dead tree, such as strong winds, it can remain standing with the root anchored, potentially for decades (Dahlström et al., 2005). Standing dead trees are referred to as snags and are commonly defined as a dead tree over 10cm diameter at breast height and over 2m tall (Cline et al., 1980; Dahlström, 2005). The density and production of snags decreases with the age of a stand of trees, but the mean size of snags increases with stand age (Cline et al., 1980; Franklin et al., 1987; Raphael and Morrison, 1987).

The causes of tree mortality and large wood creation are varied, often showing species specific patterns (Bendix and Cowell, 2010; Lofroth, 1998) and can be either episodic or chronic. Chronic sources include individual mortality and treefall (Martin and Grotefendt, 2006), as well as litterfall and windthrow (Brown, 1997). These chronic sources are the primary large wood recruitment pathways in temperate riparian forests, and have a greater prominence in older, mature forests (Lienkaemper and Swanson, 1987). Episodic sources include large floods (Pettit and Naiman, 2005), mass wasting (Benda and Sias, 2003), debris torrents (Young et al., 2006), wind storms (Bahuguna et al., 2010), ice storms (Kraft et al., 2002) and large scale mortalities, such as insect or fungal epidemics and fire (Bendix and Cowell, 2010; Nakamura et al., 2000).

Some episodic events are capable of delivering large wood to the river system from more distal sources including; wood torrents along tributaries (Young et al., 2006), floatation of wood from the more distal floodplain during overbank events, landslides and avalanches (Lofroth, 1998). Chronic sources can be observed in almost all forested systems, whereas episodic sources are dependent on environment and climate. In most systems, at least some large wood enters the forest ecosystem through wind throw (death and fall of trees), or through death and eventual fall of standing dead trees (Nakamura et al., 2000).

All of these processes combine to form a spatially and temporally complex disturbance cascade, delivering, storing and remobilising wood within the system (Gurnell et al., 2002). There will be a substantial spatial variation in the dominant processes throughout a drainage network and between networks as the geomorphology and environment favours or inhibits the various processes (Montgomery et al., 2003). In large river networks headwater channels will have wood delivery to the channel dominated by chronic inputs of wood through tree mortality and fall, windthrown wood and through landslips, whereas in lowland channels lateral channel migration will dominate wood recruitment through bank erosion (Bormann and Likens, 1979).

A new stand of trees can be created as a result of a stand replacing disturbance (either natural or anthropogenic); when this occurs saplings of pioneer species will colonise the area and the new trees will persist as an even-aged stand for up to a hundred years (Warren et al., 2009). During this period the creation of dead wood will be dominated by self-thinning; density dependent mortality of the trees due to competition between individuals, and will deliver relatively small wood to the system, but at a potentially high load in terms of individual pieces (Frelich and Graumlich, 1994; Lorimer and White, 2003). As a stand ages forest succession will take place and other species become established through gap phase regeneration following individual tree mortality, episodic factors will become increasingly important in the delivery of dead wood (Bendix and Cowell, 2010; Chen et al., 2006; Sear et al., 2010; Young et al., 2006), including fire (Warren et al., 2009) and tropical cyclones (Frelich and Graumlich, 1994; Ziegler, 2002). Hedman et al (1996) showed for rivers in the Southern Appalachian mountains in the USA, that stand age and succession rather than channel size were the primary controls on wood loads,

Frequent small-scale disturbance events such as floods, wind storms and minor landslips dominate forest dynamics in old-growth and secondary growth forests in the North Eastern United States, Upper Mid-West United States and UK (Lorimer, 1977). Intermediate, partial disturbance events which deliver volumes of dead wood, without causing complete stand replacement, such as ice storms and microburst wind disturbance associated with thunderstorms are relatively common in the North-Eastern United States (Keller and Swanson, 1979)

Where a stand is managed the effects of logging can be seen on large wood creation; a logging operation will typically result in a newly created stand of even-aged trees post-logging, with the effects mentioned above. During and immediately after logging the large wood remnants dominate in-channel wood loadings and lead to higher in-channel loads than are seen in comparable old-growth forests (Gomi et al., 2001). Often a buffer strip of riparian forest is left after logging operations, but it has been shown in the short term that the quantities of wind thrown large wood to the river can double from the outer zone of such buffer strips, in turn reducing potential future recruitment due to stand thinning (Martin and Grotefendt, 2006).

4.2.4. Recruitment to the channel

The process of a piece of dead wood entering the aquatic environment of the river from the terrestrial environment is referred to as *recruitment* (Beechie et al., 2000; Bragg, 2000; Downs and Simon, 2001; Fetherston et al., 1995; Martin and Grotefendt, 2006; May and Gresswell, 2003). If a tree, or part of a tree on the floodplain falls to the ground it will either fall directly into the channel or remain in a riparian or distal part of the floodplain. Subsequently whole dead trees or other dead wood on the floodplain can be recruited into the active river channel by processes such as channel migration or debris torrents.

Direct input of large wood to the channel through wind throw is largely stochastic in natural/semi-natural systems and will occur to a greater or lesser degree in all river channels (Downs and Simon, 2001; May and Gresswell, 2003; Warren et al., 2009). In river systems with minimal lateral channel migration, such as mountain headwater streams, it can be the principal source of wood recruitment to the channel (Gurnell et al., 2002). In headwater streams with steep hillslopes it has been shown that windthrown trees preferentially fall towards the channel compared to moderately steep slopes in which there is no preferential fall direction (Sobota et al., 2006), potentially resulting in higher in-stream wood loads for channels with steep hillslopes.

The floodplain acts as both an important source for large wood to the channel, as well as temporary storage or a sink. Substantial amounts of wood can lie immobile on the flood plain, either as a result of falling directly onto it, or as a result of being rafted onto the floodplain by over bank flows and then deposited (Gurnell et al., 2002), this wood can be subsequently mobilised by overbank flows and recruited into the river (Gurnell et al., 2002; Millington and Sear, 2007). In systems with high levels of lateral channel migration this stored wood can be recruited to the channel as the banks and floodplain are eroded, with the same process also undercutting living trees causing them to fall into the channel (Montgomery et al., 2003). Although wood can be recruited from distal sources, such as hillslopes, the majority of large wood is recruited from only a distance 1-2 times the average tree height away from the active channel (Bragg, 2000).

Large wood on the floodplain can become buried by progressive over bank sedimentation, as well as becoming submerged under organic debris from the forest (Gurnell et al., 2002; Montgomery et al., 2003). In this way wood can be stored in the flood plain for decades in an anaerobic environment, slowing decay, and can be remobilised by channel migration and erosion of the flood plain years later (Gurnell et al., 2002; Latterell and Naiman, 2007). Hyatt & Naimen (2001) showed by dendochronology of large wood in the Queets River, in the Pacific NW of the United States, that some pieces were well in excess of 1000 years old, having been previously buried in the flood plain and re-excavated by bank erosion relatively recently.

The type of system will determine the relative importance of different recruitment processes for wood delivery. Recruitment can be a function of the general rate of lateral erosion (Gurnell et al., 2002). It has been postulated in lowland rivers there will be less wood supplied than either piedmont meandering or piedmont braided, due to the lower levels of bank erosion (Downs and Simon, 2001).

Braided rivers typically have higher rates of bank erosion than meandering rivers, and channel migration and avulsions are generally restricted to an active channel belt or braidplain (Gurnell et al., 2002; Jones et al., 2011). Within the braidplain and a buffer zone along it, trees will rarely attain maturity following a previous disturbance before the channel migrates again and undercuts, or uproots them (Gurnell et al., 2002). Thus, although the channel is more active the erosion will be recruiting tree specimens that are smaller and less geomorphologically significant than meandering rivers (Downs and Simon, 2001) and lack sufficient large wood to create stable jams (Beechie et al., 2006). Figure 4.4 shows how different logjam types form in braided channels compared to meandering channels. However in some braided river systems such as the Tagliamento River in Italy where the dominant woody vegetation is willow and black poplar, trees can rapidly colonise emergent bars and stabilise islands by growing new trees from deposited wood pieces (Bertoldi et al., 2011; Bertoldi et al., 2009). In such a system the large wood is acting as a living 'ecosystem engineer' and affecting the processes of channel migration and island formation (Gurnell, 2014).

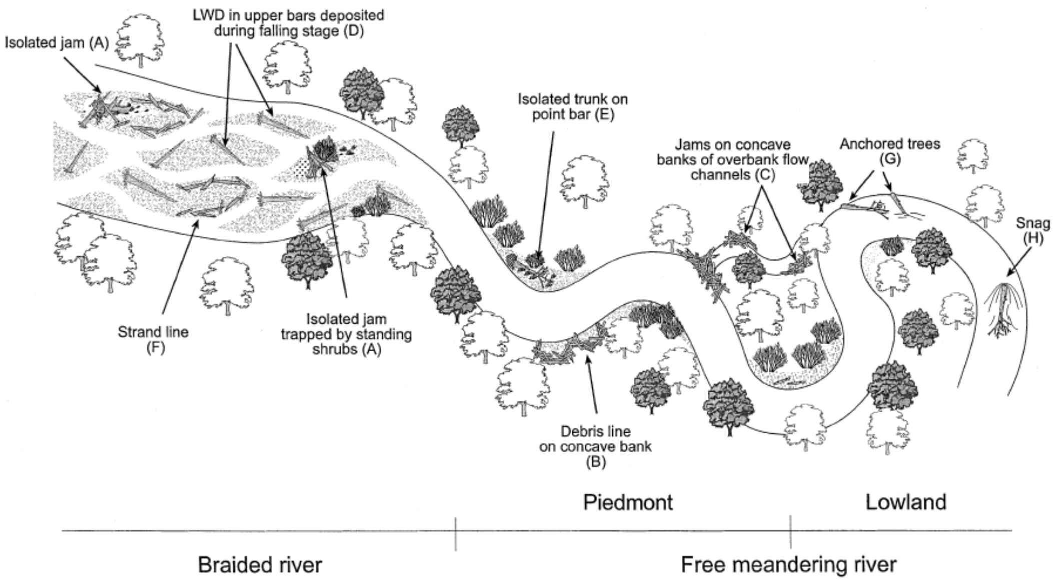


Figure 4.4 – the typology of logjam formation in different large river types, showing braided rivers are characterised by isolated logjams typically deposited during falling flood stages, whereas in meandering rivers logjams typically form at trapping points (from Gurnell et al., 2002. Their Figure 5)

For both meandering and braided rivers erosion and recruitment can be due to natural processes, but it is important to note that these can be accelerated by upstream flow adjustment or alterations typically due to urbanisation and channelisation (Downs and Simon, 2001). It is also possible for a river to be dynamically unstable and show increasing channel width due to either an altered hydrological regime or due to rapid base level change (Downs and Simon, 2001). This can occur due to climatic variations, but is more likely to be seen upstream or downstream of anthropogenic disturbances which have altered the sediment regime and/or the hydrology (Downs and Simon, 2001). The most dramatic example of which is upstream of straightened channels, which can lead to a zone of upstream migrating knick points causing rapid base-level lowering and channel incision and widening (Downs and Simon, 2001). The accelerated erosion rates will be accompanied by an increase in large wood recruitment to the channel.

4.2.5. Decay Rates

As well as mobile wood transported out of a reach, wood is removed from a river system as a result of organic decay and mechanical breakdown (Warren et al., 2009). Decay rates of wood are important as they govern the long term geomorphic impact of large wood on a fluvial system, where slower decaying species can have an impact over a longer period (Lofroth, 1998).

Decay rates of large wood will vary substantially between different systems and will depend on environmental factors such as microclimate and sediment transport regime, as well as factors such as the size of the piece of wood and the species (Lofroth, 1998; Mattson et al., 1987). Decay rates for wood in the aquatic environment are sparse; Table 4.2 shows decay rates for upland terrestrial forest plots which illustrate variability in annual decay rates across species, dead wood type and geographical region.

Table 4.2 shows rate of general decay in a terrestrial environment between species is highly variable, with conifers decaying slower than hardwoods, and supports other findings in the aquatic environment that the rate of decay between species can vary by as much as a factor of ten (Collins et al., 2002; Hyatt and Naiman, 2001; Warren et al., 2009). In general conifers also decay slower in an aquatic environment than most hardwoods (Warren et al., 2009), Oak being one of a few exceptions as a slow decaying hardwood species (Harmon et al., 1986). Although differences in species makes comparisons between geographical regions difficult Table 4.2 does show species within the oak genus decay at broadly similar rates in Cumbria, UK as they do in the Appalachian Mountains, USA. Figure 4.5 shows a conceptual model of branch wood decay in upland terrestrial plots from Swift et al., (1976) based on field data from Cumbria, UK, this shows that decay proceeds at different rates based on its type, from deadwood in the canopy through to litter.

Tree Species	Type	Decay Rate (% yr^{-1})	Location	Climate	Reference
Oak	Canopy	8.4	Cumbria, UK	Average temp 9oC, annual precipitation 3200mm/yr	(Swift et al., 1976)
Oak	Litter	6.5	Cumbria, UK	Average temp 9oC, annual precipitation 3200mm/yr	(Swift et al., 1976)
Ash	Canopy	1.9	Cumbria, UK	Average temp 9oC, annual precipitation 3200mm/yr	(Swift et al., 1976)
Ash	Litter	15.2	Cumbria, UK	Average temp 9oC, annual precipitation 3200mm/yr	(Swift et al., 1976)
Birch	Canopy	12.2	Cumbria, UK	Average temp 9oC, annual precipitation 3200mm/yr	(Swift et al., 1976)
Birch	Litter	13.8	Cumbria, UK	Average temp 9oC, annual precipitation 3200mm/yr	(Swift et al., 1976)
Hazel	Canopy	9.3	Cumbria, UK	Average temp 9oC, annual precipitation 3200mm/yr	(Swift et al., 1976)
Hazel	Litter	24.4	Cumbria, UK	Average temp 9oC, annual precipitation 3200mm/yr	(Swift et al., 1976)
Red Maple	Snag	6.8	Great Smokey Mountain NP, Southern Appalachians, USA	Average temp 13oC, annual precipitation 1470mm/yr	(Harmon, 1982)
Flowering dogwood	Snag	4.0	Great Smokey Mountain NP, Southern Appalachians, USA	Average temp 13oC, annual precipitation 1470mm/yr	(Harmon, 1982)
Hickory	Snag	6.0	Great Smokey Mountain NP, Southern Appalachians, USA	Average temp 13oC, annual precipitation 1470mm/yr	(Harmon, 1982)
Black Tupelo	Snag	10.1	Great Smokey Mountain NP, Southern Appalachians, USA	Average temp 13oC, annual precipitation 1470mm/yr	(Harmon, 1982)
Sourwood	Snag	3.1	Great Smokey Mountain NP, Southern Appalachians, USA	Average temp 13oC, annual precipitation 1470mm/yr	(Harmon, 1982)
Pitch Pine	Snag	5.7	Great Smokey Mountain NP, Southern Appalachians, USA	Average temp 13oC, annual precipitation 1470mm/yr	(Harmon, 1982)
Virginia Pine	Snag	3.6	Great Smokey Mountain NP, Southern Appalachians, USA	Average temp 13oC, annual precipitation 1470mm/yr	(Harmon, 1982)

Scarlet Oak	Snag	5.7	Great Smokey Mountain NP, Southern Appalachians, USA	Average temp 13oC, annual precipitation 1470mm/yr	(Harmon, 1982)
Chestnut Oak	Snag	11.0	Great Smokey Mountain NP, Southern Appalachians, USA	Average temp 13oC, annual precipitation 1470mm/yr	(Harmon, 1982)
Eastern Hemlock	Snag	4.6	Great Smokey Mountain NP, Southern Appalachians, USA	Average temp 13oC, annual precipitation 1470mm/yr	(Harmon, 1982)
Douglas Fir	Snag	3.1	Cascade Range, Oregon, USA	Montane/Alpine climate	(Graham, 1981)
Douglas Fir	Litter	1.2	Cascade Range, Oregon, USA	Montane/Alpine climate	(Graham, 1981)
Western Hemlock	Snag	8.6	Cascade Range, Oregon, USA	Montane/Alpine climate	(Graham, 1981)
Western Hemlock	Litter	2.1	Cascade Range, Oregon, USA	Montane/Alpine climate	(Graham, 1981)
Southern Beech	Litter	1-37	Tierra Del Fuego, Argentina	Oceanic, average temp 5oC, annual precipitation 3000mm/yr	(Frangi et al., 1997)

Table 4.2 – Annual decay rates by tree species, dead wood type and geographic region. This shows annual decay has a great deal of variability between species with hardwood species tending to decay faster than conifers, and with litter tending to decay faster than canopy dead wood or snags. Variability between species makes inter-regional comparisons difficult; however for oak species there are broadly similar decay rates observed for Cumbria in the UK and Southern Appalachians in the USA.

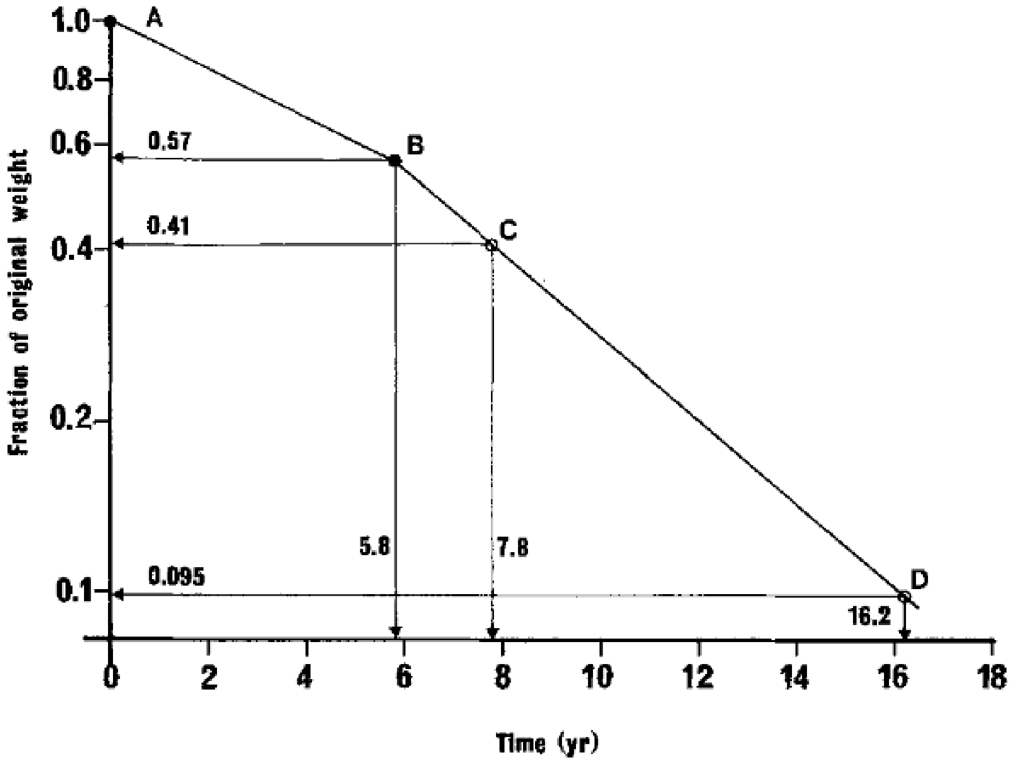


Figure 4.5 – Conceptual model showing pattern of decomposition of branch-wood from Swift et al., (1976). Decay curves are based on regression equations for total branch-wood from field data from Cumbria, UK. Four identifiable stages of decomposition are indicated on the curves; A branch death; B branch fall; C animal invasion; D arbitrary termination point (equivalent to RD=0.05 $g.cm^{-3}$) where the wood is assumed to be incorporated into the soil horizon or litter layer. From Swift et al., (1976), their Figure 2.

It could be expected that large wood delivered to streams via windthrow may already be in an advanced state of decay Swift et al. (1976) found around 40% of total decay for dead wood branches occurs in the canopy before the branch reaches the forest floor in upland plots in Cumbria. As decay is most active on the surface of large wood (Dahlström et al., 2005), larger pieces could be expected to decay slower due to their lower surface area:volume ratio compared with smaller pieces (Hyatt and Naiman, 2001), for upland forest decay in Tierra Del Fuego, Argentina the average decay rate for Southern Beech (Nortofagus Pumilio) was found to be ~1% yr^{-1}, whereas for small <1cm branches this rate was 37% yr^{-1}(Frangi et al., 1997).

Burial in sediments, and thus the sediment transport regime, is an important influence on rates of wood decay (Hyatt and Naiman, 2001). It has been found that wood buried in an anaerobic environment in floodplain sediments can survive relatively decay free for centuries, before being exhumed by floodplain erosion and reintroduced to the active channel (Gurnell et al., 2002; Warren et al., 2009). Hyatt & Naiman (2001) found that 2.7% of wood in the Queets River was in excess of one thousand years old as a result of this process.

The hydrological regime is important in controlling wood breakdown rates; decay is highly dependent on the cycle of wetting and drying, where exposure to frequent wetting and drying will increase decay rates (Gurnell et al., 2002), also the frequency of high flows will govern how often a piece of wood is exposed to forces which can lead to mechanical breakdown (Braudrick et al., 1997). During a large flood there will be a significant increase in the amount of stream power and forces acting on large wood within the channel (Shields Jr and Alonso, 2012), this can lead to increased log-to-log interactions (Warren et al., 2009), as well as interactions between pieces of large wood and in-stream obstructions as well as the channel margins (Warren et al., 2009). Collisions between large wood and other obstacles can result in mechanical abrasion and breakage of large wood pieces and will be an important component of breaking down wood pieces in the system (Benda and Sias, 2003; Hyatt and Naiman, 2001). Dahlström et al (2005) found the majority of large wood in a boreal forest stream had broken or eroded ends indicating extensive mechanical abrasion of wood.

The resistance of large wood to these forces will depend on species specific factors as well as environmental factors (Record, 1914). In general the strength of wood increases with dryness (Record, 1914). The weight of dry wood per unit volume, which is species dependant, is directly proportional to increases in crushing strength parallel to the grain, fibre stress at elastic limit in bending and shearing strength along the grain (Hyatt and Naiman, 2001; Keller and Tally, 1982; Record, 1914).

Differences in decay rates between species was attributed by Dahlström et al (2005) to differences in living and dead wood species composition in a Boreal forest stream in Sweden where the riparian forest was dominated by Birch (Betula spp.), but the in-channel large wood was dominated by Pine (Pinus Sylvestris), this was attributed to the rapid decomposition rates of birch compared to pine in the channel as well as different levels of mortality.

4.2.6. Large wood Residence Time

Studies using tree ring dating and C^{14} dating have shown that stable in-stream large wood can have residence times in excess of 100 years (Collins et al., 2002; Dahlström et al., 2005; Hyatt and Naiman, 2001). However Hyatt & Naiman (2001) found through a detailed aging study of large wood in the Queets River in the US Pacific North-West that most large wood (conifers and hardwoods) would decay or move out of the system within 50 years, but that burial of wood in flood plain sediments would dramatically increase the residence time of large wood to as much as several hundred years (Collins et al., 2002; Gurnell et al., 2002).

The sediment transport regime is an important factor governing the residence time of large wood. It has two main impacts, the first is the deposition of sediment around an immobile piece of large wood which can stabilise it and mean it is likely to remain in place for a longer time period (Dahlström et al., 2005), furthermore the decay rate of wood is slowed when buried in sediment (Collins et al., 2002; Gurnell et al., 2002; Hyatt and Naiman, 2001; Montgomery et al., 2003).

There still remain relatively few studies on the dynamics and residence times of large wood in river systems over long periods; most information comes from studies of dendrochonology of wood in a system (Dahlström et al., 2005; Powell et al., 2009), or from wood budget calculations (Wohl and Goode, 2008). It remains difficult to calculate accurate and meaningful residence times however. In order to calculate residence times a steady state has to be assumed, which is unlikely to be the case in any rivers impacted by anthropogenic disturbances. In rivers which have experienced anthropogenic disturbance there may have been changes in the input rates, species composition and size of wood over time (Braudrick et al., 1997). The behaviour of individual pieces of large wood within a system over time remains to be fully illustrated (Powell et al., 2009; Wohl and Goode, 2008).

Within this thesis I aim to quantify large wood loadings in a UK forest stream through a survey, and to also track the mobility of large wood pieces in reaches of varying morphology and floodplain connectivity in order to help address these gaps in the literature.

4.3. Mobility of large wood

Large wood in river channels can be highly mobile (Warren and Kraft, 2008). The mobility of an individual piece of large wood is governed by a balance of forces. Stream power & water depth act to move the piece through drag forces and floatation respectively (Bocchiola et al., 2006a; Braudrick and Grant, 2001; Shields Jr and Alonso, 2012). These entrainment forces are balanced against resistance to movement generated by the mass of the piece and the interaction of the piece of large wood and the geomorphology of the river (Gurnell et al., 2002; Shields Jr and Alonso, 2012). Due to these controls mobility of wood is strongly related to discharge (Berg et al., 1998; Daniels, 2006), with the magnitude and sequence of discharge events a partial control on wood mobility and dynamics (Haga et al., 2002). If large wood is present in a river channel it will typically only be mobile during flood events (Bocchiola et al., 2006a; Bocchiola et al., 2006b; Gurnell et al., 2002) and within a flume wood has been shown to be resistant to being mobilised in a simulated 1 in 10 year flood event (Braudrick et al., 1997).

Wood size is the dominant control on mobility (Daniels, 2006; Wohl and Goode, 2008). Two key relationships in wood mobility are the ratios between piece diameter:flow depth and piece length:channel width (Braudrick et al., 1997; Gurnell et al., 2002). There will be a critical threshold in the piece diameter:flow depth ratio above which the piece will not move and as flow depth increases so the probability of entrainment increases (Braudrick et al., 1997; Shields Jr and Alonso, 2012). The piece length:channel width ratio describes the likelihood of the piece becoming trapped within the channel or against an obstruction; with a greater length trapping becomes more likely (Bilby, 1984; Bocchiola et al., 2006b; Lienkaemper and Swanson, 1987; Millington and Sear, 2007). Larger sized pieces also have increasing mass with increasing length which acts to resist mobilisation by buoyancy and drag forces. As well as likelihood of entrainment, the size of wood also partly controls transport distance with smaller wood found to move further than longer pieces (Bilby, 1984; Daniels, 2006; Lienkaemper and Swanson, 1987; Millington and Sear, 2007).

Once a piece of wood is mobilised it will continue to move downstream until it is deposited in a stable location (Bilby, 1984; Braudrick et al., 1997; Ehrman and Lamberti, 1992; Haga et al., 2002; Millington and Sear, 2007). In large rivers where the channel width exceeds the longest piece length this can be of the order of kilometres (Bertoldi et al., 2013; Haga et al., 2002; Latterell and Naiman, 2007). Locations of deposition can be a channel feature representing a change in the hydraulic geometry of the river, such as shallow riffles, narrow sections of channel, or a more sinuous reach, bar heads (Bertoldi et al., 2013; Gurnell, 2014) as well as obstructions in the channel, such as boulders (Faustini and Jones, 2003), or other pieces of wood (Gurnell et al., 2002; Millington and Sear, 2007). The relationships between form and mobility have been described, however the precise nature of interactions between obstructions such as boulders and wood on the levels of wood retention remain to be fully illustrated and quantified (Gurnell et al., 2002).

Deposited wood often accumulates into depositional features such as log jams, whose abundance and distribution will be a function of the transport mechanism. Although there have been a number of studies describing logjam features and categorising their influence on hydraulics (e.g. Gregory et al., 1985; Wallerstein and Thorne, 1997), there is not a systematic method for physical characterisation and classification of logjams across all fluvial systems (Braudrick et al., 1997). As well as being a product of large wood transport, logjams are also a control on mobility, where if they remain stable they can trap other mobile pieces of large wood (Daniels, 2006; Gurnell and Sweet, 1998). Wood within logjams is more likely to be stable than randomly orientated pieces of wood in the channel (Gurnell et al., 1995; Lienkaemper and Swanson, 1987). The density and organisation of in-stream wood is an important control on the transport length of mobile wood (Ehrman and Lamberti, 1992; Gurnell et al., 1995). Studies have shown that in channels which are either too large or with insufficient wood loadings for logjams to form, in the absence of major large wood structures individual pieces of large wood can be highly mobile and capable of transport lengths of hundreds of metres (Bilby, 1984; Daniels, 2006; Gurnell and Sweet, 1998; Millington and Sear, 2007). Furthermore during floods wood can be "blown out" of one logjam and subsequently trapped by the next logjam downstream (Thomas and Nisbet, 2012).

Although there have been short-term flume and field based studies of large wood mobility within flood events there is less field data on the long term mobility of wood pieces within channels (Wohl et al., 2010). Direct field measurements of wood transport remain relatively rare in all river types (Bertoldi et al., 2013). Almost all field studies to date have been in the Pacific Northwest of the USA and there is a great need for more field studies in diverse locations (Latterell and Naiman, 2007; Wohl and Cadol, 2011; Wohl et al., 2010). This thesis aims to address this gap by conducting a 3 year study tracking the mobility of tagged large wood pieces, with a study set up such that measurement can continue to be taken after this period to generate a true long-term data set of wood mobility in UK forest streams.

4.3.1. Wood Characteristics

The absolute size of a piece of large wood is not the only characteristic that governs mobility. The orientation of wood relative to flow direction is important for drag forces acting on it, with wood orientated normal to flow experiencing highest drag forces (Shields Jr and Alonso, 2012). The presence of a root wad on a piece of large woody debris has been found to be very important in stabilising it and inhibiting movement, increasing the chances of it snagging on other wood (both living and dead) as well as geomorphic features and obstructions in the channel (Collins et al., 2012; Gurnell et al., 2002).

The general regularity of the wood shape is important in how likely a piece of wood is to become trapped, with irregularly shaped wood being more likely to become trapped than a cylindrical shaped piece of a similar size (Gurnell et al., 2002). Branched logs experience higher drag forces at higher velocities than single trunks (Hygelund and Manga, 2003; Shields Jr and Alonso, 2012). Variations in likelihood of trapping and entrainment with irregularity of shape are important in terms of the species composition of riparian forests. Conifers tend to have more cylindrical trunks with a greater mass of their total wood in the main trunk when compared to broadleaf species

(Gurnell et al., 2002). Furthermore, due to this greater central trunk mass, conifers tend to shatter and shed branches when they topple, whereas broadleaf trees with a more complex structure can be cushioned by this open form when toppling (Gurnell et al., 2002). These factors combine and tend to lead to conifers producing more cylindrical pieces, whereas broadleaf tree species yield more complex and open structured large wood (Record, 1914). This would indicate a potential for large wood from conifers to be more mobile than that of broadleaf species.

Density is another important factor governing large wood mobility. Density is affected by species, age of wood as well as stage of decomposition and water-logging (Gurnell et al., 2002). Density regulates the propensity for the piece of wood to float and thus governs the likelihood that the wood will be entrained in the flow, with denser wood being more likely to sink and less likely to move (Record, 1914). The weight of actual wood material, defined as that making up the cell material, is broadly the same across species and is about one and a half times as heavy as water (Record, 1914), however the density between different species can vary widely, due different amounts of wood material per unit volume (Piégay and Gurnell, 1997). Most species have a specific gravity lower than water and so will float, although a few, mostly non-native species (e.g. teak, eucalyptus) have a dry density greater than water (Zanne et al., 2009). Generally conifers tend to be less dense than broadleaf species when the oven dry mass to fresh wood volume is compared (Braudrick et al., 1997). A mature, unmanaged riparian forest is likely to have a mix of different wood densities, due to differing levels of decay and water-logging in dead wood on the forest floor as well as mixed species composition, thus making it hard to predict density of large wood pieces *a priori*.

4.3.2. Types of Transport

Braudrick et al (1997) used a flume experiment to investigate and describe the different mechanisms of large wood transport in rivers given different morphology and wood loading rates. They found individual logs in a stream can move by rolling, sliding or floating and found that depth of flow was the primary factor determining which type of movement dominated, with flotation being the primary method of movement where flow was deep enough (Braudrick et al., 1997; Haga et al., 2002). Sliding was observed to occur as a debris flow, where a large mass of logs moves together and is capable of remobilising previously deposited logs and pushing them downstream (Braudrick et al., 1997). A further study by Bocchiola et al (2006) showed initial log mobility is either by rolling or sliding where type of mobility is a function of log orientation to flow and flow depth.

Interactions between logs were found to be important; as well as being pushed off bars and struck by moving logs, previously deposited logs are remobilised when other logs redirected flow and eroded bars (Braudrick et al., 1997). Remobilisation often occurred as a result of a deposited log pivoting or rolling into deeper water until it floated (Braudrick et al., 1997).

Where the input rate of large wood to a stream is low then transport can be expected to be un-congested, where there is little or no contact between individual logs. However where the input rate is higher transport can become congested, where wood is moving in a pulse of pieces of wood moving as a mass occupying the channel width (Collins et al., 2002; Gurnell et al., 2002; Piegay, 2003). Where wood is arranged into logjams these will be subjected to high drag (Shields Jr and Alonso, 2012) and hydrostatic forces (Wohl, 2011) during flood events and may cause the structure to partly break up and increase susceptibility to the structure failing (Shields Jr and Alonso, 2012) with racked pieces mobilised as congested transport.

4.3.3. River Size

The majority of work done on establishing the influence of wood on channel form has been done in small forest streams, as a result less is known about wood in large rivers (Braudrick et al., 1997; Daniels, 2006; Millington and Sear, 2007). The mobility and transport distance for any given size of wood will tend to increase with increasing channel size (Gurnell, 2003; Gurnell et al., 2002; Piegay, 2003). However in large rivers the channel is capable of mobilising all wood delivered to it, and thus ratios of channel dimensions relative to the size of individual pieces of wood delivered to it are no longer the primary control on mobility (Gurnell et al., 2002). The greater size of the river means that the controls are instead a combination of the hydrology, wood density/buoyancy, sediment transport regime, and general geomorphic complexity, such as the presence of bars and planform type (Piégay and Gurnell, 1997).

Most river networks will exhibit a greater discharge and larger channel as stream order increases; this means that most rivers will show a progressive downstream change to more organised accumulations of wood and in the largest rivers a possible absence of accumulations in the highest stream orders. Small first and second order headwater channels will be transport limited such that mobile wood is rare and large logjams rarely form (Marcus et al., 2002), through third and fourth order channels where wood cycles down the channel (Marcus et al., 2002), to a larger channel where all wood moves and tends to organise into logjams at depositional locations, for example on meander bends or emergent bars (Collins et al., 2002). As a result of increasing mobility in larger rivers the distribution of wood tends to be random in first to third order streams and tends to more organised accumulations in larger channels (Bisson et al., 1987; Lienkaemper and Swanson, 1987; Máčka and Krejčí, 2010; Robison and Beschta, 1990).

Studies indicate there are lower wood loads in larger rivers than smaller ones (Lienkaemper and Swanson, 1987). However Collins et al (2002) argue that the general assumption wood has less geomorphic effect in larger rivers and is more easily transported is actually a reflection of the greater anthropogenic influence on lowland rivers. Anthropogenic controls have been suggested to reduce large wood loads by 50% (Krejčí and Máčka, 2012) and studies of pristine rivers and historical evidence shows the dominant impact of large raft-jams on large low gradient rivers in the absence of human intervention (Church, 1992). Francis et al (2008) suggested many large European rivers would have been island braided during much of the Holocene and following human induced deforestation shifted to a meandering planform in the absence of willow and poplar large wood to drive channel dynamics, such as are currently observed on the Tagliamento River in Italy (Bertoldi et al., 2013; Gurnell, 2014).

4.3.4. Attempts at classifying transport relationships

Gurnell (2003) proposed channel size should be categorised according to its relationship to dominant wood size, such is the importance of wood in ecological and geomorphological terms. The Gurnell (2003) categorisation shares similarities with earlier work classifying channels based on the size of their bed material (Church, 1992; Gurnell, 2003). Gurnell (2003) proposes that small channels are those where the width is less than the median wood piece length, medium channels have a width greater than all but the upper quartile of wood length and large channels are those where the width is greater than the length of all pieces of wood delivered to it (Gurnell, 2003; Gurnell et al., 2002; Piégay and Gurnell, 1997).

This method is useful to compare rivers in broad terms and it does give indications of the trapping potential of a river reach. For example a small channel is likely to see a lot of wood jammed in the channel in random orientations where it has fallen into the channel due to low mobility, whereas a medium channel could be expected to mobilise more wood and for this to accumulate behind larger immobile pieces of wood forming logjams (Gurnell et al., 2002). However the system is simplistic as it makes no concessions to general geomorphic complexity or sediment transport regime, both of which have been shown to be important in trapping and stabilising otherwise mobile large wood (Wohl and Goode, 2008).

It has been argued that many of the mechanisms of large wood transport are analogous to sediment transport with both involving supply, entrainment, transport and deposition (Braudrick et al., 1997). However there are some important differences in the physics involved, mainly due to the shape, size and density differences between large wood and sediment particles. Large wood is commonly rod-shaped and therefore the forces acting on the cross-sectional area will be greater than for broadly spherical sedimentary particles (Braudrick et al., 1997). Furthermore the greater size of large wood and its elongated shape means that it has a much higher probability of encountering the boundaries of the channel (Braudrick et al., 1997).

Wood is also generally less dense than water and so will tend to float, but more importantly it will tend to continue to move once entrained until it encounters a narrower/shallower section of channel or an obstruction (Bilby, 1984; Braudrick et al., 1997; Millington and Sear, 2007). With sedimentary particles the shear stress acting on the particle is important for entrainment and spatial and temporal variability in shear stress means that transport length for sediments is fairly short (Gurnell et al., 2002). As yet there are not enough studies on large wood mobility in relation to hydraulic regime to develop methods of classification for wood or streams which are transferable across systems (Montgomery, 2003), so classifications in the literature are either highly generalised or are suitable only for the study site they were derived from.

4.3.5. Flood Events

The importance of overbank flood events in recruiting large wood to the channel has already been mentioned in section 4.2.4. However flood events are also the primary means of moving and redistributing large wood, with major flood events likely to lead to a significant redistribution of large wood within a river channel and its riparian zone, particularly in small and medium sized rivers (Braudrick et al., 1997). Braudrick et al (1997) looked at previous studies on in channel large wood movement and concluded that where no large magnitude flood events occurred (greater than 1-in-10 year events) the wood movement observed was the result of the break-up and decay of existing debris jams or the remobilisation of wood introduced to the channel during the inter-event period (Gurnell et al., 2002). In this way the current pattern of wood distribution in a channel at any point would be a reflection of the immediate hydrological history, and whether there is a signature of a significant redistribution event, or a major input event (Braudrick et al., 1997).

4.4. Log Jams

Logjams, also called debris dams can occur in a channel where there is a balance between immobile pieces of wood, which can anchor the jam as *key members* or *key pieces*, as well as large wood which is otherwise mobile in the channel, which can be trapped by the key member and form the *racked members* in the jam and increase its size (Wohl and Jaeger, 2009). As already described in section 4.3 there are a number of factors governing how mobile wood is in a channel. The presence of stable, key members is a prerequisite for jams and, although the distribution of potential key members and of log jams will be largely stochastic (Collins et al., 2002), there are a number of factors which increase the probability of a key member occurring at particular types of location or belonging to particular classes and types of large wood, such as size and orientation within the channel (Wohl and Goode, 2008).

4.4.1. Trapping Points

Wood which is floating and moving independently of other pieces of wood as uncongested transport is likely to be deposited on riffles, shallows over bars and sections of narrow channel (Braudrick et al., 1997; Millington and Sear, 2007). In channel obstructions are also effective trapping points, including immobile in-channel wood, bank vegetation, standing trees, boulders and exposed roots (Bilby, 1984; Bilby and Likens, 1980; Daniels, 2006; Millington and Sear, 2007). Channels with high geomorphological complexity will have more trapping points and potentially higher reach retention (Sheldon and Thoms, 2006), dependent on a supply of mobile wood to trap. The orientation of stable in-stream wood has been hypothesised as important in governing trapping efficiency (Bocchiola et al., 2006b).

Millington and Sear (2007) note the literature suggests trapping sites are more important than the hydrological regime in governing transport distance and reach retention. However the hydrological regime is a component in determining location and volume of wood trapped, especially in systems where there is re-deposited wood on a floodplain and where the floodplain itself is a trapping site (Gurnell, 2003).

Trapping points become particularly important in medium-sized streams, defined as those where the average channel width is greater than the length of all but the upper quartile of wood delivered to it (Gurnell et al., 2002). In such systems there will be a trend for in-channel wood to be mobile until it encounters a trapping point (Millington and Sear, 2007). Pristine rivers have been found to have high reach retention (Naiman, 1982) whereas channelized rivers have low retention and act as conduits transporting wood rapidly through them (Bilby and Likens, 1980; Millington and Sear, 2007). River restoration has also been shown to increase reach retention by restoring geomorphological complexity to previously channelized reaches (Millington and Sear, 2007).

Regardless of the type of trapping point it will act as a focus for wood deposition and formation of a logjam, furthermore once a logjam has formed it will be an efficient trapping site for other mobile wood in the channel (Millington and Sear, 2007; WFPB, 1997).

4.4.2. Key members

A key member is defined as an independently immobile piece of large wood in the channel, capable of acting as a trapping point for other pieces of mobile wood in the channel and instigating a log jam (Abbe and Montgomery, 2003). In order to act as a key member a piece of large wood not only needs to be stable, but also have a high probability of mobile wood colliding with it and for that wood to become trapped as a racked member in a logjam. Some of the factors determining whether a piece of large wood will become a key member are highly stochastic, and difficult to predict, such as the orientation of tree fall (Sobota et al., 2006). However factors which increase the likelihood that a piece of large wood will be stable in the channel or riparian zone and also factors increasing the potential for trapping and retaining mobile wood will increase the probability of a piece of large wood acting as a key member.

The most intuitive control on stability is that of size; Abbe & Montgomery (2003) found that wood which was in excess of half bank full width and with a diameter greater than half bank full depth were likely to form key members. They also found that key members could form from shorter pieces of wood, but the diameter needed to be larger (Abbe and Montgomery, 1996; Collins and Montgomery, 2002). Collins & Montgomery (2002) found from a survey of large wood that key pieces were generally longer and wider than racked pieces. Key pieces can be of any tree species provided the specimens can reach a sufficient size (Montgomery et al., 2003).

Shape is one important control, as widely branching, or multiple stem hardwoods are more prone to form snags and act as potential key members than the longer, more cylindrical large wood associated with conifer species (Collins et al., 2002). The presence of a root wad on a piece of large wood is a fundamental control on stability as it raises the centre of mass compared to a log, as well as providing the wood with a generally more complex shape, so resisting entrainment (Collins and Montgomery, 2002). Furthermore a tree with a root wad attached may retain some roots anchored into the soil or bank increasing its stability (Hyatt and Naiman, 2001).

Decay resistant species are important as they can form stable key members with long residence times; a study on the Queets River found that decay resistant species can remain unmoving for 70-100 years (Keller and Tally, 1982; Montgomery, 2003). In small river channels in old growth forests key members have been found to be stable for hundreds of years (Braudrick et al., 1997)

Often wood mobility will be a function of the hydrological regime with large wood becoming mobile or immobile depending on the current discharge (and thus wood diameter:flow depth ratio) of the stream (Gurnell et al., 2002). Therefore the deposition of wood in channel may be temporary and as a result will have a lesser potential to retain and trap mobile material at high flows due to the potential for being mobilised/remobilised itself (Kreutzweiser et al., 2005). In these cases the sediment transport regime is important for the stabilisation of wood which has been temporarily deposited (Gurnell et al., 2002). The deposition of sediment around an immobile piece of large wood can stabilise it, particularly by partially burying part of the wood, which raises the entrainment threshold of the wood and potentially allows a temporarily deposited piece of large wood to develop into a key member (Gurnell, 2003).

To date the potential for large wood to act as a key member has focused on the stability of the piece and has largely neglected factors which may affect trapping potential and thus raise the likelihood of logjam formation. There is a great need for more studies looking at the architecture of logjams and the function and characteristics of key members beyond raw size. This thesis aims to address this gap by examining the orientation, size and shape of logjam key pieces in a small forested river to try and identify trends in piece type that are correlated with pieces acting as key members.

4.4.3. Log Jams and Channel Size

In small rivers, defined as having a channel width less than the length of wood delivered to it, a significant proportion of the wood delivered to the channel is likely to remains where it falls (Gurnell et al., 2002; Montgomery et al., 2003). Such key members are likely to only trap small wood and organic debris, and unlikely to result in significant accumulations of large wood and wood jams, as the system is transport limited with respect to large wood. In medium and large rivers the function of key members becomes much more influential (Montgomery et al., 2003).

Medium and large rivers with stable key members can trap and retain smaller, mobile debris, which can range in scale from organic debris up to substantial pieces of large wood (Montgomery, 2003; Montgomery et al., 2003). Only a small proportion of the wood delivered to a river needs to be capable of forming key members in order to instigate the retention of material (Montgomery et al., 2003). Such key pieces are important for altering channel morphological processes (Gregory et al., 1985).

In larger rivers log jams can account for the majority of wood in a system; Collins et al (2002) reported over 90% of the wood in the Nisqually River, Washington State, USA was held in jams.

4.4.4. Persistence of Log Jams

Some jams have been shown to persist in-situ for 200 years (Dahlström et al., 2005) and have been shown to be resistant to large magnitude flood events (Dahlström et al., 2005; Gregory et al., 1985; Piégay and Gurnell, 1997). Wood in jams has also been shown to be older than loose logs (Dahlström et al., 2005). Jams can potentially trap pieces of wood for years to decades in the active channel and this can be extended to centuries where the jam leads to avulsions away from the channel burying the wood in floodplain sediment where it can then be exhumed in the future (Collins et al., 2002).

Logjams can persist in the same location for years (Dahlström et al., 2005; Gregory et al., 1985). Conversely logjams in the same location frequently change type and size due to collapse, decay and addition of newly recruited racked wood and fine organic material to existing jams (Gregory et al., 1985; Millington and Sear, 2007; Sear et al., 2010).

Furthermore a number of studies looking at fine scale large wood dynamics have shown that although local quantities of wood remain stable there can be high rates of turnover of individual pieces (Latterell and Naiman, 2007; Marcus et al., 2002; van der Nat et al., 2003).

There may be an element of cyclical behaviour in logjam type and complexity for jams which persist at the same location. Shields and Alonso (2012) showed the highest drag and greatest overall force acting on a logjam structure were for simple structures, i.e. analogous to non-porous weirs. It is hypothesised that logjams form complex structures which are then subject to higher forces and broken up over time to simpler ones (Shields Jr and Alonso, 2012). It has also been shown that material broken up from a logjam will often be transported and subsequently deposited at the next logjam downstream (Thomas and Nisbet, 2012), thus increasing the complexity of that structure.

4.4.5. Classification of Log Jams

A number of classification schemes have been developed to describe logjams in various settings. To date the application of these classification schemes are restricted to systems similar to those they were developed in.

Gregory et al (1985) classified logjams in the Highland Water catchment, UK (Figure 4.7) to allow for comparisons between logjams and surveys, and to enable the monitoring of logjam evolution and change over time.

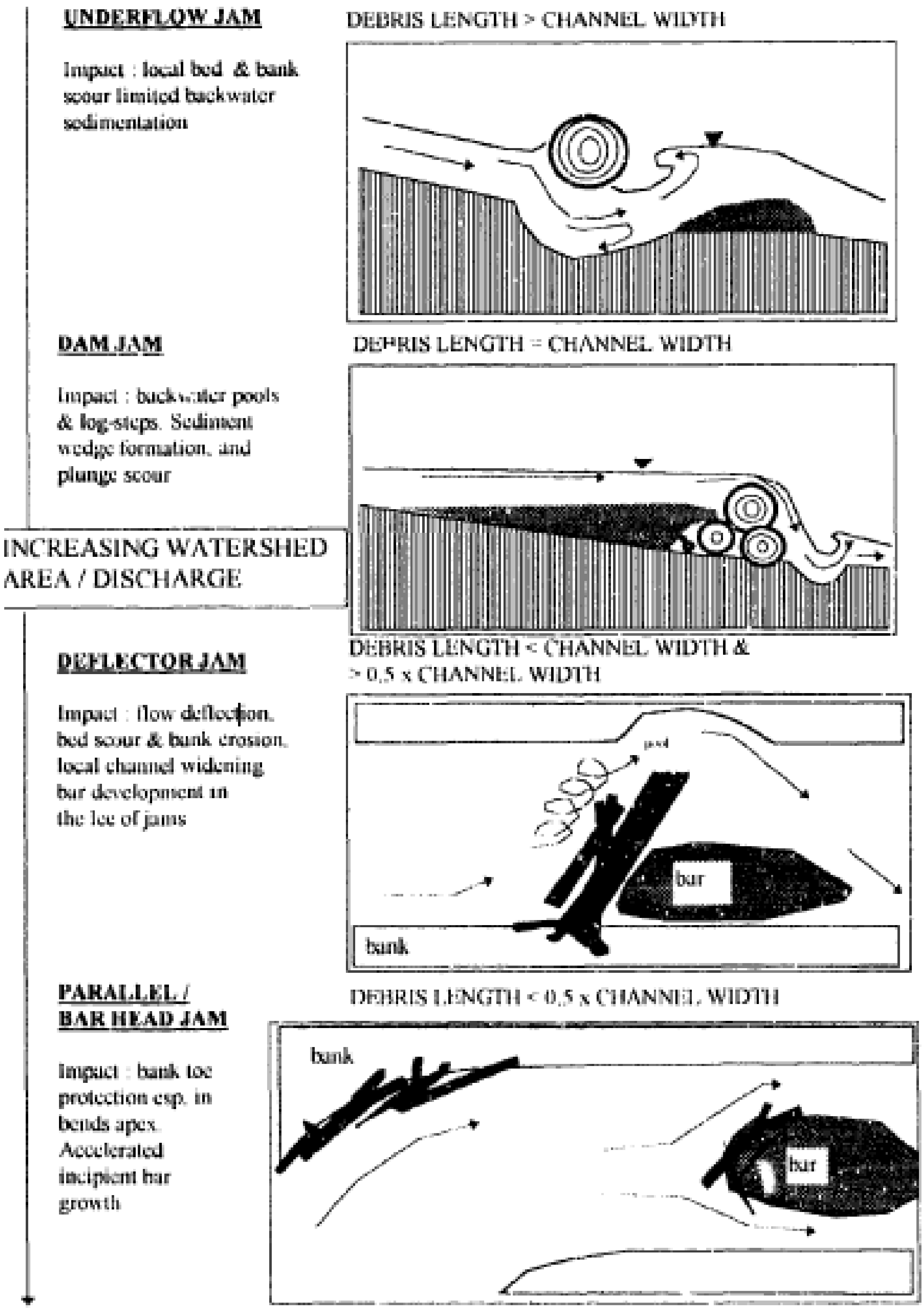


Figure 4.6 – logjam classification system of Wallenstein & Thorne (1997) developed for rivers of the Northern Mississippi, USA.

The dams were classified as either *active*, completely barring the channel and resulting in a pronounced step in water profile, *complete*, spanning the channel but not inducing a step at low/medium flows, *partial*, which doesn't span the channel, and *high water* which includes at least one piece of wood bridging across the bank tops, but does not fill the channel (Gregory et al., 1985; Piégay and Gurnell, 1997). Wallerstein et al (1997) developed a similar classification system for forest rivers in the Northern Mississippi, USA (Figure 4.6) based on how the logjam influenced hydraulics as parallel, deflector, underflow or dam logjams. Dam jams are analogous to Gregory's active jams (Gregory et al., 1985; Wallerstein et al., 1997).

Gregory et al (1985) found that active dams were the most stable during a time experiment, suggesting that these dams will play an important role in trapping mobile wood (Abbe and Montgomery, 1996; Abbe and Montgomery, 2003; Montgomery et al., 2003).

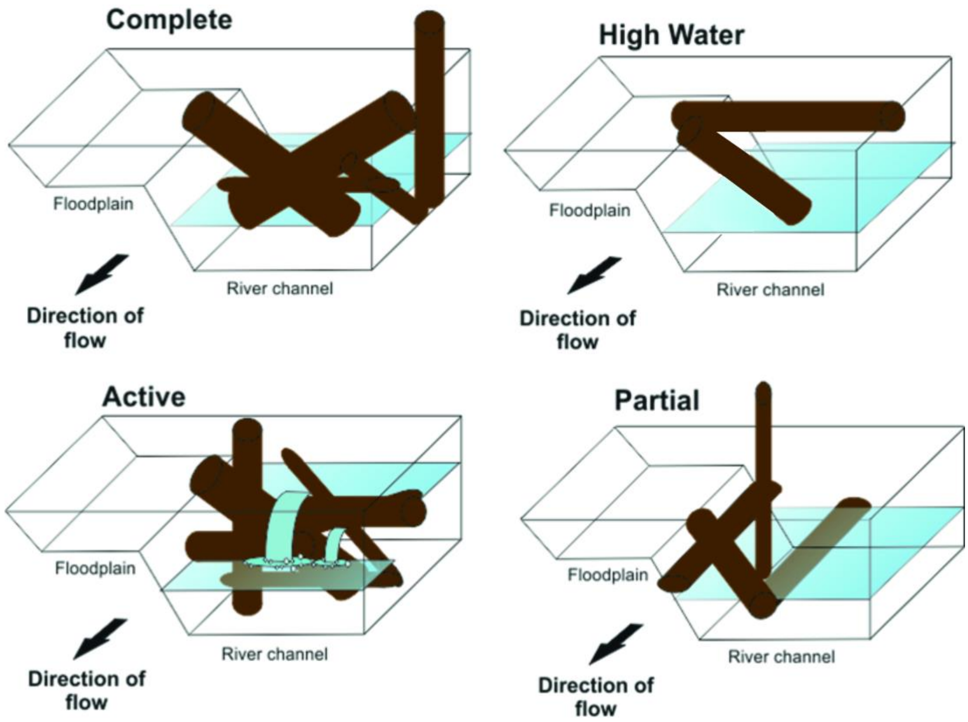


Figure 4.7 – logjam classification system developed by Gregory et al (1985) for rivers in the New Forest, UK.

Abbe & Montgomery (1996) also classified jams on the braided Queets River in the North Western US. They classified them as either bartop jams (BTJ), bar apex jams (BAJ) or meander jams (MJ), where BTJ's are chaotic and unstable and BAJ's and MJ's were morphologically effective, creating associated pools, bars and floodplain patches. Abbe & Montgomery (2003) expanded on this earlier work and described 10 types of jams based on the key member(s) and their orientation with racked and loose wood (Table 4.3).

Types	Distinguishing characteristics
In-situ (autochthonous)	Key member has not moved down channel.
Bank input	Some or all of key member in channel.
Log steps	Key member forming step in channel bed.
Combination	In-situ key members with additional racked large wood.
Valley	Jam width exceeds channel width and influences valley bottom.
Flow deflection	Key members may be rotated, jam deflects channel course.
Transport (allochthonous)	Key members moved some distance downstream.
Debris flow/flood	Chaotic large wood accumulation, key members uncommon or absent, catastrophically emplaced.
Bench	Key members along channel edge forming bench-like surface.
Bar apex	One or more distinct, key members downstream of jam, often associated with development of bar and island.
Meander	Several key members buttressing large accumulation of racked large wood upstream. Typically found along outside of meanders.
Raft	Large stable accumulation of large wood capable of plugging even large channels and causing significant backwater.
Unstable	Unstable accumulations composed of racked large wood upon bar tops or pre-existing banks.

Table 4.3 – wood accumulation typology from Abbe & Montgomery (2003)

Although these logjam classification schemes remain useful tools, and in the case of the Gregory et al (1985) system are still finding application nearly 30 years after being published, their usage is restricted. A logjam classification scheme which links logjam form and function and is scale invariant from forest rivers through island braided systems remains to be illustrated.

4.5. Conclusions

Large wood and logjams within forest rivers affect hydrology, geomorphology and ecology at a range of scales. As well as forming discrete geomorphological features such as pools, wood can be the primary influence on channel form. The range of erosional and depositional features associated with logjams has been described in a variety of settings. The impacts of changing patterns of erosion and deposition in the presence of large wood have also been described and quantified at reach and catchment scales. The features of individual logjams which cause geomorphological and habitat complexity however remain to be illustrated.

Wood enters forest rivers as fallen trees, through windthrow, bank erosion and from fluvial transport. Wood persists in the river system for times ranging from minutes up to decades, or even centuries, before leaving the system through fluvial transport or decay and disintegration. Despite these processes being documented in the literature the quantification and rates for many processes remain to be illustrated and are especially lacking in a UK context. A lack of published field studies hampers understand of wood transport lengths and factors influencing log mobility over inter-flood time scales.

In this thesis using experimental field studies I will address these gaps in the literature by:

- Conducting a survey of logjams within a forest river in order to examine the link between the characteristics of the key structural pieces around which the accumulation forms and the characteristics of the resultant logjam.

- Analysing data from the logjam survey to examine links between the characteristics of logjams and the likelihood of there being geomorphological and habitat diversity in association with the structure.
- Tagging and tracking the movement of pieces of large wood within a forest river. Data on mobility will help to constrain the size at which wood becomes functionally immobile in flood flows and to quantify transport lengths of large wood. Analysis of wood characteristics will enable factors governing mobility, apart from piece size, to be illustrated.

5. Controls on the distribution and geomorphic performance of large wood accumulations

5.1. Abstract

Large wood is an important part of many river systems, influencing the hydrological, ecological and geomorphic conditions of the channel and floodplain. Linkages have been shown between high wood loads and improved biotic conditions in river channels and this has been an important driver for introducing large wood into degraded river systems as part of river restoration projects; however such projects have met with mixed success and do not always achieve restoration aims. Little research has been done into the link between logjam form and function contributing to limited guidance in the grey literature and a lack of scientific basis for some engineered logjam designs. The research addresses the importance of reach scale geomorphological constrictions in controlling logjam location and clustering. The structure and architecture of logjam types are investigated in connection to the characteristics of their key structural piece of large wood. Finally correlations between logjam characteristics and the likelihood of habitat provision being associated with the structure are examined. This study has shown that (i) the location of logjams within a small headwater catchment has only a very weak correlation with geomorphological constrictions and thus I hypothesise logjam location is determined predominantly by the stochastic distribution of wind-thrown large wood acting as key structural pieces in logjams; (ii) the characteristics of key structural pieces can be linked to logjam size, type and function; (iii) the likelihood of a logjam having habitat or sedimentary structures associated with it is linked to logjam type and is predominantly determined by hydraulics and the creation of a step in the water profile. These findings can be incorporated into river restoration planning to help improve the success of engineered logjams creating new in-stream habitats.

5.2. Introduction

Research into logjams over the past thirty years has revealed their important geomorphological and hydrological effects. We use a definition here of logjams as in-channel accumulations of large wood, commonly packed with fine woody material and other organic matter. Logjams interact with flow to create local variations in shear stress, promoting heterogeneity in erosion and deposition (Abbe and Montgomery, 1996; Montgomery et al., 2003), resulting in effects such as increasing overbank sedimentation (Jeffries et al., 2003) and mediating crevasses and avulsions (Phillips, 2012; Sear et al., 2010). The heterogeneity of geomorphic forms created by large wood in forest rivers provides diverse habitats for a wide range of organisms (Gurnell et al., 2002). Scour associated with logjams acts to sort gravels suitable for salmonid spawning (Buffington et al., 2004), and the formation of pools acts to provide refuges for juvenile fish (Peterson, 1982). Logjams trap and retain coarse particulate organic matter (Harper et al., 1999), providing important temporally and spatially regulated food sources for aquatic organisms (Gurnell et al., 2002). Variations in hydraulic roughness associated with logjams cause complex flow patterns which act to dissipate energy and increase the travel time of flood waves and reduce flood peaks (Gregory et al., 1985; Thomas and Nisbet, 2012), as well as decreasing the average grainsize (D_{50}) of bed material due to a reduced competence to transport bedload in the channel (Buffington et al., 2004). Although the general effects of logjams have been documented in the literature there is a great deal of variability in typology, with individual logjams displaying variations in their geomorphic, hydrological and ecological effects, these variations are likely to be due to differences in logjam form (Jones et al., 2011). Context is also important with variations likely in hydraulics and geomorphological features associated with wood in different regional environments and in channels of different sizes (Montgomery, 2003).

Although not recommended as a long-term, holistic river restoration strategy (Beechie et al., 2010), engineered logjams (ELJs) are still very popular (Abbe and Brooks, 2011). The insertion of artificial logjams is advocated only as an initial step (Abbe et al., 2003; Collins and Montgomery, 2002) as part of a long-term regeneration strategy based on a model of biophysical landscape development alongside riparian forest planting

(Collins et al., 2012). The use of logjams in river restoration and rehabilitation projects often attempt to harness their ecological benefits to increase abundance and/or biodiversity of target species and taxa as a short term intervention; such schemes have met with mixed success (Stewart et al., 2009) and are not thought to be widely effective at meeting restoration aims (Larson et al., 2001; Sear, 1994; Whiteway et al., 2010). Logjam placement schemes to increase species abundance will typically fail due to insufficient understanding of the natural processes at work (Ward et al., 2001), either they are designed to increase habitat provision for species whose abundance are not habitat limited (Sweka et al., 2010), or where the target species is habitat limited the engineered logjams fail to create enough new habitat (Rosenfeld and Hatfield, 2006). If habitat is not increased post restoration this indicates insufficient knowledge in designing and implementing logjams.

It has been shown that reaches with ELJs exhibit fewer pools than expected for equivalent wood loading in forested streams (Larson et al., 2001) and although species richness is often increased after addition of wood, species density does not tend to increase (Miller et al., 2010). There is therefore an urgent need to understand the characteristics of individual logjams which lead to provision of habitat and geomorphic diversity to enable more effective river restoration design. Current river restoration manuals have little information on the function of ELJs in relation to their placement or structure. The readily available guidance for using wood in restoration is primarily focused on ensuring stability of introduced wood with cables and stakes (e.g. Fischenich and Morrow Jr, 1999; Lewis, 2008; Mott, 2006), the placement of wood to protect against bank erosion (Lewis, 2010), or engineering individual logs to deflect flow (Seehorn, 1992). Brooks (2006) is one of the most comprehensive guides for using ELJs in restoration and has a detailed evaluation framework for projects, however only one type of ELJ structure is described in detail, similarly in a set of conceptual design guidelines for the Scottish Environmental Protection Agency several jam types are described but no details of their structure or function are provided (Herrera Inc, 2006). Despite a focus on engineering for stability the lifespan of ELJs can be relatively short and prescribed structural designs can be inappropriate or counterproductive in some geomorphic settings (Frissell and Nawa, 1992).

Component pieces of wood in a logjam can be classified as key pieces, racked pieces or loose pieces. Key pieces structurally anchor the accumulation and are usually independently stable in the channel. Racked pieces have previously been mobile in the channel and have been trapped by the logjam, loose pieces are those associated with the logjam, but not structurally connected. The importance of very large key pieces in stable logjam formation has been illustrated in a number of different environments (e.g. Abbe and Montgomery, 1996; Keller and Swanson, 1979; Nakamura and Swanson, 1993; Sear et al., 2010), particularly in larger rivers where only the largest pieces of wood will remain immobile (Collins et al., 2012; Marcus et al., 2002). Key pieces have been shown to be larger than racked pieces (Fetherston, 2005 in Collins et al., 2012; Montgomery and Abbe, 2006) and often have an attached rootwad (Fetherston, 2005 in Collins et al., 2012) which increase long-term stability (Abbe and Montgomery, 2003; Wallerstein and Thorne, 2004). Other studies have hypothesised the complexity of branching (Abbe and Montgomery, 2003) and partial burial in sediment as important in controlling large wood stability (Brooks et al., 2004), but the importance of these remains to be fully illustrated.

In order for logjams to form there needs to be a balance between stable pieces of large wood in the channel which can act as key structural pieces in a logjam and a supply of mobile pieces of wood which can be trapped and act as racked pieces. A potential key piece needs to be both stable and able to trap and retain other pieces of large wood in the channel in order to form a logjam. As a key piece needs to be able to trap other pieces of wood it is possible that factors such as the orientation of the key piece relative to the channel, along with its size will increase the likelihood of "collisions" between the key piece and mobile wood in the channel. Collisions between mobile and immobile wood pieces are a pre-requisite for trapping potential racked pieces. A piece of wood imbedded in the channel and angled onto a bank top for example could be expected to collide with more pieces of mobile wood than one bridging both banks which rarely interacts with flowing water. The orientation and position of large wood is an important factor which has hitherto received little attention compared with studies quantifying wood loadings (Jones et al., 2011).

The relative size of a piece of wood to the river channel (ratios of piece-length:river-width and piece-diameter:river-depth) is important in governing mobility (Collins et al., 2012). Gurnell et al (2002) describes how the importance of various factors to mobility change as the size of the river increases relative to the size of wood supplied to it. In medium sized rivers (river width less than the upper quartile of wood length delivered to it) with high geomorphic diversity, the majority of wood will be capable of being mobilised by the river and will become trapped in narrower or shallower sections of channel and in more sinuous reaches, therefore the formation of logjams is likely to occur more frequently in areas of channel constriction and higher geomorphic complexity (Gurnell et al., 2002). In small rivers (river width smaller than median wood length) a significant proportion of the wood delivered to the stream may remain immobile as even during flood discharges the river is not competent to transport it; in such transport limited systems geomorphic trapping points are likely to be less important than for large rivers (Gurnell et al., 2002). Previous studies on small rivers have found logjams are formed, change form and function and disappear on sub-decadal timescales (Gregory et al., 1985; Gurnell and Sweet, 1998; Sear et al., 2010), over periods of <2 years the majority of logjams will not retain the same form and function, with persistence at a site typically accounting for just 14% of logjams (Sear et al., 2010). Beaver (Castor canadensis) constructed logjams have significant and well documented geomorphic effects (Gurnell, 1998; Naiman et al., 1988; Naiman et al., 1986) and these have also been shown to have residence times of a decade or less (Butler, 2012).

5.2.1. Aims

The important role logjams play in increasing geomorphic and habitat diversity has long been recognised. Logjams create habitat diversity and therefore it is important to understand what controls logjams location and type and how logjam type is related to habitat creation, in order to understand in which locations these features will have the greatest impact on habitat provision. Data are sparse on the factors controlling the locations of logjam formation; furthermore the characteristics of individual logjams which lead to changes in geomorphic diversity through altered patterns of sediment erosion and deposition remain to be illustrated in the literature.

This paper aims to explore the relative importance of key pieces of wood compared with variations in reach-scale channel geomorphology in controlling logjam location; we hypothesise that in small headwater streams where the channel is not competent to mobilise all the wood delivered to it, large key pieces of wood will act as stable anchor points and these will be more important for logjam formation than channel constrictions. Furthermore we hypothesise that large, complex shaped key pieces will form larger logjams.

The following questions were addressed: (i) do river channel and catchment characteristics of a reach predict logjam frequency within it? (ii) Are there material differences in the architecture of different logjam types? (iii) Are the characteristics of the key structural piece related to the size of a logjam?

The properties of individual logjams which lead to geomorphic diversity or creation of new habitats remain to be illustrated, despite the widespread use of logjams in river restoration with the aim of increasing habitat diversity and improving biological conditions in degraded river systems. Therefore the second aim of this chapter is to investigate whether there are any physical attributes of logjams and/or their key pieces which are correlated with a greater likelihood of geomorphic features or habitat in association with the logjam.

The following further questions were addressed: (iv) are the characteristics of the key piece related to the hydraulic effectiveness of logjam formed around it? (v) Is the type of logjam and/or its physical characteristics related to the likelihood of habitat provision being associated with the structure?

5.2.2. Study Area

This research was conducted on the Highland Water, a third and fourth-order tributary of the Lymington River, flowing through the New Forest National Park, Hampshire, Southern England (Figure 5.1). The stream is low energy and meandering with cohesive bank material and a bed substrate of coarse-grained alluvial gravels. The river flows through a mix of Tertiary clay, silt and gravel alluvial deposits (Gurnell and Sweet, 1998) overlain by humus rich forest soils (Sear et al., 2010). The underlying geology is Eocene Barton clay resulting in a flashy hydrological regime (Gurnell and

Sweet, 1998; Piégay and Gurnell, 1997). The floodplain sediments range from <0.5m thick in the headwaters to 1.2m thick in the lowland floodplains (Sear et al., 2010). The shallow soils lead to a shallow rooting depth for floodplain trees, forcing horizontal spread of the root network and making the trees susceptible to wind-throw (Brown, 1997).

The woodland is characterised by a dominance of *Betula pubescens* (beech) with *Quercus robur* (sessile oak), *Fraxinus Excelsior* (ash), *Alnus glutinosa* (alder), *Betula pendula (birch)* and some *Ilex aquifolium* (holly) (Jeffries et al., 2003). Ground vegetation is sparse, particularly in areas of beech dominated woodland, but includes *Pteridium aquilinium* (bracken), *Rubus spp*. (bramble) and *Rubus fruticosus* (blackthorn) (Peterken et al., 1996).

Streams in the New Forest have been subject to engineering since the 1840s, with many periodically straightened and dredged to improve drainage, particularly in plantation enclosures (Tubbs, 2001). These works have led to downwards cutting of the bed and headward erosion into mires (Tubbs, 2001), the deeper, straightened channels have reduced geomorphic diversity and are often disconnected from their floodplains leading to habitat fragmentation (Sear et al., 2006). Recently management focus has shifted to river restoration, with a number of projects since 2005 undertaken to reconnect the engineered channels with their floodplains including stabilising knickpoint recession, raising bed levels and re-meandering the streams (Millington and Sear, 2007; Oakley, *pers. comm*., 6th July 2011).

The New Forest remains one of the few areas within Europe with relatively unmanaged lowland forest river channels, this enables the study of large wood and logjam distribution and function in a relatively natural setting (Gurnell and Sweet, 1998).

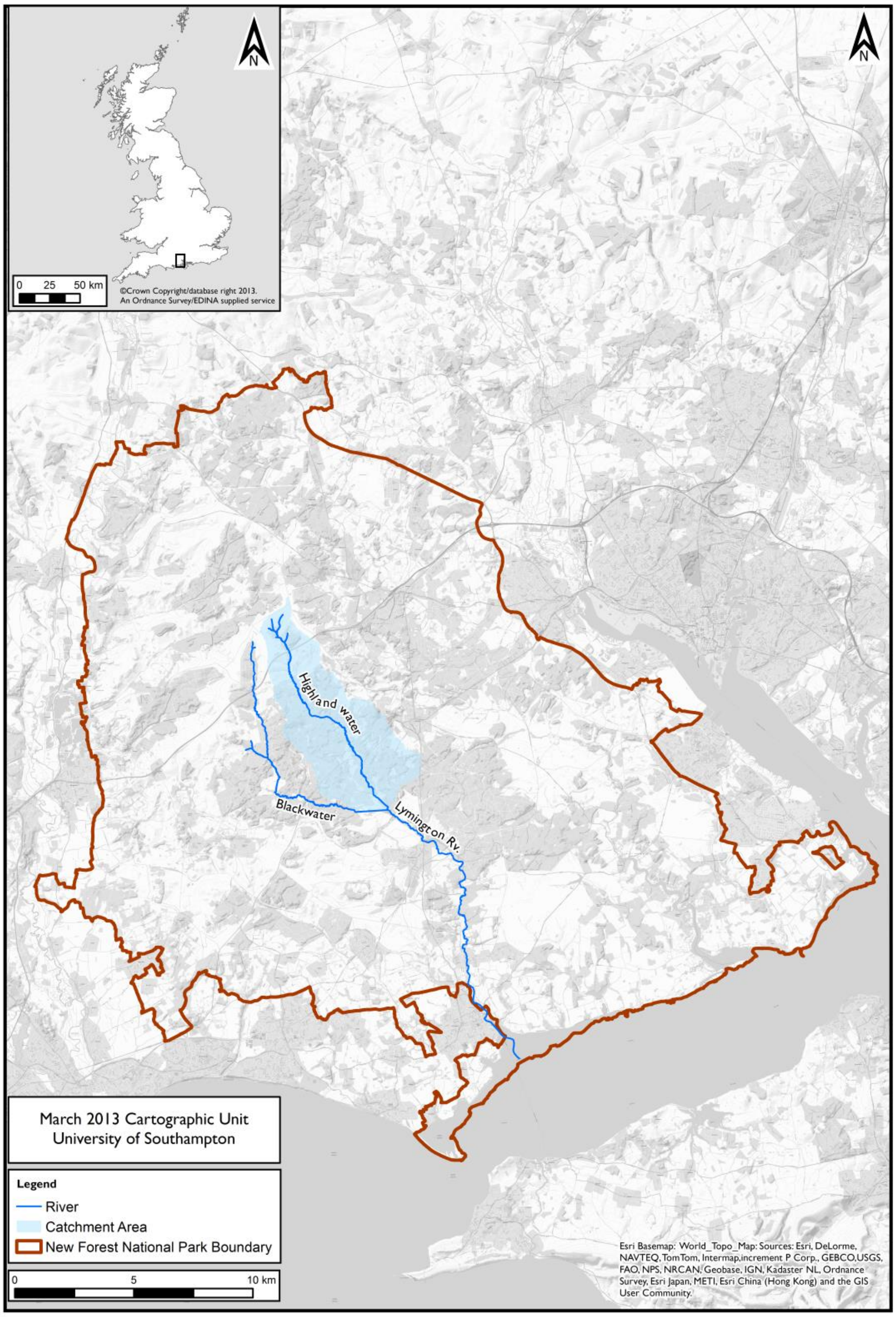


Figure 5.1 – Map showing the location of the study catchment

5.3. Methodology

The locations of logjams were mapped between June and July 2010 and again in November 2011 along a 11.5km length of river. Locations were recorded using a handheld palmtop computer (PDA) with a geographical positioning system (GPS; accurate +/-2.0m) and cross referenced using an Ordnance Survey 1:10000 map. Logjams were classified according to Gregory *et al.* (1985) based on relative hydraulic influence (Figure 5.2); Active logjams span the channel and cause a step in the water profile even at low flows; Complete logjams span the channel but have a high porosity and thus do not cause a step in the water profile; Partial logjams do not span the channel and High water logjams span the channel but only at the bank top. The number of key, racked and loose pieces of large wood was tallied for each jam as was the volumetric extent of the jam within the channel, measured along three axes (downstream, across stream and height).

For data collection we define large wood as both at least 1m in length and 0.1m in diameter. Although we do not link criteria directly to the channel dimensions these minimum values are commonly used in the literature and facilitate comparisons with other studies (Wohl et al., 2010).

To allow estimates of the proportion of total large wood within logjams the 11.5km survey length was divided into 500m reaches (n=23) and for each reach every piece of large wood within the channel was recorded as either a logjam piece or loose and tallied into a size class matrix with class divisions at 1m in length and 0.1m in diameter.

In order to calculate catchment characteristics a digitised river centreline was obtained (Sear et al., 2010) along with a 10m digital elevation model (DEM). Spot measurements were taken every 100m along the channel for bank height and channel width.

5.3.1. Sliding Window

In order to analyse how the distribution of logjams varied with channel and catchment characteristics a sliding window approach was used, with a reach length of 150m, sliding 50m (n=231). Meleason et al. (2007) demonstrated a sliding window approach gave more representative values for wood loading than selecting 'representative' reach lengths which tend to mask areas of very high or low wood loadings. A window length of 150m is sufficiently short to allow local variations in reach characteristics to be observed, but long enough to encompass ≥2 meander wavelengths (Dury, 1981), and 4-

6 pool-riffle sequences (Leopold et al., 1964), given an average bankfull width of the study river of ≈5m.

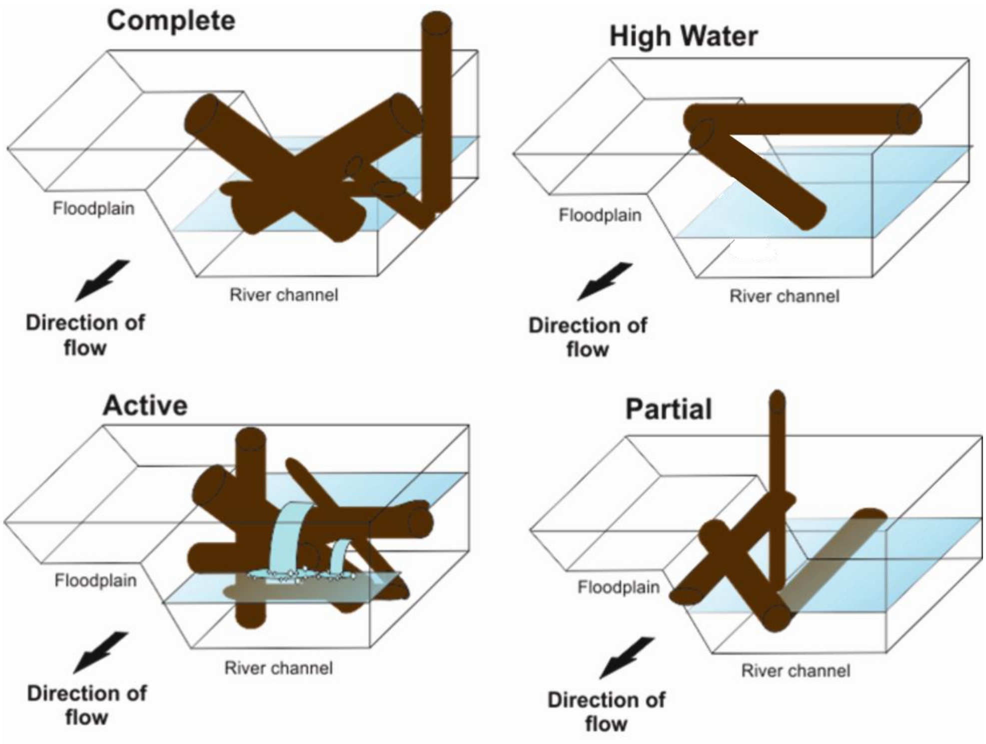


Figure 5.2 - Gregory (1985) logjam classification scheme, Complete jams fill the channel but do not cause a step in the water profile at low flows; Active jams fill the channel creating a complete barrier to water and sediment and cause a step in the water profile; High Water only span the channel at the bank top and Partial do not span the channel.

Reach sinuosity was calculated using ArcMap with a digitised river centreline layer as a route event, using the straight line distance along a reach to calculate sinuosity. Catchment area draining to each window was calculated using an automated watershed model built in ArcGIS and a 10m resolution DEM, using the point locations of each window as the catchment outflow point (snapped pour point in ArcGIS) to generate the drainage network and catchment for each window. Average bankfull width and bank height, along with total quantity of large wood within each window was calculated from field data. Satellite imagery in GoogleEarth (Google, 2012) was used to generate a percentage forest cover map of the riparian zone along the study reach.

A modelling strategy of backwards elimination was adopted for all multivariate analyses (Agresti and Finlay, 2007); statistical models were initially built with all of the variables indicated in the literature as potential controls as model predictors, these initial models are assessed and those variables indicated as poor predictors (having a p-value greater than α=0.05) removed. The reduced statistical models are then re-run, the goal being to develop the most parsimonious model whilst still explaining a high variance.

5.3.2. Key Structural Piece & Geomorphic Features

In November 2011 additional data were collected on the key structural piece (Table 5.1). Geomorphological and habitat features associated with each logjam were noted as either present or absent, these were; pool, riffle, bar, exposed sediment, backwater, cover/shade.

5.4. Results

5.4.1. Wood Loadings

Within the in the 11.5km study reach 1803 pieces of large wood were recorded, this represents a mean loading of 156.78 pieces per km. Median dimensions for each size class from a matrix and the formula for the volume of a cylinder were used to derive a total wood loading volume of 167.7m^3, representing a mean loading of 14.58m^3/km or 29.16m^3/ha, based on channel length and mean channel width (Table 5.2).

Large wood was preferentially located in logjams with 70.1% of total large wood pieces (1315 pieces) found in logjams; across the 23 reaches this proportion varied between 40.0% and 90.2%. Figure 5.3 shows the size distribution of key pieces, racked pieces and loose pieces, demonstrating key pieces are longer than racked and loose, and typically at least one bankfull channel width in length. The proportion of racked pieces declines with increasing size, a racked piece is normally one which has been moving in the channel and has been trapped by a logjam, so this indicates increased length

Length	Metres*
Diameter	Metres*
Orientation	0-180^{o} Determined using compass to nearest 45^{o} where 0^{o} is aligned to flow direction and facing downstream, 180^{o} is aligned to the flow direction and facing upstream
Position	Informal description of position and environment; Bridge, Partial Bridge, Active Bridge, Ramp, Parallel, Submerged, Upright. Based on Wohl & Goode (2008) & Jones & Daniels (2008)
Decay	5-class decay system of Robison & Beschta (1990)
Species	Oak, Beech, Holly, Alder, Birch, Ash, Conifer, Hawthorn, Unknown
Branching Order	Greatest branch order recorded. Order 1 is a main trunk, with each lateral branch having an order number one greater than its parent branch (Wilson, 1966), analogous to Strahler stream order
Rootwad	Yes/No
Buried/Partially buried in sediment	Yes/No
Alive	Yes/No
Trapped by living trees	Yes/No
Habitat Features	Yes/No
Geomorphic Features	Yes/No

Table 5.1 - measurements taken of logjam key structural piece with the units/classification system used. * - In order to convert length and diameter to dimensionless units for analysis, length was divided by reach average channel width and diameter by reach average bankfull depth.

corresponds to lower mobility and only smaller pieces under 14m in length are transported in the flow. The proportion of key pieces increases substantially when length exceeds average bankfull width of 5m.

5.4.2. Sliding Window Analysis

In order to answer the first question on the importance of reach scale geomorphological controls in logjam location a multivariate general regression analysis was performed (Agresti, 2002; Sokal and Rohlf, 1994) using the sliding window characteristics for; width, depth, sinuosity, drainage area, wood load (pieces) and % riparian forest cover as predictors. Using ArcGIS the locations of logjams from the 2011 survey were mapped onto each sliding window using the spatial join function within ArcGIS, the number of logjams within each window was tallied and these logjam counts used as the model response. Only the 2011 data on logjams were used as they are contemporaneous with the measured reach characteristics for the sliding window.

A model with all predictors has R-squared = 15.47% (adjusted for degrees of freedom), only sinuosity ($p=0.026$), bankfull width ($p=0.012$) and wood load ($p<0.001$) were found to be statistically significant at $\alpha=0.05$. The parsimonious model with these three predictors has R-squared = 16.79% (adjusted for degrees of freedom), with the regression equation:

$$Logjams/150m = 1.10 + 0.95S - 0.25W_{bf} + 0.10P \qquad 5.1$$

where S=sinuosity, Wbf=average bankfull width & P=wood load in number of pieces. All predictors were found to be statistically significant; sinuosity $\beta=0.945 \pm 0.434$, $p=0.030$; bankfull width $\beta=-0.246 \pm 0.090$, $p=0.007$; woodload (pieces) $\beta=0.098 \pm 0.014$, $p<0.001$.

	Sinuosity	Width (m)	Depth (m)	Catchment Area (km^2)	Wood Load (n/150m)	Forest Cover (%)
Mean	1.32	5.00	1.19	12.58	23.6	71.60
Minimum	1.00	2.15	0.30	2.75	5.4	0
Maximum	3.46	8.25	2.25	48.32	47.4	100

Table 5.2 - summary of reach characteristics from Highland Water, November 2011 survey.

5.4.3. Logjam Architecture

Table 5.3 shows there is a large range in logjam characteristics. In order to answer the second question and establish if there are differences in the physical characteristics of Gregory logjam types a One-way Anova analysis was performed between logjam volume and Gregory logjam type (Sokal and Rohlf, 1994). Logjam volume is approximately log-normally distributed, so a log-transform was performed on these data. This test is statistically significant, but only explains a small amount of the variance in volume ($p<0.001$, $R^2 = 9.58\%$). Using Tukey pairwise comparisons (Stevens, 2007) Partial logjams were shown to be significantly smaller than all other types (at 95% confidence interval); however the other logjam types were not shown to be significantly different from each other.

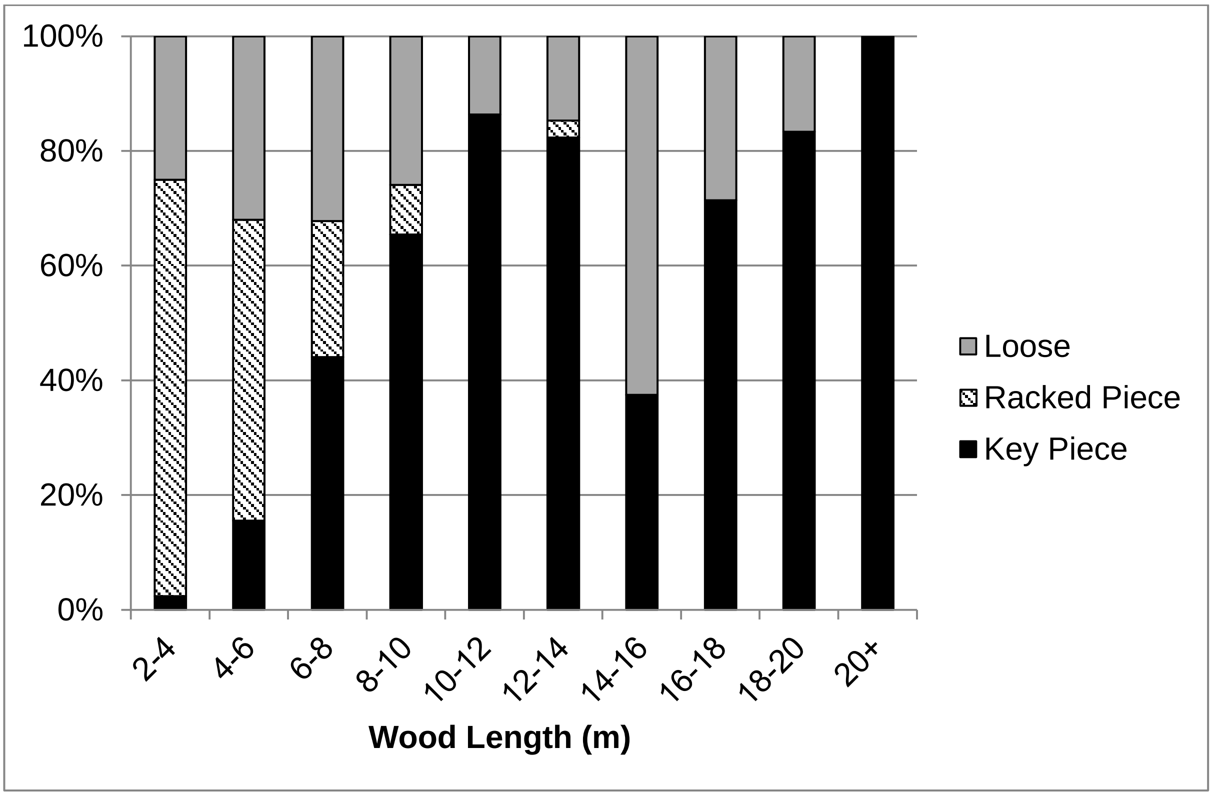


Figure 5.3 - Distribution of large wood size fractions.

The size of different logjam types was further tested by analysing the total number of large wood pieces in a logjam against Gregory logjam type. Total large wood pieces are not normally or log-normally distributed, so a non-parametric Kruskal-Wallis test with Dunn's simultaneous pairwise comparisons was used (Sokal and Rohlf, 1994). This showed a significant relationship between the variables ($p<0.001$); Dunn's pairwise comparisons (at 95% confidence interval, Bonferroni z-value=2.68) show Active and Complete logjams have significantly more pieces of large wood than Partial or High water logjams (z=4.98-6.15), and there is no significant difference in the number of pieces between Active and Complete logjam types (z=0.21).

	Length (m)	Width (m)	Height (m)	Volume (m^3)	Channel Area (m^2)	Pieces of Large Wood
Mean	4.01	3.59	1.09	18.77	15.10	6.53
Minimum	0.50	0.50	0.50	0.50	1.00	2.00
Maximum	15.00	10.00	3.00	180.00	120.00	62.00

Table 5.3 - summary of logjam characteristics from Highland Water, November 2011 survey

5.4.4. Key Structural Piece

Figure 5.4 shows the distribution of key piece position, with Ramps, Upright and Bridge key pieces having the highest frequency and the other types occurring less frequently in the study reach. This suggests logjams are preferentially formed around Ramp key pieces (large wood which is resting on one banktop and the channel bed); as well as Upright key pieces (living trees and dead snags). Other key piece orientations only account for a small proportion of total logjams.

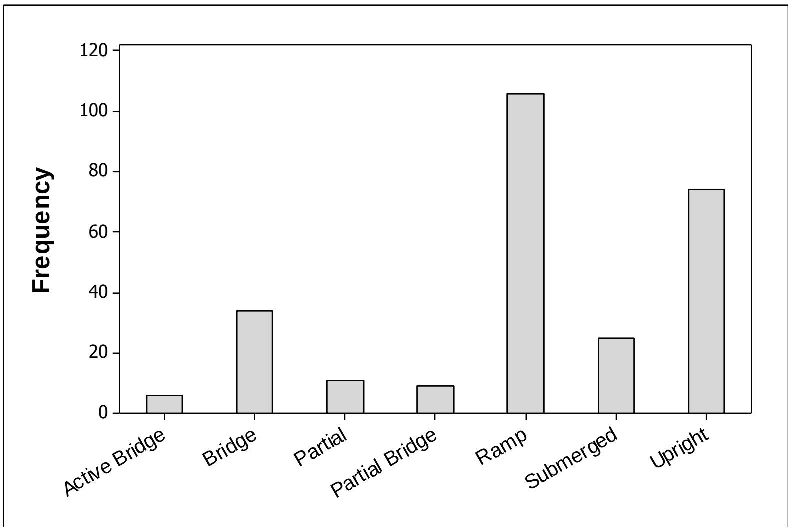


Figure 5.4 - Distribution of key structural piece classification.

Figure 5.5 shows the dimensionless size distribution of key structural pieces. The majority of key structural pieces are in excess of one bankfull width in length, key structural piece diameter however follows an approximately log-normal distribution, and is similar to the overall size distribution for all large wood in the river.

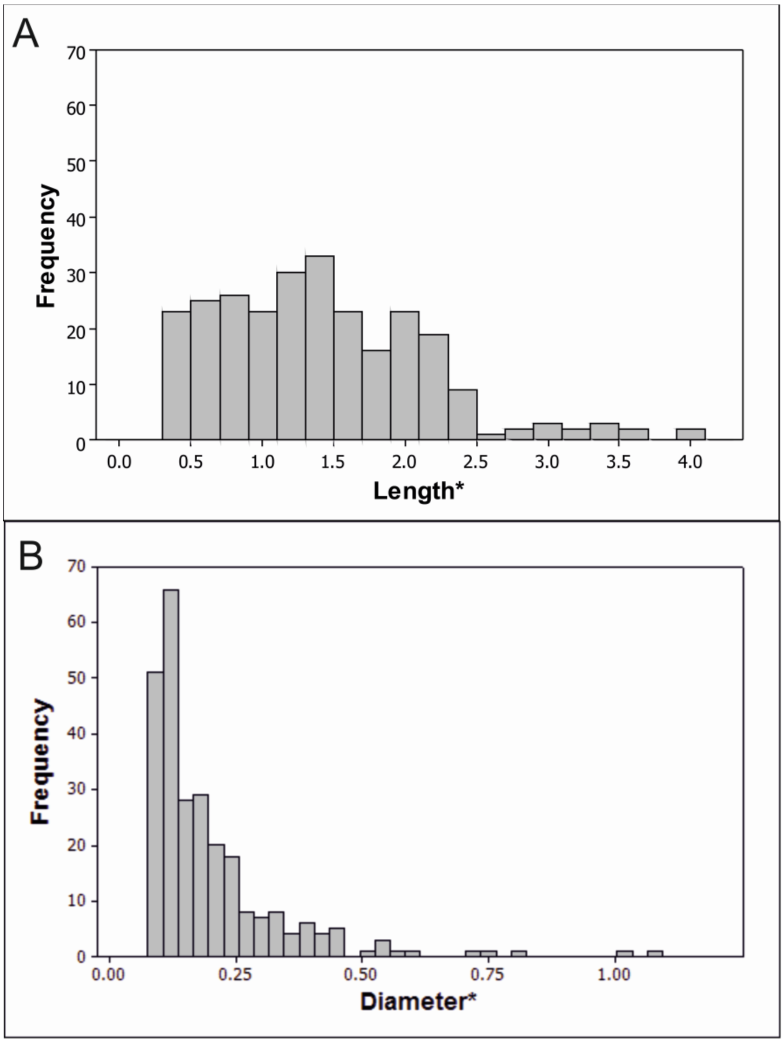


Figure 5.5 - dimensionless size distribution of logjam key structural pieces, a) piece length is divided by river width and b) piece diameter is divided by bankfull depth.

In order to answer the third question and identify if key structural piece characteristics are related to logjam size a general multiple linear regression analysis was performed with the total number of pieces of large wood in the logjam as the response and all key structural piece characteristics from Table 5.1 as predictors. A log transform was performed on the response data (number of pieces of wood in logjam) as the residuals of an initial model were not normally distributed, with a bias towards very large positive residuals and with the model performing poorly at large fitted values. The log transform data for number of large wood pieces are approximately normally distributed. The model has an r-squared =7.10% (adjusted for degrees of freedom), with only the diameter of the key piece and living key piece (y/n) statistically significant predictors ($p=0.008$ & $p=0.009$ respectively), furthermore the model is not particularly sensitive to any of the predictors (β=-0.21-0.80). Although the residuals for the log model are approximately normally distributed, the mean is not at zero and there remains a bias for large positive residuals. These results indicate that key piece characteristics are poor predictors of the total number of pieces of wood within a logjam.

To further examine the relationship between the key piece and logjam size a general multiple linear regression analysis was performed with a log transform of logjam volume as the response and all key structural piece characteristics from Table 5.1 as predictors. As with large wood pieces, neither the source data nor the model residuals were normally distributed, so a log transform was performed where log-volume data were approximately normally distributed. The model has an r-squared of 27.46% (adjusted for degrees of freedom); a more parsimonious model was built excluding those predictors shown to be not statistically significant. This model has an r-squared of 26.53% (adjusted) with key piece; length, diameter, alive (y/n) and classification as predictors. An upright key piece is used as the 'reference event' (Agresti, 2002) and all classifications apart from Partial bridge ($p=0.071$) and Parallel ($p=0.074$) are statistically significant at α =0.05. The model is most sensitive to key piece length given the 18.5m range of measurements ($\ln(\beta)=0.26$, $\beta=1.30$, $p=0.007$), the model is relatively insensitive to key piece classification (β=1.84-9.21) and key piece alive ($\beta=1.30$) particularly given the large range in model response (logjam volume), see Table 5.4.

Predictor	Coef (β)	SE Coef	Wald Statistic (z)	P	Odds Ratio	95% CI Lower	Upper
Logit (High Water/Partial)							
Constant	-3.568	0.650	-5.49	0.000			
Key Piece Orientation (Ref: upright)							
0°	-0.518	1.139	-0.45	0.649	0.60	0.06	5.56
45°	2.214	0.746	2.97	0.003	9.15	2.12	39.52
90°	2.811	0.575	4.88	0.000	16.62	5.38	51.35
135°	1.625	0.972	1.67	0.095	5.08	0.76	34.10
Key Piece Length*	0.801	0.196	4.08	0.000	2.23	1.52	3.27
Logit (Complete/Partial)							
Constant	-2.255	0.469	-4.81	0.000			
Key Piece Orientation (Ref: upright)							
0°	0.787	0.508	1.55	0.121	2.20	0.81	5.95
45°	1.708	0.603	2.83	0.005	5.52	1.69	17.99
90°	1.932	0.447	4.33	0.000	6.91	2.88	16.57
135°	1.747	0.666	2.62	0.009	5.74	1.56	21.17
Key Piece Length*	0.644	0.176	3.65	0.000	1.90	1.35	2.69
Logit (Active/Partial)							
Constant	-2.832	0.573	-4.95	0.000			
Key Piece Orientation (Ref: upright)							
0°	0.411	0.698	0.59	0.556	1.51	0.38	5.93
45°	1.666	0.726	2.29	0.022	5.29	1.27	21.98
90°	2.371	0.524	4.52	0.000	10.70	3.83	29.92
135°	2.374	0.721	3.29	0.001	10.74	2.61	44.10
Key Piece Length*	0.601	0.197	3.06	0.002	1.82	1.24	2.68
Logit (High Water/Complete)							
Constant	-1.314	0.657	-2	0.046			
Key Piece Orientation (Ref: upright)							
0°	-1.305	1.170	-1.12	0.264	0.27	0.03	2.68
45°	0.506	0.717	0.71	0.480	1.66	0.41	6.76
90°	0.878	0.572	1.54	0.124	2.41	0.78	7.38
135°	-0.123	0.970	-0.13	0.899	0.88	0.13	5.92
Key Piece Length*	0.157	0.147	1.07	0.286	1.17	0.88	1.56
Logit (Active/Complete)							
Constant	-0.578	0.606	-0.95	0.340			
Key Piece Orientation (Ref: upright)							
0°	-0.376	0.757	-0.5	0.619	0.69	0.16	3.03
45°	-0.042	0.715	-0.06	0.953	0.96	0.24	3.89
90°	0.438	0.536	0.82	0.413	1.55	0.54	4.43
135°	0.626	0.730	0.86	0.391	1.87	0.45	7.82
Key Piece Length*	-0.043	0.158	-0.27	0.787	0.96	0.70	1.31
Logit (High Water/Active)							
Constant	-0.736	0.732	-1	0.315			
Key Piece Orientation (Ref: upright)							
0°	-0.929	1.265	-0.73	0.462	0.39	0.03	4.71
45°	0.548	0.827	0.66	0.508	1.73	0.34	8.75
90°	0.440	0.635	0.69	0.488	1.55	0.45	5.39
135°	-0.749	1.008	-0.74	0.457	0.47	0.07	3.41
Key Piece Length*	0.199	0.170	1.17	0.241	1.22	0.87	1.70

Table 5. 4 -Ordinary logistic regression table for the changes between logjam types associated with key piece length and orientation, coefficient (β) is effect size

In order to analyse specifically how logjam volume varies in response to key piece classification alone a one-way Anova test was performed against log transformed logjam volume and key piece classification (R-squared=15.87%, $p<0.001$). At 95% confidence intervals logjams with Active bridge and Bridge key pieces are larger than other logjams. Logjams with Upright key pieces were smaller than all other logjams, with Submerged, Partial bridge and Parallel key pieces also showing a trend as smaller than Ramp, Bridge and Active bridge key piece logjams.

5.4.5. Hydraulic effectiveness

Gregory logjam types in Figure 5.2 are based on the hydraulic influence of the structure; therefore comparisons between logjam types can be used as a proxy for hydraulic effectiveness in order to answer the fourth question. Figure 5.6 illustrates the variations in the distribution of key piece type between the different logjam types. Each class of logjam is characterised by ≥45% of logjams formed by a single positional type of key structural pieces; Active and Complete logjams are a high proportion of Ramp key pieces, High Water logjams have a majority of Bridge key pieces and Partial have a high proportion of Upright key pieces. For all logjam types the majority of key pieces are either orientated vertically or perpendicular, with a high degree of variability in orientation percentage between the logjam types. A Chi-squared analysis was performed (Agresti, 2002) showing a significant relationship between key piece classification and Gregory logjam type (Chi-Sq = 118.466, df = 15, $p < 0.001$), as there are only five Active bridges these were included in the Bridge category for the analysis. There is little difference in the key piece position distribution between Active and Complete logjams; this is unsurprising as the only difference between these logjam types is their porosity, typically a function of the packing with fine organic material. Active and Complete logjams are characterised by a statistically significant higher proportion of Ramp key pieces (58%, Figure 5.6a) and fewer Upright and Parallel key pieces than is expected from the chi-squared distribution. High Water and Partial logjam types display marked differences in key piece position between each other and compared to Complete/Active logjams. High Water logjams have a very high proportion of Bridge key pieces (56%) and relatively few Ramps and Submerged key pieces, Partial logjams have a high proportion of Upright (45%) and Submerged key pieces and fewer Ramps or Bridge key pieces than is expected from the chi-squared distribution. Figure 5.6b indicates there is a high proportion of key pieces perpendicular to the channel and upright and relatively few key pieces orientated parallel or angled to the flow.

In order to identify if other characteristics of the key structural piece, in addition to position, are associated with Gregory logjam type an ordinal logistic regression analysis

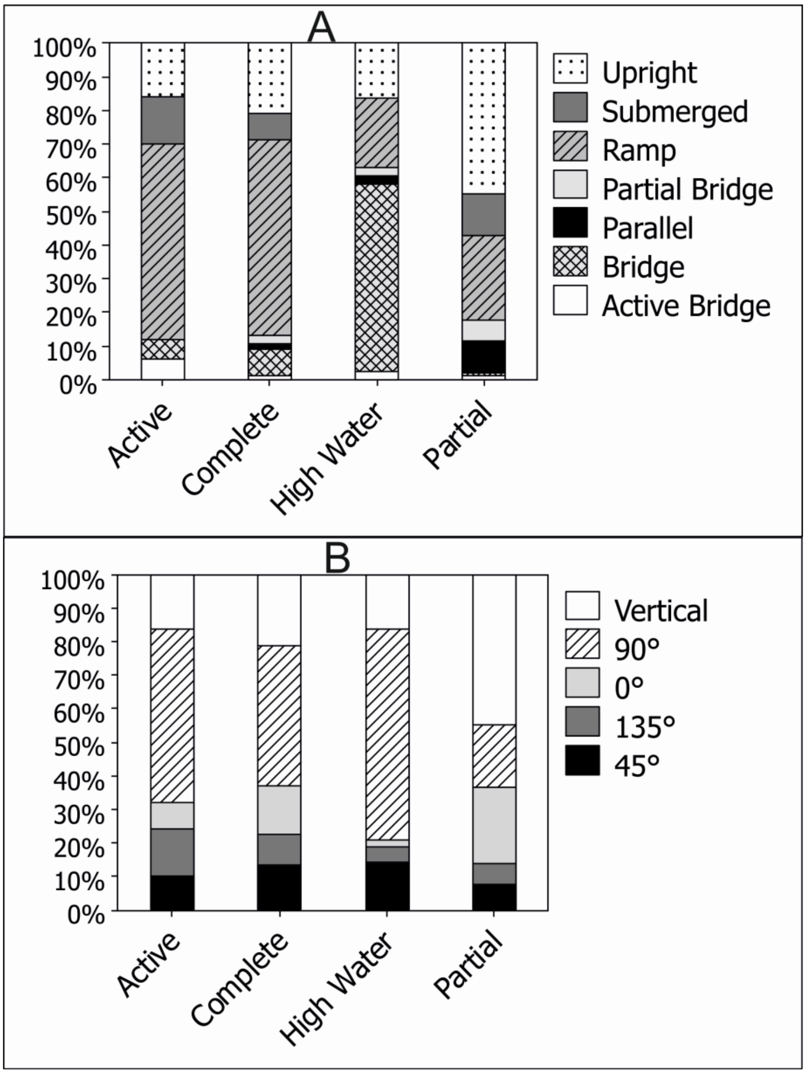


Figure 5.6 - a) position of key structural piece for each class of Gregory logjam, b) Angle of key structural piece (relative to the flow direction) for each class of Gregory logjam.

was performed (Hosmer and Lemeshow, 2000) using logjam type as the response. Length and diameter of the key piece are converted to dimensionless units by dividing key piece length by reach average bankfull width and key piece diameter by reach average bankfull depth. The model with all key piece predictors included indicates at least one of the predictors is statistically significant ($D = -308.538$, $G = 90.889$, $df = 24$, $p<0.001$). The Wald statistic, and associated p-values compared for the individual logits identifies only key piece orientation relative to the channel and key piece length as statistically significant predictors of logjam type at $\alpha=0.05$.

A more parsimonious model with only key piece length and orientation included is statistically significant ($D=-315.322$, $G = 77.321$, $df = 15$, $p < 0.001$), indicating at least one predictor is significant. The individual logits show there is a difference between Partial logjams and the other three types; showing an increase in length is statistically significant giving an increase in odds ratio of 1.82-2.23 for a change from a Partial jam ($p\leq0.002$); this means with an increase in key piece length of 1m the odds of the logjam being High water, Active or Complete compared to Partial is approximately doubled. Using a reference event of a vertical key piece a change in key piece orientation relative to the flow direction to either 45°, 90° or 135° increases the odds ratio of the logjam being a logjam type other than Partial, indicating vertical key pieces are closely associated with Partial logjams. A change from a vertical to a perpendicular (90°) key piece increases the odds ratio of the logjam being High water rather than Partial by 16.62 ($p<0.001$), indicating a much higher proportion of perpendicular key pieces are associated with High water logjams than Partial logjams. The individual logits for other pairs (Active/Complete, Active/High water, Complete/High water) do not show any statistically significant predictors. This result indicates there are differences in the key pieces of Partial logjams compared to key pieces for all types of channel spanning jams, but no statistically significant difference between the key pieces for the three types of channel spanning jams (High water, Complete, Active).

5.4.6. Geomorphic and Habitat Features

Binary logistic regression (Hosmer and Lemeshow, 2000) was used to investigate if there is any correlation between the likelihood of logjams having sedimentary structures and habitat features associated with them and the characteristics of the logjam and its key structural piece in order to answer the fifth question. The logistic regression was coded as sedimentary geomorphic feature(s) being present if the logjam was associated with one or more of: pool, riffle, bar or exposed sediment; it was coded as habitat features being present if the logjam was associated with one or more of: pool, riffle, bar, exposed sediment, cover/shade or backwater.

For habitat features the statistically significant predictors were found to be Gregory logjam type ($p \leq 0.035$), key piece dimensionless diameter ($p=0.002$) and presence of a key piece rootwad ($p=0.007$), with the overall model based on these variables a significant predictor of habitat presence ($D=-114.633$, $G=90.143$ $df=5$, $p<0.001$)(Table 5.5). Presence of a rootwad on the key piece has an odds ratio (OR) of 2.81, key piece diameter has OR=179.92, indicating an increase in key piece diameter of 1m would lead to the associated logjam being nearly 180 times more likely to have habitat associated with it. For Gregory logjam type High water logjams were coded as the reference event; Partial logjams had OR=2.43, Complete OR=17.61 and Active OR=122.53, all with $\beta>0$ indicating these OR represent positive changes to habitat presence over a High water logjam.

						95% CI	
Predictor	Coef (β)	SE Coef	Z	P	Odds Ratio	Lower	Upper
Constant	-2.3335	0.6014	-3.880	0.000			
Type_Gregory (Ref: High Water)							
Active	4.8084	1.0911	4.410	0.000	122.53	14.44	1040.01
Complete	2.8682	0.5415	5.300	0.000	17.61	6.09	50.89
Partial	0.8895	0.4229	2.100	0.035	2.43	1.06	5.58
Key Piece D*	5.1925	1.7084	3.040	0.002	179.92	6.32	5119.74
Key Piece Rootwad (Ref: N)							
Y	1.0331	0.3775	2.740	0.007	2.81	1.34	5.89

Table 5.5 - Binary logistic regression table for presence of habitat features associated with logjams using key piece characteristics as predictors

For geomorphic features the statistically significant predictors were found to be key piece dimensionless diameter ($p<0.001$), and Gregory logjam type, although not all logjam types were significantly different from each other (Active and Complete logjams, $p<0.001$; Partial logjam, $p=0.246$) and with the overall model a statistically significant predictor of geomorphology ($D=-128.11$, $G=104.774$, $df=4$, $p< 0.001$) (Table 5.6). The model shows that only Active and Complete logjams are significant predictors of geomorphic features, of these Active logjams have a very large OR=166.63, and Complete a smaller OR=7.72. Partial logjams are not statistically significant within the logjam classification and thus appear to be fairly similar to High water logjams in terms of association with geomorphic features. Key piece dimensionless diameter has an OR=245.06 indicating an increase in key piece diameter of 1m will increase the odds of the associated logjam having a geomorphic feature by 245 times.

						95% CI	
Predictor	Coef (β)	SE Coef	Z	P	Odds Ratio	Lower	Upper
Constant	-2.090	0.500	-4.18	0.000			
Type_Gregory (Ref: High Water)							
Active	5.116	1.083	4.72	0.000	166.63	19.94	1392.68
Complete	2.044	0.459	4.45	0.000	7.72	3.14	19
Partial	0.506	0.436	1.16	0.246	1.66	0.71	3.9
Key Piece D*	5.502	1.416	3.88	0.000	245.06	15.26	3935.29

Table 5.6 - Binary logistic regression table for presence of geomorphic features associated with logjams

5.5. Discussion

The wood loadings reported here are comparable with other studies of streams in deciduous, temperate forest streams (Harmon et al., 1986). The wood loading values for the Highland Water are at the low end of what would be expected for a semi-natural stream in deciduous woodland (Harmon et al., 1986) and are low compared to general deadwood values for European beech forests (Christensen et al., 2005). Given the relative absence of management in the last ten years and the semi-natural state of much of the riparian woodland in the Highland Water it would be expected the wood loading values would be comparatively high. There are two explanations as to why the wood loadings are lower than expected; using average volume of size fractions to estimate wood loadings may under-estimate where the distribution is dominated by a few very large pieces of wood as is the case in some reaches (Meleason et al., 2007). Furthermore the legacy of previous woodland management practises, including harvesting and de-snagging of channels, as well as overgrazing limiting forest regeneration (Bond, Unpublished; Grant and Edwards, 2008) may be evident in lower measured wood loadings.

There was a significant difference found between sizes of racked and key pieces in logjams with racked pieces shorter (Figure 5.3), this is has been shown in other systems such as large rivers in the US Pacific North-West (Montgomery et al., 2003). Racked pieces made up the greatest proportion of wood pieces in size fractions less than one channel width in length (>5m) and there were virtually no racked pieces in excess of two channel widths in length (>10m). This indicates the system is transport limited and that with increasing piece length there is a reduced likelihood of mobility and subsequent trapping in logjams. The proportion of key pieces increases with increasing length, almost all key pieces are longer than the average channel width and between one to two channel widths in length the proportion of large wood acting as key pieces increases from of 43% to 85%.

The locations of logjams were not found to be particularly sensitive to variations in reach scale geomorphology or drainage area. Although a regression model was shown to be significant with sinuosity, channel width and wood loading all good predictors of logjam frequency, over 80% of the variance was not explained by the model, indicating that although the variables are significant predictors the model is not particularly sensitive to them. Previous studies have indicated geomorphic variability is important for controlling log mobility and constrictions acting as foci for logjam formation, however in a system transport limited with respect to large wood such as the Highland Water, existing logjams and stable piece of large wood are the most important trapping points for mobile wood in the channel; this has already been reported with respect to small wood (Millington and Sear, 2007). Although some wood is recruited to the channel by channel migration and erosion, wood input to the channel is dominated by wind throw (Brown, 1997). The inputs of large stable pieces of wood are stochastic and chronic (Wallerstein and Thorne, 2004), and the location of such pieces are the dominant process governing the locations of logjams in a transport limited system such as the study site.

Stable pieces of large wood in the channel are potential key pieces in a logjam, we have shown there is a great deal of variability in the size and orientation of logjam key pieces and that not all pieces of wood that could be key pieces develop a logjam around them. The results show specific characteristics of large wood acting as key pieces are closely associated with types of logjam around them; large pieces of wood falling across the channel as Bridges either fail to form a logjam, or tend to form one that only traps pieces of wood near to the bank top. Conversely wood that is orientated with one end in the channel and one end on a bank (Ramp) tend to form large channel spanning jams. Bridge key pieces have been hypothesised as being particularly stable and staying in place until they decay (Jones et al., 2011), this is partly due to the rarity of flood events which generate an excess depth of water over bankfull discharge needed in order to float the bridging log and instigate transport. Living, upright trees have been shown to be important sites for trapping mobile wood (Sear et al., 2010); however such trapping points tend to form Partial logjams, typically trapping wood only against the bank from which the tree is growing, rather than across the whole channel. Results show whether the key piece is alive is not a statistically significant variable (p=0.067), as the regression model appears to be a reasonable solution for the observed variance on most diagnostic measures, it seems likely, given previous studies (e.g. Sear et al., 2010) suggesting alive key pieces are associated with logjam formation, that this study is not sufficiently powerful to return a statistically significant result for these variables, potentially indicating a greater sample size is needed. Wallerstein and Thorne (2004) described how windblown trees have a random fall orientation and position, whereas those recruited through bank erosion tended to topple forwards into and across the channel or slump backwards as they are undermined. As the study river has channel widths much smaller than average tree height this is likely to mean trees recruited to the channel through bank retreat will form Ramp or Upright orientated large wood. The dominance of Ramp and Upright key pieces shown in Figure 5.4 is likely to be a combination of a greater delivery of these pieces through bank erosion but also the preferential trapping of mobile pieces of wood by stable wood orientated in this way. Thus their dominance of the key piece distribution is a combination of the greater abundance of piece of wood orientated in this way throughout the system and the greater effectiveness of such pieces of wood to act as foci for logjam formation.

The relationships between key pieces and the logjams formed around them cannot be completely explained by stability alone particularly given the variations in logjams formed around Bridges compared to Active bridges. Rather, the patterns of different logjam types forming around particular types of key piece are likely to be due to a combination of key piece stability and the potential for these key pieces to trap and retain mobile wood in the channel; key pieces which are both stable and have a large cross-sectional area presented perpendicular to the free-surface of the flow during the rising and falling limb of a flood wave (e.g. Ramps) will have the greatest chance of generating collisions with mobile pieces of wood in the channel, this enhances their chances of trapping racked pieces of wood and forming logjams which fill the channel.

The logjam classification system introduced by Gregory et al. (1985) has been assumed to be a proxy for both size and geomorphic effectiveness of logjams. We have shown Partial logjams are smaller in extent and have less pieces of wood than other types, and that High water logjams have less pieces of wood than Active or Complete. We did not find any significant difference in either the extent or the number of pieces of wood between Active and Complete logjams, this similarity between the core architecture of Complete and Active logjams indicates the two types may be very similar to each other. The only difference between Active and Complete logjams is their degree of porosity which is primarily a function of the amount of packing with leaf litter and fine woody debris, with the lower porosity of Active logjams creating a step in the water profile. The input of leaf litter is a seasonal process in deciduous woodland and is likely to show temporal variability with the highest input of material and thus lowest porosity occurring from October to December after leaf fall and before leaf packs break down and are mobilised in winter floods (Sear et al., 2010). Hence all channel spanning jams exhibit similar architecture and some individual logjams will shift between Active and Complete types on a seasonal basis.

Geomorphic and habitat features associated with logjams have been shown to be significantly correlated with the Gregory logjam type, with Active logjams displaying very high odds of having such features associated with them compared with other logjam types and Complete logjams showing a small increase in odds. Conversely logjam size, measured as either volumetric extent or number of pieces of wood showed only weak relationships to presence of geomorphic and habitat features. This result demonstrates hydraulics and the presence of a channel spanning jam are much more important in logjam function than logjam size. Individual large jams would be expected to contain some pieces of wood which would deflect flow and create vortices and slack water zones which would add heterogeneity to the pattern of sediment erosion and deposition and create geomorphic features (Montgomery, 2003), however these findings illustrate such effects are less noticeable at a catchment scale compared with individual logjam hydraulics.

The findings in this study can be synthesised to give insight into best practise recommendations for using engineered logjams as part of river rehabilitation. Most current guidance prioritises the stability of introduced large wood (e.g. Fischenich and Morrow Jr, 1999; Lewis, 2010), and advocates the placement of wood either as bridges (e.g. Lewis, 2008), angled to the flow and anchored to the bed/banks as a form of groyne (e.g. Lewis, 2008; Mott, 2006; Seehorn, 1992), or parallel to the bank to provide erosion protection (e.g. Lewis, 2010). This guidance does not reflect the natural distribution of logjam key pieces which we have shown is dominated by Ramps and Upright pieces. We have shown Bridge, Parallel and Submerged key piece have relatively poor performance in regards to forming channel spanning jams. If the objective of a restoration project is to increase the abundance of a target species which is habitat limited the key outcome should be an increase in the provision of habitat suitable for all life stages of the species as well as habitat suitable for their food sources. In the case of salmonids this is typically achieved by increasing geomorphic heterogeneity which increases habitat diversity; we have demonstrated for restoration of headwater streams this will have a higher chance of success if channel spanning jams are installed. Restoration guidelines based on these findings are that engineered channel spanning logjams should be anchored with Ramp orientated key pieces at least one, but preferably over two channel widths in length, typically orientated approximately perpendicular to the flow direction in order to mimic processes that occur naturally. Alternatively in systems with naturally high wood loadings and where river managers wish to increase the frequency of logjams the channel can be seeded with potential Ramp and Upright orientated key pieces a minimum of two channel widths in length this will lead to a variety of logjam types gradually forming at these points by trapping mobile pieces of wood in the channel.

5.6. Conclusion

This chapter has demonstrated the importance of a key structural piece in the formation of logjams and in small forest streams has shown the characteristics of key pieces are linked to the types of logjams that form around them. Logjam location within the catchment is not predominantly linked to changes in channel dimensions, sinuosity or catchment area, rather the stochastic distribution of wind-thrown pieces of large wood act to anchor the majority of logjams. We have shown key piece size and orientation to be the most important factors linking key pieces to logjam types with Ramp orientated key pieces associated with channel spanning jams and Upright key pieces preferentially forming Partial logjams. Factors previously hypothesised as important for increasing key piece stability, such as a complex branching structure are not identifiable as significant in small streams.

Gregory logjam types have long been used to classify logjams in small streams and remain an appropriate and useful method of differentiating between logjams of differing form and function. We have shown these classifications display a weak relationship to logjam size with Partial jams smaller than other types. There is a strong relationship between Gregory logjam type and logjam function; Active and Complete logjams have much higher odds of having geomorphic and habitat features associated with them, conversely there are only weak relationships between logjam size and function. These findings illustrate that hydraulics are much more important than logjam size in determining logjam function.

River rehabilitation guidelines have been proposed for more effective habitat creation using logjams as an initial measure, with large Ramp orientated key pieces identified as more likely than other types of key piece to create a stable logjam which generates habitat diversity. These findings and recommendations will need to be tested in the field at reach to catchment scale to confirm logjam structure and function can be predicted from key piece type.

6. The influence of geomorphology on large wood dynamics in a low gradient headwater stream

6.1. Abstract

The mobility of large pieces of wood over inter-annual timescales within natural channels remains poorly quantified. The stability of individual pieces of large wood is a prerequisite for a variety of wood-mediated ecological and hydromorphology benefits, notably the provision of suitable habitats for macroinvertebrates and fish.

In this study individual pieces of naturally occurring large wood were tagged and surveyed over a 30 month period within a third and fourth order lowland forest river. Individual pieces of wood were found to be highly mobile, with 75% of pieces moving during the survey period, and a maximum transport length of 5.6km. Multivariate analyses identified piece length as an important factor in explaining likelihood of movement ($p=0.068$) and transport distance ($p=0.063$). Graphical analysis shows a threshold of 1.5 channel widths in length for piece mobility with fewer pieces moving above this size; binary logistic regression shows as piece length increases, the odds ratio of movement decreases by 1.56 for each dimensionless length unit. Species type, branching complexity, location and diameter were found to be important in determining mobility in some contexts.

Where logjams persist over multiple years in the same location they were shown to be reworked with component pieces being transported away and newly racked pieces added to the structure. Due to similar architecture recurring logjams can appear to be stable but are in fact dynamic features with a turnover of racked pieces.

The findings of this study have implications for river management policy. Currently river managers determine, largely based on personal experience, whether or not to remove pieces of large wood from a river or from against structures to prevent damage to infrastructure if these pieces move during floods. By quantifying what sizes of wood are likely to be transported during floods, removal can be targeted to these pieces ensuring effective river management and potentially saving money.

6.2. Introduction

Large wood within forested streams is recognised as a crucial component of vibrant and healthy aquatic ecosystems. As both individual pieces of wood and as accumulations (logjams), large wood acts to trap and store sediment (Brummer et al., 2006) and organic matter (Bilby, 2003; Collins et al., 2002), dissipate flood wave energy (Gregory et al., 1985; Sholtes and Doyle, 2011; Thomas and Nisbet, 2012) and create and maintain greater geomorphic diversity (Abbe and Montgomery, 1996; Gurnell et al., 2000; Sear et al., 2010) which in turn provides habitat and refuges for a variety of aquatic and terrestrial organisms (Collins et al., 2012). Wood acts as an autogenic ecosystem engineering component (Jones et al., 1994) to: increase the frequency and depth of pools in the presence of logjams which are important refuges for salmonids (Abbe and Montgomery, 1996; Collins et al., 2002; Montgomery et al., 1995), cause variations in hydrodynamics caused by flow over wood leads to deposition of gravels suitable for salmonid spawning (Wheaton et al., 2004), and provide habitat and a food source for a variety of macro-invertebrates (Benke and Wallace, 2003; Harmon et al., 1986).

Many habitat creation effects of in-stream large wood are a result of wood mediated variations in local hydraulics which result in altered patterns of sediment erosion and deposition and changes to local geomorphology. For large wood to influence local erosion and deposition patterns it needs to be retained in a stable position long enough for the hydraulic changes it imposes to alter local geomorphology (Millington and Sear, 2007), therefore the stability of large wood in a system is a prerequisite for many ecological benefits (Millington and Sear, 2007). Historically, large wood has been removed from many rivers (Brooks et al., 2004) but more recently river restoration programmes have used artificial emplacement of large wood in an attempt to improve the ecohydromorphological conditions in impaired aquatic ecosystems (e.g. Brooks et al., 2004; Collins et al., 2002; Reich et al., 2003; Shields Jr et al., 2006). Given the importance of large wood stability in relation to proving ecological benefits it is necessary to understand the factors influencing and controlling large wood mobility, both to understand natural systems and for design of artificial wood emplacement schemes.

The stability of large wood has been linked to the concept of 'reach retention'; the ability of a river to trap and retain organic and inorganic matter (Millington and Sear, 2007) such as mobile wood in the channel. Reach retention and trapping of large wood is dependent on the geomorphological complexity of the channel (Braudrick et al., 1997; Millington and Sear, 2007; Sheldon and Thoms, 2006), the frequency of in-channel obstructions such as boulders (Bocchiola et al., 2006b) and logjams (Bilby and Likens, 1980; Bocchiola et al., 2006b; Daniels, 2006; Ehrman and Lamberti, 1992; Millington and Sear, 2007). The abundance of logjams in a channel is proportional to the wood loading (Wohl and Cadol, 2011) and thus in any channel the mobility of large wood should be inversely proportional to the complexity of the channel planform and the wood loading to the channel.

Previous studies of wood mobility have found wood length to be important, with ratios of large wood length to channel width of 1:1 found to define a threshold of mobility below which wood is highly mobile (Braudrick et al., 1997; Gurnell et al., 2002; Lienkaemper and Swanson, 1987). Some studies finding the importance of piece length in wood mobility have used uniform, straight pieces of dowel with equal density as tracers (e.g. Bocchiola et al., 2006a; Braudrick et al., 1997; Ehrman and Lamberti, 1992) thus controlling for other potential variables which may be important and potentially underestimating the importance of these factors. A threshold of wood length to channel width of 1:1 has a physical basis in defining the upper size limit for freely mobile wood within a confined channel, as below this length wood can easily rotate within the channel in response to drag and lift forces to a preferential position for transport (Braudrick et al., 1997). However the large wood length to channel width threshold of 1:1 does not have a physical basis as a minimum threshold for functional immobility in unconfined channels connected to their floodplain. Wood has a density around half that of water (Zanne et al., 2009) and thus floats at or just below the free surface. During a high flow event in a channel connected to its floodplain, the water depth across the floodplain may increase to the point where large wood can float above the top of the channel banks, at this point geomorphological constrictions within the channel are not restricting mobility and transport initiation is purely a function of the balance between buoyant and drag forces acting on the large wood piece to mobilise it and its resistance to these forces, of which piece length is only one determinant along with diameter (Haga et al., 2002) and density (Gurnell et al., 2002). Large wood of length approximately equal to or greater than channel width would be increasingly likely to become trapped or wedged in channel constrictions and against upright trees (Bocchiola et al., 2006b), but only wood with a sufficient submerged weight to resist the largest drag and lift forces produced by the river will be immobile under all river discharges. Keim et al (2000) monitored a series of artificially inserted conifer "key pieces" in third order Oregon streams with dimensionless length L* (piece length divided by channel width) of 0.9-1.7 and found that every piece moved during a three year period, concluding these logs were not large enough to be stable in high flows. Conceptually the wood delivered to a given stream can therefore be divided into three broad size classes; wood sufficiently large to be functionally immobile due to its weight, intermediate wood for which mobility is dependent on local geomorphology and smaller wood which is highly mobile independent of local geomorphology (Gurnell et al., 2002).

Despite the importance of understanding large wood mobility in natural environments direct field measurements of wood transport remain relatively rare (Bertoldi et al., 2013), there are relatively published few short-term (<10 years) monitoring data sets (Wohl et al., 2010) and limited research into wood dynamics (Daniels, 2006). Therefore wood remains an incompletely quantified component of river systems (MacVicar and Piégay, 2012). Although flume studies using wooden dowels to simulate large wood pieces are valuable in understanding some of the mechanisms of transport (e.g. Bocchiola et al., 2006a; Braudrick et al., 1997) field studies in varied settings are needed to better understand reach scale wood transport and retention in natural rivers (Collins et al., 2012; Latterell and Naiman, 2007). Wohl et al (2010, p.623) conclude "datasets .. of wood dynamics through time are extremely valuable in understanding temporal variations in wood recruitment, retention and function, and there is a great need for more of them".

The primary controls on large wood mobility are the balance between hydrodynamic forces acting on the log and the stabilising effect of gravity anchoring the piece (Curran, 2010) and the presence of trapping points within the river; thus in any given location mobility is a balance between river discharge and large wood size (Bilby, 1984;

Bocchiola et al., 2006b). The balance of large wood size and river discharge means wood that is stable in a given flow event may be mobilised under a larger discharge. Individual pieces of large wood may therefore be stable on annual or intra-annual time scales, but mobilised on annual or decadal time scales depending on hydrological conditions (Collins et al., 2012). Individual pieces of large wood may be mobilised by moderate discharges shortly after recruitment to the aquatic environment and be subsequently deposited during this event at a trapping point, before being subsequently remobilised by a larger event. In the time between entering the fluvial system and decaying or being exported out of it, a single piece of large wood can move through the system in a series of staggered steps in response to discharge events of varying magnitude (Latterell and Naiman, 2007). During an inter-flood period the distribution of large wood within any given reach will consist of newly recruited, unorganised large wood as well as wood with longer residence times which has already been subjected to varying magnitude high flow events. In order to understand dynamics of large wood in natural conditions it is necessary to conduct repeat surveys in study reaches to identify pieces of wood as they move through this cycling.

There is a clear research gap in knowledge of large wood mobility in small, low gradient river channels over multiple years. In order to address this gap in the literature this study is conducted over three winters on third and fourth order river reaches of gradients 0.005-0.008 m/m and average widths of 4-5m. To date there are no such published studies in the literature.

6.2.1. Aims:

The overall aim of this study is to examine the relationships between large wood mobility and river discharge in a small lowland forested river. The key objective is to understand which pieces of wood in a river channel are more mobile and once mobilised what their transport distance is before being re-deposited.

In order to address this key objective data will be collected on the position of individual tagged pieces of wood, as well as the physical characteristics and geomorphological setting of each piece. Analysis of these characteristics along with piece mobility will enable specific objectives to be addressed. Specific objectives are; i) to examine the effects of geomorphology in governing large wood mobility and in trapping mobile wood in the channel, ii) to identify through a multivariate analysis those factors, including size, which contribute to the mobility of pieces of large wood, iii) to estimate transport distances for mobilised large wood.

6.3. Methods

6.3.1. Study Site

See section 5.2.2 for full site description.

6.3.2. Study Reaches

Five study reaches of the Highland Water, each of approximately 150 metres stream length, were chosen as study sites for large wood mobility (Table 6.1, Figure 6.1). These five reaches are representative of the broad range of geomorphological planform types found within the Highland Water, this allows comparisons to be made between two natural sinuous reaches, a reach in which the sinuous planform has been restored, and two artificially straightened and channelized reaches.

Site A is a second order tributary of the Highland Water with a semi-natural meandering planform, there is evidence of bed downcutting. The floodplain is a mixed, mature woodland.

Site B is a restored third order reach, connected to its floodplain and experiences frequent over bank inundation. The floodplain has only sparse tree cover with the riparian zone dominated by stands of alder (*Alnus glutinosa)*.

Site C is an artificially straightened 3rd order reach flowing through a former plantation enclosure. The channel is considerably wider than would be expected (See Table 6.1 for a sequential list of mean channel widths) and is disconnected to its floodplain which is dominated by an even aged (~80 year old) cohort of beech (*Fagus sylvatica*) with sparse undercroft vegetation.

Site D is a semi-natural 3rd order reach flowing through mixed mature woodland with an extensive network of ephemeral floodplain channels.

Site E is an artificially straightened 4th order reach flowing through a former beech plantation. The channel is substantially over deepened and over wide and is completely disconnected from its floodplain, which has a mixed woodland of mature beech and oak with isolated other species and virtually no undercroft vegetation.

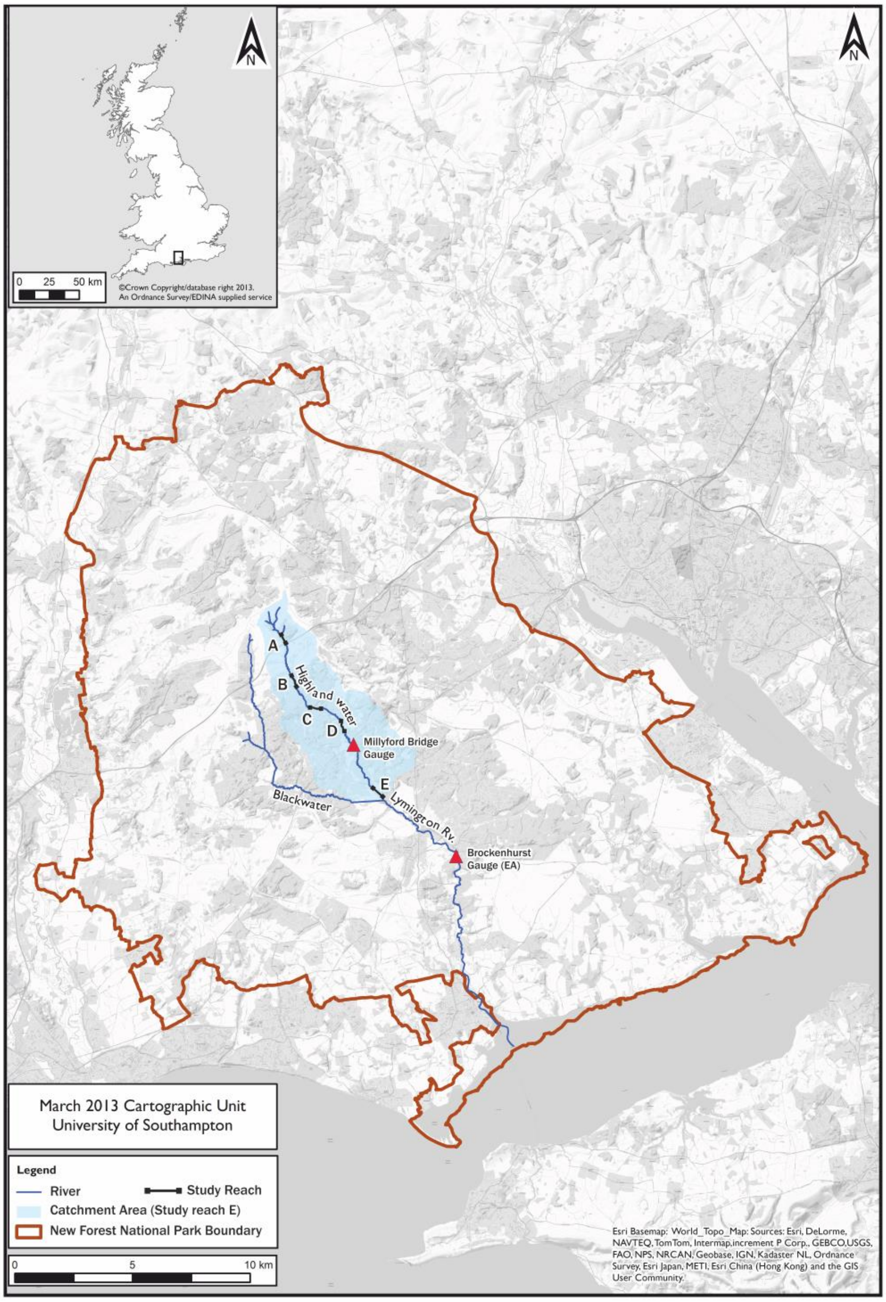


Figure 6.1 - location map showing study reaches, catchment area draining to further downstream study reach and location of hydrometric gauges

	Average bankfull Width	Average bankfull depth	Sinuosity	Strahler Reach Order
Reach A	3.00	0.80	1.75	2^{nd}
Reach B	3.50	1.15	1.62	3^{rd}
Reach C	4.75	1.04	1.02	3^{rd}
Reach D	3.36	1.28	1.54	3^{rd}
Reach E	4.38	1.78	1.01	4^{th}

Table 6.1 – summary reach characteristics.

6.3.3. Large Wood tagging

Within each of the study reaches during August 2010 each piece of large wood (defined as wood having both a diameter at least 10cm and a length at least 1m) was tagged using numbered, aluminium tree tags and secured with galvanised tree tag nails. One tag was secured to each end of the piece of wood, or as close to the end as practicable.

A coordinate grid was established using an electronic total station, a network of 1m wooden stakes driven into the floodplain surface act as reference points within the coordinate grid, the absolute position of the stakes was established using handheld Global Positioning System (GPS). A relative coordinate system using a total station gives a point accuracy with an error of <0.01m, compared to a mean error of ≤2m using GPS under a forest canopy (Hasegawa and Yoshimura, 2003). Each tag was surveyed into the coordinate grid, in addition a palm top personal data assistant (PDA) was used to record descriptive variables of each piece of wood (Table 6.2). Length measurements were taken with a 10m tape and wood diameter measurements with a tree caliper.

Following the initial survey the position of each piece of large wood was surveyed into the coordinate grid again in October 2010, May 2011, November 2011 and May 2013, with the exception of reach A which was not surveyed in November 2011 and reach E which was surveyed in January 2012 and not in May 2013 due to river restoration works scheduled near this location. In each repeat survey the coordinate grid was re-established using the reference stakes inserted in the initial survey giving a <0.1m error margin on points between surveys. The survey grid covered a distance of approximately 200-300m downstream from the end of each reach, dependent on the degree of blocking of line of sight for the total station by vegetation. If all pieces of large wood were not located inside the survey grid for a given reach a walking survey was conducted from the end of the reach, where logs were located their position was recorded using handheld GPS (accurate to ~2m) (Hasegawa and Yoshimura, 2003). Prior to each survey any newly delivered, untagged pieces of large wood within each reach were tagged and information (Table 6.2) about each new piece recorded using a PDA.

Movement of large wood was calculated by subtracting the surveyed Northing and Easting coordinate points for each tag from the coordinates measured in the previous survey and using Pythagoras theorem to calculate the straight line distance between the start and end points. A straight line method was used as wood was observed during substantial overbank flow to move in a predominantly down-valley direction, floating at or near to the free surface and largely bypassing sinuous meander bends, following the path of ephemeral meander-neck cut-off channels. Instantaneous data collection was not possible and so the travel paths taken by mobile wood pieces were unknown. Given the field observations of movement pathways it was therefore determined that potentially underestimating actual transport distance by using the shortest distance between start and end points was more robust than potentially overestimating transport distance by assuming wood moving during floods had followed the low-flow thalweg. The largest error in point location recorded in any survey was 0.15 metres, therefore any movement calculated as 0.3m or less was considered within the possible margin of error and not counted as movement.

Large wood length and diameter were converted into dimensionless units (length, L* and diameter, D*) by dividing length by local channel width and diameter by local channel depth; channel width and depth measurements were obtained from Chapter 5.

Characteristic	Units/Categorisation
Length	metres
Diameter 1[a]	metres
Diameter 2[a]	metres
Branching Order	Greatest branch order recorded. Order 1 is a main trunk, with each lateral branch having an order number one greater than its parent branch (Wilson, 1966), analogous to Strahler stream order
Fractured end	Root wad, broken, sawn/axe cut, eroded, N/A
Species	Elm, Ash, Oak, Alder, Beech, Holly, Conifer, Birch, Willow, Unknown
Living	Yes/No
Sprouting	Yes/No
Total Length with branches	metres
Rootwad Length	metres
Rootwad Diameter	metres
Decay Class	1-5 using decay class system of (Robison and Beschta, 1990)
Rooted in bed/bank/floodplain	Yes/No
Location of large wood	Informal description
Function of large wood relative to logjam	Key/Racked/Loose/Isolated (where Isolated indicates piece is not in a logjam)
Fine wood racked	Yes/No
Volume of fine wood racked	m^3
Orientation relative to main channel	0-180^o Determined using compass to nearest 45^o where 0^o is aligned to flow direction and facing downstream, 180^o is aligned to the flow direction and facing upstream
Partially buried/anchored with sediment	Yes/No
Magnetic Orientation	0-360^0
In channel length	metres
Geomorphological effect	Yes/No for 15 geomorphological effects in association with large wood.

Table 6.2 – information collected for each piece of wood on the position, orientation, environment and physical characteristics. [a] – diameter was collected at both ends of each piece in order to allow volume to be estimated more accurately.

6.4. Results

Figure 6.2 show the variability in flow discharge during the study and demonstrates the inter-annual variability in discharges. There is a pattern for higher winter discharges than in summer, with both an elevated baseflow and high periodic flood discharges. The winters of 2010-2011 and 2011-2012 were relatively dry and contributed to a widespread drought during the summer of 2012. Conversely the winter of 2012-2013 was exceptionally wet and the hydrograph shows the gauging station repeatedly being drowned out by flood events >10m^3/s in the period November 2012-January 2013.

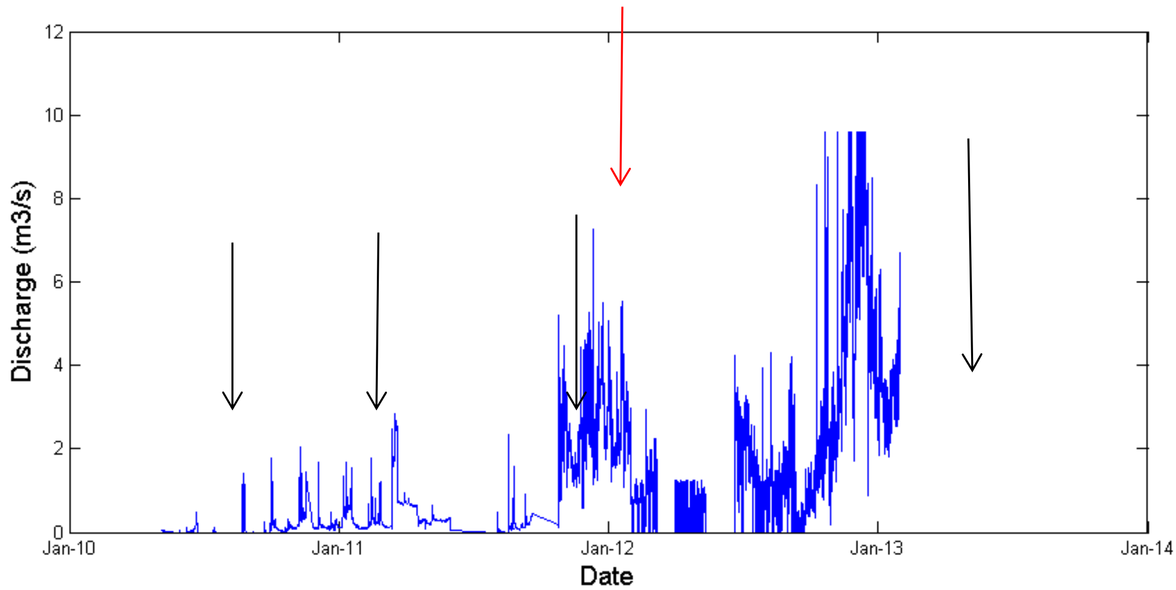


Figure 6.2 – Hydrograph of flows at Millyford Bridge gauging station for the duration of the study, black arrows shows the 4 survey intervals, with the red arrow showing the final survey for reach 5.

Table 6.3 shows the percentage of large wood moving during the 30 months of the survey and between each survey, separated into individual reaches and reach classes, showing that over 3 winter flood seasons around three-quarters of the wood in fluvial and riparian environment moved to at least some degree.

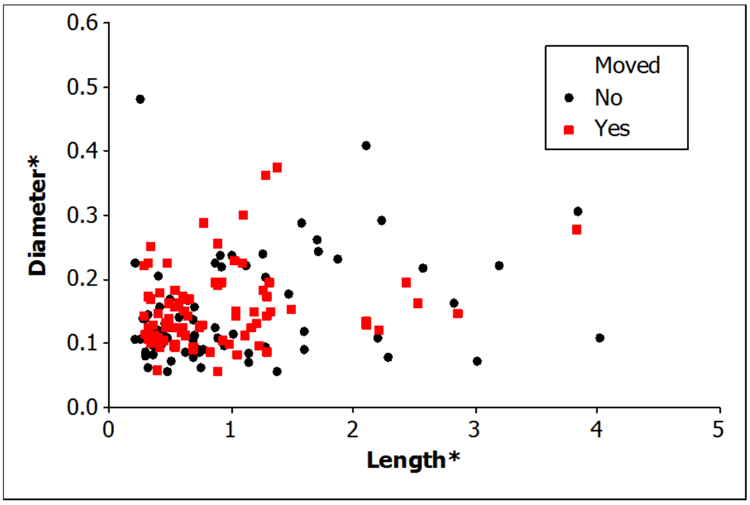
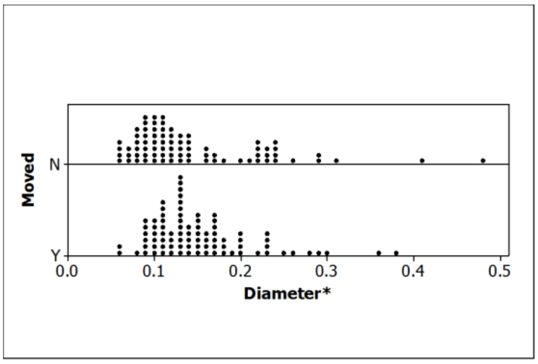


Figure 6.3 – a dimensionless plot of log size showing movement.

Figure 6.3 shows a binary measure of log movement during the 30 month study, these data are corrected to remove tagged logs which could not be found in subsequent surveys due to possible disintegration (having one of the two highest decay classes) or burial (where their previously recorded location has been submerged in sediment). Figure 6.3 shows a larger diameter relative to the channel is not a noticeable impediment to movement and that although increasing length is correlated with fewer logs of a given size moving, logs up to four times the channel width are still capable of some degree of movement.

	Whole study – August 2010 – March 2013		August 2010 – May 2011		May 2011 – October 2011		October 2011 – March 2013	
	Moved	Not Moved	Moved	Not Moved	Moved	Not Moved	Moved	Not Moved
All Sites	75.5%	24.5%	50%	50%	62.1%	37.9%	61.5%	38.5%
Channelized Reaches	67.2%	32.8%	29.4%	70.6%	61.8%	38.2%	70.4%	29.6%
Semi-Natural & restored reaches	80.0%	20.0%	61.3%	38.7%	62.3%	37.7%	63.3%	36.7%
Reach 1 (headwater)	69.2%	30.8%	66.7%	33.3%	N/A	N/A	45.5%	54.5%
Reach 2 (restored)	83.8%	16.2%	61.1%	38.9%	75.0%	25.0%	63.0%	37.0%
Reach 3 (channelized)	75.8%	24.2%	32.1%	67.9%	72.7%	27.3%	70.4%	29.6%
Reach 4 (semi-natural)	83.3%	16.7%	57.6%	42.4%	51.2%	48.8%	75.0%	25.0%
Reach 5 (channelized)	56.0%	44.0%	26.1%	73.9%	45.5%	54.5%	N/A	N/A

Table 6.3 – showing percentage of large wood moving in each survey across all sites (row 1), grouped by reach type (rows 2 & 3) and for each reach individually.



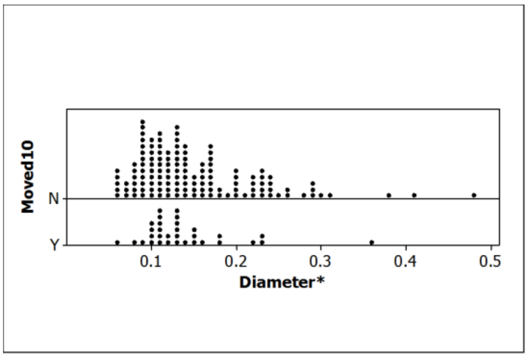
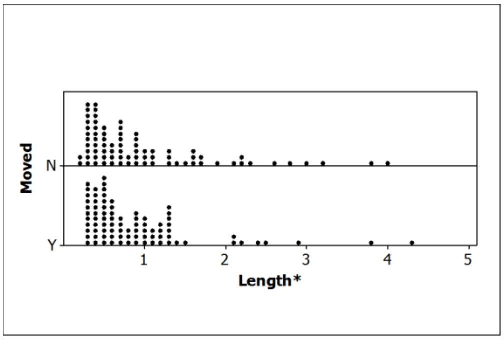


Figure 6.3 – Dot plots showing log movement against dimensionless diameter (D*).



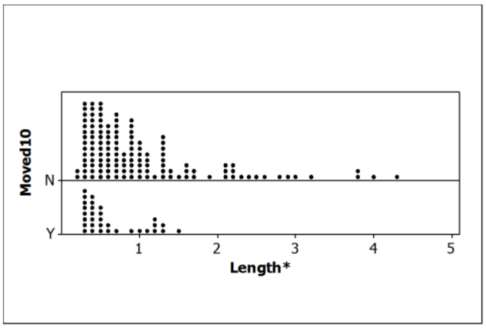


Figure 6.4 – Dot plots showing log movement against dimensionless Length (L*).

Figures 6.3 and 6.4 show the dimensionless diameter and length of logs that have moved (Figures 6.3A and 6.4A) and logs moving 10 metres or further (Figures 6.3B and 6.4B), these figure show that there is not a strong relationship between size and movement, however for logs moving further than 10m there are much stronger relationships between size and mobility, with few large logs moving more than 10m, with thresholds at approximately 1.5 L* and 0.25 D*, these relationships were explored using binary logistic regression (Hosmer and Lemeshow, 2000).

For large wood moving any distance binary logistic regression shows little relationship between movement and L* (p=0.495, n=162) or D* (p=0.507, n=162). For large wood moving 10 metres or further binary logistic regression does show a significant relationship between movement and L* (G=9.663, DF=1, p=0.002, n=162) with an odds ratio of 3.31 for decreasingly likelihood of movement with an increase of 1 in L*. There is not a significant relationship between movement and D* (p=0.344), however the test does have strongly performing goodness-of-fit tests and a high odds ratio (17.82), suggesting there may be a relationship but the sample size (n=162) is insufficient to identify it (Hosmer and Lemeshow, 2000).

6.4.1. Multivariate analysis

The modelling strategy followed for all multivariate regression analyses was to initially build a model including all explanatory variables believed to be important, then to exclude the covariate with highest p-value, and re-run the analysis as an attempt to generate the most parsimonious model possible, where all covariates included have $p<\alpha$ and which still explains a high level of variance. A level of $\alpha=0.1$ was used for the analyses.

Binary logistic regression was used to determine which characteristics of large wood pieces from Table 6.2 were correlated to the large wood movement and large wood moving ten metres or further. Several nominal categorical variables had categories which were poorly represented in the sample (n<5) making them unsuitable for inclusion in the model (Hosmer and Lemeshow, 2000) in these cases multiple categories were combined to ensure n>5 for all categories. Modified coding was used for: branching order (single trunk, branched), species (unknown, conifer, broadleaf), decay class (1+2, 3, 4+5), with function, orientation and location combined into a single nominal category (in main channel, key logjam piece, racked logjam piece, on floodplain).

A binary logistic regression model of large wood moving (Table 6.4) is statistically significant (G=18.767, DF=4, p=0.001). The model shows dimensionless length (p=0.068) and species type (p=0.001/0.051) are good predictors of mobility. Branching complexity was left in the model, despite having a p-value greater than α, as it is indicated in the literature as an important variable controlling mobility, furthermore the model including it has a very low overall p-value and the covariate has a relatively high odds ratio and individual p-value near to α. The measure of association for concordant pairs for the model, which can be thought of as analogous to r-squared is 64.5%.

Predictor	Coef	SE Coef	Z	P	Odds Ratio	95% CI Lower	Upper
Constant	-0.527	0.493	-1.07	0.285			
Length *	0.446	0.244	1.83	0.068	1.56	0.97	2.52
Species							
BRD	-0.783	0.401	-1.95	0.051	0.46	0.21	1.00
CON	-2.380	0.690	-3.45	0.001	0.09	0.02	0.36
Branching							
SGL	0.608	0.435	1.40	0.163	1.84	0.78	4.31

Table 6.4 – Binary logistic regression table for large wood mobility with large wood characteristics as predictors. Tests that all slopes are zero: G=18.767, DF=4, P-value=0.001. Species is coded as UNK-unknown as reference value, with BRD-broadleaf and CON-conifer, Branching is coded as BRN-Branching as reference value and SGL-single stem.

Predictor	Coef	SE Coef	Z	P	Odds Ratio	95% CI Lower	Upper
Constant	-0.440	0.667	-0.66	0.509			
Length*	1.739	0.553	3.15	0.002	5.69	1.93	16.82
Species							
BRD	-1.149	0.504	-2.28	0.023	0.32	0.12	0.85
CON	-1.184	0.612	-1.94	0.053	0.31	0.09	1.01
Branching							
SGL	1.245	0.555	2.24	0.025	3.47	1.17	10.31

Table 6.5 – Binary logistic regression table for large wood moving 10 metres or more, with large wood characteristics as predictors. Tests that all slopes are zero: G = 19.875, DF = 4, P-Value = 0.001. Species is coded as UNK-unknown as reference value, with BRD-broadleaf and CON-conifer, Branching is coded as BRN-Branching as reference value and SGL-single stem.

A binary logistic regression model of large wood moving 10 metres or more (Table 6.5) is statistically significant (G = 19.875, DF = 4, P-Value = 0.001) and has dimensionless length (p=0.002), species type p=0.023/0.053) and branching complexity (p=0.025) as statistically significant predictors. The measure of association for concordant pairs for the model, analogous to r-squared, is 74.2%.

6.4.2. Geomorphological complexity

In order to analyse how geomorphological complexity affects the likelihood of large wood transport the five reaches were separated into two classes; channelized reaches, comprising reaches 3 and 5 and semi-natural and restored reaches (reaches 1, 2 and 4), see Table 6.1 for summary reach characteristics. Table 6.3 shows mobility is higher in the semi-natural and restored reaches (80%) than in the channelized reaches (67.2%). The semi-natural and restored reaches display low variability between surveys with between 61.3-63.3% mobility, in channelized reaches there is greater variability in mobility between the low flow periods of October 2010 to May 2011 (26-32%) and the greater mobility during the subsequent high flow periods (60-70%). Figure 6.5 shows plots of log movement against dimensionless length and diameter for the two classes of reach, this shows log size is an important factor in movement within both types of reach, however in channelized reaches there is no movement for large wood either of length* greater than 2.5 or diameter greater than 0.2.

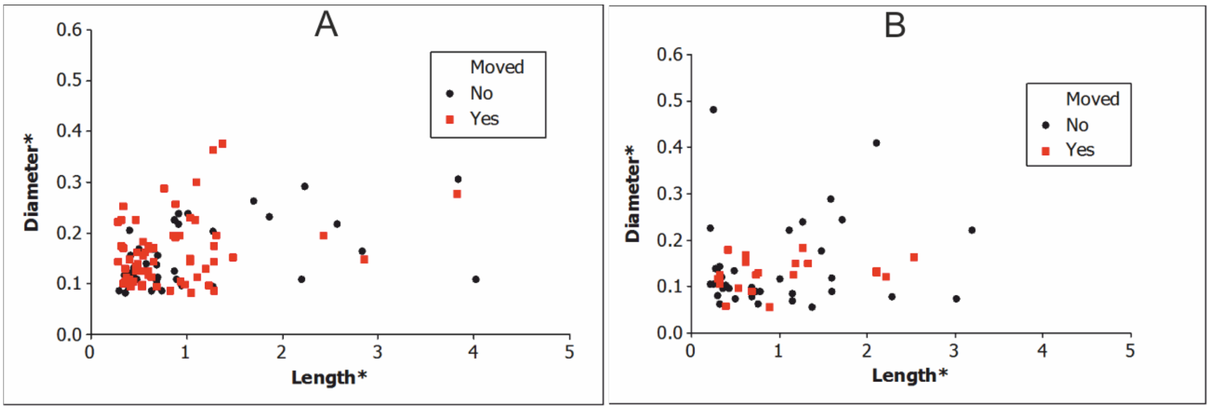


Figure 6.5 – movement of large wood relative to large wood size for A) semi-natural and restored reaches only, B) channelized reaches only.

Binary logistic regression was used to analyse which characteristics of large wood are correlated with both movement and movement of ten metres or more in the two separate reach classes. Table 6.6 shows large wood moving in semi-natural and restored reaches (G = 28.935, DF = 8, P-Value < 0.001) showing dimensionless length (p=0.036), diameter (p=0.064), branching complexity (p=0.021), starting location (p=0.024/0.732/0.615) and species type (p=0.187/0.009) are all significant predictors of large wood movement in these reaches. Table 6.7 summarises the best performing model for large wood showing any degree of movement in channelized reaches (G=8.106, DF=3, P=0.044), with none of the individual covariates showing p-value less than α. These two tables show there are substantial differences in the degree to which large wood characteristics govern mobility in geomorphologically homogenous and complex reaches.

Predictor	Coef (β)	SE Coef	Wald Statistic (z)	P	Odds Ratio	95% CI Lower	95% CI Upper
Constant	-1.124	1.017	-1.11	0.269			
Length*	0.790	0.376	2.10	0.036	2.20	1.05	4.61
Diameter*	-7.857	4.23	-1.86	0.064	0.00	0.00	1.56
Location (Ref; Floodplain)							
In Channel	2.804	1.240	2.26	0.024	16.51	1.45	187.66
Key Piece	0.250	0.730	0.34	0.732	1.28	0.31	5.36
Racked Piece	-0.286	0.568	-0.50	0.615	0.75	0.25	2.29
Species (Ref: Unknown)							
Broadleaf	-0.799	0.606	-1.32	0.187	0.45	0.14	1.47
Conifer	-2.087	0.795	-2.63	0.009	0.12	0.03	0.59
Branching (Ref: Complex branching)							
Single Stem	1.794	0.777	2.31	0.021	6.01	1.31	27.56

Table 6.6 – Binary logistic regression table for large wood moving in semi-natural reaches, with large wood characteristics as predictors. Tests that all slopes are zero: G = 28.935, DF = 8, P-Value < 0.001.

Predictor	Coef (β)	SE Coef	Wald Statistic (z)	P	Odds Ratio	95% CI Lower	95% CI Upper
Constant	1.530	0.664	2.30	0.021			
Length*	-0.474	0.432	-1.10	0.273	0.62	0.27	1.45
Location (Ref: Floodplain)							
Key Piece	0.238	0.776	0.31	0.759	1.27	0.28	5.80
Racked Piece	-1.678	0.728	-2.31	0.021	0.19	0.04	0.78

Table 6.7 – Binary logistic regression table for large wood moving in channelized reaches, with large wood characteristics as predictors. Tests that all slopes are zero: G = 8.106, DF = 3, P-Value = 0.044.

Binary logistic regression was also performed for large wood moving ten metres or more during the 30 month study, in channelized reaches (Table 6.8) the only factor found to be a statistically significant predictor was dimensionless length (p=0.092). For large wood moving ten metres or more in semi-natural and restored reaches (Table 9) dimensionless length (p=0.018), branching complexity (p=0.015) and species type (p=0.006/0.030) were found to be statistically significant predictors, with location and dimensionless diameter not found to be statistically significant predictors (p>0.800) despite being good predictors of initial movement (Table 6.7).

Predictor	Coef (β)	SE Coef	Wald Statistic (z)	P	Odds Ratio	95% CI Lower	Upper
Constant	0.530	0.613	0.86	0.388			
Length*	1.363	0.810	1.68	0.092	3.91	0.80	19.13

Table 6.8 – binary logistic regression table for large wood moving 10 metres or further in channelized reaches. Tests that all slopes are zero: G = 4.291, DF = 1, P-Value = 0.038.

Predictor	Coef (β)	SE Coef	Wald Statistic (z)	P	Odds Ratio	95% CI Lower	Upper
Constant	-0.568	0.866	-0.66	0.512			
Length*	1.663	0.703	2.37	0.018	5.28	1.33	20.94
Branching (Ref: Complex branching)							
SGL	1.748	0.716	2.44	0.015	5.75	1.41	23.37
Species (Ref: Unknown)							
BRD	-1.777	0.645	-2.76	0.006	0.17	0.05	0.60
CON	-1.502	0.693	-2.17	0.030	0.22	0.06	0.87

Table 6.9 – Binary logistic regression table for large wood moving 10 metres or further in semi-natural and restored reaches. Test that all slopes are zero: G = 19.407, DF = 4, P-Value = 0.001. Species is coded as UNK – unknown as the reference event, with BRD-Broadleaf and CON- conifer, Branching is coded as BCH – complex branching as the reference event with SGL – single stem.

6.4.3. Maximum Transport Distance

Of the 162 pieces of large wood tagged a total of 86 mobile pieces of wood were surveyed in new locations giving a minimum transport distance for each piece. Some of these pieces were recorded as having moved in surveys 2 and/or 3, but were then not subsequently found at these new locations in later surveys, suggesting the possibility of further movement in excess of the measured transport distance. The range of transport distance was 0.36m to 5600m; mean transport distance of recorded movement was 148m and a median transport distance of 5m. Of the pieces of large wood with the largest transport distance there were 3 pieces moving around 500m, one moving 700m, one piece moving 2250m and the longest recorded transport distance was 5600m. All the furthest moving pieces were shorter than 2m in length and close to cylindrical in shape.

Figure 6.6 shows the relationship between large wood dimensionless length and transport distance, demonstrating the envelope for 95% confidence in maximum transport distance shows an exponential decrease in maximum transport distance with increasing large wood dimensionless length. Analyses show no patterns or statistically significant relationships between large wood dimensionless diameter and transport distance, furthermore there are no statistically significant differences in transport distances between the channelized and semi-natural reaches. Figure 6.7 shows the transport distance for pieces of large wood in each of the five study reaches; the three furthest moving pieces of large wood were from

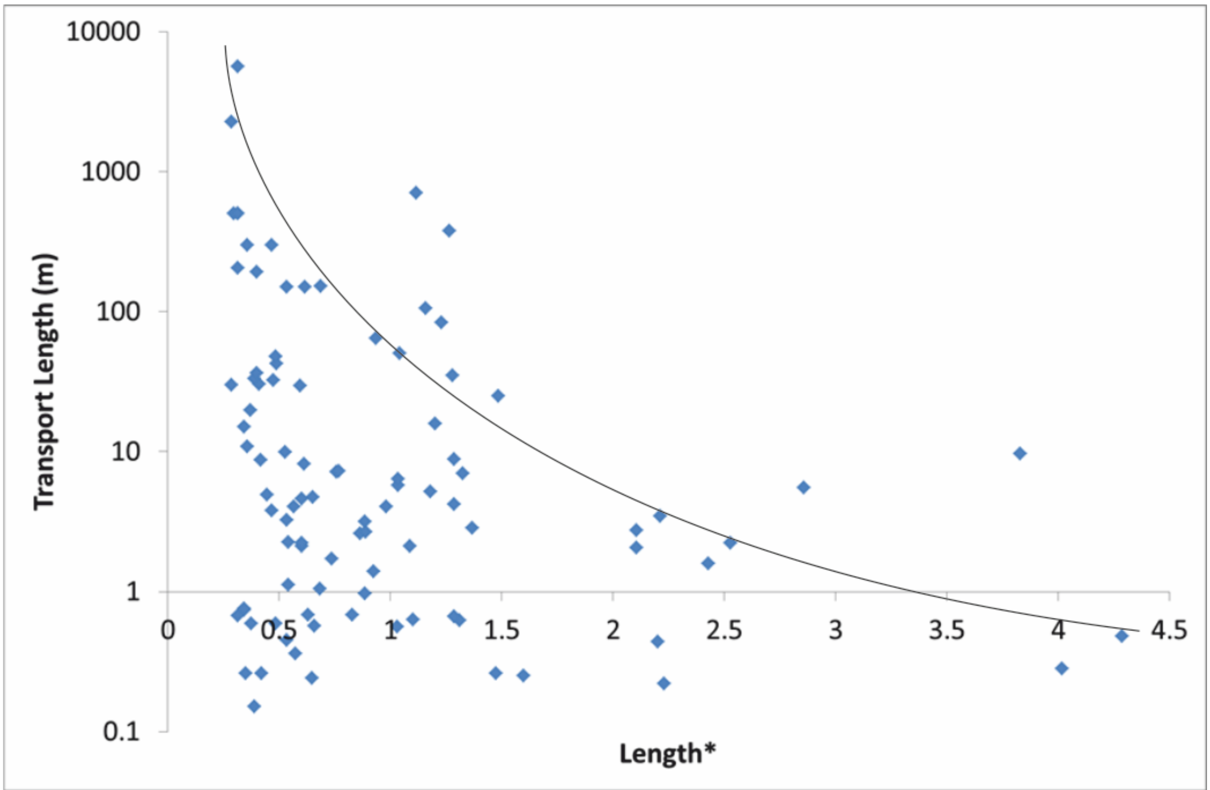


Figure 6.6 – showing the relationship between large wood dimensionless length and transport distance for mobile large wood, the solid line shows the 95% confidence envelope for the relationship between maximum transport distance and dimensionless length.

reach B (restored) and the three other pieces of wood moving 500m or more were from reach C (channelized), this figure shows the vast majority of transport distances are very short across all reaches.

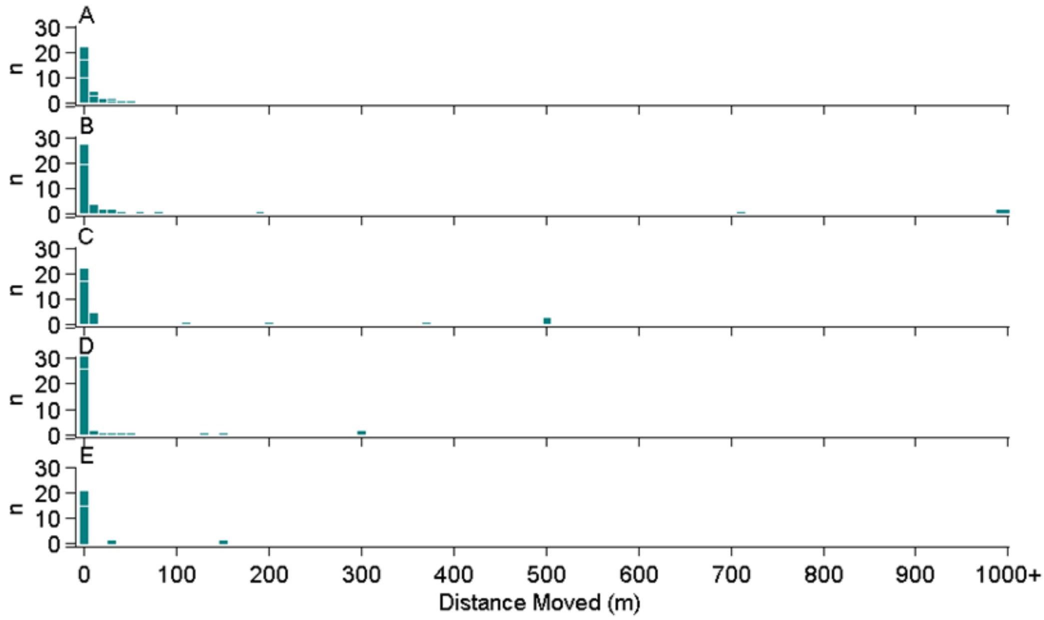


Figure 6.7 – composite histogram showing large wood transport distances across all five study reaches (A-E)

A multivariate general linear regression analysis was performed for transport distance of all large wood against the characteristics of large wood pieces deemed from the literature and observations to be important to mobility and transport distance (from table 6.1). The most parsimonious model has an r-squared (adjusted) = 7.17%, with dimensionless length (p=0.063), species type coded with unknown as a reference event and conifer (p=0.001) and broadleaf (p=0.532) and dimensionless diameter (p=0.259) as coefficients. Dimensionless diameter was used as a coefficient despite a p-value greater than α=0.1, the model performs better with it included and it has a very high coefficient (620.513) and an adjusted sum of squares of 0.74%. The adjusted sum of squares show species type is the largest contributor to the prediction of variance within the model. The individual regression equations are:

Unknown	$TL = 8.42 - 105.50L^{*} + 620.51D^{*}$	6.1
Species Type		
Broadleaf	$TL = 64.85 - 105.50L^{*} + 620.51D^{*}$	6.2
Conifer	$TL = 399.21 - 105.50L^{*} + 620.51D^{*}$	6.3

6.5. Discussion

6.5.1. Relationship between discharge and mobility

The monitoring period for this study encompassed two winter flood seasons which experienced lower than average rainfall and thus lower winter baseflow and less frequent and lower magnitude flood events, leading to drought conditions in much of Southern England. During the final winter flood season there were periods of sustained, heavy rainfall leading to widespread regional flooding and elevated winter baseflow and frequent high magnitude discharge events which drowned out the local gauging station (Figure 6.2). Despite such inter-annual variability in flood frequency and magnitude there was not a great deal of variability in the percentage of logs mobilised between the survey periods (Table 6.3), across all reaches the overall mobility was 75% over 30 months and annual mobility rates were 50-62%. Mobility varies between the two types of reaches; Semi-natural and restored reaches, characterised by higher sinuosity and lower bank heights show almost no inter-annual variability in the proportion of large wood mobilized by varying discharge regimes ranging from 61.3% to 63.3%, compared to channelized reaches, characterised as straight with high bank heights, where 29.4% of large wood mobilised during the driest winter flood season and 70.4% mobilised during the winter with highest flows. Across all individual reaches the mean annual mobility in semi-natural and restored reaches is 61.9% with a standard deviation of 10.5%, compared to channelized reaches with a mean of 49.4% and a standard deviation of 21.5%.

The trend for channels which frequently inundate their floodplains to display much lower variability in annual mobility despite inter-annual variations in flood magnitude and sequencing is contrary to the findings of other studies which have suggested discharge magnitude and timing is an important control on wood dynamics (e.g. Bilby, 1984; Haga et al., 2002), in this study such a control is found only in channelized reaches. Previous studies have suggested that pristine rivers are highly retentive of organic matter (Naiman, 1982), whereas channelized reaches have very low reach retention and act to flush material through the system (Bilby and Likens, 1980; Gregory et al., 1991; Millington and Sear, 2007), these findings suggest a complex pattern of mobility where channel planform and complexity is but one control.

Mobility rates are highly dependent on setting with previous studies reporting annual mobility rates for large wood ranging from 0.8% (Berg et al., 1998) to 95% (van der Nat et al., 2003). Comparisons between mobility studies are difficult due to variations in reporting of reach characteristics and criteria for including large wood, however in broadly similar sized watersheds other studies have found mobility rates of; 89% over 4 months (Daniels, 2006), 17-84% over 20 months (Benke and Wallace, 1990), 0.8-31% annual mobility (Berg et al., 1998), 18% mean annual mobility (Grette, 1985 in Berg et al, 1998) and less than 10% annual mobility (Lienkaemper and Swanson, 1987). The mobility rates reported in this study are higher than most other reported data sets; however all the above studies are from the US and the majority from forests of the Pacific NW, where differences in forest composition and possibly less disturbance to channels and more wood congestion are possible reasons for lower mobility. The mobility levels here do not support the estimate of Gregory (1992) that only 35% of the annual input of wood to New Forest streams is exported out of the system, these findings suggest large wood mobility rates in such temperate lowland rivers are higher than has previously been assumed and may reflect other systems with stable large wood loadings, but a high turnover of individual pieces (Marcus et al., 2002; van der Nat et al., 2003).

Large wood mobility is governed by the balance between stabilising factors anchoring the piece of wood in place and destabilising factors acting to entrain the large wood in the flow. Large wood is mobilised by a combination of buoyant and drag forces and these have been shown to vary with discharge (Shields Jr and Alonso, 2012), with drag reaching a maximum just after flow overtops a piece of large wood as a standing surface wave develops, thereafter as depth increases drag decreases towards a steady state (Shields Jr and Alonso, 2012). Shields Jr and Alonso (2012) suggest in small flashy streams, such as the Highland Water, the maximisation of the sum of applied forces on a piece of large wood will occur early in a flood event, with buoyant forces maximised due to unsaturated wood and drag forces elevated as the piece of wood is overtopped.

Differences in channel geometry between the channelized and semi-natural reaches are a factor in the higher variability of large wood transport within channelized reaches given sequences of high or low discharge. Within the semi-natural reaches fairly moderate discharges equivalent to around 2m^3/s at the gauging station, result in at least some degree of overbank flow, whereas in the channelized reaches even the largest discharges recorded during the monitoring period were confined in-bank. The equation for unit stream power (ω), shows how stream power per unit width varies with slope, discharge and channel width:

$$\omega = \frac{\rho g Q S}{b} \tag{6.4}$$

Where ρ is density of water, g is acceleration due to gravity, Q is discharge, S is channel slope and b is channel width (Bagnold, 1966). Within a confined, boxed shaped channel, such as the channelized reaches, during a flood event discharge will vary whilst slope and width will remain constant, thus unit stream power is proportional to discharge. In a smaller channel connected to its floodplain, such as the semi-natural and restored reaches, unit stream power shows a non-linear relationship to discharge; unit stream power will be approximately proportional to discharge while the flow is contained in bank, however during overbank flow the channel width increases substantially as the floodplain is inundated and unit stream power for overbank flow shows lower increases, or decreases with discharge compared to flow confined in-bank (Figure 6.8). Where flows over the floodplain becomes sufficiently deep, portions of flow may shift to a predominantly down-valley direction, bypassing the channel sinuosity, in this instance the 'channel' slope will increase leading to a greater increase in unit stream power for the overbank portion of flow.

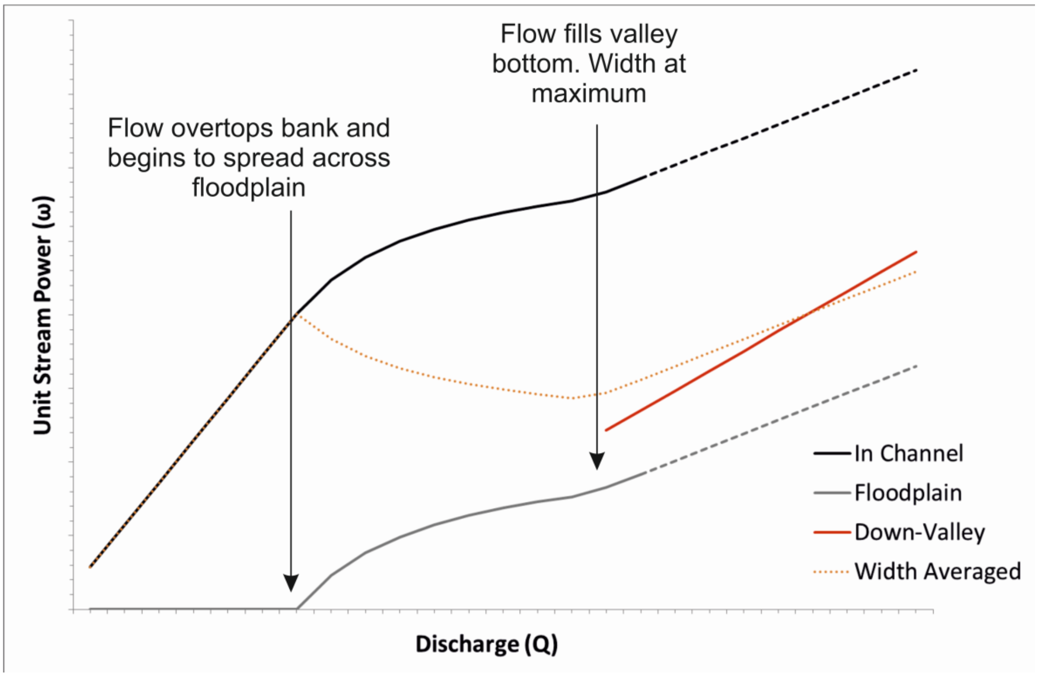


Figure 6.8 – conceptual relationship between unit stream power and discharge in semi-natural/restored channels.

Figure 6.8 shows a conceptual relationship between discharge and unit stream power in meandering channels connected to their floodplain. At low discharges all flow is confined in bank and unit stream power has a broadly linear relationship to discharge. As flow overtops the banks at a given discharge, water spreads over the floodplain, increasing the channel width but only slightly increasing the flow depth. As water spread across the floodplain the unit stream power for the portions of flow within the channel and on the floodplain increases only slowly with increasing discharge, but due to the increasing overall width of the wetted perimeter the width averaged unit stream power decreases. Once the valley floor is completely inundated width approaches a constant with unit stream power and discharge once more displaying a linear relationship. If flow switches to a predominantly down-valley direction, it will bypass channel sinuosity and channel slope will increase, at this point unit stream power will increase (red line). In the semi-natural and restored reaches of the Highland Water the majority of high flow events have peak discharges in the region between the two arrows.

Figure 6.8 shows conceptually in a channel connected to its floodplain unit stream power within the confines of the channel shows little increase once the flow is out of bank, so pieces of wood within the channel will not be substantially more likely to move in larger discharge events compared to moderate overbank flow events; all but the very largest high flow events during the study period had peak discharges which did not result in the inundation of the whole floodplain width and thus fall in the central zone of Figure 6.8. Conversely in channelized reaches where all discharges are in-bank wood within the confines of the channel will be subjected to higher flow velocities with higher discharge events and thus would be subject to higher drag forces and crucially a greater overall range of flow velocities and drag forces and thus be more likely to move given larger discharge events.

6.5.2. Controls on large wood stability

Binary logistic regression shows across all reaches and survey periods dimensionless length, species type and branching complexity are good predictors of large wood movement and movement of 10 metres or further. Movement is less likely with increasing dimensionless length, this is due to both the increasing weight of the piece of large wood providing resistance to buoyant and drag forces acting to entrain the wood, but also geometric factors with pieces of wood longer than the channel width more likely to become lodged in channel constrictions (Bocchiola et al., 2006b) and increasingly only able to be transported near to parallel to the flow direction. This finding confirms previous studies suggesting piece length is an important control on stability (e.g. Berg et al., 1998; Braudrick et al., 1997; Curran, 2010; Lienkaemper and Swanson, 1987; Máčka and Krejčí, 2010). Branching complexity is also a geometric constraint on movement with single stems less likely to become stabilised at trapping points such as channel constrictions and against other pieces of wood (Montgomery et al., 2003) and with more complex structures subject to lower combined forces due to complex interactions of wakes from individual branches causing variations in lift (Shields Jr and Alonso, 2012). Species type was coded as unknown, conifer and broadleaf, with conifers more likely to move and to move 10 metres or more than broadleaves. In the context of large wood mobility broad species type can be a proxy for density and specific gravity which has been found to be important in other studies (Ehrman and Lamberti, 1992; Gurnell et al., 2002); European broadleaf species (e.g. Birch, Beech, Oak, Ash) have a density in the range 0.525-0.585g/cm^3 (oven dry mass) (Brzeziecki and Kienast, 1994 in; Zanne et al, 2009; Schutt et al., 1994 in: Zanne et al, 2009), whereas conifers (e.g. Douglas fir, Pines, Spruces) have lower densities in the range 0.370-0.453g/cm^3 (oven dry mass) (Alden, 1997 in: Zanne et al, 2009; Brzeziecki and Kienast, 1994 in: Zanne et al, 2009). Large wood with a lower density and thus a lower specific gravity such as conifers will be more buoyant in water and thus will float and become entrained more readily (Shields Jr and Alonso, 2012). In a similar study in 5 streams of the Colorado Rockies Wohl & Goode (2008) found using a binary logistic regression analysis that dimensionless length, dimensionless diameter and location were statistically significant predictors of movement, it is possible their inclusion of pieces of 5cm or larger in diameter, rather than the 10cm minimum used here introduces a highly mobile small size fraction of wood and thus makes diameter easier to identify as a significant covariate. Studies have suggested other large wood variables are important for stability, however in this study no correlations were found between movement and either; dimensionless diameter (Curran, 2010; Wohl and Goode, 2008), root wad presence/absence (Collins et al., 2012; Curran, 2010; Montgomery et al., 2003), decay class (Gurnell et al., 2002) or large wood location (Curran, 2010; Gurnell et al., 2002; Wohl and Goode, 2008).

6.5.3. Geomorphology

An analysis looking at both the characteristics of large wood as predictors of movement and the geomorphological complexity of the reach shows different patterns between channelized reaches and semi-natural or restored reaches. For channelized reaches binary logistic regression reveals no individual characteristics of large wood to be significant covariate predictors of movement, although there are indications in the best performing model that dimensionless length and location of large wood may be important, with racked logjam pieces showing a higher likelihood of movement than wood in other locations; this correlation contradicts other studies suggesting large wood redistributed into logjams is more likely to be stable (Gurnell et al., 1995; Lienkaemper and Swanson, 1987). This correlation could be due to racked pieces of large wood having previously been mobile where the movement has only been stopped by the logjam and thus its current stability is primarily a function of the integrity of the logjam, therefore high flow events which subject the structure to high drag (Shields Jr and Alonso, 2012) and hydrostatic forces (Wohl, 2011) may cause the jam to partially break up and become more susceptible to racked pieces being mobilised (Shields Jr and Alonso, 2012). The break up and re-formation of logjams in the same location (Collins et al., 2012) as well as the cycling of large wood material downstream from one logjam to another was observed during the study (see logjam evolution below) and suggests that although local large wood loadings within logjams can be constant there is mobility of individual racked pieces.

For large wood moving 10 metres or more only dimensionless length was found to be a statistically significant predictor. The lack of relationship between large wood characteristics and movement within channelized reaches can be explained as a geometric problem; as the channels are deep, box shaped and fairly regular, there are no geomorphological constrictions to trap large wood; furthermore this channel shape and disconnection from the floodplain means discharge is directly proportional to flow velocity within the range of events observed. Wood within the channel will become fully submerged in all magnitudes of flood events and thus will not be subjected to a range of magnitudes of force acting to float the piece of wood; it will be either subject to minimum forces during baseflow, or the maximum force during flood events. Conversely drag forces in these confined channels will increase with increasing magnitude of flood events. The degree of decay, species and diameter of large wood are largely related to the specific gravity and weight of the piece, which is acting to resist flotation, which are less important in this setting, explaining why they are not found to be significant predictors. Conversely the dimensionless length of a piece of wood governs how it can be transported in the flow with pieces of length* equal or greater than one only able to be transported parallel to the flow due to geometry, furthermore the longer a piece of wood relative to the channel width the more likely it will be resting on one or both bank tops providing further stability (Bocchiola et al., 2006b) and that some portion of the wood will not be submerged during even the largest flood events.

Within semi-natural and restored reaches binary logistic regression shows dimensionless length, dimensionless diameter, location, branching complexity and species type are statistically significant predictors of movement. Location is more important in these reaches than in channelized reaches for two reasons, the first is statistical in that by having wider riparian and floodplain zones there are more pieces of wood in varying environments in these reaches, the second is large wood on the floodplain can only be mobilised during overbank flows corresponding to large discharge events, so will be mobilised less readily than wood in the channel. Large wood on the floodplain will experience a greater range of forces acting to float it; due to the wide floodplain water depth will increase far slower with increasing discharge than in confined channels, and so differing magnitude flood events will produce differing depths of wood piece submergence. A prerequisite for wood movement is a sufficient depth of flow over the floodplain in order to cause the wood to float (Haga et al., 2002), in a multivariate analysis such as this one the importance of large wood characteristics which affect specific gravity and thus resistance to flotation such as a species type (Zanne et al., 2009) will be picked up as significant covariates alongside location. Furthermore due to low flow velocities over the floodplain drag forces will be minimised contributing to the importance of buoyancy as a control on mobility in this environment compared to the confined channelized sections. Dimensionless length and branching complexity control how likely a piece of wood is to be resistant to drag forces (Shields Jr and Alonso, 2012); a longer piece of wood with branches will be more likely to be trapped by upright trees or wedged at geomorphological constrictions in the channel or floodplain (Bocchiola et al., 2006b).

For large wood moving 10 metres or further in semi-natural or restored reaches binary logistic regression shows dimensionless length, branching complexity and species type are statistically significant predictors or movement. In contrast to the analysis of all movement in these reaches dimensionless diameter and location are not statistically significant predictors. In geomorphologically complex reaches, with a sinuous planform there is a greater possibility of large wood being mobilised from a location and subsequently trapped or deposited a short distance away at a channel constriction, channel bend or against riparian vegetation. Factors which influence the potential for a mobile piece of large wood to become deposited or wedged at trapping points, such as dimensionless length and branching complexity (Bocchiola et al., 2006b) are thus important in controlling whether large wood can move a substantial distance once mobilised. These findings support the conclusions of a flume experiment by Bocchiola et al (2006b) that the distance large wood moves is a random variable whose expectation and variance is dependent on stream power, inter-obstacle spacing and piece length. Although the location of large wood is an important factor in determining initial movement it has no direct influence on the distance the wood will move once mobilised thus explaining why it was not found as a significant factor for wood moving 10m or further.

6.5.4. Logjam evolution

Tracking the positions of individual pieces of large wood enables the component pieces of logjams within the study reaches to be monitored. This showed logjams which persist at the same location over several flood seasons can be reworked and despite containing typically the same key pieces anchoring the logjam and appearing to have the same or similar structure and architecture some individual racked pieces of wood will be transported out of the structure and replaced with other mobile pieces trapped when they encounter the logjam. Figure 6.9 shows one such example with two photographs taken on 19/03/2011 (Figure 6.9A) and 04/10/2011 (Figure 6.9B), the solid arrows show pieces of large wood which have been transported out of the logjam (solid white arrows Figure 6.9A) and newly trapped by the logjam (solid white arrows Figure 6.9B) between the two photographs. The piece of large wood indicated in Figure 6.9A by white broken lines are tagged pieces which were previously recorded as racked pieces in a logjam over 100m upstream in the September 2010 survey, demonstrating some pieces move down the river system from one logjam to another. Although logjams and local quantities of large wood remain stable there is a high turnover of pieces of large wood (Latterell and Naiman, 2007; Marcus et al., 2002; van der Nat et al., 2003), this supports the findings of Marcus et al (2002) who suggest third and fourth order streams cycle wood through the system whilst maintaining wood loadings.

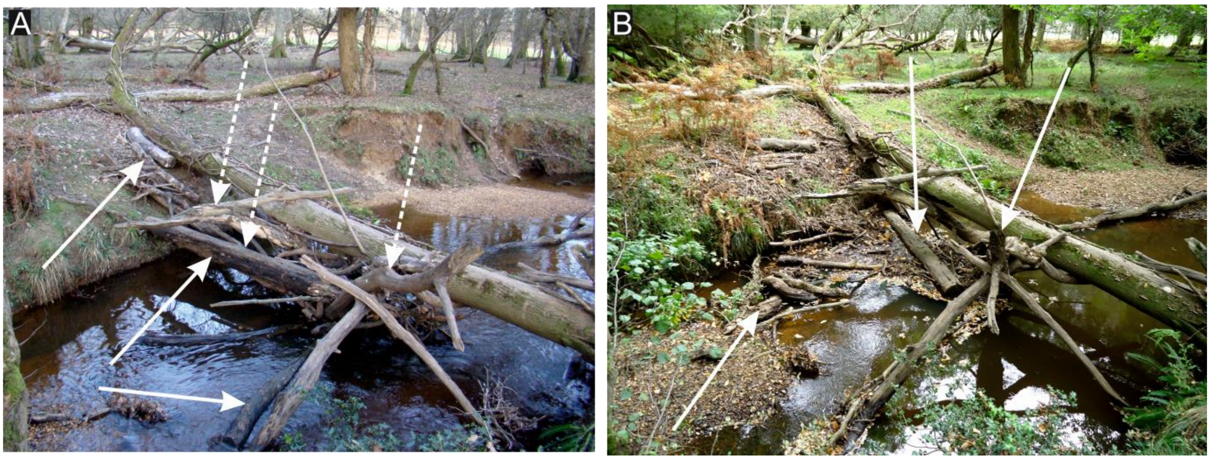


Figure 6.9– reworking of component racked pieces of large wood in a logjam, A – 19/03/2011, B – 04/10/2011. White arrows show wood transported into the logjam (A) and newly trapped by the logjam (B).

6.5.5. Transport distance

The proportion of large wood confirmed as moving from its initial location over the three winter flood seasons of the study was 75.5%, however only 70.1% of these mobile pieces of wood were located and surveyed in a new position; representing 53.1% of the total large wood. The proportion of tagged logs recovered is comparable with other large wood tagging studies (Latterell and Naiman, 2007) and although a relative success, the use of physical tags and markers relying on visual identification to relocate them has limitations (MacVicar et al., 2009). Visual identification is difficult where large wood can be buried in sediment, incorporated into the heart of logjams, submerged in deep water, or where algae can discolour tags or markers. Due to the limitations of physical tags future studies would ensure a higher recovery rate of tagged material by using radio tags (MacVicar et al., 2009).

The transport distances recorded in this study for mean (148.39m), median (5.32m) and furthest movement (2250m and 5600m) are high compared to those reported in other wood mobility studies (Table 6.10). A possible explanation for longer recorded transport distances is the experimental design used where a likely maximum transport distance was not assumed apriori and a walking survey undertaken encompassing in excess of 10km of river length; such a design increases the chances of locating pieces of wood with longer transport distances compared with a design in which a fixed downstream distance or fixed distance

Location	Stream Order	Gradient	Channel Width	Annual Transport Rate	Mean Transport distance	Median Transport distance	Maximum Transport distance	Reference
This Study (High Water, UK)	Third & Fourth	0.005-0.008	4-5m	50-62%	148m	5.3m	5.6km	
Highland Water, UK – small dowels <1m length	Fourth	0.005-0.008	4-5m	N/A		48-400m	1.2km	(Millington and Sear, 2007)
Popular Creek, N.Illinois, USA		0.001	15m	83% (15 months)			77% of wood moved >0.6km	(Daniels, 2006)
Sierra Nevada, California, USA	“small headwater streams	0.021-0.078%	2.1-12.8m	0-31%	70-361m			(Berg et al., 1998)
Oyabu Basin, SW Japan		>0.577	>1.7m	92% (9 months)	200-1400m		~4km	(Haga et al., 2002)
Queets River, Pacific NW, USA	Fifth	0.006	125m				12km	(Latterell and Naiman, 2007)
Rocky Branch, New York, USA	Second	0.065	8m	25% (4 years)		35m	>300m	(Warren and Kraft, 2008)

Crow's Creek, Wyoming, USA	Second	0.055	7m	18%	(Young, 1994)
Central Rocky Mountains, Colorado, USA	5 Headwater streams	0.013-0.098	4.3-6.5m	16-23%	(Wohl and Goode, 2008)

Table 6.10 – comparison between transport distances reported in this study and other studies from the literature.

beyond the last located piece are used. The majority of the furthest moving pieces of wood were comparatively small with lengths of less than 2 metres, although a 6x0.2m piece moved 375m (reach C) and a 3.9x0.14m piece moved 700m (reach B). Long transport distances for some longer pieces of wood indicates that although an increasing dimensionless length decreases the likelihood of a piece moving a substantial distance, it does not preclude such transport.

The multivariate analysis of transport distance using general linear regression identifies; dimensionless length, dimensionless diameter and species type as statistically significant predictors, although the model only predicts 7.17% of the variance in transport distance. The low predictive power of the model is a combination of some important factors being poorly represented in the variables collected and the large stochastic element in the transport of wood down a complex river channel (Bocchiola et al., 2006b). The relationship between large wood size and mobility has also been illustrated in other settings (e.g. Bilby, 1984; Daniels, 2006; Lienkaemper and Swanson, 1987; Millington and Sear, 2007); however the model used here indicates that although statistically significant, length and diameter only have a low predictive power. Species type is indicated by the model to be the most important individual predictor of transport distance variance; this is due to the large difference in density between wood from conifers and broadleaves (Zanne et al., 2009), resulting in greater buoyancy in conifers. The importance of buoyancy in transport distance can be explained by a greater likelihood of highly buoyant pieces of wood moving over the top of channel obstructions such logjams and a greater likelihood of moving out of bank over shallow floodplain flows, Haga et al (2002) note the importance of the critical floating depth and suggest transport distances can be substantial where flow depth is greater than large wood diameter. There are other factors which affect the density of large wood, notably decay (Gurnell et al., 2002) which is not found to be statistically significant here and degree of water-logging (Shields Jr and Alonso, 2012) which was not represented in the study. Further studies which either directly measure density in the field, through collection of small cores and lab analysis, or which represent these factors experimentally rather than visually should find a statistically significant relationship between density factors and transport distance.

Variables related closely to the geometric complexity of large wood relative to the channel are either found to have low predictive power (dimensionless length) or are not statistically significant (branching complexity), this can be explained by the high degree of connectivity between the river channel and floodplain along much of the study river. The high connectivity between channel and floodplain results in ready inundation of the floodplain during the high flow events which mobilise more wood. When the floodplain is inundated alternative flow paths develop which bypass the geomorphological complex of the channel and any planform sinuosity, resulting in a change to a wide, shallow flow moving in a predominantly down-valley direction. The sparse under-croft vegetation in the study environment results in fewer trapping locations for large wood moving via alternative overbank flowpaths during overbank flows.

Figure 6.10 – upstream end of reach B during high discharge event (left) and at baseflow (right), arrow shows direction of flow. In left image flow over the floodplain is up to 0.3m deep and in places is moving via alternative flow-pathways in a predominantly down-valley direction, channel planform is sketched onto image as white line.

No statistically significant differences were found in transport distance between the two different classes of reach, this finding can be explained by considering the patchwork nature of reach types in the study catchment; channel engineering was typically undertaken in forest plantations which are spread throughout the catchment leading to the channel changing in sinuosity along its length (Figure 6.11). The variability in sinuosity typically results in pieces of large wood transported in excess of 300m having moved through sections of both high and low sinuosity; in order to compare the effects of channel geomorphological complexity on long transport distances, further studies are needed in which mobility is compared between sinuous and near straight rivers with downstream study lengths of at least 5km and with consistent sinuosity.

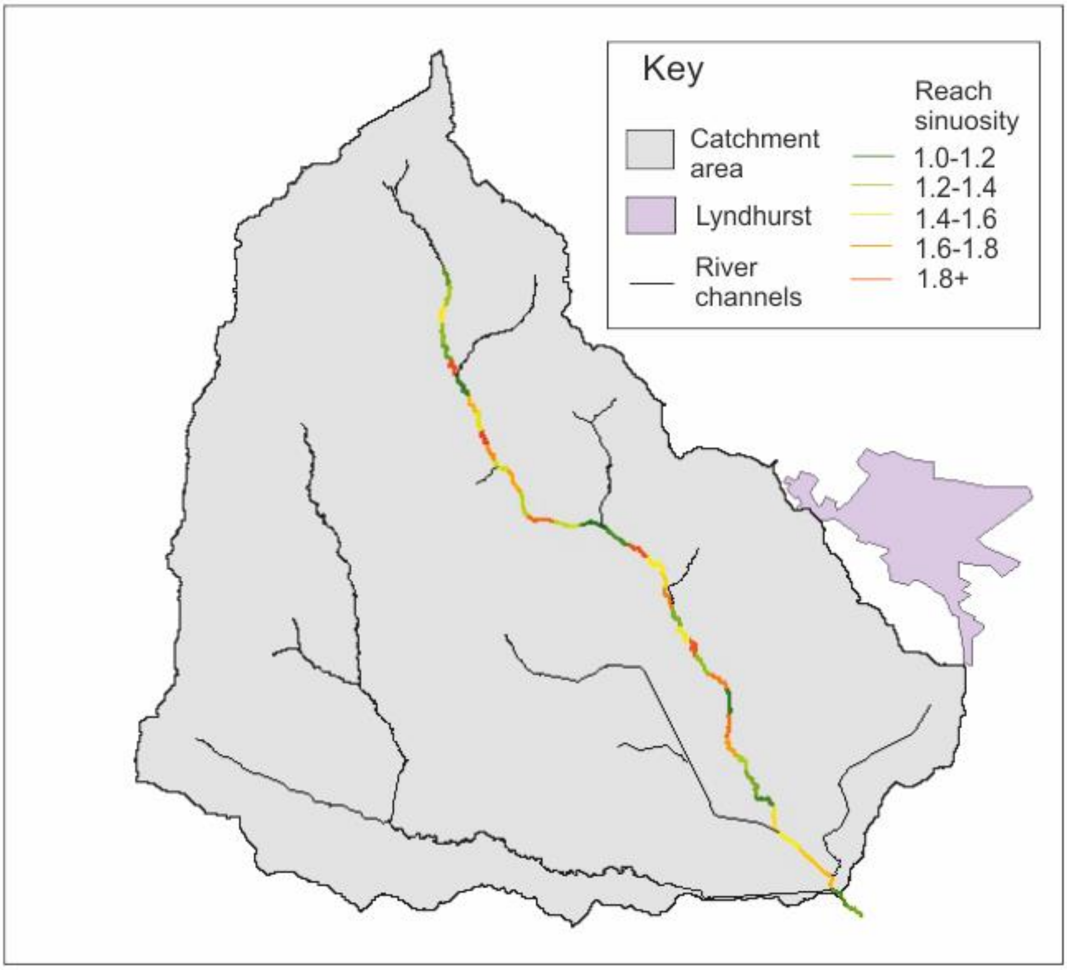


Figure 6.11 – variability in reach sinuosity along the main study channel.

6.6. Conclusion

This study has demonstrated large wood in small forest rivers can be highly mobile with over 75% of pieces moving during a two and a half year study. Transport distances for mobile large wood were found to be longer than expected with several pieces moving in excess of 500m and a furthest recorded transport distance of 5.6km.

Multivariate analyses show dimensionless length to be an important factor explaining mobility and transport distance in all contexts with dimensionless length over 1.5 leading to lower likelihood of mobility, they also show dimensionless diameter, branching complexity, species type and location can all be important factors in explaining mobility and transport distance but this depends on context. Statistically significant models were found for all multivariate analyses but the majority of variance in both likelihood of mobility and transport distance remains unaccounted for by the variables collected. Density of wood was identified as an important variable which would need to be specifically measured in addition to proxy measurements in future studies.

In common with many large wood mobility studies using physical tags there was a challenge in locating and resurveying large wood that had moved from its original location with only around 70% of mobile pieces recovered. The transport distances reported here suggest that in other studies where a low proportion of tagged logs have been recovered, such pieces may have been transported far out of the study area and the distance downstream in which a search for large wood is conducted may have been too short. Future large wood mobility studies should use radio tags or some other remote method of relocating or tracking mobile wood in order to improve recovery rates and to capture pieces moving substantial distances of 1km or more.

Logjams formed around a stable key piece of large wood can persist for several years and through multiple high discharge events, we have shown that although such logjams may have the same function and structure and may ostensibly appear the same, the component racked pieces of wood are reworked and moved down the system.

7. Hydraulic resistance properties of logjams during flood flows

7.1. Abstract

Logjams affect in-stream hydraulic resistance of channels and can contribute to slowing flood wave travel time. Although many studies have examined the hydraulic resistance effects of grain roughness and form elements there are only a handful of studies in the scientific literature which attempt to partition the hydraulic resistance contribution of logjams.

Results from this study show logjams in small, moderate gradient forest rivers under high flow conditions contribute an average of 65% of total flow resistance. Logjams can contribute 75-98% of total hydraulic resistance where they induce spill over the logjam structure or underflow beneath it. The results of this study along with studies of logjams in step-pool and low gradient rivers shows a continuum of increasing hydraulic resistance with increasing water slope. The relationship between water slope and hydraulic resistance suggests energy dissipation related to logjams mediated steps in the water profile, and thus spill, is the most significant contributor to wood related hydraulic resistance.

Values presented in this study provide valuable quantitative parameterisation of roughness coefficients for logjams under high flow conditions derived from direct field measurements; which are important for hydrological modelling of logjams and are hitherto sparsely reported in the literature. Values reported here will be used to parameterise Manning's *n* values for engineered logjams in hydrological modelling in Chapters 9 and 10.

7.2. Introduction

The effects of in-stream wood on morphology (Collins et al., 2002; Gurnell et al., 2002; Jeffries et al., 2003; Montgomery et al., 2003), ecology (Dolloff and Warren, 2003; Piégay and Gurnell, 1997) and floodplain connectivity (Sear et al., 2010) have been documented in a range of river environments. Conversely, the effects of wood accumulations on hydraulic resistance have received less attention in the literature, and studies describing and quantifying the flow resistance of large wood have been largely restricted to step-pool channels (Curran and Wohl, 2003; David et al., 2011; Dust and Wohl, 2012; Yochum et al., 2012) with high slopes and large, low gradient rivers (Shields Jr and Gippel, 1995). Step-pool and low gradient sand bed channels represent two opposite ends of a spectrum of wood assemblage hydraulics (Kitts, 2011). Step-pool channel hydraulics are dominated by water spilling over wood, classified by Gregory et al (1985) as 'active' and Wallerstein et al, (1997) as 'overflow' logjam types. Low gradient sand-bed river hydraulics are dominated by single pieces of wood and accumulations filling only part of the active channel and deflecting flow, classified by Gregory et al (1985) as 'partial' and Wallerstein et al, (1997) as 'deflector' logjams.

There is a need to quantify hydraulic resistance associated with wood in a variety of accumulation types within gravel-bed channels of moderate slope, in order to better understand hydraulics within forest river channels and to parameterise hydrological models. To date only Gregory et al., (1985) and Kitts (2011) have described flow resistance in this environment but there is a need to extend this work to empirically calculate logjam hydraulic resistance at high flows, in order to parameterise hydrological flood modelling. There is a general lack of empirical documentation of how logjams affect flow patterns at bankfull or flood stages (Manners et al., 2007). There is also a wider research need for reach scale investigations of vegetation and roughness (Robert, 2011).

Total flow resistance is a composite of resistance due to; skin friction, eddy losses and flow distortions around obstacles (Einstein and Barbarossa, 1952; MacVicar, 2013), Einstien & Barbarossa (1952) detailed an approach whereby total flow resistance can be partitioned into component parts related to specific features. Although resistance partitioning remains a subjective exercise due to variability between field sites and the application of uniform flow resistance equations to non-uniform rivers (Ferguson, 2007; MacVicar, 2013), partitioning remains useful to understand the relative importance of different elements in total flow resistance (David et al., 2011).

There are published formulae for calculating flow resistance due to grains (e.g. Ferguson, 2007; Griffiths, 1981; Rickenmann and Recking, 2011; Zimmermann, 2010), drag forces and pressure differences upstream and downstream of form elements and bends (Henderson, 1989; Shields Jr and Gippel, 1995), however macro-scale form elements, such as logjams, are still difficult to incorporate and are generally only dealt with at a reach scale (MacVicar, 2013).

Methods to predict the flow resistance contribution of large wood and logjams have relied on two approaches; either broad crest weir equations, or a drag force approach. In step-pool channels wood is treated as a step-element with resistance calculated using broad-crest weir equations assuming the dominant resistance component is spill (Dust and Wohl, 2012; Yochum et al., 2012). In low gradient rivers a drag force approach is used to calculate flow separation around a cylinder and frictional shearing between the flow and the object assuming the wood can be treated as a boundary roughness component (David et al., 2011; Kitts, 2011; Shields Jr and Gippel, 1995); such single log models dominate the literature but natural logjams are poorly described as cylinders (Manners et al., 2007). The drag force approach was refined by Manners et al, (2007) for deflector logjams to include logjam porosity, showing that assumptions of non-porosity can overestimate drag coefficients by 10-20%. In a system in which there are a mix of isolated pieces of large wood and accumulations of varying sizes and hydraulic influence, a single unifying equation for calculating flow resistance remains elusive. Hydraulics are complicated by flow separation, vortex shedding, overlap in wakes causing a decrease in drag coefficient (Manga and Kirchner, 2000) and jam porosity (Manners et al., 2007).

In systems where some portion of total flow resistance cannot be calculated and is un-measurable it is possible to partition total resistance (Equation 7.1).

$$f_{total} = f_{measurable} + f_{unmeasurable} \qquad 7.1$$

Total flow resistance (f_{total}) is measured in the field and the contribution to total resistance of components which can be predicted or estimated, such as grain roughness, are calculated ($f_{measurable}$). The remaining resistance component which cannot be independently estimated or calculated ($f_{unmeasurable}$) is the difference between total resistance measured and the sum of predicted or estimated resistance components

(Equation 7.2); in this way flow resistance due to large wood can be characterised in the absence of predictive equations (Curran and Wohl, 2003).

$$f_{unmeasurable} = f_{total} - f_{measurable} \qquad 7.2$$

7.3. Aims

This study aims to quantify the hydraulic resistance associated with in-stream large wood during high flow events. Objectives are to:

- Quantify the range of hydraulic resistance values for forest streams with varying wood loads and varying types of logjam assemblages. These results will be used to parameterise hydraulic resistance values for engineered logjams in hydrological modelling reported in Chapters 9 and 10.
- Describe the relationships between hydraulic resistance and logjam type and wood loading.
- Compare the hydraulic resistance values measured and calculated to other published data. This will place results in the wider context of other studies.

7.4. Methodology

7.4.1. Study Site

See section 5.2.2 for full site description and Figure 5.1 for a map of the overall study area.

Hydraulic resistance measurements were conducted using a dilution method at the reach scale, with reach lengths of 20-35m (5-8 channel widths). Five study reaches were selected to give a range of wood loads in both straight and meandering sections (Table 7.3). Using a Geodimeter Total Station at low flows with a point density of 0.1m, cross-sectional profiles were produced for; the upstream end, downstream end and the centre of each reach, in addition the thalweg length was measured along each reach. These measurements were used to calculate reach length (L_{af}), bed slope (S) and channel width (B).

In order to sample bed grain size a Wolman (1954) pebble count was performed in each reach sampling 100 pieces of gravel at 5 locations, selecting the grains blind to reduce sampling bias. A random pebble count sampling strategy does have limitations in terms of sampling bias towards larger grains and sampling the surface material only, however it is widely used and is recognised as a valid technique in circumstances where a bulk sampling strategy is deemed impractical (Manners et al., 2007).

To calculate the reach averaged velocity a salt dilution method was used (Figure 7.1) where the average velocity of the centroid of a tracer cloud approximates the mean flow velocity (Whiting, 2003). A tracer method was chosen as it is particularly suited to situations where flow is clogged with vegetation or obstructions (Whiting, 2003). A salt solution is introduced to the river at the injection point, conductivity is measured at points A and B and the time of peak conductivity recorded at both points. Peak conductivity corresponds to the centroid of the salt tracer cloud. Mean flow velocity can be calculated using the travel time of salt tracer (Δt) and the distance between A and B (Laf). Lmix represents a distance sufficiently far upstream to allow full mixing of salt tracer throughout the water column and laterally through the cross-section after the salt solution injection point, Lna is a length of river unaffected by wood upstream of the measured reach, note that Lmix<Lmix+Lna.

The aim of this study is to quantify the hydraulic resistance within reaches during high flow events; therefore data collection was restricted to periods immediately following large rainfall events. The catchment exhibits a highly “flashy” hydrological response to rainfall due to the underlying clay geology where in excess of 80% of rainfall can be delivered to the channel network via runoff; as a result of the catchment hydrology and small catchment size the time between the peak of a rainfall event and peak flow within the river is approximately 3-4 hours. A minimum rainfall intensity of between 4-8mm/hr for 2-3 hours is needed to raise the stream discharge from the baseflow of 0.2 m^3/s to a minimum threshold of 0.8 m^3/s herein defined as a “high flow event”. The study reaches were channelized at some point between the 1840’s and 1920’s based on historical sources for the onset of channel engineering in the Forest (Tubbs, 2001) and the current age of plantation trees. The number of suitable rainfall events during a year is relatively low, given the relative scarcity of such rainfall events and the necessity the peak of the event occurs between 4am and 11am in order that the peak discharge can be gauged during daylight hours. High flow gauging was conducted on 12/11/2011, 17/04/2012 and 01/05/2012.

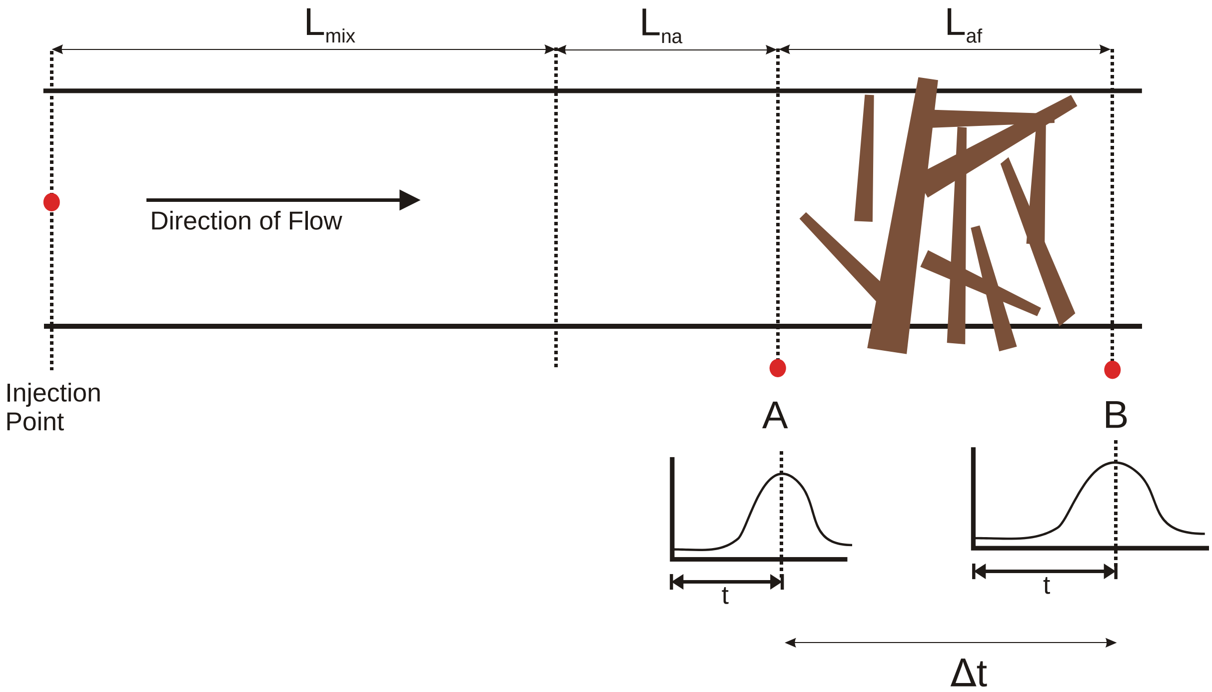


Figure 7.1 -Salt Dilution method. A salt solution is introduced to the river at the injection point, conductivity is measured at points A and B and the time of peak conductivity recorded at both points. Δt is travel time of salt tracer, Laf is the distance between A and B, Lmix is mixing length for salt tracer in flow, Lna is a length of river unaffected by wood upstream of the measured reach.

Salt tracer was made by dissolving 500g of NaCl in 5l of water to make a 100g/l solution, this was injected using a gulp injection method to the centre of the channel at a point sufficiently upstream of the study reach to allow full lateral mixing (L_{mix} in Figure 7.1). Conductivity of the flow was measured at the start and end point of a study reach (A & B in Figure 7.1) using two electronic Wissenschaftlich Technische Werkstaten GmbH (WTW) Cond3310 conductivity meters using an autologging feature recording conductivity every second. During the first gauging exercise of 12/11/2011 WTW Cond3310 gauges were not available and two analogue gauges and a pair of stopwatches were used to record the concentration peak. As the electrical conductivity of the water is linearly related to the concentration of dissolved salts (Whiting, 2003), the time at which the centroid of the tracer cloud passes a detection point will correspond to the peak value in conductivity at that point. The time from injection to peak concentration was recorded for both detection points, and by subtracting the time to peak at point A from point B a reach travel time for the tracer, and thus flow velocity could be obtained. The energy slope was measured across the reach using a dumpy level to record water surface height at the start and end of the reach, and the water depth was read off a gauging board to calculate hydraulic radius. The reach averaged velocity can be calculated using either a simple distance over time equation (Equation 7.3), or an equation based on conductivity measurements (Equation 7.4)

$$\bar{v} = \frac{\partial x}{\partial t} \tag{7.3}$$

$$\bar{v} = 0.5 \int_{0}^{\infty} (Con_d(t) - Con_b)\partial t \tag{7.4}$$

Where Con_d and Con_b are the tracer conductivity at the downstream end of reach and the background concentration respectively.

The technique is predicated on the tracer being well mixed laterally with the flow at the sampling location(s) (Ward, 1973; Whiting, 2003). The uncertainty of the method is about 5%, which is comparable with other field methods for gauging stream discharge/velocity (Whiting, 2003). In order to control lateral mixing the injection point was calculated to be far enough upstream from the start of the reach to allow for a minimum of 95% mixing at detection points based on equations from Ward (1973), who derived the following equations for salt dilution mixing in open channels, for straight channels (Equation 7.5) and meandering channels (Equation 7.6) respectively.

$$\frac{X_m}{B} = \frac{K_1}{0.02}\frac{B}{d} \qquad \textbf{7.5}$$

$$\frac{X_m}{B} = \frac{K_1}{0.06}\frac{B}{d} \qquad \textbf{7.6}$$

Where, X_m is the mixing length and K_1 is equal to $e_z t/B^2$ where e_z is a dispersion coefficient in z-direction, B is channel width and d is channel depth (Ward, 1973). Values of e_z were determined from the original paper; the equations assume that dispersion in the horizontal direction is achieved asymptotically at infinite lengths so the desired degree of mixing must be specified, as must the injection position(s).

For this study a 95% degree of mixing was used along with a central injection point to give a balance between practical mixing lengths of less than 50m, compared with 100-200m for 99% mixing and accuracy of conductivity readings.

Reach averaged velocity allows the Darcy-Weisbach friction factor *f*, or Manning's *n* to be calculated.

Darcy-Weisbach friction factor (*f*)

$$f = \frac{8gRS_0}{\bar{v}^2} \qquad \textbf{7.7}$$

Manning's n

$$n = \left(\frac{1}{\bar{v}}\right) R^{\frac{2}{3}} S_0{}^{\frac{1}{2}} \qquad \textbf{7.8}$$

Where g is acceleration due to gravity (9.81m^{3}s^{-2}), R is hydraulic radius and v is mean velocity (m^{3}s^{-1}). In both Equations 7.7 and 7.8 a slope component (S_0) is needed to calculate the roughness coefficient, in most conditions this would be measured as the bed slope, however in conditions where the flow is not uniform, such as flow over logjams, it is recommended that the energy slope be used instead (Ferguson, 2007). The water surface slope was measured at the start and end of each reach following each dilution gauging using a dumpy level and survey staff (with uncertainty of ±0.02m).

7.4.2. Potential impacts on Wildlife

In order to conduct responsible field based science it is important to consider the effects of proposed field methods upon the natural environment. The goal should be to minimise the duration and magnitude of any impacts. High levels of dissolved salt can have detrimental effects on aquatic wildlife (Hart et al., 1991). Most biota have osmoregulatory mechanisms which can be put under strain at high salinities, sub-lethal osmotic stress typically occurs at lower concentrations of salt than those at which toxic effects are observed (Beadle, 1969; Padhye and Ghate, 1992; Sanzo and Hecnar, 2006). When introducing pulses of salinity into an aquatic environment it is important to establish concentrations at which biota will experience stress or mortality and to endeavour to maintain levels below this. A summary of literature values of chronic and toxic effects of salt on freshwater biota is presented in Table 7.1. Although data are not specific to Southern UK taxa, they are useful as a guide for levels of salt toxicity at a class level. Generally fish are fairly resistant to salinity with sub-lethal effects on adults only observed above 10000mg/l , (Hart et al., 1991; Hogan and Nicholson, 1987; Parry, 1966), however amphibians are rarely good regulators of salt (Alvarado, 1979), and some macroinvertebrates display adverse effects to salt concentrations above 1000mg/l (Beadle, 1969).

Biota	Sub-lethal Effects (NaCl mg/l)		Toxicity (NaCl mg/l)		Reference
Fish	>10000	In adult fish			(Hogan and Nicholson, 1987)
Fish	9000-20000	Upper limits for fertilisation and hatching of eggs			(Hogan and Nicholson, 1987; Parry, 1966)
Amphibians	1030	Chronic mortality observed	1646-4206	Rarely good regulators of salt	(Alvarado, 1979; Padhye and Ghate, 1992)
Amphibians	2636-5109	Sub-lethal effects in tadpoles	3000	Concentration at which tadpole mortality observed.	(Sanzo and Hecnar, 2006)
Algae	5800	Short term depressions in activity observed above 5800mg/l.	>14500	Observed to recover from osmotic stress up to 14500mg/l	(Moore et al., 1985; Reed et al., 1984)
Macro-invertebrates	1000	Adverse effects observed above 1000mg/l	>9000	Osmoregulatory mechanisms fail above 9000mg/l	(Beadle, 1969)
Waterfowl	3000	Some evidence of lower breeding success			(Hart et al., 1991)

Table 7.1– values for NaCl toxicity for a value of biota.

Based on the information in Table 7.1 the concentration of NaCl in the river channel should not exceed 1000mg/l in order to avoid any adverse effects on wildlife. An injection concentration of 500g in 5l of water was selected; assuming instantaneous injection into a discharge of $1m^3/s$ would equate to a mixed concentration of 500mg/l.

7.5. Results

Based on field measurements hydraulic resistance values for each reach were calculated (Table 7.2). Calculations were done for both Darcy-Weisbach friction factor f (Equation 7.7) and for Manning's n (Equation 7.8). Boxplots of calculated Manning's n roughness coefficients are shown in Figure 7.2. In Equations 7.7 and 7.8 the slope S_0 is the energy slope measured as the difference in water surface elevation along the reach. Hydraulic radius R is calculated from measured cross-sections and water depth measured during gauging, mean velocity v is calculated as the time taken for the tracer cloud to travel along the reach over the length of the reach.

Summary reach characteristics are shown in Table 7.3.

Reach	Logjam Category[a]	Manning's n			Darcy-Weisbach f			Number of measurements
		Min	Mean	Max	Min	Mean	Max	
1	Active	0.137	0.240	0.362	1.694	4.676	10.585	8
2	Partial	0.027	0.059	0.132	0.089	0.582	2.154	8
3	Complete	0.077	0.113	0.182	0.585	1.960	5.318	8
4	Partial	0.041	0.107	0.199	0.186	1.966	4.571	8
5	High Water	0.059	0.158	0.412	0.426	4.637	19.824	8

Table 7.2 – summary of calculated hydraulic resistance values for each study reach based on field data. [a] – Logjam categories based on Gregory et al (1985).

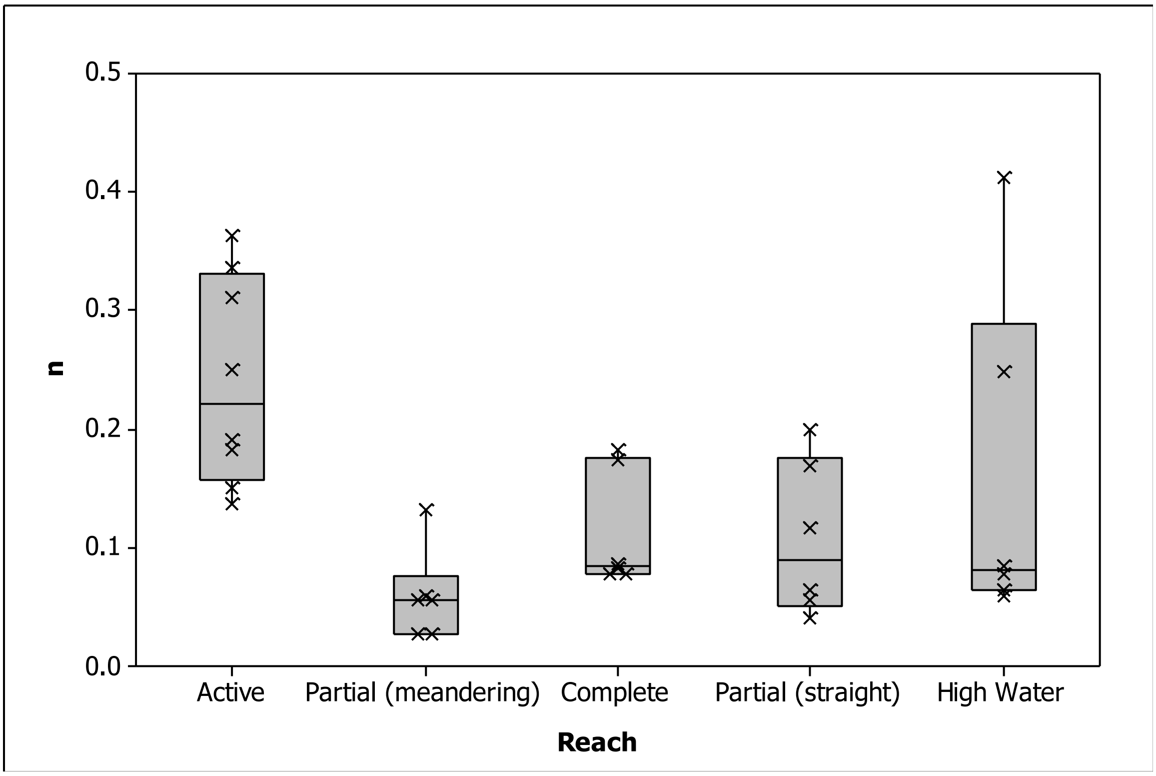


Figure 7.2 - boxplot showing measured total hydraulic resistance values for five study reaches.

Reach	Length (m)	Sinuosity	Bed Slope (m/m)*	Water Slope (m/m)*	D_{84} (m)	Wood Load (m^3/m^2)	Surface Area:Wood Area	Logjam Category[a]	Description
1	25.89	1.00	0.004	0.007-0.011	0.0021	0.139	0.095	Active	Compact channel spanning active logjam in straight channel
2	36.31	1.36	0.013	0.002-0.005	0.0026	0.005	0.039	Partial	S-shaped meander bend with partial logjam against outer bend
3	24.36	1.00	0.023	0.004-0.007	0.0023	0.011	0.078	Complete	Two connected, and loosely formed complete logjams in straight section
4	23.57	1.00	0.012	0.005-0.007	0.0025	0.004	0.022	Partial	Three large loose logs parallel to flow at low flow margin (partial logjam) in straight section.
5	21.62	1.04	0.014	0.005-0.012	0.0030	0.042	0.067	High Water	High Water logjam of very large fallen tree across low radius meander bend.

Table 7.3 – summary study reach characteristics.

a – Logjams were categorised based on Gregory et al., (1985). * - Note that due to minor form elements the bed slope measured between the start and end of some of the reaches is steeper than the overall long profile bedslope of approximately 0.006m/m, reach 3 is notable for having a gravel deposit at the start of the reach deposited in association with the logjam, reaches 2 and 5 have similar, but smaller (~h=0.2m) gravel deposits at the start of the reaches and reach 4 has a deeper section of water at the terminus of the reach, possibly a relic pool.

7.6. Discussion

The dilution gauging results show the active logjam has the highest hydraulic resistance; however there is a degree of overlap between the measured hydraulic resistances of the other study logjams (Figure 7.2). This variability can be partly explained through roughness partitioning, where the overall hydraulic resistance is divided into component roughness contributions. Curran and Wohl (2003) found resistance due to spill over logjams or other obstructions to be a key component in overall hydraulic resistance in mountain streams, in this study individual logjam hydraulics are highly sensitive to water depth. The complete logjam (reach 3) has no spill at baseflow discharge (~0.2 m^3s^{-1}), when measured at 1.6 m^3s^{-1} the jam is inducing a small amount of spill ~0.05m in height. The partial logjam (reach 4) does not induce spill at baseflow discharge, however a single large piece of wood does cause a small spill step at 0.8 m^3s^{-1} before becoming drowned out and submerged at discharges over 1 m^3s^{-1}. The high water logjam (reach 5) causes very high flows (1.6 m^3s^{-1}) to be pushed under the main bridging log towards the bed and causes a slight step in the water profile at these discharges. The changes in local hydraulics with varying water depth can be seen clearly in Figure 7.2 for complete (reach 3) and high water (reach 5), where there is a two phase grouping of individual roughness measurements, low at $n \approx 0.1$, where the logjams are not inducing spill or underflow and high $n > 0.2$ where the complete logjam induces areas of spill and the high water jam behaves as an underflow jam (Figure 7.3).

Figure 7.3 shows that high water logjams (upper part of Figure 7.3) such as experimental reach 5 may have little interaction with flow during lower discharges; even those approaching bankfull (top left, Figure 7.3). At higher discharges when the height of the water surface is above the height of the base of the log flow can become ponded behind a bridging log (top right, Figure 7.3), forcing flow under the log (underflow). This can cause transitions from sub-critical to super-critical flow and back again as a hydraulic jump dissipating energy. As flow is forced towards the bed erosion of bed sediments may occur.

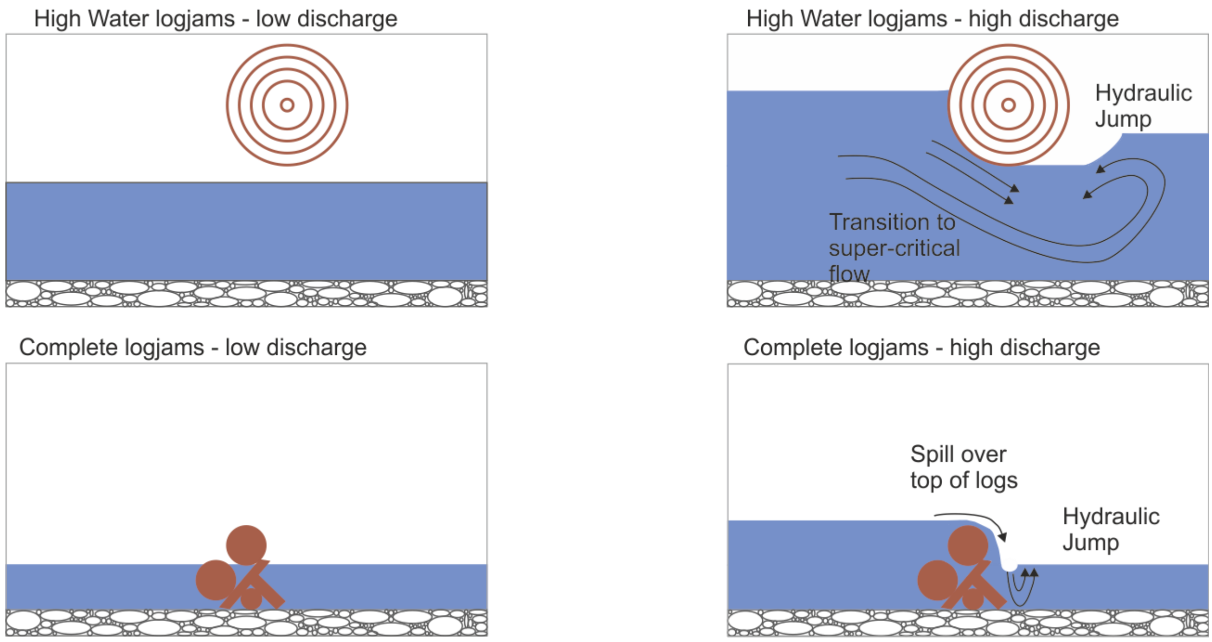


Figure 7.3 – Behaviour of High water and Complete logjams at varying discharges. Diagrams are drawn side-on to flow direction, which is left to right.

Complete logjams (lower part of Figure 7.3) may be sufficiently porous that hydraulic resistance is insufficient to cause flow to become ponded behind the logjam (bottom left, Figure 7.3). At higher discharges greater water depth and resultant increase in hydraulic resistance may cause water to become ponded behind the dam and lead to water spilling over the structure. Increased energy dissipation is seen in higher calculated friction factors for cases where underflow or spill are observed at logjams.

7.6.1. Roughness Partitioning

Einstien and Barbarossa (1952) describe how total hydraulic resistance in the form of an empirical roughness coefficient can be partitioned into the sum of component sources of resistance. One application of roughness partitioning is to estimate the roughness contribution from a component which is difficult to calculate or directly measure, such as large wood. To estimate the contribution from a non-measurable component, such as large wood, the difference between total hydraulic resistance measured in the field and the sum of roughness contributions from all other components is calculated. The sum of roughness contributions such as grain roughness and form roughness are estimated using empirical equations. Generally roughness partitioning in the literature is performed using Darcy-Weisbach friction factor *f* (e.g. Curran and Wohl, 2003; Kitts, 2011; Wilcox et al., 2006), although it is an equally valid approach for Manning's *n*. In this partitioning exercise Darcy-Weisbach friction factor is used as it is dimensionally correct (Hey, 1979) and allows direct comparison with other studies using this method.

The individual roughness components included in the partitioning depends greatly on the study environment; a term for grain scale roughness is universally included (e.g. f_{grain}), other terms which are variously included or excluded are f_{bend}, f_{wood}, f_{form} and f_{spill}.

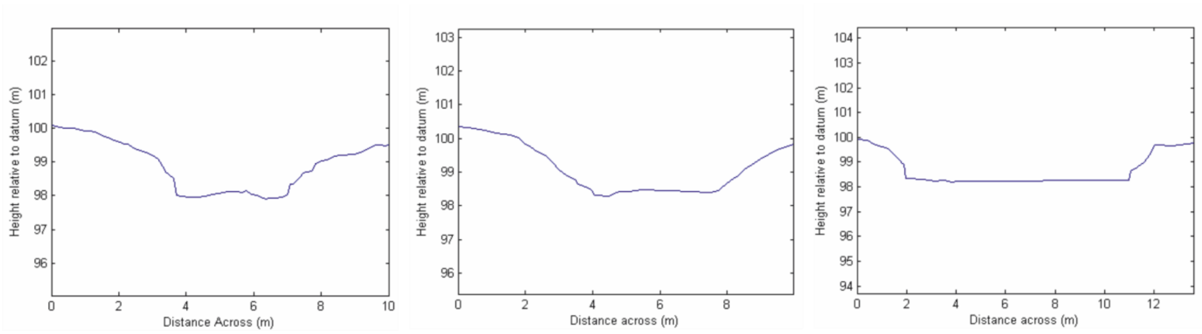


Figure 7.4 – cross-sectional profiles for reach 4 (partial logjam, straight channel), cross-sections are drawn as if looking upstream. From left to right, Figure shows cross-sections from upstream end to downstream end of reach. These cross-sections illustrate there are only minor bedform irregularities within study reaches.

In this study the main sources of hydraulic resistance are; grain or bed roughness, large wood and form roughness (Equation 7.9). The study reaches are in relative narrow channels with only minor bed forms and no large bar or pool elements; Figure 7.4 shows three cross-sectional profiles for reach 4 demonstrating the bed surface is largely regular. Although bedforms can provide substantial resistance in gravel bed rivers (Hey, 1988; Parker and Peterson, 1980; Prestegaard, 1983) where bedform amplitude is less than wood diameter bedform contributions to total flow resistance will be negligible (Manga and Kirchner, 2000). The major form roughness element within this study is the presence of meander bends within reaches 2 and 5, therefore the f_{form}

component is represented with an equation for headloss at bends (Equation 7.12) and f_{form} in Equation 7.9 replaced with f_{bends}.

$$f_{total} = f_{grain} + f_{bends} + f_{wood} \qquad 7.9$$

In this study empirical equations have not been used to estimate the contribution of wood to total hydraulic resistance. Previous studies of wood hydraulics in channels dominated by active, step forming logjams have used broadcrest weir equations (Curran and Wohl, 2003; Wilcox et al., 2006), and in channels where wood is primarily loose pieces or deflector jams a drag force approach has been used (Manga and Kirchner, 2000; Shields Jr and Gippel, 1995; Shields Jr and Gray, 1992), including the effects of porosity on drag force (Manners et al., 2007). Flow resistance due to wood is complicated by factors including: surface waves created by drag (Wallerstein et al., 2002), overlapping wakes created by individual wood pieces (Manga and Kirchner, 2000), flow divergence/convergence around objects (Manners et al., 2007; Wallerstein and Thorne, 1997), local accelerations and decelerations in flow (Abbe and Montgomery, 1996) and degree of spill over accumulations (Wilcox et al., 2006), with the size and arrange of wood pieces significantly affecting the hydraulics of individual jams (Daniels and Rhoads, 2004; Manners et al., 2007). In the Highland Water there is a range of logjam architecture and a universal empirical approach has not yet been described to calculate hydraulic resistance across logjam types. In this study wood is treated as a non-measurable component and hydraulic resistance contribution is assigned as the difference between total resistance and resistance calculated due to grain and bends.

A number of empirical equations have been proposed to calculate mean flow velocity based on bed grain size (e.g. Bathurst, 2002; Griffiths, 1981; Hey, 1979). Grain based resistance equations make the assumption that total flow resistance can be parameterised by small scale properties of the bed and a unique relationship exists between mean depth and mean velocity for a given roughness (Ferguson, 2007). Mean velocity equations are typically parameterised using data from flumes or channels in which form roughness is likely to be a minimal component of total roughness. Empirical coefficients used in grain roughness equations may therefore account for total roughness in a system in which grain roughness is the overwhelmingly dominant, but not the only roughness component. It is important to be aware that the empirical origins of grain roughness equations can lead to limitations in their accuracy when using a roughness partitioning approach. The Variable Power Equation (VPE) of Ferguson (2007) (Equation 7.10) has been shown to perform well in calculating grain roughness compared to other approaches for both shallow and deep flows over coarse gravel bed rivers (Ferguson, 2007; Rickenmann and Recking, 2011).

$$\left(\frac{8}{f}\right)^{1/2} = \frac{a_1 a_2 (R/D_{84})}{\left[a_1^2 + a_2^2 (R/D_{84})^{5/3}\right]^{1/2}} \tag{7.10}$$

Where a_1 and a_2 are constants with values of 7.5 and 2.36 respectively where D_{84} is used for grainsize (Ferguson, 2007). The Darcy-Weisbach friction factor f can be converted to the Manning's n roughness coefficient using Equation 7.11.

$$n = R^{1/6} (f/8g)^{1/2} \tag{7.11}$$

Bend resistance was approximated using headloss coefficients in line with a roughness partitioning approach using by Shields & Gippel (1995); where headloss due to bends in sub-critical flow with deflection angles of 90º-180º can be approximated using

$$\sum f_{bends} = \sum \rho \left[\frac{2B_i}{r_c}\right] \alpha \bar{v}^2 \tag{7.12}$$

Where ρ is density of water, B_i is channel width at bend i, r_c is the radius of curvature of the bend, α is a kinetic energy correction factor assumed to be equal to 1.15 in uniform flow (Henderson, 1989; Kitts, 2011).

A summary of the roughness partitioning values for each reach are shown in Table 7.4. The values reported for f_{wood} in Table 7.4 represent $f_{unmeasurable}$ and comprise all roughness elements which are not attributable to bed or meander planform roughness, therefore this component includes; direct drag from individual pieces of wood, spill resistance as a result of the wood and energy dissipation as a result of increased sediment erosion and transport.

Other studies (e.g. Kitts, 2011; Shields Jr and Gippel, 1995; Wilcox et al., 2006) have further partitioned roughness to attempt to isolate spill resistance and drag associated with individual pieces of wood, however such a detailed approach is beyond the scope of this study. Wilcox et al (2006) showed the order in which individual roughness components were calculated and summed has a large effect on the values for the non-measured component; with a limited data range across five individual reaches further partitioning would likely be subject to large uncertainty. In the current calculations there are two measurements in which the empirically derived component roughness elements for grain and bend roughness are slightly in excess of the measured roughness, suggesting that the empirical equations used are overestimating these roughness components. It has previously been noted that the equation for head loss at bends may overestimate total loss by a factor of three (Henderson, 1989; Shields Jr and Gippel, 1995).

Comparing the resistance attributable to the combined effects of large wood between the reaches shows a difference in the behaviour of active logjams and the other types. The active logjam (reach 1) has the largest f_{wood} values in each of the individual measurement periods, with the exception the largest discharge for which measurement were taken (1.6 m^3s^{-1}, 17/04/2012). During the largest discharge event the high water logjam (reach 5) has the largest f_{wood} value; with the discharge of 1.6 m^3s^{-1} flow depth was sufficient to cause underflow at the high water logjam. Through all measurement periods the reaches with the largest f_{wood} value were those where the logjam was either causing spill or underflow.

Reach	Darcy-Weisbach components	f_{min}	f_{mean}	f_{max}
1 – Active	f_{total}	1.694	4.676	10.585
	f_{grain}	0.230	0.247	0.277
	f_{bends}	n/a	n/a	n/a
	f_{wood}	1.464	4.430	10.348
2 – Partial/ Meandering	f_{total}	0.089	0.582	2.154
	f_{grain}	0.303	0.307	0.314
	f_{bends}	0.128	0.138	0.142
	f_{wood}	-0.040	0.138	1.712
3 – Complete	f_{total}	0.585	1.960	5.318
	f_{grain}	0.248	0.272	0.315
	f_{bends}	n/a	n/a	n/a
	f_{wood}	0.337	1.687	5.003
4 – Partial	f_{total}	0.186	1.966	4.571
	f_{grain}	0.282	0.296	0.346
	f_{bends}	n/a	n/a	n/a
	f_{wood}	-0.102	1.670	4.289
5 – High Water	f_{total}	0.426	4.637	19.824
	f_{grain}	0.297	0.306	0.308
	f_{bends}	n/a	n/a	n/a
	f_{wood}	0.118	4.331	19.527

Table 7.4 – Summary of mean values for hydraulic resistance partitioning for five study reaches. Summary characteristics for these five reaches are details in Table 7.3.

The percentage of f_{wood} roughness component was between 86%-98% of the total roughness of the active logjam reach (1) and 58%-94% of total roughness of the complete logjam reach (3) across all measurement periods, whereas for other reaches there was much greater variability with ~0%-94% for the two partial logjam reaches (2 and 4) and 28%-99% for the high water logjam reach (5). In all cases where the f_{wood} component was greater than 75% of the total roughness either spill or underflow was observed for the logjams in question. For the high water logjam a mean f=0.6 was found for flow passing under the main bridging log (at discharge of 1.3 m^3s^{-1} or less), but mean f=8.7 where flow depth was sufficiently high (at discharge of 1.6 m^3s^{-1}) to cause flow to cause a step in the water profile and for flow to be forced under the bridging log as underflow. In previous studies the importance of spill as a roughness component has been documented (Curran and Wohl, 2003; Kitts, 2011). Where high flow resistance is observed in the presence of underflow this can be attributable to flow being directed towards the bed and leading to scour around the logjam. Considerable sediment transport has been postulated as leading to increased resistance even in situations where resulting bedforms formed by a mobile bed have negligible form roughness (Bathurst, 1985; Charlton et al., 1978 in Bathurst, 1985; Griffiths, 1981).

7.6.2. Relationship to wood load

Figure 7.5 shows calculated Darcy-Weisbach friction factor f against the measured woodload within the reach for each velocity measurement in this study. Figure 7.5 shows there is a weak relationship of increasing hydraulic resistance with increasing wood load, however the linear regression only has r^2=24.6. A relationship between increasing wood density and increasing shear stress was also found by Manga and Kirchner (2000). In this study there is only a small sample size of five logjams and associated wood loads, it is possible with a greater sample size that the relationship would be stronger.

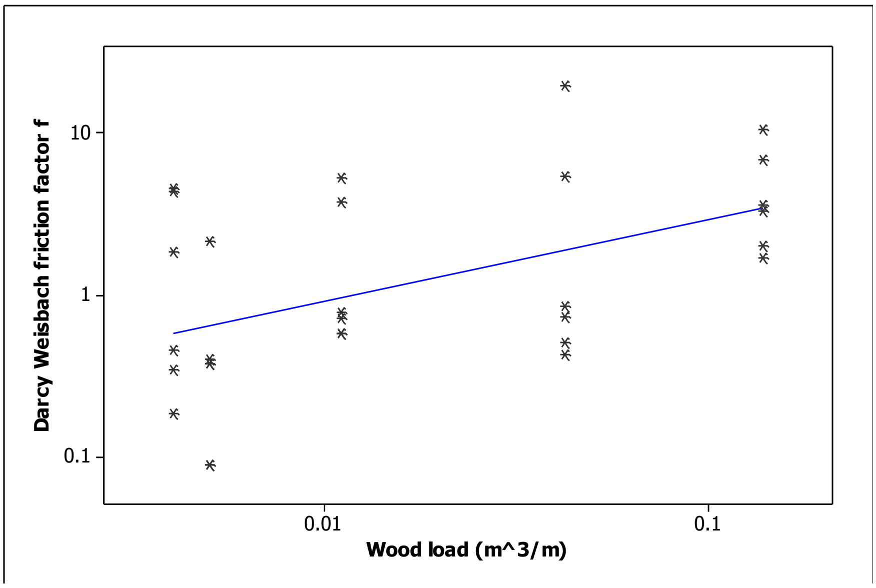


Figure 7.5– log/log plot of reach large wood load against measured Darcy-Weisbach friction factor. These illustrates there is a trend for increasing hydraulic resistance with increasing wood load/wood density, although sample size is relatively small.

7.6.3. Comparisons with other studies

Comparing roughness values between different studies can prove problematic for three reasons, the first is due to differences in the study environments (MacVicar, 2013); roughness is sensitive to slope (Hey, 1979), connectivity to the floodplain (Sear et al., 2010) and the degree of organisation of large wood into structures (Manners et al., 2007). The second is differences in the empirical equations used to partition roughness where f_{grain} is calculated using a wide variety of equations, which deliver different values (Rickenmann and Recking, 2011). The third is the sensitivity of roughness measurements to reach length where a discrete feature such as a logjam is being measured. Curran & Wohl (2003) show spill resistance associated with abrupt changes in velocity can be the largest source of roughness, if this spill is occurring at a single discrete location, then as the distance across which mean velocity is measured ("x" in Figure 7.1) tends to zero, the measured hydraulic resistance, and the f_{wood} component become very large, conversely as the measurement distance x becomes very large then hydraulic resistance tends towards f_{grain}.

Figure 7.6 shows the results of this study along with results from three other studies which partitioned roughness with an in-stream wood component; Kitts (2011), Shields & Gippel (1995) and Curran and Wohl (2003). Kitts (2011) measured hydraulic resistance across logjams in the same river as this study, as well as measurements in a flume. Shields & Gippel (1995) measured roughness in low gradient sand bed rivers in Tennessee, USA and New South Wales, Australia, while Curran and Wohl (2003) measured roughness in high gradient step-pool channels in the Cascade range in Washington, USA. Figure 7.6 suggests a relationship of increasing hydraulic roughness with increasing slope, a pattern also observed by Kitts (2011). This relationship also reflects the dominant type of wood accumulation (Kitts, 2011), with partial, deflector type jams dominating the rivers studied by Shields & Gippel (1995) and active, overflow jams dominating the step-pool channels of Curran & Wohl (2003). The reach in this study and in Kitts (2011) are of intermediate slope and have a variety of accumulation types present, with active jams causing a step in the water profile and thus increasing the friction slope relative to the bed slope and representing measured reaches with the highest slope and friction factor.

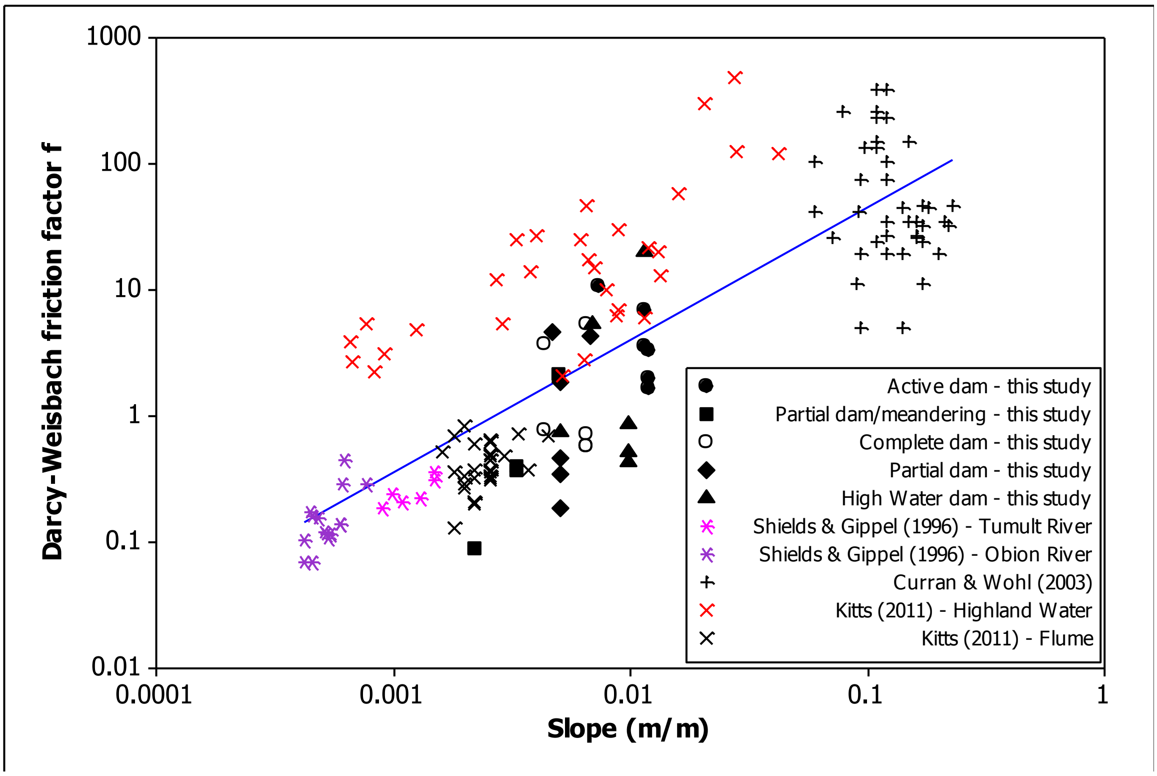


Figure 7.6 –Measured Darcy-Weisbach friction factor f vs slope from this study and values reported by Kitts (2011), Curran & Wohl (2003) and Shields & Gippel (1995). This shows a continuum of values with increasing hydraulic resistance with increasing slope across a variety of river types influenced by large wood.

A linear regression line has been shown in Figure 7.6, however despite having r^2=67.7% a linear relationship is unlikely to be correct as it cannot be extrapolated much beyond the measured data as slope is not a continuous variable but has a range from 0-1. Where mean velocity is measured over a constant distance ("x" in Figure 7.1) as slope gets larger the momentum of the flow due to gravity will increasingly dominate and the influence of roughness elements on retarding flow velocity will decrease, thus we would expect the graph of slope against friction factor to follow a form of Gaussian distribution. When extrapolated to extremes the relationship between friction slope and hydraulic resistance is therefore complex and hard to characterise, although within the current scope of field data of natural channels there does appear to be a continuum. There is a need for further field and flume studies reporting hydraulic resistance values across a range of environments to attempt to replicate the results of Curran & Wohl

(2003), Shields & Gippel (1995), Kitts (2011) and this study and to test the slope/roughness continuum concept postulated by Kitts (2011).

Figure 7.7 shows Darcy-Weisbach friction factor *f* plotted against discharge for: this study, Kitts (2011), Curran and Wohl (2003) and Shields and Gippel (1995). The hashed line drawn onto Figure 7.7 illustrates a conceptual relationship between values indicating as discharge increases to very large values friction factor is asymptotically approaching zero. Extrapolating relationships between studies is problematic as logjam type varies between each study river, however Figure 7.7 also indicates a relationship of decreasing friction factor with increasing discharge within each study. A within study decrease of friction factor with increasing discharge is notable in this study, Curran and Wohl (2003) and Kitts (2011) – Flume data.

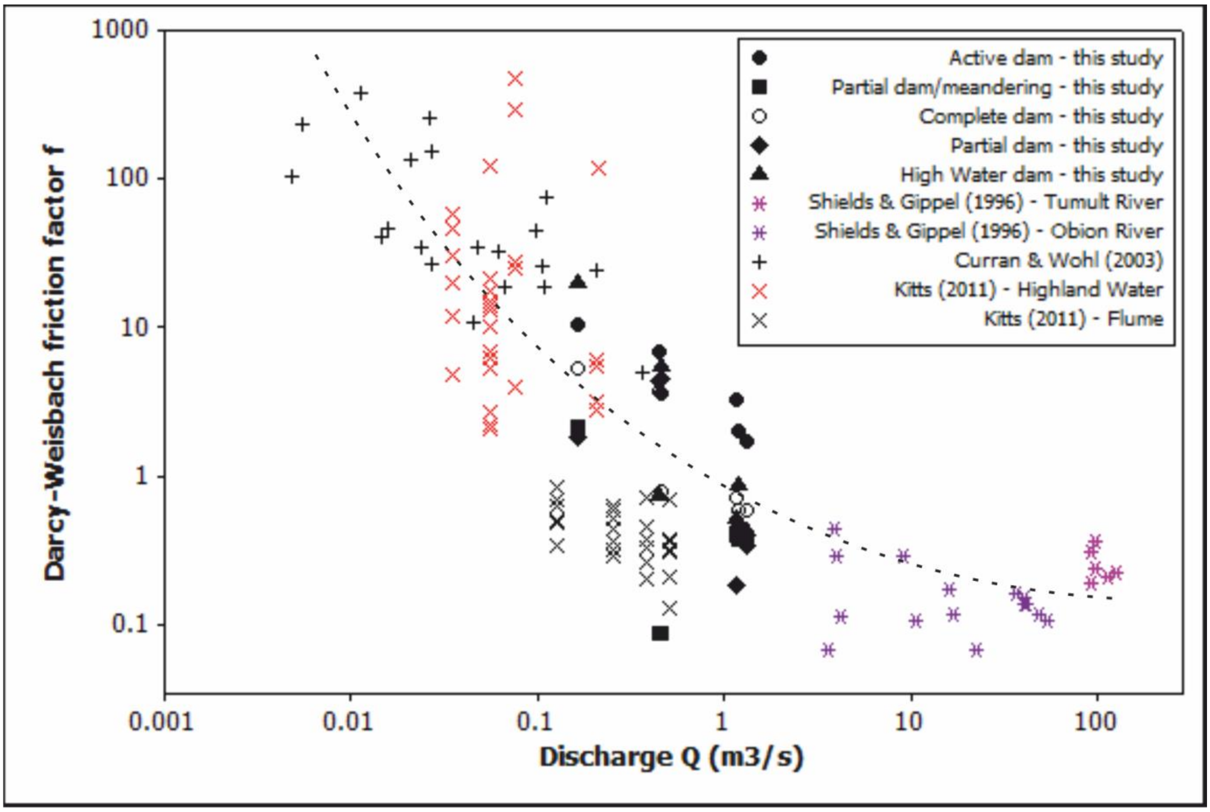


Figure 7.7 - Measured Darcy-Weisbach friction factor f vs slope from this study and values reported by Kitts (2011), Curran & Wohl (2003) and Shields & Gippel (1995). These data show that as discharge increases friction factor decreases.

Calculated friction factor values from this study are lower than those reported by Kitts (2011) in study streams within the same area. There are two possible reasons for this discrepancy; the first is that mean velocity was measured by Kitts (2011) along relatively short lengths of up to 20m (Kitts, *pers. comm.*, 31st July, 2013) which may lead to higher calculated flow resistance values in the presence of spill, due to steeper water slope values. Secondly the measurements in this study, although within the same river as Kitts (2011) were further downstream and in reaches disconnected from the floodplain. Measurements in this study were taken during high flow events of >$0.8m^3s^{-1}$ whereas measurements by Kitts (2011) were taken during low to medium flows of 0.2-$0.3m^3s^{-1}$ (Kitts, *pers. comm.*, 31st July, 2013). Variations in hydraulic resistance values measured between this study and Kitts (2011) could therefore reflect a decrease in f with increasing discharge and thus increasing flow depth (Figure 7.7). The results presented here support a theory postulated by Gregory et al (1985) that as discharge and thus flow depth increases the resistance effects of logjams gradually become "drowned out" in a similar way to increasing flow depth over gravel beds decreases the influence of the bed on flow resistance (Bathurst, 1985; Wilcock, 1996).

Along with data from Kitts (2011) the results of this study add important data to the measured values for wood mediated hydraulic resistance in channels of intermediate slopes and contribute to understanding the continuum of flow resistance values across a range of environments.

7.6.4. Error propagation

Field measurements naturally include an element of uncertainty as to the accuracy of collected data. In studies such as this one small errors in field data can propagate through calculations of hydraulic resistance and become compounded by small errors in other data, resulting in potentially large magnitude errors in final results.

In this study potential sources of field data error have been identified and the margin of potential error estimated from field tests, these are summarised in Table 7.5. Error propagation of these potential errors has then been computed using a 2^{3-1} fractional factorial design experiment with resolution III, that is each variable has been allowed to vary to a higher and lower value and the resulting error for all combinations of higher and lower values have been calculated. High and low error values for hydraulic radius, water slope and velocity were calculated and then the experimental design used to recalculated friction factor. A summary of error propagation and uncertainty is shown in Figure 7.8, this shows maximum levels of uncertainty are -60% to +40% of calculated values, however the majority (68%) of results have an error below ±40% and around half of results have an error below ±25%. Uncertainty analysis shows majority of calculations (80%) potentially underestimate friction factor.

Measurement	**Source of error**	**Uncertainty magnitude**	**Resulting uncertainty in Darcy-Weisbach friction factor calculations**
Water height	Dumpy level & staff	±0.02m	Water slope (S_0)
Water depth	Dumpy level & staff	±0.02m	Hydraulic Radius (R)
Cross-sectional profile	Total Station	Negligible	Hydraulic Radius (R)
Time of tracer cloud passage	Salt dilution gauging	±13%	Reach mean velocity (v)

Table 7.5 – sources of measurement error likely to propagate through hydraulic resistance calculations

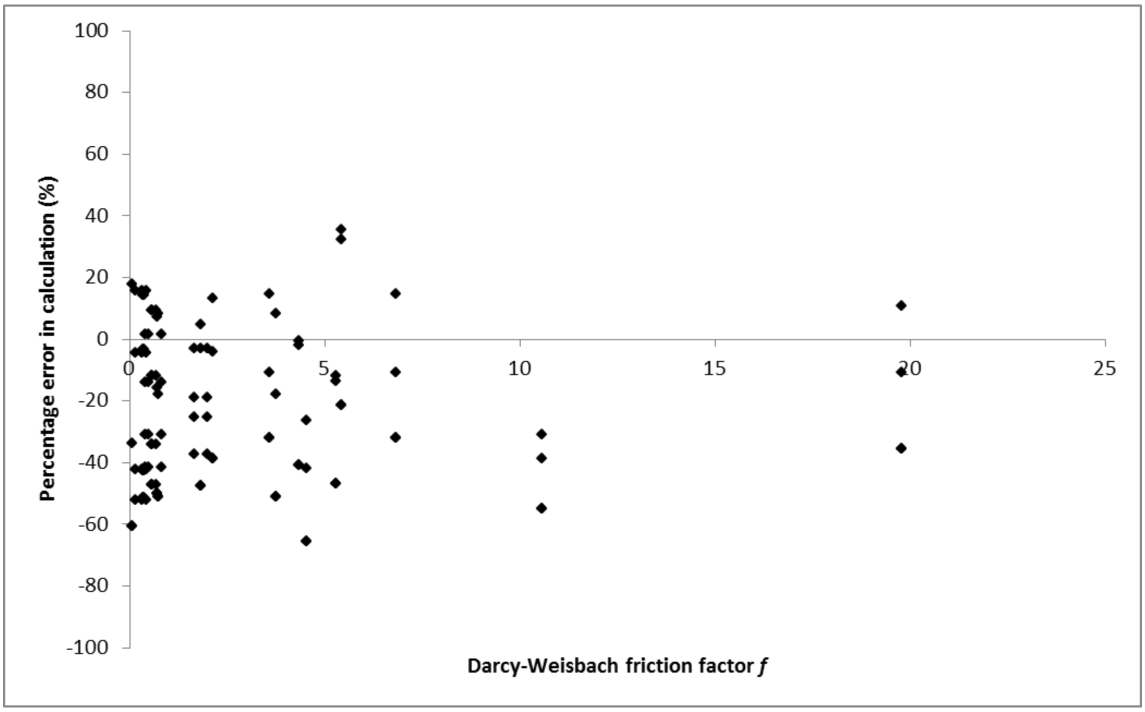


Figure 7.8 – results of a fractional factorial design experiment of error propagation on friction factor calculations. Sources of error are mean velocity (v), water slope (S0) and water depth (hydraulic radius, R).

7.7. Conclusion

Flow resistance calculations for five study reaches in a lowland forest channel give Darcy-Weisbach friction factor (*f*) ranging from 0.09 to 19.8. These values fit into a continuum of hydraulic resistance values relative to slope in the published literature, being higher than those found in low gradient sand-bed rivers which range from 0.1-0.6 and lower than found in step-pool channels which range from 5 to 380. The values reported in this study reflect the hydraulic resistance of a variety of naturally occurring logjam formations typical of those found in lowland rivers.

Hydraulic resistance was found to be higher where logjams caused flow to spill over the structure, or to underflow where flow accelerates underneath the structure, this effect was observed for the high water logjam which had a mean *f*=0.6 where flow was not in contact with a large bridging log and mean *f*=8.7 where flow was forced under the bridging log, similarly a partial logjam had a mean *f*=0.3 in the absence of spill and a mean *f*=3.6 where at least one piece of wood was causing a small spill step.

Partitioning hydraulic roughness is a subjective process, but analysis herein and previous published results in the same study area reveals large wood to be the dominant contributor to hydraulic resistance within lowland forest rivers accounting for 75-98% of total resistance in the presence of logjams causing spill or underflow and a median 65% of total resistance overall across all logjams measured.

This study begins to quantify the hydraulic resistance associated with four broad logjam types found in forest channels. Further studies are needed reporting hydraulic resistance values for a range of large wood loadings and large wood accumulation types in order to move towards a general empirical approach for predicting mean velocity in forest rivers as a form roughness element based on known in-stream wood loadings rather than a feature based approach where hydraulic effects need to be known a priori.

It is important to understand and quantify hydraulic resistance in forest river channels in the presence of large wood in order to understand how large wood loadings can influence catchment scale hydrology and to aid in parameterising flood models.

8. Developing a conceptual model of riparian forest succession following restoration

8.1. Conceptual riparian forest succession within the context of the thesis

Within the context of this thesis the conceptual model of riparian forest growth described in this chapter will be used in hydrological modelling in Chapters 9 and 10. The goal with this hydrological modelling is to look at the immediate and long term effects of river restoration on flood risk and flood hydrology. In order to understand the long term effects on flood hydrology it is necessary to understand and be able to predict how a floodplain forest will develop and change over time. The hydrological model used simulates land cover by using hydraulic resistance coefficients and therefore in order to parameterise model scenarios it is necessary to know the complexity of the land cover. The conceptual model proposed within this chapter allows estimates at 25 years, 50 years and 100 years post-restoration, of the relative complexity of forest cover in term of stem density and dead wood abundance on the floodplain along with levels of input of dead wood to the river channel. The conceptual model will allow informed estimates of hydraulic resistance coefficients to be parameterised in hydrological models, including the one used in this thesis.

8.2. Introduction

The riparian ecotone is the interface between the aquatic and terrestrial environments (Fetherston et al., 1995) and although relatively small in area has many important influences on conditions in both environments (Lowrance, 1998), particularly in the case of riparian forests (Gregory et al., 1991). A riparian forest extends laterally from the channel edge up to the furthest point of river inundation, including all floodplain forest and wetlands (Fetherston et al., 1995), and includes an unusually diverse array of species and environmental processes (Naiman et al., 1993). Riparian forests influence form and processes within the channel (Gregory et al., 1991); acting as a source of dead large wood both to the floodplain surface leading to geomorphic complexity and to the river channel where it has important influences on geomorphology, hydrology and ecology. Forested floodplains are a source of particulate organic matter to the channel (Gurnell et al., 2002), provide shade (Montgomery et al., 2003) and increase bank stability through root stability (Beechie et al., 2006; Shields Jr and Gray, 1992) and decreased erosive power of the channel (Fisher et al., 2010; Gregory et al., 1985; Manga and Kirchner, 2000). Forested floodplains develop a complex patchwork of erosion and deposition as flood water moves between tree boles and emergent roots (Jeffries et al., 2003; Sear et al., 2010). During flooding floodplain forests act as a sink for fine sediment (Fetherston et al., 1995; Jeffries et al., 2003) and flood deposited large wood piles on the floodplain (Steel et al., 1999) which progressively enriches soils and creates unique habitats for birds, mammals and macroinvertebrates (Naiman and Décamps, 1997; Steel et al., 1999).

As knowledge of floodplain forest benefits has increased, policy and practice both in the US and Europe has turned towards encouraging riparian forest restoration and protecting riparian forests (Broadmeadow and Nisbet, 2004; Naiman and Décamps, 1997; Nislow, 2005). Riparian forests can be used to provide ecological and ecosystem services benefits whilst minimising changes to land use e.g. through buffer strips (Nislow, 2010). Riparian forests can reduce runoff and attenuate flood waves (Gregory et al., 1985), compared to modest attenuation effects of grass covered floodplains (Darby and Thorne, 1996) and thus can potentially form part of flood control. Riparian forests also deliver large wood to the river channel to improve ecological conditions, reduce delivery of diffuse pollution through trapping of fine sediment runoff from agricultural land (Cooper et al., 1987; Daniels and Gilliam, 1996; Lowrance et al., 1997) and remove nitrogen and phosphorous from runoff and sub-surface flow (Lowrance, 1992; Lowrance et al., 1997; Peterjohn and Correll, 1984; Sutton-Grier et al., 2013; Wang et al., 2012).

In order to be able to predict the influence of restoring floodplain forests it is necessary to understand the development of a new stand of floodplain forest trees over time. However, comprehensive studies of riparian tree growth are few (Naiman et al., 1998). The complexity of forest ecosystems makes it challenging to develop conceptual models of forest growth (Botkin et al., 1972), which is particularly the case in riparian forests (Robertson and Augspurger, 1999). Riparian forest differs from upland plots in the presence of allogenic disturbances from the fluvial system with flooding and erosion leading to a harsh, non-equilibrium environment which limits seedling establishment (Naiman and Décamps, 1997) and leads to rare natural community types (Nislow et al., 2002). Site specific erosion and deposition as well as lateral channel migration lead to destruction of land as well as creation of new emergent land surfaces (Naiman and Décamps, 1997). In the presence of active erosion there can be chronic stress reducing community structure at the eroding edge leading to the possibility of retrogression where succession is not unidirectional (Decamps et al., 1988; Kupfer and Malanson, 1993). Death of large tree specimens through flooding or bank erosion in mature stands opens up the canopy and allows colonisation of and gap phase regeneration by pioneer species (Fierke and Kauffman, 2005; Keeton et al., 2007; Kupfer and Malanson, 1993).

Structurally riparian forests have greater vertical and horizontal variations than all successional stages in upland forests (Alaback, 1982), in mature riparian forests there are large trees, large snags, massive fallen logs with relatively open multi-levelled canopies and a diverse understory (Naiman et al., 1998). Primary succession in upland plots is characterised by rapid establishment of pioneer species which subsequently prevent the growth of secondary species until their death, in riparian forests this is complicated by allogenic disturbance regimes which can destroy pioneer individuals and allow other trees to colonise subsequent gaps (Robertson and Augspurger, 1999). Where landforms are created and destroyed by erosion, patterns of succession may be different to upland areas dominated by a recurrent cyclical disturbance regime (Kupfer and Malanson, 1993), Van Pelt et al (2006) contend the heterogeneity observed in riparian forest stands in the Pacific North West is due to succession proceeding via multiple pathways determined both by original conditions and subsequent shifts in the fluvial disturbance regime.

Figure 8.1 shows a simplified conceptual model proposed by Naiman et al (1998) in which they characterise a four stage development for riparian forests. Following initial establishment there is a second phase of stem exclusion where all growing space is occupied and species or specimens with a competitive advantage can expand into space occupied by other specimens, out competing and eliminating them. New plant colonisation is mostly excluded and vertical sorting and stratification occurs. In the third phase an understory develops through the establishment of shade tolerant species and gap phase regeneration following mortality of large trees leading to multiple canopy levels. In the final stage there is an old growth assemblage where mortality opens up gaps in the canopy as an autogenic regeneration process (Naiman et al., 1998). Growth rates and time between stages will vary with species composition, disturbance regime and sites. As riparian forests age wood is likely to play a greater role in the aquatic environment (Kasprak et al., 2012) and studies have found correlations between in-stream large wood loadings and the age of the dominant canopy trees (Brooks et al., 2012).

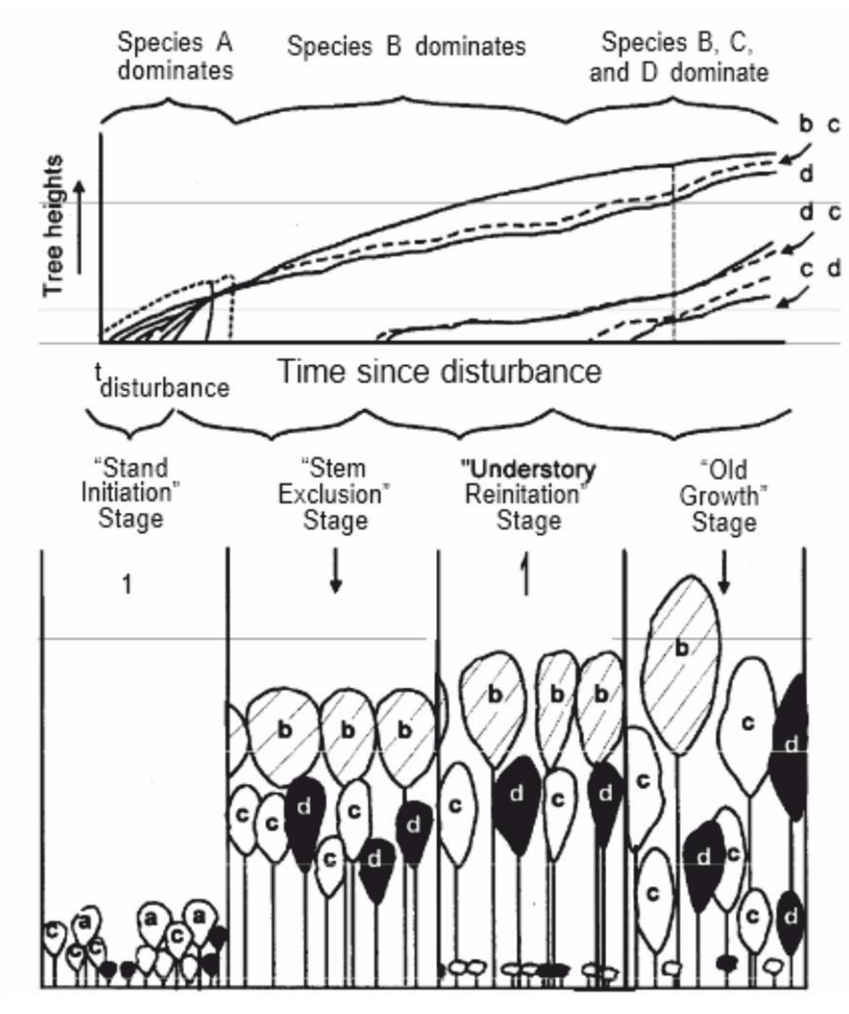


Figure 8.1 - Four stages of forest development on a 'bare earth' site, e.g. following a disturbance. (from Naiman et al, 1998)

The use of numerical models to predict upland forest plot growth and harvest yields is well established in a variety of settings (e.g. Busing and Solomon, 2004; Mikac et al., 2013; Randle, 2000), however growth of riparian forests has received less attention. Riparian forests are challenging to model explicitly due to allogenic disturbance (Hanson et al., 1990), most forest models do not address impacts of forest fragmentation or dispersal mechanics such as transport of seeds by the river or by birds (Hanson et al., 1990). Despite challenges in modelling complex riparian areas it is established that where appropriate old growth reference conditions do not exist vegetation simulation models can be useful in understanding riparian forest dynamics (Kasprak et al., 2012).

The use of forest growth models to predict harvest yields is well established in a variety of contexts. Numerous models have been developed to predict growth in forest plots (Botkin et al., 1972; Huber et al., 2013; Phipps, 1979; Shugart and West, 1977; Wykoff et al., 1982) and forest succession (Pearlstine et al., 1985; Shugart and West, 1977). However comprehensive studies of riparian forest growth are few (Naiman et al., 1998). Conceptual models of riparian forest succession have been proposed (Fonda, 1974; Hawk and Zobel, 1974; Nierenberg and Hibbs, 2000; Pabst and Spies, 1999) and some numerical models developed (Decamps et al., 1988; Nuttle and Haefner, 2007; Phipps, 1979), including models for seed dispersal in riparian environments (Hanson et al., 1990). However the application of the majority of models is to predict forest growth and yields, often under varying controlling conditions, such as changing climate (Huber et al., 2013; Klopf and Hasenauer, 2012; Mikac et al., 2013; Randle, 2000), whereas in order to understand changing in-stream large wood loads a dead wood component and not just a growth component is needed.

Basic riparian recruitment models have been developed (Beechie et al., 2000; Bragg, 2000; Malanson and Kupfer, 1993; McDade et al., 1990; Van Sickle and Gregory, 1990), however these do not include log dynamics such as decay and fluvial transport. Modelling which includes a range of forest dynamics, including dead wood processes, are few. Kupfer and Malanson (1993) used a successional model to predict forest growth and adjusted output at each model time step to simulate changing successional patterns on the eroding bank of a meander bend. Although it incorporated growth dynamics as a result of bank erosion, the Kupfer and Malanson (1993) approach did not take account of recruitment of wood to the channel. Meleason (2001) developed the STREAMWOOD model which simulated wood inputs and decay to streams in the Pacific Northwest. Although STREAMWOOD includes comprehensive log dynamics, inputs of wood to the stream are through a simplified forest gap model and only include coniferous species from the Pacific Northwest. Although STREAMWOOD represents a good approach to simulating in-stream wood dynamics the lack of detailed forest growth and species included means it is not suited to simulating riparian forest succession following restoration in a UK context where broadleaf and mixed forests are the norm.

Lester et al., (2003) and Nislow (2010) describes the use of a riparian forest growth model (NE-CWD) in New England to understand expected in-stream wood loads for different forest types, environments and management. This NE-CWD model predicts forest growth and includes dead wood and riparian dynamics. NE-CWD can be used to predict; forest composition, forest biomass and both in-stream and floodplain dead wood biomass. The combination of forest, riparian and dead wood dynamics makes NE-CWD an ideal framework to investigate the long-term effects of riparian forest restoration on in-stream deadwood volumes and floodplain forest complexity.

In previous applications of NE-CWD in-stream wood loads were shown to be much higher than those found in natural managed forest streams, with highest accumulation rates found for 100-150 years after stand initiation (Nislow, 2010). Low loads in natural streams are a legacy of previous deforestation and forest management which have exerted a strong long-term influence on structure and function of ecosystems (Bragg, 2000; Jones et al., 1999; Nislow, 2010).

8.3. Methods

In order to derive predictions of in-stream large wood loads and the complexity of floodplain surfaces over time following a programme of riparian forest regeneration, a numerical modelling approach was adopted to simulate forest growth and succession. Numerical models of riparian forest growth are comparatively rare worldwide and none exist for a UK context (Broadmeadow, 2012), therefore a numerical model for the North-Eastern United States was used which incorporates growth, dead wood and riparian dynamics (Lester et al., 2003). The purpose of this modelling exercise is to investigate the directionality and magnitude of changes in live wood populations and dead wood biomass, rather than to deliver quantitative, site specific predictions of forest composition at a given time.

Modelling output will be used to develop a conceptual model of riparian forest stand development over time. The development of a conceptual model of riparian forest growth following restoration is essential in order to predict the long term effects of restoration on hydrology, ecology and geomorphology. In this study specifically the conceptual model developed will be used to design land cover scenarios for hydrological modelling in Chapters 9 and 10. These scenarios will be used to predict the long term effects (up to 100 years) of restoring floodplain forests on flood hydrology at the catchment scale.

The model outputs will be validated using a literature review of dead wood biomass values provide quantitative estimates of biomass.

8.3.1. Model Description

The upland and riparian Northeastern Coarse Woody Debris (NE-CWD) model was developed between the United States Department of Agrictulture (USDA) Forest Service Northern Research station and the University of Massachusetts, Amherst. The model is unusual in modelling both upland and riparian elements of forest dynamics. NE-CWD is an extension of an upland large wood model called NE-WOOD, itself a derivation of an original stem growth model NE-TWIGS created by Hilt & Teck (1989) to simulate individual tree growth within stands and predict forest yields (Lester et al., 2003).

The model incorporates live tree dynamics such as seedling regeneration, ingrowth and tree growth and death at the individual tree/subject level. Dead wood dynamics such as snag fall rates, log breakage and decomposition are incorporated to predict residence times of dead wood. Using living and dead wood dynamics the model is able to predict forest biomass and dead wood biomass over time. In addition to live and dead wood dynamics the model also incorporates riparian dynamics with the input of riparian logs through bank erosion and the transport of in-stream large wood by river flow.

Functions for ingrowth (growth of existing trees within the model), diameter growth and mortality are derived from NE-TWIGS (Hilt and Teck, 1989). Snag fall rates are based on forest inventory data from Massachusetts, Maine and New England and snag fall angles and log breakage rates are based on data from Bragg et al (2000). Bank erosion functions are based on data from Idaho (Meleason, 2001 in Lester et al, 2003), decomposition and decay rates were derived from values cited in the literature for the North-eastern USA (Lester et al., 2003).

The model is run using a Monte Carlo approach in which 100 variant models are run and the replicated runs are averaged, expressing output on a per unit area basis, with the exception of riparian elements which are expressed on a per reach length basis. The model calculates yearly timesteps and output is written for every 5 years of model simulation.

8.3.2. Limitations

NE-CWD was designed to simulate forest growth in the North-eastern states of the USA and thus any quantitative predictions are likely to be inapplicable to other geographical settings due to variations in soil type, temperate, climate and elevation leading to variations in tree growth and mortality (Liu and Malanson, 1992). The dynamics for snag fall, bank erosion and log breakage however are based on a wide variety of studies and are assumed to be fundamental processes which are not dependent on climatic variables (Liu and Malanson, 1992). NE-CWD output has been shown to be insensitive to variations in these parameters (Nislow, *pers. comm.*, 24th January 2012), indeed Sobota et al (2006) showed that riparian tree fall directionality is determined largely by valley slope rather than climate variations.

Although inter-continental variations would be expected in live tree growth rates and dead wood dynamics, there is also a great degree of intra-regional variation in such functions which are an inherent limitation on the quantitative predictive power of forest growth models. Boddy and Swift (1983) found order of magnitude variations in dead wood decay rates in South East England between 1.8 – 144.5 years for turnover of material. Lombardi et al, 2011 found deadwood biomass in Slovenian and Italian forests varied between 19-145 m^3/ha for broadly similar live wood biomass values, whilst Christensen (2003) suggest relationships between deadwood and living tree volumes may only be casually connected to regional variability in climate and wind strength.

NE-CWD contains full dynamics for twelve tree species which are not native to the Southern UK and are not present in the New Forest. However the most abundant tree species within the New Forest have similarities to tree species included in the NE-CWD model; European beech (*Fagus sylvatica*) is the same genus as the American beech (*F. grandifolia*), pedunculate oak (*Quercus robur*) is the same sub-genus as white oak (*Q. alba*), silver birch (*Betula pendula*) is the same genus as yellow birch (*B. allaghaniensis*) and Scots pine (*Pinus sylvestris*) is the same sub-genus as Eastern white pine (*P. strobes*). NE-TWIGS simulates tree growth at the genus and sub-genus level for broadleaf species, i.e. it treats all species within a genus as having the same growth rate. Therefore within the limitations of the original growth model variations in species within the same genus are an acceptable limitation.

Data for the tree growth functions from NE-TWIGS are based on a 30 year forest inventory analysis and snag fall rates are based on twenty years of data, therefore predictions beyond these time frames may not accurately reflect observed ecosystem assemblages.

NE-CWD assumes model parameters are temporally invariant; this may restrict application where relationships and dynamics of species may be expected to change over time due to climate change.

Transport of in-stream wood is simulated as a binary removal function; if the model calculates a log is small enough to be transported it is removed from the simulation. There is no representation of any subsequent re-deposition of wood, or transport into the modelled reach of mobile wood from upstream. Therefore the output values for dead wood biomass in-stream only represents the total biomass of stable pieces of wood and should be considered a minimum estimate for dead wood biomass. It is probable that some, if not all, transported in-stream wood will be trapped and deposited within the catchment (Braudrick et al., 1997; Gurnell et al., 2002). Pieces of stable large wood and logjams in a river have been shown to be effective trapping locations for mobile wood in the channel (Braudrick and Grant, 2001; Gurnell et al., 2002; Millington and Sear, 2007; Chapter 5; Chapter 6). As formation of logjams requires a balance between mobile and immobile wood in the channel (potential racked and key pieces respectively) the output of stable wood indicates the abundance of potential large wood trapping locations and thus potential logjam sites. Therefore although in-stream dead wood output may not be an accurate estimate of total in-stream dead wood biomass it is a useful measure of the likely relative abundance of logjam features in small and medium sized forest streams (Gurnell et al., 2002).

8.3.3. Model runs

8.3.3.1. Tree composition

The New Forest floodplain woodland is characterised by a dominance of *Fagus Sylvatica* (beech) with *Quercus robur* (sessile oak), *Fraxinus Excelsior* (ash), *Alnus glutinosa* (alder), *Betula pendula (*birch*)* and some *Ilex aquifolium* (holly) (Jeffries et al., 2003). Ground vegetation is sparse, particularly in areas of beech dominated woodland, but includes *Pteridium aquilinium* (bracken), *Rubus spp*. (bramble) and *Rubus fruticosus* (blackthorn) (Peterken et al., 1996). Within some enclosures there are mono-species forest plots of *Pinus sylvestris* (Scot’s pine), although these are largely restricted to hillslopes (Tubbs, 2001).

Model runs were set up to simulate two forest types which characterise current New Forest bottomland forest composition; i) beech and ii) mixed beech, birch and oak. Further scenarios were set-up to simulate plantation type forests and a mixed composition forest including conifers as a contrast to the broadleaf runs; these were iii) pine and iv) mixed beech, birch, oak and pine. All model runs were conducted using input parameters scaled to the Highland Water, Southern UK (see chapters, 5 and 6 for site description).

8.3.3.2. Plot parameters

Stream width is set to 4.3m, which corresponds to the mean stream width of the Highland Water (Chapter 5); with the stream running from short-edge to short-edge of a rectangular model plot.

Distance from the stream edge to the edge of the plot, perpendicular to the channel is set as a minimum of 30m. A sensitivity exploration of model parameters showed that below 30m plot width deadwood biomass volumes became dependent on plot width, i.e. 30m from the channel is the total area which potentially contributes dead wood to the channel in the model for the given stream size.

Total model plot size is 0.4ha. This corresponds to a roughly square plot with the values for plot width and channel width.

Plot slope angle is set to 0.260 degrees corresponding to the valley slope of the Highland Water (Chapter 5).

For each model run the plot was populated with a single tree per species at the minimum diameter at breast height recognised by the model (13cm), this is necessary in order for the model to allow ingrowth through seedling propagation. It is not possible to start a model run from “bare earth”, at least one tree is needed to generate seeds for subsequent in-growth.

8.4. Results

Figures 8.2, 8.3, 8.4 and 8.5 show summary results from the four model simulations.

All model runs show a number of similarities in forest composition at 25 years, 50 years and 100 years from a bare earth scenario (post-restoration); prior to ~25 years there is negligible deadwood biomass either on the floodplain or in-stream. After 100 years all scenarios are at, or are asymptotically approaching, a maximum live wood biomass value, although this maximum value varies between scenarios. All scenarios initially show establishment of a large number of the smallest tree size class which approach a maximum number of specimens at between 25 and 50 years, before

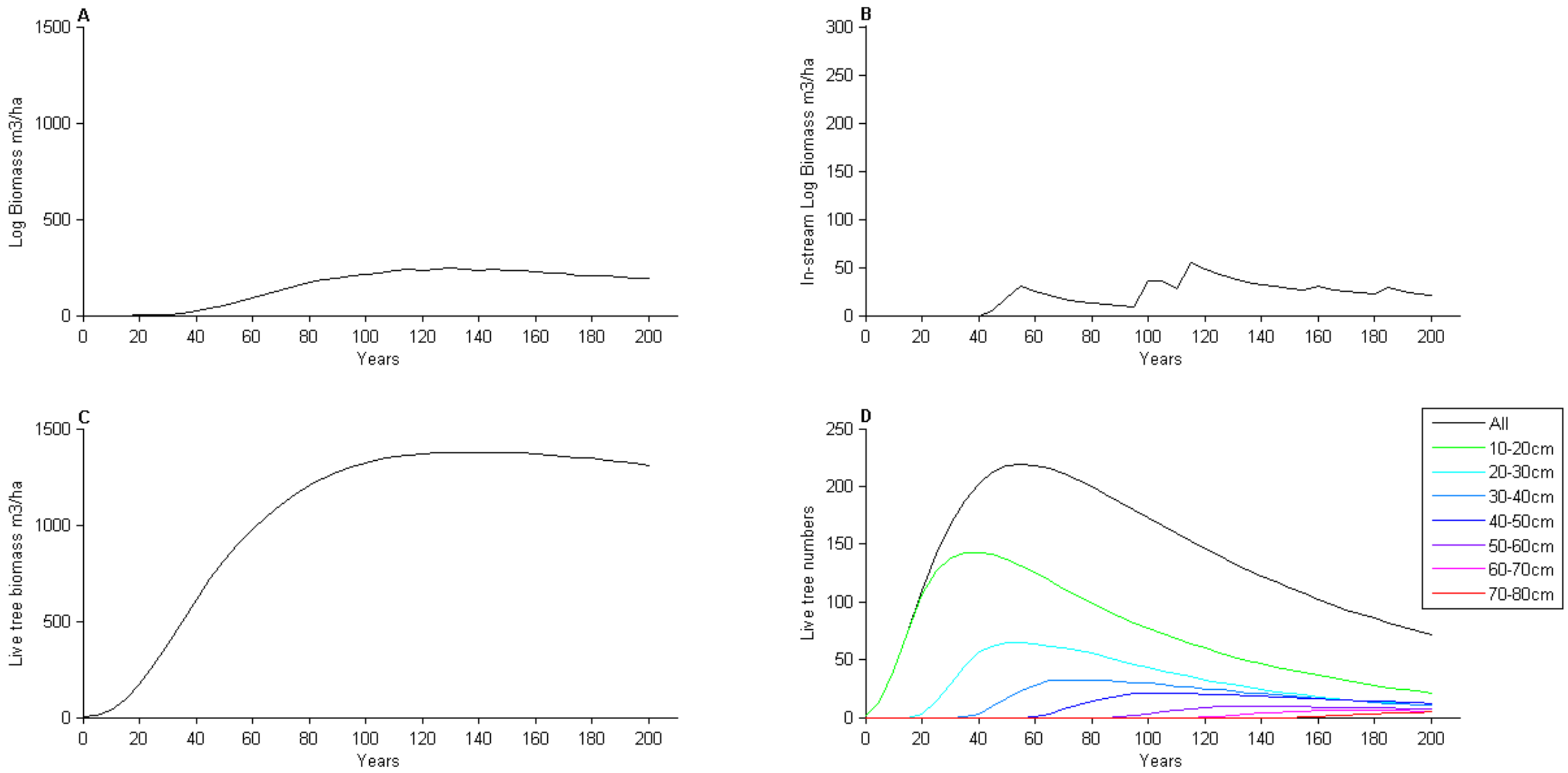

A
Log Biomass m3/ha
Years
B
In-stream Log Biomass m3/ha
Years
C
Live tree biomass m3/ha
Years
D
Live tree numbers
Years
All
10-20cm
20-30cm
30-40cm
40-50cm
50-60cm
60-70cm
70-80cm

Figure 8.2 – NE-CWD model results for Beech forest establishment;

A – dead wood biomass, initially low as trees are of insufficient size to produce large deadwood and mortality is low, after ~40 years tree mortality begins to increase through competition and deadwood volumes peak at around 120 years, after this point there is a balance between deadwood input rate through mortality and removal rate through decay, B – in-stream dead wood biomass, (note difference in y-axis scale to A), follows broadly the same pattern as floodplain deadwood, however values are much lower than floodplain deadwood due to removal of smaller pieces of wood through fluvial transport, C – live tree biomass, increases steadily through succession and reaches an equilibrium value at 100 years, D – number of live trees per size class, initially the stand is rapidly colonised by small trees, as these mature they being to compete and die off, surviving trees increase in size and the stand becomes increasingly dominated by large mature trees, subsequent growth of small trees is limited to gap phase regeneration.

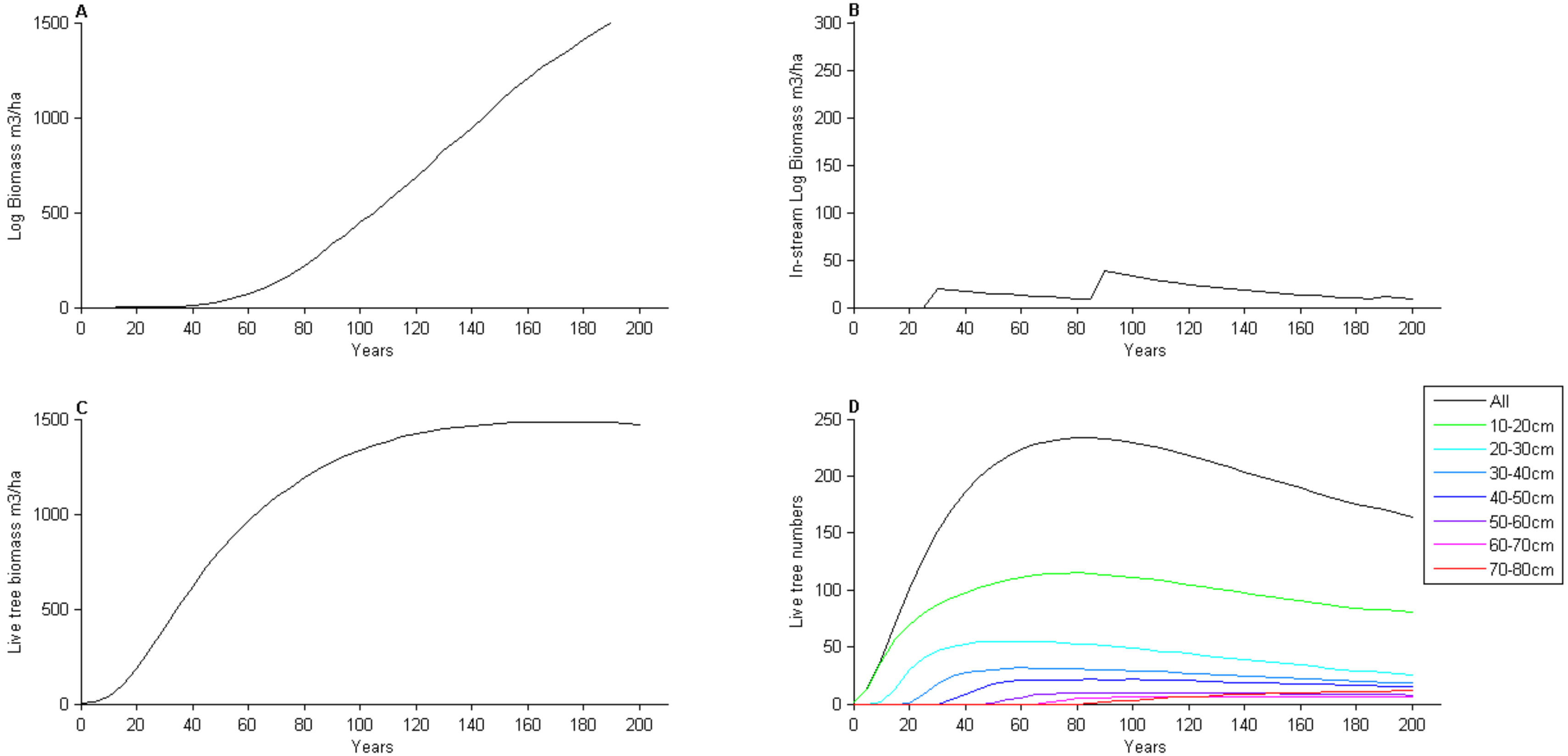

A
Log Biomass m3/ha
Years
B
In-stream Log Biomass m3/ha
Years
C
Live tree biomass m3/ha
Years
D
Live tree numbers
Years
All
10-20cm
20-30cm
30-40cm
40-50cm
50-60cm
60-70cm
70-80cm

Figure 8.3 – NE-CWD model results for Pine forest establishment;

A – dead wood biomass, initially low as trees are of insufficient size to produce large deadwood and mortality is low, after around 40 years tree mortality begins to increase through competition however during the model run decay rates never equal mortality rates and thus deadwood continues to accumulate, B – in-stream dead wood biomass, (note difference in y-axis scale), Although there is a high input of deadwood throughout the model run in-stream deadwood remains at a low equilibrium value $<50m^3$/ha, this is due to deadwood being dominated by relatively small pieces of wood <30cm dbh which are removed from the stream through fluvial transport, C – live tree biomass, increases steadily through succession and reaches an equilibrium value at 100 years, D – number of live trees per size class, the stand is initially colonised by a large number of small trees, however few of these trees mature into very large specimens (>50cm dbh) as a result the stand remains dominated by a large number of relatively small trees of dbh <30cm.

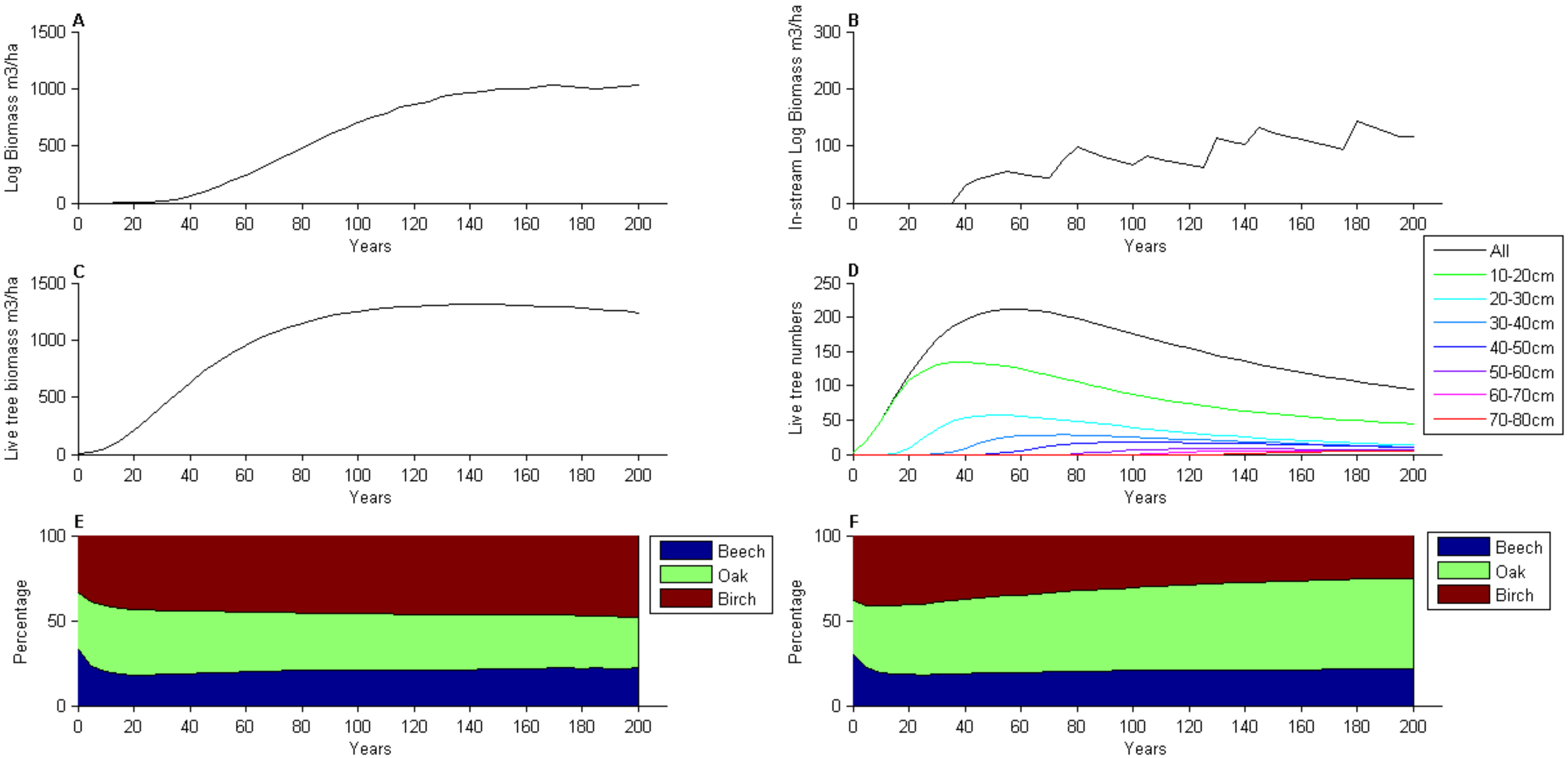

A
Log Biomass m3/ha
Years
B
In-stream Log Biomass m3/ha
Years
C
Live tree biomass m3/ha
Years
D
Live tree numbers
Years
All
10-20cm
20-30cm
30-40cm
40-50cm
50-60cm
60-70cm
70-80cm
E
Percentage
Years
Beech
Oak
Birch
F
Percentage
Years
Beech
Oak
Birch

Figure 8.4 – NE-CWD model results for Mixed broadleaf forest establishment (mixed beech, birch & oak);

A – dead wood biomass, initially low as trees are of insufficient size to produce large deadwood and mortality is low, after ~40 years tree mortality begins to increase through competition and deadwood volumes peak at around 150 years, after this point there is a balance between deadwood input rate through mortality and removal rate through decay, B – in-stream dead wood biomass, (note difference in y-axis scale), follows broadly the same pattern as floodplain deadwood, however values are much lower than floodplain deadwood due to removal of smaller pieces of wood through fluvial transport, C – live tree biomass, increases steadily through succession and reaches an equilibrium value at 100 years, D – number of live trees per size class, initially the stand is rapidly colonised by small trees, as these mature they being to compete and die off, surviving trees increase in size and the stand becomes increasingly dominated by large mature trees, subsequent growth of small trees is limited to gap phase regeneration, E – percentage biomass per species, shows birch is initially quicker to colonise the stand, but proportions of biomass remain steady after 15 years, F – percentage live tree numbers per species, shows birch initially colonises quicker and has a greater proportion of pioneer specimens, however oak increasing dominates the stand as the model progresses.

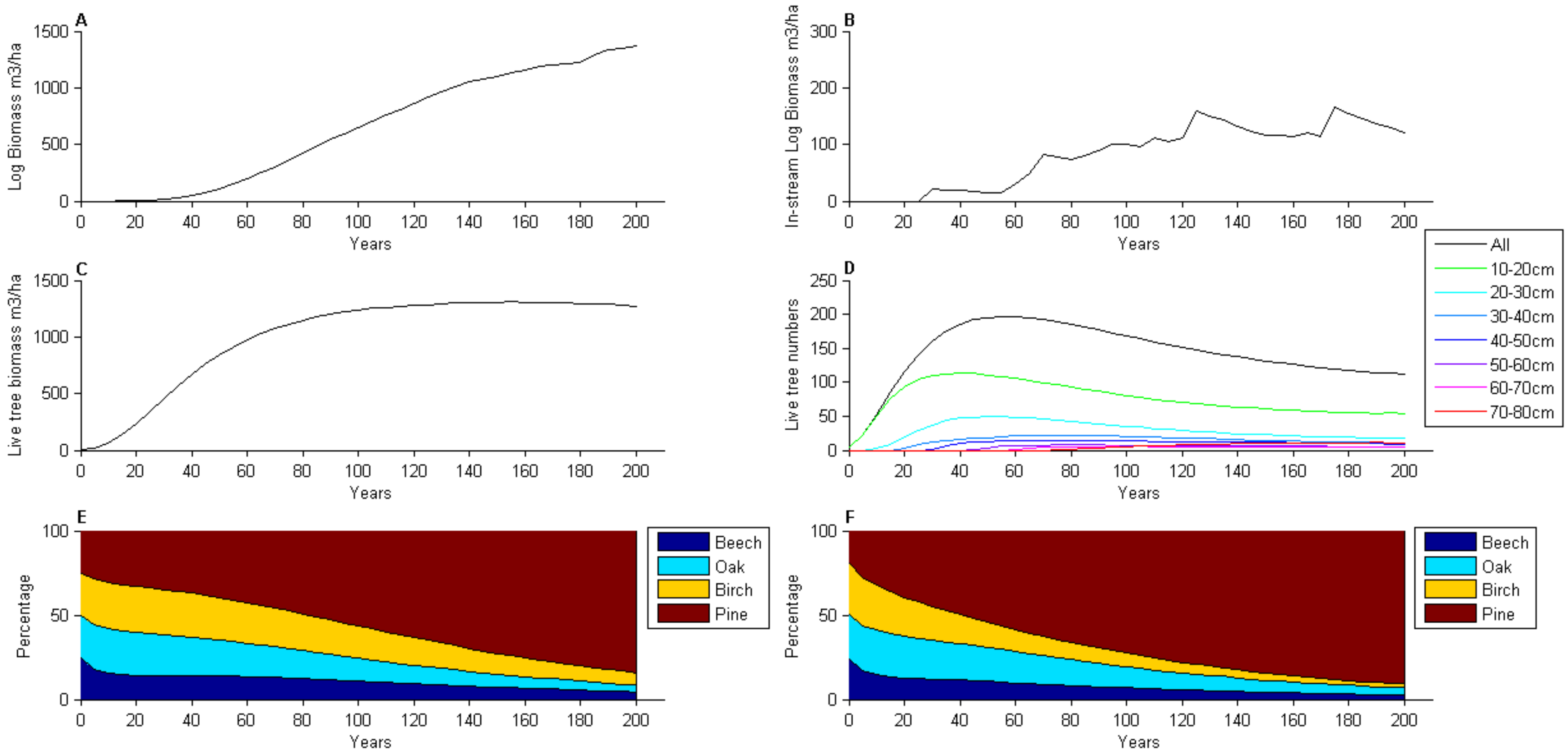

A
Log Biomass m3/ha
Years
B
In-stream Log Biomass m3/ha
Years
C
Live tree biomass m3/ha
Years
D
Live tree numbers
Years
All
10-20cm
20-30cm
30-40cm
40-50cm
50-60cm
60-70cm
70-80cm
E
Percentage
Years
Beech
Oak
Birch
Pine
F
Percentage
Years
Beech
Oak
Birch
Pine

Figure 8.5 - NE-CWD model results for Mixed forest establishment (mixed beech, birch, oak and pine);

A – dead wood biomass, initially low as trees are of insufficient size to produce large deadwood and mortality is low, after around 40 years tree mortality begins to increase through competition, however deadwood continues to increase throughout the model run as decay rates remain lower than mortality rates, other model runs indicate this high input of deadwood is due to pine, B – in-stream dead wood biomass, (note difference in y-axis scale), initially low this increases as mortality of large trees increases after 60 years and reaches an equilibrium at around 120 years, although pine wood contributes high volumes of deadwood (a) this is of relatively small sizes and is thus removed from the channel by fluvial transport, such that in-stream deadwood follows a different pattern to floodplain deadwood, C – live tree biomass, increases steadily through succession and reaches an equilibrium value at 100 years, D – number of live trees per size class, initially the stand is rapidly colonised by small trees, as these mature they being to compete and die off, surviving trees increase in size and the stand becomes increasingly dominated by large mature trees, subsequent growth of small trees is limited to gap phase regeneration, E – percentage biomass per species, shows beech is slower to colonise the stand and that pine increasing dominates to the point it comprises ~75% of living biomass after 200 years, F – percentage live tree numbers per species, shows beech initially colonises the stand slower, however pine increasing dominates the stand as the model progresses.

numbers drop to a half or a third of the maximum after 200 years. As the individual trees mature and move into larger size classes with increasing model run time the plot becomes increasing dominated by larger trees. All model runs appear to be approaching an equilibrium state at around 200 years where successful in-growth of new trees is dependent on gap phase regeneration following the death of larger specimens, so that both live wood biomass and numbers of trees within each size class are constant.

Figures 8.2 and 8.4 are the most similar to the forest composition in the New Forest, and thus are of most interest in developing a conceptual growth model. Figure 8.2 shows that beech forests succession following restoration is characterised by an initial high number of small trees in the 10-20cm dbh category, at around 50 years the total number of trees reaches a peak of 220 trees/ha, which drops to 160/ha at 100 years and to just 75/ha at 200 years. The very largest trees of 70-80cm dbh do not appear until around 150 years. Figure 8.4 shows a similar pattern of growth to Figure 8.2, however deadwood values are much higher. An analysis of individual dead wood results shows that birch contributes substantially more deadwood per unit area than beech or oak and accounts for the higher dead wood values compared to the beech results in Figure 8.2, a study in Baltic forests suggested that birch does have a high mortality rate in mature forests and produces abundant deadwood (Laarmann et al., 2009). Figure 8.4E and 8.4F show that birch is quicker to colonise the stand initially and in the first 20 years accounts for nearly half of all trees in the stand, as the model progresses beyond 50 years oak increasingly dominates the stand and by 200 years over half the trees in the plot are oak.

8.5. Discussion

The results of four forest modelling scenarios using the NE-CWD model show broad similarities in the way a forest develops from a bare earth scenario; all model results show an initial rapid growth of small tree specimens, maturation of these initial trees which are thinned out through competition and mortality, eventually approaching a broad dynamic equilibrium state where the whole plot is forested and new tree specimens appear only through gap phase regeneration. Figures 8.2 – 8.5 show there is a difference in the behaviour of forests with and without pine; pine grows faster than broadleaf varieties and produces more deadwood. Model runs of only broadleaf varieties (Figures 8.2 and 8.4) show live tree and deadwood biomass asymptotically approaching a maximum value, conversely for model runs containing pine (Figures 8.3 and 8.5) although the live wood biomass asymptotically approaches broadly similar values as for broadleaf simulations, the deadwood biomass is still linearly increasing at the end of the 200 year model runs at which point it is ~1.5x the maximum values observed for the broadleaf simulations. Although this difference is quite marked for the floodplain deadwood the same pattern is not observed for in-stream deadwood where all scenarios show year-on-year variance around a mean of 25-100 m^3/ha. Simulations containing pine do not display elevated in-stream deadwood biomass compared to broadleaf model runs, despite having substantially more floodplain deadwood, due to the removal of small in-stream deadwood pieces through fluvial transport. Within the NE-CWD model for the combination of slope and channel size only deadwood in excess of ~30cm dbh will remain immobile; Figure 8.3D shows trees of this diameter or greater are relatively few even in the later stages of a pine model run, thus there are few pine trees of sufficient size to generate stable in-stream wood upon death, compared with broadleaf forest plots of similar age (e.g. Figures 8.2D and 8.4D), leading to lower levels of in-stream deadwood biomass compared to floodplain deadwood biomass.

For broadleaf forest runs (Figures 8.2 and 8.4) the following characterisation of the forest succession can be made:

At 25 years there is negligible floodplain deadwood and no in-stream deadwood, there are around 130 trees/acre, of which 120 are in the 10-20cm dbh size class and the live tree biomass is approximately 300 m^3/ha. This pattern of small trees and no deadwood is similar to that described by Bretz Guby and Dobbertin (1996) who recorded a high abundance of 'polewood' (dbh 10-30cm) with very low deadwood volumes for early successional Swiss forests, and Laser et al (2009) who found just 2.5 m^3/ha of in-stream deadwood in young riparian forest stands in Maine, USA.

At 50 years deadwood biomass is approximately 100 m^3/ha, with in-stream biomass around 25 m^3/ha. Tree numbers are at a maximum value of 220 trees/acre; of which 130 are in the 10-

20cm dbh and 60 in the 20-30cm dbh size categories. Total live wood biomass is around 900 m^3/ha. Liu and Malanson (1992) found the first 50 years of riparian forest stand development is a transitional period towards establishment of mature trees. Deadwood values are still low in both the floodplain and channel ; an absence of deadwood in young forests plots has also been observed in the Pacific North West (Van Pelt et al., 2006), where significant logs are not created during the first 50 years.

At 100 years deadwood biomass has reached an equilibrium value of around 200 m^3/ha, with in-stream dead wood biomass in the range 25-50 m^3/ha. Live tree numbers are declining with about 170 trees/acre, of which around a third are in the size categories 30-60cm dbh. Total live wood biomass is approaching an equilibrium value of around 1300 m^3/ha.

The complexity of forest ecosystems makes it difficult to develop conceptual model of forest growth (Botkin et al., 1972), and this is especially true for riparian forests with additional allogenic disturbances (Hanson et al., 1990). Figure 8.6 shows a proposed simplified conceptual model for broadleaf riparian forest succession following restoration developed from the broadleaf model runs (Figures 8.2 and 8.4).

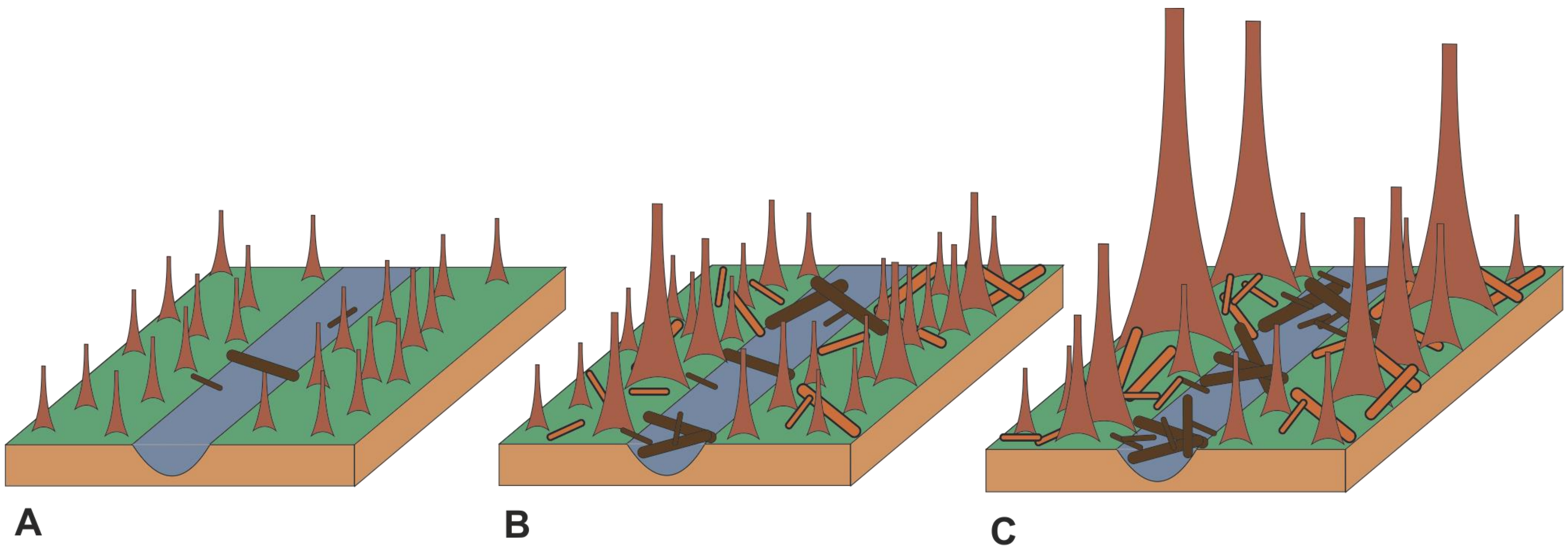


Figure 8.6 – conceptual model of broadleaf riparian forest succession following forest restoration to a bare earth site.
A (25 years) – an even aged cohort of trees grows up in the first few decades post-restoration, there is little competitive pressure, almost no large deadwood and in-stream deadwood is restricted to background levels representing wood transported in from upstream. B (50 years) – The forest reaches a maximum number of live tree specimens, at this point competition is increasing and beginning to limit seedling in-growth, biomass for deadwood and in-stream deadwood is starting to rise. C (100 years) – mature forest, live tree biomass is at equilibrium and is at its maximum value, although the number of trees has declined from the peak values seen in B, forest biomass composition is dominated by fewer, larger trees. Seedling in-growth is very limited and restricted to gap-phase regeneration upon the death of larger trees. Deadwood biomass both on the floodplain and in the river channel is at, or asymptotically approaching, maximum values.

The conceptual model proposed in Figure 8.6 also shares similarities with observed riparian forest growth in other environments; the model expands on theories put forward by Naiman et al (1998) (Figure 8.1) to explicitly include deadwood and the fluvial environment into a model of riparian forest succession. Van Pelt et al (2006) described a 300 year vegetation chronosequence for mixed riparian forests of the Pacific North West as being initiated by small fast growing trees, followed by intense in-stand competition in which over 90% of stems die off. Overbank sedimentation promotes the development of a floodplain terrace, smaller trees die off and mature late successional trees become established, eventually leading to a complex multi-level, multi-species forest which appears at 200-250 years. Deadwood was observed to be absent in young forests, as trees are not of a sufficient size to generate significant logs, with large logs not appearing until well into the second century post establishment (Van Pelt et al., 2006). Nanson and Beach (1977) also describe early riparian forest succession in British Columbia Canada as characterised by dense even aged stands with tree density reaching a maximum at around 200 years.

8.5.1. Comparison with values from the literature

Validation of riparian forest model output is problematic in that there are very few studies reporting in-stream deadwood biomass in the context of forest stand age, or in comparison with live tree biomass, and none from the Southern UK (see Table 4.1 for a list of values from the literature). However even in the absence of field validation, forest modelling of processes, directionality and composition are still recognised as heuristically useful (Hanson et al., 1990). Studies in the literature on aspects of in-stream wood dynamics and processes do report total biomass estimates but without context these can only be used to constrain the upper bounds for observed in-stream deadwood loads. Figure 8.7 shows a range of literature values for in-stream deadwood biomass in association with riparian forest types similar to those used for model scenarios, along with the range of in-stream wood loads found in the Highland Water (Chapter 5). As already mentioned output from NE-CWD is likely to underestimate total in-stream dead wood biomass as the model simulates smaller pieces being completely removed from the system by fluvial transport. Given limitations in the model results for in-stream dead wood, the literature and field data in Figure 8.7 shows that results are of the same order of magnitude.

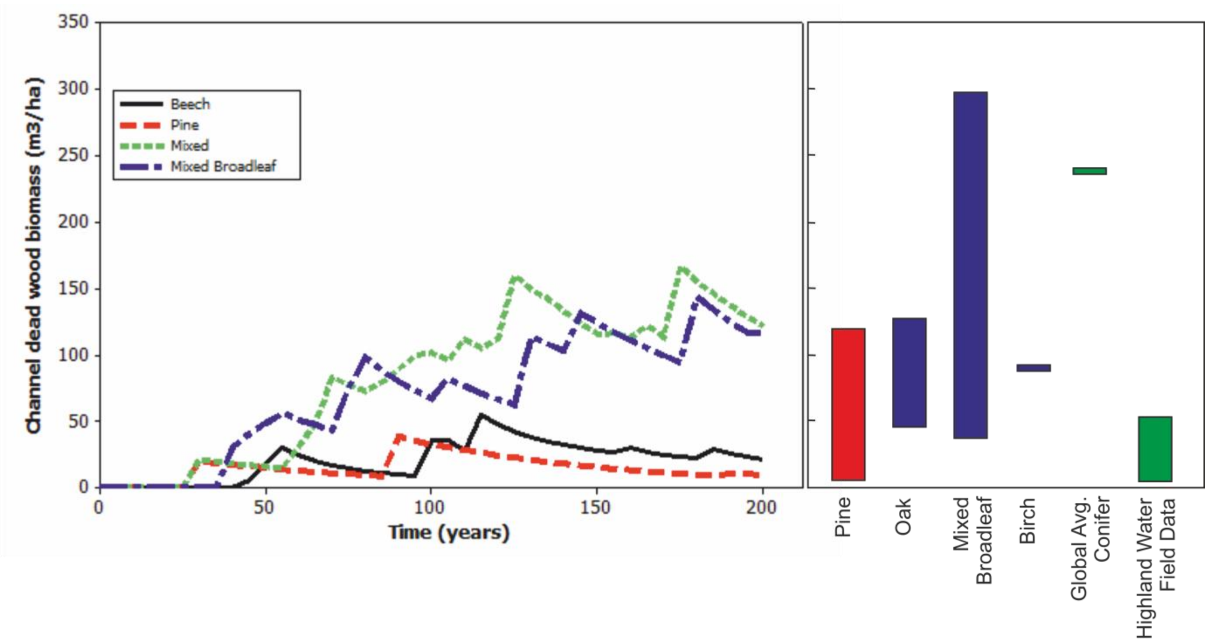


Figure 8.7 – comparison of NE-CWD output for in-stream dead wood biomass over time and data from the New Forest (see Chapter 5) along with values of in-stream deadwood from the literature (Gurnell et al., 2002; Harmon et al., 1986) (see Table 4.1 for full details of literature values).

Compared with the scarcity of in-stream dead wood biomass values in the context of riparian forest stand age there are a number of studies reporting values for live tree and deadwood biomass in beech dominated forests, which can be checked against modelling output. Figure 8.8 shows the average live tree and deadwood biomass from Monte-Carlo runs plotted against each other for each five yearly step in the modelling simulation of riparian beech forests, showing a hysteresis as the forest approaches maximum living wood biomass. Literature values for European beech forests (see Appendix A for a list of data points and references) are plotted onto to this figure showing despite a high degree of variance in deadwood values the literature values are of the same order of magnitude as the modelling results and appear to follow a similar relationship of increasing deadwood biomass with increasing living biomass. Furthermore, data from the Pacific North West (Figure 8.9) (Van Pelt et al., 2006), although geographically and climatically dissimilar, shows similar patterns of growth curves for pioneer species and for evergreens as observed in modelling results. In the multi-species sequence in Figure 8.9 the pioneer species (salix and alnus) are analogous to polewood (dbh 10-30cm) in NE-CWD mono-species runs, showing rapid establishment and peak values around 50 years from initiation before a decline as the forest matures. Acer is analogous to large individual specimens (>40cm dbh) in NE-CWD mono-species model runs and appears later in the model runs around 100 years from initiation and has a very similar shaped curve asymptotically approaching a maximum value as the forest reaches maturity.

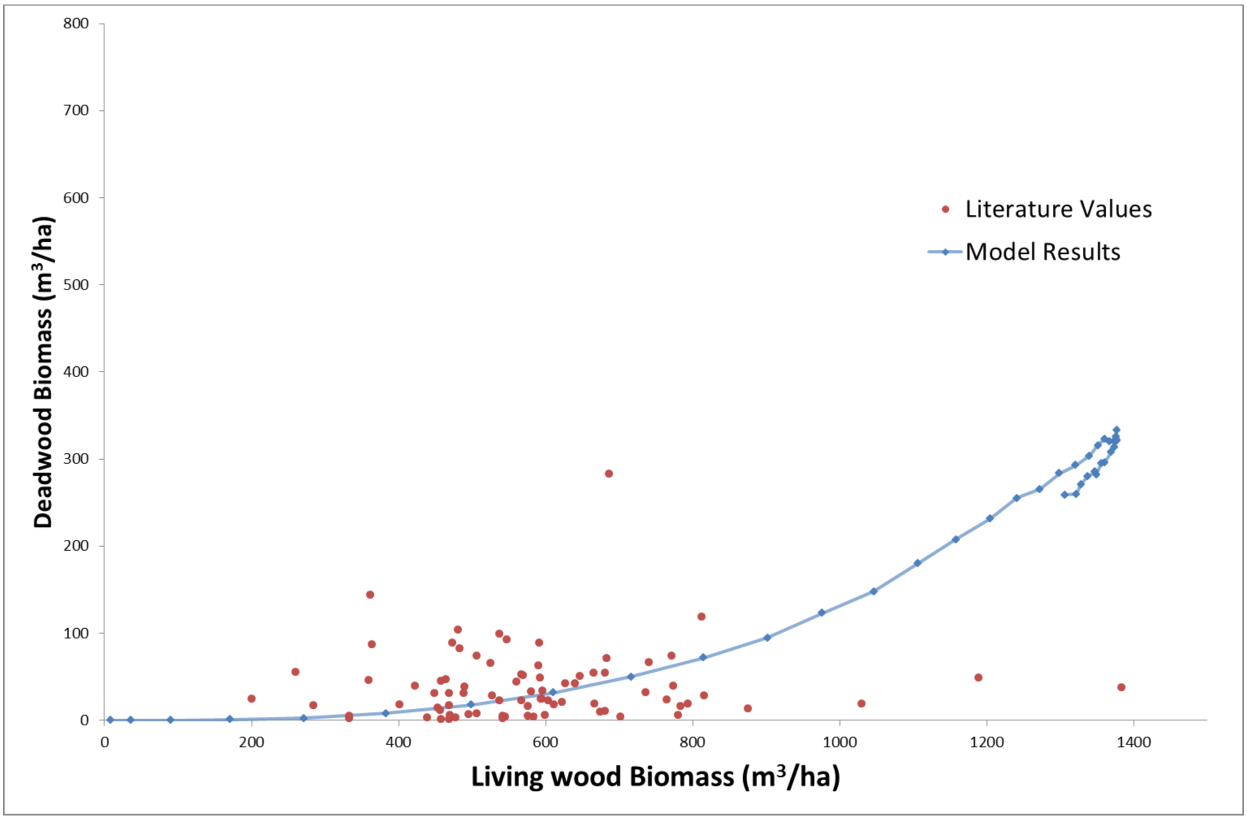


Figure 8.8 – Comparisons between modelling output over a 200 year simulation and literature values for beech forest live wood and deadwood biomass. Literature values from Lombardi et al (2010) and studies reported by Christensen et al (2005).

It is not possible however to validate NE-CWD results against European data for the later stages of model runs. There are only three values in the literature for live wood biomass in excess of 900 m^3/ha which the model suggests will characterise mature beech forests after around 60 years of unmanaged growth and these three values from Italy have comparatively low deadwood biomass relative to the model predictions. There is also only one Slovenian forest which has deadwood biomass in excess of 150 m^3/ha. It is possible the literature does not contain forest inventory results for any sufficiently mature forests, or that given extensive past management that such intact unmanaged forests do not exist in Europe (Lombardi et al., 2010). Given that some studies report values for forest reserves established over 100 years prior to measurement (e.g. Christensen et al., 2005) it is more likely the NE-CWD model is overestimating the potential productivity of stands. The potential overestimation in NE-CWD for beech forests could either be due to climatic variations between the North-Eastern US and Europe. Alternatively overestimation may be due to the process of development and validation of the original NE-CWD model growth coefficients. Growth coefficients were derived as relative to maximum possible tree growth defined as the rate of the fastest growing 10% of trees within a plot, and with data from forestry plots. It is possible the forestry plots used were subject to various degrees of management and optimisation of tree growth for harvesting (e.g. thinning) and thus the data on beech trees represents a higher growth potential than in natural European forests.

Given the lack of supporting literature values it is not possible to verify the quantitative output of the NE-CWD model for model results of 75 years growth or more, however the directionality and magnitude of the model results and the successional stage of the forest model are supported by the literature.

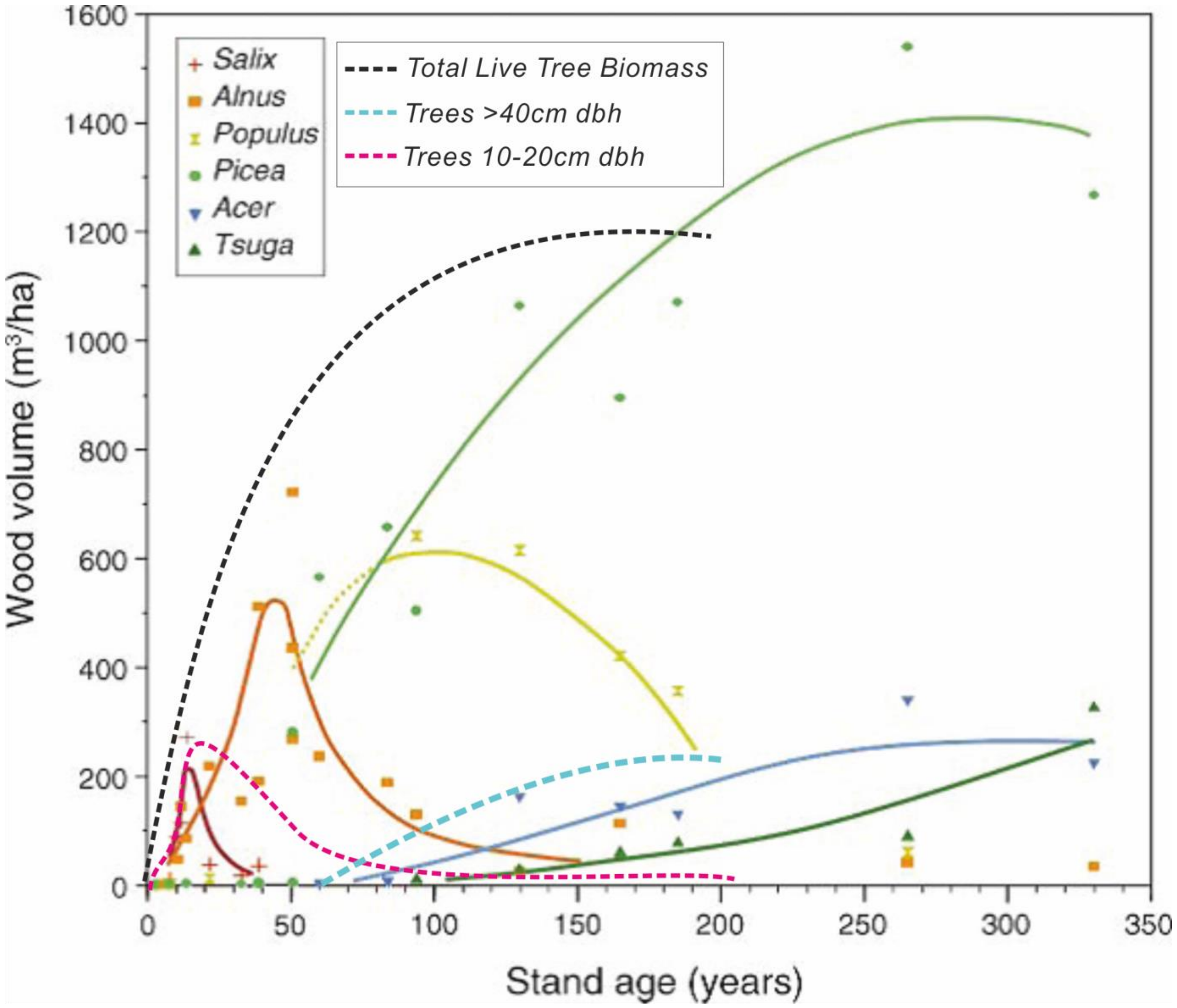


Figure 8.9 – Field measurements of living biomass for six main tree genera in a 300 year chronosequence for forests of the Pacific North West, USA showing similarities in species curves with results from NE-CWD. Thick hashed lines show model curves from NE-CWD model runs. Modified from Van Pelt et al, 2006.

8.6. Conclusion

Results from a USDA Forest service numerical model predict how a riparian forest stand will develop over time. Results shows early succession stands are dominated by abundant polewood with negligible deadwood either on the floodplain or in the river. In later successional phases the polewood matures to larger trees and competition thins the total number of trees per unit area, as trees die off the amount of deadwood increases on the floodplain, but deadwood within the river channel increases slower than that on the floodplain due to removal of smaller deadwood through fluvial transport. In mature successional phases there is a dynamic equilibrium of live wood biomass, floodplain deadwood and in-stream deadwood; live wood biomass is balanced by gap-phase regeneration of seedlings upon the death of a large tree and deadwood biomass is an equilibrium where decay rate is roughly equal to the rate of input of deadwood. A three-phase conceptual model of riparian forest succession is proposed and broadly validated with data from the literature. The conceptual model needs to be critically tested against unmanaged broadleaf riparian forest plots of varying ages. The conceptual model proposed herein provides a valuable tool for exploring the impacts of potential riparian forest restoration on hydrology and nutrient cycling over time.

9. Peak flow hydrology simulation in Lymington catchment using OVERFLOW. I: Model set-up and calibration

9.1. Abstract

OVERFLOW is a reduced complexity hydrological model designed as an exploratory tool for investigating the effects of spatially distributed channel and catchment management strategies during major rainfall events. The model is designed to provide guidance as to potential impacts of different spatially distributed catchment intervention and land use management measures on flood peak timing, depth and duration. Simplifications and assumptions incorporated in the model allow for rapid model run times and thus the investigation of multiple land cover scenarios. The model is designed to identify which types and spatial arrangements of interventions produce reductions in peak discharge and time over peak. By running multiple combinations and extent of interventions the most promising scenarios for reducing downstream flood risk can be identified. Where applied as part of a site-specific investigation to inform placement of interventions, OVERFLOW can form the first part of a modelling strategy. Firstly OVERFLOW could be used to run multiple scenarios identifying those which are indicated as substantially reducing flood peaks. Scenarios identified through OVERFLOW modelling could then be studied in more depth using more strongly physically-based, computationally intensive, models of the type conventionally used for flooding investigations to derive quantitative predictions of flood hydrology.

The model rationale and architecture are described, along with the processes of setting up a digital elevation model, channel network, hydraulic geometry and hydraulic roughness parameters for a study catchment. The model is calibrated for a flood event of 30th March 2006 on the Lymington River at Brockenhurst and an example model investigation is presented. The calibration hydrograph shows excellent agreement to the measured flood event with a Nash-Sutcliffe of 0.98.

9.2. Use of OVERFLOW in the context of the overall thesis

The model description and calibration of OVERFLOW described in this chapter forms the first of two chapters describing a heuristic hydrological modelling exercise to compare the effects of spatially distributed land use on the directionality and magnitude of flood peak change between different scenarios.

The model set-up and calibration is used in an investigation described in detail in Chapter 10. The purpose of this chapter is to provide a rationale for conducting a heuristic modelling exercise and for using the OVERFLOW model to conduct the investigation. The overarching aim of the investigation is to give insight into the use of engineered logjams and changing forest cover to alter flood hydrology as part of a flood mitigation programme.

9.3. Introduction

Historically, land management priorities related to flood risk focused on increasing hydraulic connectivity within a catchment and in increasing channel capacity and reducing hydraulic resistance of channels and floodplain; the goal being to convey rain falling on the catchment into the river network as quickly as possible and to minimise the travel time of water through the drainage network (Sear et al., 2000). More recently such an approach has been shown to be counterproductive, by decreasing flood wave travel times & increasing peak flood flows. In addition channelized river systems disconnected from their floodplains have associated ecological and morphological problems which need to be addressed to improve the ecological status of the river, particularly in the EU as part of the Water Framework Directive.

Urban flood risk is perceived to be increasing and is likely to increase in the future through climate change, as a result alternatives to increasing the extent and height of existing flood defence schemes are being explored (Johnson and Priest, 2008). There is considerable interest in the use of spatially distributed diffuse land use interventions in river catchments as an alternative method for mitigating downstream flood risk (Defra, 2005a; Defra, 2007; Johnson and Priest, 2008). Land management changes at the reach scale such as planting riparian woodland buffer strips have been shown to be effective at attenuating both runoff generation and runoff velocity (e.g. Anderson et al., 2006; Carroll et al., 2004; Heathwaite et al., 1990; Marshall et al., 2009), and small scale land use changes, such as improved pasture and restoration of riparian vegetation have been shown to reduce peak river flows during extreme rainfall events (e.g. Liu et al., 2004; Marshall et al., 2009; Wilkinson et al., 2010).

Changing land cover from open pasture or scrub vegetation to forests can increase interception (Robinson et al., 2003), increase infiltration (Bracken and Croke, 2007), increase temporary storage capacity (Ghavasieh et al., 2006), attenuate runoff (Broadmeadow and Nisbet, 2009) and slow conveyance (Lane et al., 2007). The effects of vegetated land cover on hydraulic connectivity and runoff generation have been shown at multiple scales from patch to catchment (Bracken and Croke, 2007; Bull et al., 2000; Lasanta et al., 2000). There is not a linear relationship between the total area of land cover change and extent of runoff attenuation, and the spatial arrangement of land cover change within a large catchment can be more important than the total proportion of the catchment which is changed (Cammeraat and Imeson, 1999; Fitzjohn et al., 1998; Ludwig et al., 2005). Logjams in rivers have been shown to slow the passage of flood waves (Gregory et al., 1985; Sear et al., 2006; Thomas and Nisbet, 2007; Thomas and Nisbet, 2012), by increasing channel hydraulic resistance (Curran and Wohl, 2003; Kitts, 2011; Shields Jr and Gippel, 1995). Logjams also increase connectivity with the floodplain, forcing water out of bank onto the floodplain earlier in flood events than for non-logjam reaches (Sear et al., 2010). Increased connectivity with the floodplain allows water to be stored in non-critical floodplain areas and reduces flood risk at vulnerable downstream urban locations (Liu et al., 2004). Although logjams have been shown to reduce downstream flooding this has so far only been demonstrated in relatively small reach scale modelling studies (Sholtes and Doyle, 2011; Thomas and Nisbet, 2012).

The extent to which local effects can be transferred to larger, catchment scales remains uncertain (O'Connell et al., 2007; Parrott et al., 2009; Pattison and Lane, 2012b). The interactions between multiple small scale changes creates potential problems, although each one may reduce local peak flows multiple changes may result in alterations to sub-catchment response times, potentially synchronising or desynchronising sub-catchment flood waves resulting in greater variability in downstream flood peak response (Odoni and Lane, *In Prep;* Pattison et al., 2008) with the relative timing of sub-catchment flood waves explaining between 10-20% of the downstream flood peak magnitude (Lane, 2003) Such interactions necessitate a catchment-scale evaluation of land use change with respect to flood hydrology and flood risk.

Catchment scale modelling exercises with multiple land-use scenarios present three major challenges to conventional hydrological models such as ISIS (Halcrow, 2001) or HecRAS (HEC-RAS, 2010) (Odoni and Lane, *In Prep*). Firstly the analysis needs to be spatially explicit as the downstream hydrograph response depends on the responses of other parts of the catchment, ruling out the use of effective runoff models with approaches such as width functions or instantaneous unit hydrographs (Liu et al., 2003; Odoni and Lane, *In Prep;* Olivera and Maidment, 1999). The second problem is related to the quantity of model runs needed to investigate multiple land use types at multiple locations. Assessing all possible combinations would require $(l \times N)^m$ simulations, where N is type of management intervention, m is the possible locations and l is the number of possible magnitudes or intensities of management intervention, e.g. sparse forest growth and dense forest growth (Odoni and Lane, *In Prep*). Investigating three possible interventions with three levels of magnitude/intensity for a small sized catchment (25km^2) with 250m long reaches this could be in excess of 1000 model simulations. Such a modelling exercise would represent a serious computational problem unless either the number of simulations or the complexity of the computation can be reduced (Odoni and Lane, *In Prep*). Furthermore in addition to computational time there is a substantial amount of user time involved; within ISIS for example, each case would need to be set up individually by an experienced ISIS user. Thirdly, when performing a catchment scale investigation data needed for the set-up of conventional hydrological models may often not exist, or may not be available at a sufficient spatial resolution (Odoni and Lane, *In Prep*). Parameters affecting the balance of infiltration and runoff such as local lithology, soil depth and vegetation cover, as well as channel widths and depths (or cross-sectional profiles) are problematic to measure over large spatial scales at a high spatial resolution, and may need to be estimated in setting up physically based models (Odoni and Lane, *In Prep*). Such estimation may lead to a high level of uncertainty and has implications for model complexity where a model necessitates data which are not available (Odoni and Lane, *In Prep*).

The model described here is a computationally efficient, reduced complexity hydrological model designed to allow testing of multiple combinations of spatially-distributed land-use scenarios with the aim of investigating their impact on reducing downstream flood risk.

9.4. Model Description

OVERFLOW was developed as part of the *Slowing the flow at Pickering* project by University of Durham and Forest Research (Odoni and Lane, 2010) to investigate the possibility of using spatially distributed "catchment interventions" within the Pickering Beck catchment area with the objective of minimising flood risk in the town of Pickering. At present the model remains unpublished in the academic literature, however a draft paper in preparation is included in Appendix B. OVERFLOW is a 'minimum information requirement' model (Odoni and Lane, *In Prep*), meaning it is designed to be simpler to use than traditional hydrological models and allows specialists and non-specialists to investigate flood behaviour. OVERFLOW is based on a spatially-distributed unit hydrograph approach (Du et al., 2009; Liu et al., 2003; Maidment, 1993; Maidment et al., 1996; Olivera and Maidment, 1999; Saghafian et al., 2002) using the time to equilibrium approach of Saghafian and Julien (1995). The model is designed to simulate flood events for which the catchment is at, or approaching saturation and thus have standard runoff coefficients of greater than 70%, and for rainfall events where the event duration approaches or exceeds the time to equilibrium. OVERFLOW focuses on the generation of rapid runoff routes and their contribution to peak flows and thus major reductions can be made in model complexity (Odoni and Lane, *In Prep*).

The model builds on the spatially-distributed unit hydrograph approach of Maidment et al (1996) and the subsequent work of Saghafian and Julian (1995) and Saghafian et al (2002) by allowing for time dependent evolution of travel times to reflect the gradual wetting of the catchment during the rainfall event and the greater proportion of flow delivered by rapid runoff. Furthermore the model allows evolution of flow paths as a function of time to represent time-dependent transitions from channel to overbank flow.

Equilibrium runoff for the catchment is achieved when calculated runoff is equal to the maximum potential runoff for the catchment under a given constant rainfall intensity. Equilibrium runoff will vary as a function of the rainfall intensity, topography and physical properties of the catchment. The time to equilibrium runoff under different rainfall intensities can be used to estimate 'travel time' maps of equal travel times called isochrones (Odoni and Lane, *In Prep*) following the formulation developed by Saghafian and Julian (1995).

To calculate travel time maps, runoff is assumed to occur in two simultaneous, but distinct phases, that is, in channel flow and overland flow. Typically the in-channel flow will usually have a faster flood wave celerity whereas overland flow will be shallower and slower. The wave travel time needs to be computed separately for these two components.

9.4.1. Requirements of the model

OVERFLOW has the following minimum data and hardware requirements in order to set up and run:

- A computer running MATLAB, preferably with a large RAM (>4Gb) and a fast processor (>1.80Ghz) in order to generate fast model run times of less than 10 minutes.
- A Digital Elevation Model (DEM), ideally at 5m resolution for small (<100km^2) catchments, although courser resolution can be used if needed.
- Established catchment channel network, typically derived from a detailed map (1:10000), remote sensed data and/or catchment surveys.
- Channel geometry measurements across the study catchment in order to parameterise the channel network in the model. As a minimum this would include between 5-10% of headwater streams and measurements at 1km resolution a least one channel of Shreve stream order 2 through to the channel outflow.

- Land cover information in the form of surveys, photographs or remote sensed data in order to determine appropriate hydraulic resistance coefficients.
- Rain gauge data for a rainfall event, preferably from a 15 minute tipping bucket rain gauge sited within the catchment.
- Stream discharge data at 15 minute resolution at, or near to the model catchment outflow.

9.4.2. Determination of the channel network

The channel network is inferred from the DEM by applying a single steady-state extreme, effective rainfall rate corresponding to bankfull discharge, based on channel geometry measurements from the field. Such a rainfall event will correspond to a flood with a return period of c. 2 years at the catchment outlet. A unit discharge threshold is applied to the accumulated rainfall runoff to identify the onset of channel formation across the grid.

The DEM is first pit-filled using the Planchon and Darboux (2003) method (Odoni and Lane, *In Prep*). Flow paths are initially calculated as hillslopes using Quinn et al.'s (1991) FD8 algorithm with a diffusion exponent of 3 (Odoni and Lane, *In Prep*). To identify the onset of channel formation a unit discharge threshold is then applied to the FD8 (diffusive) routed accumulated rainfall. Where the threshold is exceeded a channel head is formed at that cell and a D8 (non-diffusive, steepest downslope flow path) algorithm is applied to the cell and all cells downstream. From this a definition of which cells are hillslope and channel cells is derived (Odoni and Lane, *In Prep*).

The channel network inferred from the DEM needs to be checked against the actual channel network within the study catchment as differences can arise particularly where channels have been anthropogenically straightened, moved or are perched. If there are discrepancies between the actual and inferred channel network it will be necessary to correct the DEM, typically by "burning" channels into the DEM to force the flow paths to follow the actual drainage network.

9.4.3. Effective rainfall rate

OVERFLOW applies a spatially uniform rainfall rate across the catchment, following the approach used in other spatially-distributed unit hydrograph applications (e.g. Maidment et al., 1996; Saghafian et al., 2002) the runoff is set as the effective rainfall (Odoni and Lane, *In Prep*). Effective rainfall is a runoff percentage and is calculated as the rainfall rate minus some assumed percentage loss representing the excess of net rainfall minus losses due to evapotranspiration, interception and infiltration (Odoni and Lane, *In Prep*). During model set up the effective rainfall rate is calculated using a mass balance approach using measured rainfall and an outflow hydrograph. As with the rainfall rate the runoff percentage is assumed to be spatially uniform.

9.4.4. Variable runoff rate

During a rainfall event in a humid climatic zone a catchment will have a reduced capacity for infiltration as rainfall rate increases (Bracken and Croke, 2007) and therefore the runoff percentage will vary as a function of the rainfall rate and rainfall duration(Odoni and Lane, *In Prep*). The mass balance calculations of runoff percentage described above are based on the entire event and thus do not reflect any temporal variations in runoff percentage with rainfall rate or duration. A non-linearity is used in the runoff percentage calculation so that runoff increases very gradually with rainfall rate (Odoni and Lane, *In Prep*) as

$$P_{eff} = A(aP + bP^2) \qquad 9.1$$

Where P_{eff} is effective precipitation that becomes runoff, A is catchment area, P is precipitation rate, a and b are constants with a value between zero and one (Odoni and Lane, *In Prep*). This equation is used for both the calculation of the equilibrium time maps and the time map sampling to generate the hydrograph.

9.4.5. Isochrone calculation

The basis of isochrone maps is the concept of time to equilibrium t_e of a watershed, which can be defined as the time taken for a watershed outlet runoff to reach a steady state under some uniform rainfall rate; this will be the time taken for the most hydraulically remote part of the catchment to contribute surface runoff to the outlet. Thus, t_e is a function of the catchment characteristics and the rainfall intensity. Practically in hydrological calculations t_e is often replaced by a time to virtual equilibrium which is when discharge is 97% of steady-state discharge (Saghafian and Julien, 1995).

Saghafian and Julien (1995) showed isochrone maps of equal runoff travel time can be calculated from a formula for time to equilibrium runoff. A flood wave travel time (t_w) for a kinematic wave approximation is given by:

$$t_w = \int_{x_1}^{x_2} \frac{\gamma b_1}{(\beta - 1)\alpha^{1-\gamma}} \left(\frac{a_1}{Q_e}\right)^{\gamma} dx \qquad 9.2$$

Where

$$\gamma = \frac{\beta - 1}{\beta + b_1 - 1} \qquad 9.3$$

where x is distance along the flow path; α and β are resistance law parameters; a and b are constants based on local channel cross-sectional geometry; Q_e is upslope discharge delivered to a point. Along with a Manning resistance equation this formula can be used to calculate for both overland and channel flow (Saghafian and Julien, 1995).

Travel time maps are calculated using Equation 9.2 and a Manning resistance formula for N rainfall intensities, giving N maps of runoff generation and N isochrone maps (Odoni and Lane, *In Prep;* Saghafian et al., 2002). The hydrograph is obtained by convolving the isochrone maps (Equation 9.4) to produce N incremental runoff hydrographs, which are delayed by a time corresponding to each isochrone and then superimposed (Odoni and Lane, *In Prep*) as

$$Q_j = \sum_{k=1}^{j} E_k A_{j-k+1} \qquad 9.4$$

where j is the time step, Q is the runoff discharge, E is excess rainfall intensity; and A is the area bounded by isochrones (Saghafian et al., 2002).

The Saghafian et al (2002) approach is modified to allow for explicit effects of flow transfer from channels to floodplains, in each cell and time step this transfer is dependent on: discharge, channel geometry and channel and floodplain hydraulic resistance within the cell (Odoni and Lane, *In Prep*). This modification allows for the testing of the effects of riparian zone alterations such as floodplain forest or floodplain buffer strips (Odoni and Lane, *In Prep*).

9.4.6. Flow routing

Figure 9.1 shows the conceptual flow routing architecture of the model for each cell. Much of the flow routing follows a standard unit hydrograph approach (Odoni and Lane, *In Prep*). Where OVERFLOW differs from previous spatially-distributed unit hydrograph approaches is in its treatment of flow path evolution as a function of rainfall rate. The two novel modifications are: (1) as the volume of runoff increases within an event the model allows for headwards extension of the channel network as unit discharge threshold for channel formation is exceeded to represent a variable source area (Freeze, 1974) (see Chapter 3, Figure 3.2); (2) the development of flow path routing across floodplains to represent ephemeral floodplain channels, rather than restricting flow paths entirely to the channel network, allowing for the exploration of diffuse land management effects on flood hydrology (Odoni and Lane, *In Prep*). This approach makes OVERFLOW ideal to explore the effects of changing floodplain forest cover and floodplain complexity on flood hydrology.

The channel head extension modification is highlighted in red in the upper right of Figure 9.1. The modification is based on the first calculation of hillslope and channel cells with FD8 and D8 routing respectively, the D8 routing is allowed to extend headwards when the estimated unit discharge exceeds the channelized flow threshold in a hillslope cell adjacent to a channel cell. It is important to note that this only represents a dynamic expansion of the source area and hydraulic network, geomorphology is assumed to be constant on the scale of the event and thus incision of a river channel does not extend beyond the original network. Thus the adjustment is only from a FD8 to D8 routing; the cell velocities and travel times continue to be calculated from the full channel width (Odoni and Lane, *In Prep*).

The floodplain channel formation modification is highlighted in green in the bottom half of Figure 9.1. Simplified representations of overbank mechanisms are used due to the data uncertainties inherent in catchment scale modelling, the aim being to correct travel times to reflect situations where some flow is routed out of bank across a floodplain.

The initial step in the analysis is based on the channel cells identified using threshold discharges under each rainfall rate, and using D8 routing. Using estimated channel hydraulic geometry and hydraulic roughness and the Manning equation discharge estimates are converted to estimated flow depths (d_{ij}^{ch}) and then estimated water surface elevation (z_{ij}^{w}) based on the DEM cell elevation (z_{ij}) (Equation 9.5) (Odoni and Lane, *In Prep*).

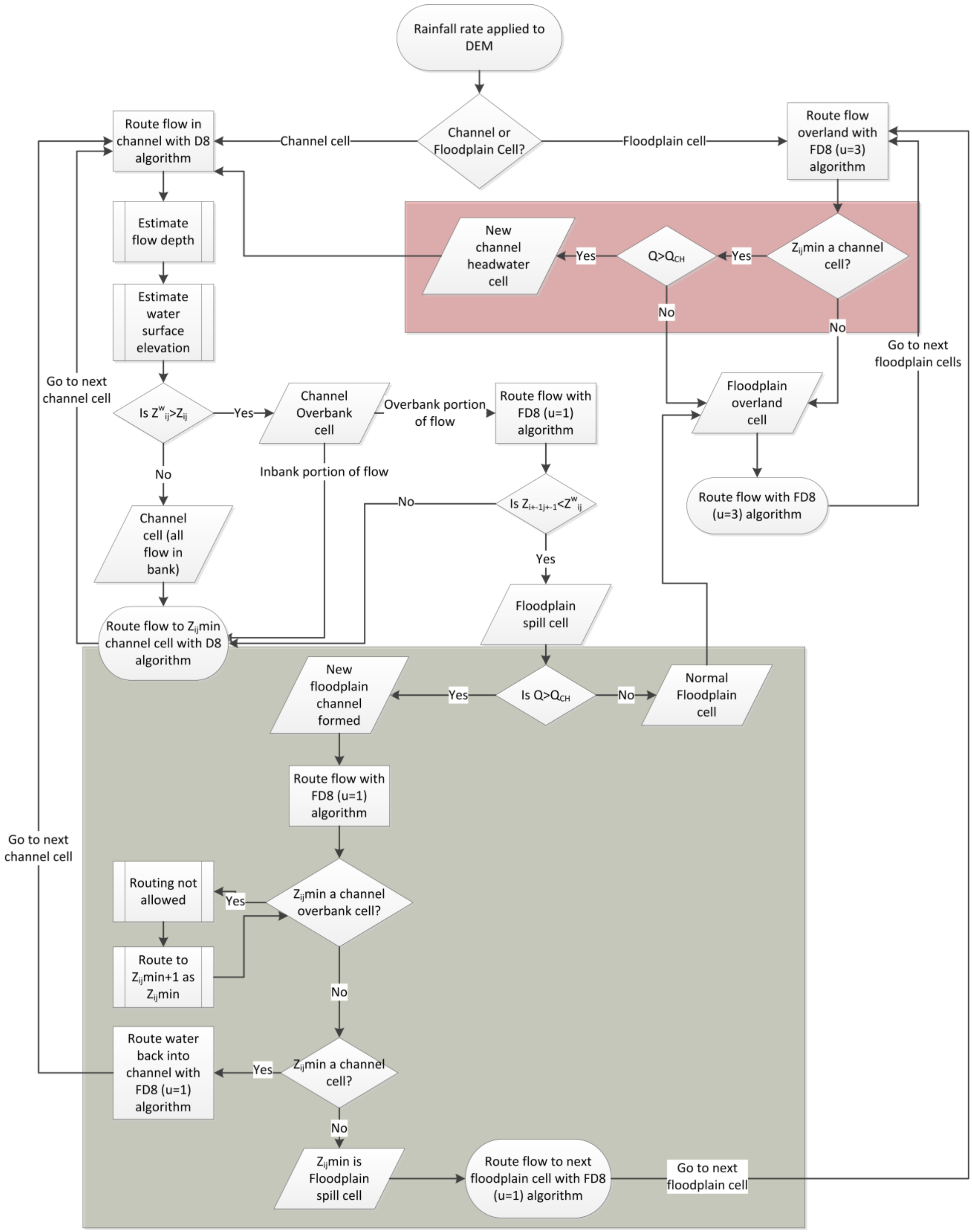


Figure 9.1 – Flow chart of the flow routing routine. Channel head modification is highlighted in red at the top right of the process flow, floodplain channel cell modification is highlighted in green at the bottom of the process flow.

$$z_{ij}^{w} = z_{ij} + d_{ij}^{ch} \quad 9.5$$

This initial water surface elevation calculation is likely to be an overestimate and lead to water flux to the floodplain too readily. The overestimate is due to the channel being a sub-grid feature (channel width less than cell size), thus the elevation of a grid cell containing a channel is a spatial average of elevations associated with it (channel and non-channel) such that $z_{ij}^{ch} < z_{ij}$ (Odoni and Lane, *In Prep*). Therefore a correction term *k* is introduced into the water surface elevation estimate

$$z_{ij}^{w} = z_{ij} + kd_{ij}^{ch} \quad 9.6$$

where

$$k = (1 + k_c s)^{-1} \quad 9.7$$

The scaling effect will show a DEM resolution dependence; *k* is defined as a function of slope as channel width is inversely proportional to the area of the cell which is floodplain, and narrower channels by implication have steeper slopes (Odoni and Lane, *In Prep*). The adjustable parameter k_c can be parameterised using high resolution remotely sensed data where available (Odoni and Lane, *In Prep*).

Channel cells are reclassified as overbank channel cells if

$$z_{ij}^{ch} > z_{ij} \quad 9.8$$

For each overbank channel cell the surrounding cells acting as floodplain spill cells are identified by

$$z_{ij} + kd_{ij}^{ch} > z_{i+x_{-1}^{+1},\,j+y_{-1}^{+1}} \quad 9.9$$

Once overbank channel cells and floodplain spill cells (collectively termed flood cells) have been identified the flow path, flow routing and flow accumulation are repeated with modification for the flood cells. The process will identify the flow paths across the floodplain for a series of local flow discharges corresponding to the set of rainfall rates, rather than the diffusive spreading of ponded water within floodplain storage areas during a flood event (Odoni and Lane, *In Prep*).

The floodplain flow routing is based on differences in water surface elevation with flow treated as a diffusion wave (e.g. Bates and De Roo, 2000; Bradbrook et al., 2004; Horritt and Bates, 2001; Yu and Lane, 2006), spreading water iteratively, but restricting routing to only two lowest neighbouring cells in any one time step (Odoni and Lane, *In Prep*). The modification to allow routing to two neighbouring cells makes flood cells more diffusive than channel cells, on the basis water surface gradient and momentum will reduce dependency of floodplain flow paths on topographic steering, but less diffusive than floodplain sheet flow given the momentum component of channel spill (Odoni and Lane, *In Prep*). The diffusive modification is accomplished with an adjustable parameter u, which represents the sensitivity of flow routing to topography, flow is partitioned in fractions (f) defined as:

$$f_{i+x,j+y} = \left[s_{ij}^{i+x,j+y}\right]^{u} \left(\sum x \sum y \left[s_{ij}^{i+x,j+y}\right]^{u}\right)^{-1} \qquad 9.10$$

For x=+1:-1, y=+1:-1, where x=0, y=0 is excluded as the source cell from which flow is being routed (Odoni and Lane, *In Prep*). The adjustable parameter u has a value between u=0 and u=3 and allows an adjustment of flow diffusion. As u becomes larger routing tends towards a steepest decent method (i.e. a D8 algorithm), as u tends towards zero routing becomes progressively more diffuse and less dependent on topography, until at u=0 routing is completely independent of topography. A value of u=1 approximates routing to only two downslope cells in any timestep, this is similar to the degree of slope dependence used in convention diffusion wave models (Odoni and Lane, *In Prep*).

Applying this to the flood cells; channel overbank cells have two portions of flow, the in channel portion and the overbank spill portion. These two flow portions are routed differently; the in bank portion ($z_{ij}^{w} \leq z_{ij}$) is routed via a D8 algorithm in line with all channel cells, however the portion corresponding to overbank flow ($z_{ij}^{w} > z_{ij}$) is routed with a FD8 algorithm with u=1 (Odoni and Lane, *In Prep*). All floodplain spill cells have

FD8, u=1 routing, a modification is introduced to calculate flow path using the lowest neighbouring cell rather than the steepest flow path, in practice these are often the same (Odoni and Lane, *In Prep*). However in the case of floodplain spill cells adjacent to the channel cell the steepest flow path is often straight back into the channel. Additional floodplain spill cells are iteratively identified as those cells adjacent to existing floodplain spill cells, with the exceptions explained below.

Within the process of routing overbank flow across the floodplain there are cells with different classifications (i.e. channel cells, channel overbank cells, floodplain spill cells and floodplain cells); the model performs a series of checks on flow routed through floodplain spill cells based on these definitions to ensure the routing proceeds in a logical manner. The first of these is that the discharge is greater than the critical discharge threshold for channel flow; if the discharge drops below this then the flow is likely to be more dependent on topographic steering so routing is returned to the FD8, u=3 algorithm. Secondly FD8, u=1 routing is modified so diffusion is not allowed from a floodplain spill cell into a channel overbank cell, to prevent water from being routed straight back into the channel. Thirdly where a floodplain spill cell is adjacent to a channel cell which is not defined as channel overbank cell water is routed back into the main channel (Odoni and Lane, *In Prep*). The iterative process of defining floodplain flow paths typically leads to stable flow paths for given rainfall rates in just one or two iterations (Odoni and Lane, *In Prep*).

9.4.7. Cell Travel Time Modifications

The cell travel times need to be recalculated based on the modified flow paths, this introduces complexities in the treatment of overbank spill cells which have two flow components; in channel flow and overbank flow, which need to be calculated separately and weighted to give a combined travel time for the cell.

The in channel flow portion is assumed to follow a D8 routing following the steepest path, the overbank portion is routed to the lowest neighbouring cell. It is important to note that although these two routes may in fact be the same the flow velocities will be different with the in channel portion exhibiting a faster velocity than the shallow water moving slowly over the floodplain (Odoni and Lane, *In Prep*). In order to recalculate the cell travel time firstly the two components of the flow need to be partitioned. The flow depth and width for in channel flow is assumed to be the channel depth and width and the length of travel the distance in the steepest path direction, the hydraulic resistance is the Manning's *n* value for the channel. For the overbank portion of flow the flow depth is inferred from the full width of the cell and the velocity from the lowest neighbour flow path and the Manning's *n* value for the adjoining floodplain cells (Odoni and Lane, *In Prep*). The model does include an option to specify the bankside Manning's *n* value as different to that of the surrounding floodplain, to represent narrow riparian buffer strips for example (Odoni, *pers. comm.*, 16^{th} May 2012).

Cell travel times for each flow component are then weighted according to the values for flow type and an overall cell travel time for each overbank channel cell calculated using:

$$t_w = \frac{V_{inbk}t_{inbk} + V_{obk}t_{obk}}{V_{inbk} + V_{obk}} \qquad 9.11$$

Where t_w is the overall weighted cell travel time, t_{inbk} and t_{obk} are the cell travel times for inbank and overbank flow components respectively and V_{inbk} and V_{obk} are the inbank and overbank flow volumes derived from the modified flow map. From these modified travel times taking into account headwards extension of the channel network and overbank flow paths a set of isochrone maps can be generated for each rainfall rate (Odoni and Lane, *In Prep*).

9.4.8. Calibrating isochrone generated hydrographs to a reference event

In order to generate an estimated outlet hydrograph the modified isochrone maps are sampled based on an observed rainfall event (Odoni and Lane, *In Prep*). This is primarily a calibration problem and is typically complicated by a poor spatial coverage of rainfall records within a study catchment (Odoni and Lane, *In Prep*). An existing river flow record is used to infer the isochrone maps that produced the measured flow within the flood event of interest. Each isochrone map is convolved with the effective rainfall rate (rainfall corrected for percentage runoff) for all observed rainfall rates, which provides a vector for each isochrone map of the time taken for runoff from each (*i,j*) grid cell to reach the catchment outlet (Odoni and Lane, *In Prep*).

A Monte Carlo calibration framework is then used to determine which isochrone maps to use for each observed rainfall rate. Initially isochrone maps are randomly sampled for each time step to produce 200 outflow hydrographs. For each time step the isochrone maps producing the best Nash-Sutcliffe index of model efficiency (Nash and Sutcliffe, 1970) for the whole hydrograph are calculated (Odoni and Lane, *In Prep*). The sampling process is then repeated refining the set of isochrone maps that can be sampled at each time step by restricting sampling to those isochrone maps that have been already identified as having a high Nash-Sutcliffe index result in the previous iteration (Odoni and Lane, *In Prep*). This iterative process is repeated until all 200 hydrograph simulations have a Nash-Sutcliffe index >0.98. Given the dependence of the modelled results upon calibration the model performance needs to be independently assessed with respect to internal flow field information (Odoni and Lane, *In Prep*).

9.5. Example Model Application

30th to 31st March 2006 event, Lymington River catchment, UK

9.5.1. Runoff percentage estimation

In common with other spatially distributed unit hydrograph models the focus of OVERFLOW is upon modelling the rapid transfer of runoff into the channel network. It is important to estimate the percentage runoff accurately for the specific event being modelled and this is done through an event-specific mass balance calculation comparing estimates of the total discharge volume in the river to the catchment average rainfall. A discharge record is inferred from the gauging station record for the Environment Agency gauging station for Lymington at Brockenhurst (Station ID:42003). The total volume of water making up the flood event can be calculated by estimating the baseflow for the river and subtracting this from the hydrograph and integrating the area under the curve. For this event the baseflow is estimated as $1.18 m^3 s^{-1}$ and flood event started at 22:15 30th March and the falling limb of the hydrograph returns to close to baseflow at 22:00 31st March. The total storm discharge volume less baseflow over this time is estimated as 407,372 m^3. The 30^{th}-31^{st} March 2006 event only represents a moderate flood event, however the data range for rainfall in the New Forest was limited to the period December 2005 to May 2006, for which the March 30^{th}-31^{st} 2006 event was the largest flood gauged.

Alternative sources of rainfall data were explored, including rainfall data from a weather station in Hurn, Dorset and BADC radar data from the Met Office. Data from Hurn showed poor agreement with available data from the rain gauge in the New Forest so was deemed to be unsuitable for application in OVERFLOW given the sensitivity of the model to rainfall rates (Byers, 2011). Data derived from Met Office weather radars showed good agreement with the onset of rainfall and the duration, but the precise rates of rainfall did not show good agreement with measured data in the New Forest and so radar data was also deemed to be unsuitable for application in OVERFLOW. Given the lack of availability of suitable rainfall data corresponding to larger flood events it was decided to use the measured event from 30^{th} March 2006.

Rainfall was estimated based on a rain gauge at Oknell Plain at the highest elevation in the catchment. Rainfall was assumed to be spatially uniform across the catchment during the event and thus by integrating the rainfall rate over the time of the event across the catchment area a total rainfall volume can be calculated. For this period the total rainfall was estimated as 14.4mm. Comparing values of total storm flow and total rainfall integrated across the catchment gives a percentage runoff of 87.7% for this event.

The main reason for such a high percentage runoff is the geology of the catchment which is underlain by Eocene Barton clays and is thus largely impermeable and hydrologically 'flashy'. The whole catchment exhibits the same underlying geology and thus is a simple example where the responses are spatially similar. In more complex catchments with variable geology it would be important to determine whether all sub-catchments were responding to rainfall in a similar manner. In situations where parts of the catchment are highly permeable it may be necessary to apply corrections to the mass balance calculations by partitioning the contributions from different sub-catchments and treating their contributions to the hydrograph separately.

9.5.2. Topographic Data

Figure 9.2 show the process for deriving the DEM and channel network from source LiDAR data. The DEM used to model the Lymington River basin is at 20m resolution; this DEM was formed from a resample of 5m resolution LiDAR data from Environment Agency Geomatics division. The DEM resolution was used to ensure that channels are a sub-grid feature; OVERFLOW has been written with small sized catchments in mind and to date a 20m DEM has been found to be optimal for all applications (Odoni, *pers. comm*., 20th November 2010), however for simulations of catchment smaller than 40km^2 or over 100km^2 it is possible to adjust the model to run at different DEM resolution.

In the source 5m resolution LiDAR data headwater channels are likely to be sub-grid features, where the overall elevation will be an average of all elevations across the cell, thus the channel depth will be somewhat underestimated in the source data. Furthermore during a resampling exercise this problem can be exacerbated as the elevation of a resampled 20m cell resolution will be some average of a 4x4 array of 5m resolution elevations, for a resultant channel cell some of the source cells will be entirely floodplain, further masking the channel depth. This problem is tackled by applying a weak weighting in the resampling code to maintain lower elevations where a channel exists as a sub-grid feature at 20m resolution. This modification ensures the flow accumulation and flow routing accurately reflects the channel network, but may result in slightly lower sub-grid bankside elevations particularly where sub-grid channels are deeply incised below the floodplain surface. The routine for resampling a 5m to 20m DEM resolution within MATLAB along with the pit-filling routine was developed by Dr Odoni as part of OVERFLOW and has been refined with subsequent application of OVERFLOW including this study (e.g. Byers, 2011; Odoni and Lane, 2010; Odoni and Lane, *In Prep*). In this application Dr Odoni wrote and ran the resampling and pit-filling routine to convert the source 5m resolution DEM to a 20m resolution DEM suitable for running OVERFLOW.

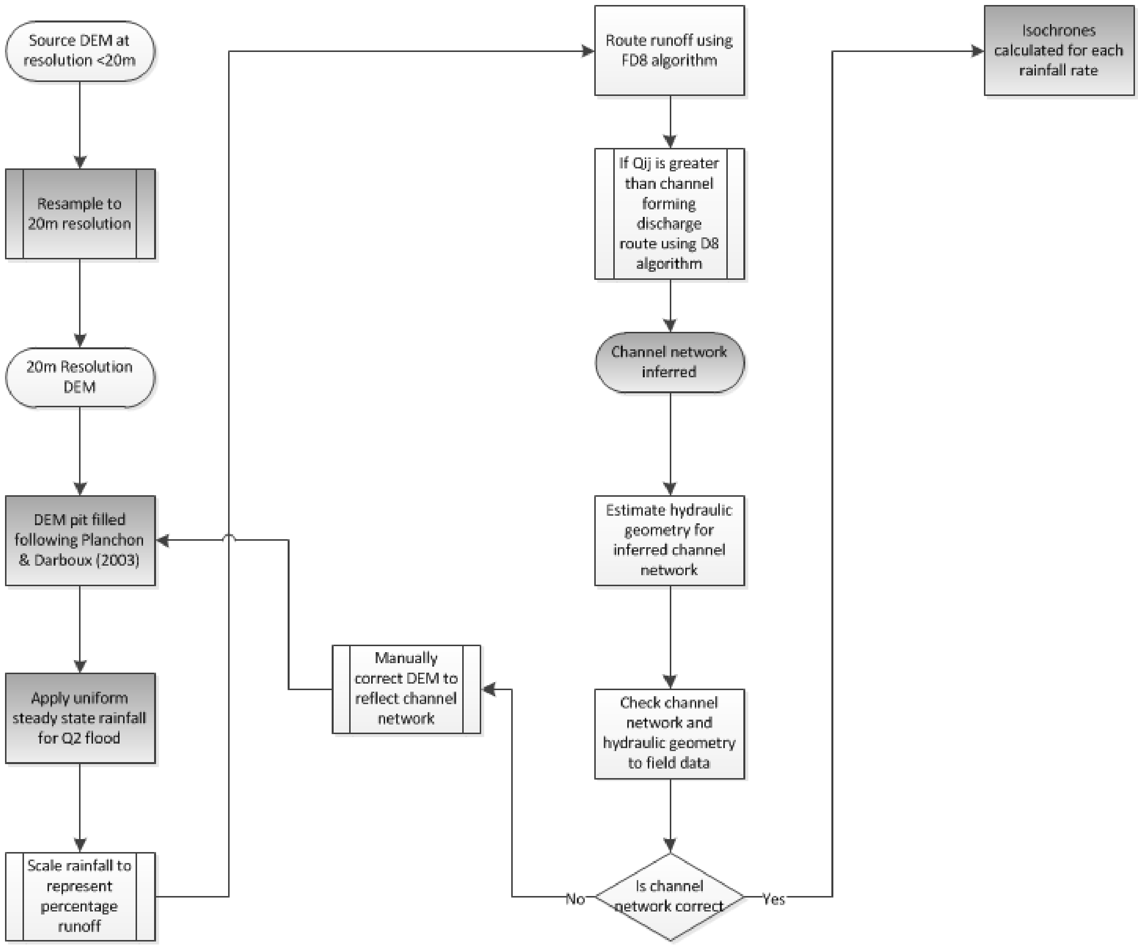


Figure 9.2 – Flow chart showing the DEM and channel network generation. See section 9.4.3. for explanation of figure shading.

9.5.3. Channel Network

In line with other applications of OVERFLOW (e.g. Byers, 2011; Odoni and Lane, 2010; Odoni and Lane, *In Prep*) the channel network is inferred from the DEM by applying a rainfall rate which will result in a bankfull discharge at the catchment outlet (7.8mm/day); this rainfall rate is approximately equivalent to the 2% flood recurrence interval. A channel formation threshold is applied to identify the transition from hillslope to channel cells, which in this case is a unit discharge of $7.5m^3s^{-1}$. The inferred network was then checked against the channel network from the Ordnance Survey 1:25000 map of the catchment and field reconnaissance notes. Due to channel management works in connection with timber plantation drainage from the 19th and 20th century a number of the channels have been moved, straightened and in some cases slightly perched and the previous channels filled in. In some parts of the catchment with heavily incised engineered channels artefacts from the DEM resampling and pitfilling routines combined with low topographic relief results in the flow accumulation and flow routing directing water into ephemeral floodplain channels, which are on the course of old, filled channels, in preference to the actual channels. Where this occurs it is necessary to go back to the resampled DEM and manually adjust the elevation of channel cells to "burn" the channel into the floodplain surface in order to correct for the flattening effect of the resampling and pit-filling routines. In this application the code for applying the steady rainfall rate and inferring the channel network is part of the set of OVERFLOW model codes, but needs to be modified for each application. Code modification was done by Dr Odoni; hydraulic geometry estimates, ground truthing of the inferred channel network and manually burning/filling of the DEM was done by the author.

Once modifications have been made to the DEM the process of pit-filling, flow accumulation and flow routing is repeated to infer a second iteration of the channel network. Checking and modification to the DEM are repeated until the inferred network matches the actual channel network.

Figure 9.2 also provides a visual guide to assistance provided by Dr Odoni at University of Bristol who wrote code for performing the technical routines shaded grey on the figure. In summary; Dr Odoni wrote the resample and pit-filling of 5m raw DEM to 20m DEM, generating an initial estimate of channel network and wrote and ran the calibration routine at University of Bristol. All other elements were performed in University of Southampton by the author, including running resample/pit-filling routines and ground truthing and correcting the 20m DEM. All data for model calibration was collected and processed by the author and decisions on all aspects of the model set up and calibration were taken by the author in consultation with Dr Odoni.

9.5.4. Hydraulic Geometry

In a model testing multiple, spatially diffuse land use changes and the response of flood hydrology the local river geometry is a vital component as it effects how connected the local channel is with the floodplain and how readily and at what discharge the water switches from in channel flow to overland flow (Odoni and Lane, *In Prep*). The generation of hydraulic geometry is based on the assumption that the 2% return period flood corresponds to bankfull discharge and is equivalent to the effective discharge for channel geometry (Odoni and Lane, *In Prep*). The discharge estimates under the 2% return period flood for each cell in the network is used with Leopold & Maddock (1953) empirical relations (Odoni and Lane, *In Prep*).

For channel widths the equation is:

$$w_{ij} = AQ_{ij}^{a} \qquad 9.12$$

And for channel depth/bank height:

$$d_{ij} = BQ_{ij}^{b} \qquad 9.13$$

Where w_{ij} is the width and d_{ij} is the bank height of the channel at cell (i,j), Q_{ij} is the 2% flood discharge for the cell, and A, B, a and b are constants. The coefficients and exponents from Equations 9.12 and 9.13 are estimated to deliver realistic values for the hydraulic geometry based on field observations and previous applications of OVERFLOW (Byers, 2011; Odoni and Lane, 2010; Odoni and Lane, *In Prep*). In channel networks without significant anthropogenic channel modifications it should be possible to use fixed values for A, B, a and b (Odoni, *pers. comm.*, 3rd July, 2012). In the Lymington River catchment, historic channel modifications mean variable values need to be used for different groups of stream orders to reflect areas with over widened and over deepened channels.

Field measurements of hydraulic geometry were taken along the Highland Water in 2001 (see Chapter 5) and additional measurements were taken in June 2012 for the Blackwater and for channel headwaters across the catchment. Calculating hydraulic geometry across the catchment was approached by fitting a least squares regression to measured field width and depth against modelled width and depth from Equations 9.12 and 9.13. By grouping stream reaches using Shreve stream order (Shreve, 1966) it is possible to use variable coefficients and exponents in Equations 9.12 and 9.13 for groups of stream orders within the channel network; Table 9.1 lists the constants used for each stream order in this study.

An analysis of catchment channel widths and depths in Table 9.1 shows coefficients are very similar for headwater channels and large trunk channels near the catchment outflow. Tubbs (2001) stated that channel headwaters have typically not been modified or channelized as they are on steeper hillslopes which were not used for tree plantations. Information from Tubbs (2001) along with the noticeably different hydraulic coefficients for channels of intermediate Shreve stream order (n=11:31) indicates channel modifications are mostly restricted to large tributary channels and not to headwaters or main trunk channels.

Shreve Stream Order	Width Coefficient (A)	Width Exponent (a)	Depth Coefficient (B)	Depth Exponent (b)	Fit to measured data - R^2
1:10	2.40	0.53	0.51	0.42	0.91
11:16	3.10	0.50	0.70	0.42	0.72
17:19	3.30	0.52	0.68	0.42	0.88
20:31	3.20	0.52	0.85	0.42	0.82
32:37	2.90	0.53	0.76	0.42	0.90
38:150	2.40	0.54	0.51	0.42	0.87

Table 9.1 : Constants used in equations 9.12 and 9.13 to calculated predicted width and depth

At confluence nodes between segments using different constants for the hydraulic geometry equations there can be abrupt changes in channel geometry of up to 0.2m between cells which is not reflected in the field where widths and depths gradually change over channel lengths of *c*.100m, therefore a manual smoothing is applied by taking the local average channel geometry from the two nearest upper and lower reach cells and applying the geometry transition across these cells (approximately 110m channel length allowing for intra-cell meandering).

In almost all applications of OVERFLOW it is recommended to ensure the channel is a sub-grid feature so that channel width<cell width, therefore the width predictions from Equation 9.12 will be smaller in magnitude than cell width (Odoni, *pers. comm*. 28th May, 2012). The channel width is used for all channel calculations and the residual cell width is distributed as floodplain, which is assumed to exist equally to either side of the channel.

9.6. Hydraulic Resistance

When using a kinematic wave approximation (Equation 9.2) in a spatially distributed unit hydrograph approach the treatment of Manning's n approximations is an important area of uncertainty (Odoni, *pers. comm*., 29th January, 2013). OVERFLOW requires cell specific estimations of Manning's n and this is based on the division between channel cells and hillslopes. Hydraulic resistance is therefore averaged across the catchment and assumed to be spatially uniform; whilst this assumption is unlikely to be correct it is justified given the restricted applications of the modelling approach where the interest is in modelling at the catchment scale (Odoni, *pers. comm*., 29th January, 2013). The values for Manning's n are also assumed to be fixed during the modelled event and flow invariant, for many channels and land cover types this is unlikely to be correct; the influence of clast size as well as bed and bank roughness elements in channels declines with rising river stage, such that Manning's n will be lower for bankfull discharge than for baseflow discharges (Bathurst, 1985). Similarly when floodplain vegetation such as grasses becomes submerged by flood water it will exert less drag on flowing water than when emergent leading to lower Manning's n values at high discharges compared to discharges just over bankfull (Anderson et al., 2006). In principle a depth dependent hydraulic resistance parameter could be incorporated into OVERFLOW, but such a modification would be likely to introduce increased uncertainty in the parameterisation of hydraulic roughness values given a paucity of depth dependent reference values for hydraulic resistance (Odoni, *pers. comm*., 16th May, 2012).

In order to calibrate OVERFLOW spatially uniform values for channel and hillslope hydraulic resistance need to be set. In Chapter 7 results for hydraulic resistance across a range of range of logjam types were reported in order to parameterise Manning's n values for logjams in OVERFLOW (see Chapter 10). Ideally the investigation reported in Chapter 7 would also have involved calculating Manning's n values for bankfull discharge based on field values for low wood loads, or non-wood influenced reaches across the catchment, this would have given channel hydraulic resistance values for calibrating OVERFLOW. However due to logistical problems explored in Chapter 7 collecting data at high flows was restricted to a handful of rainfall events and expanding the study to gauge reaches suitable for calibration was not possible. It was therefore determined to set Manning's n values for calibration based on reference to several techniques; a variable power equation based on catchment average grain size (Ferguson, 2007; Ferguson, 2010), a general approach based on adjustment for channel characteristics (Jarrett, 1985), selecting a value based on broad channel characteristics (Chow, 1959) and consultation with expert practitioners as to suitable values based on field photographs. Due to the inherent uncertainties in setting Manning's n values the goal is to choose values which are realistic and representative, rather than conduct a detailed site survey of the catchment.

9.6.1.1. Variable Power Equation

$$n = R^{1/6}(f/8g)^{1/2} \qquad 9.14$$

Where:

$$(\frac{8}{f})^{1/2} = \frac{a_1 a_2 (R/D)}{\left[a_1^2 + a_2^2 (R/D)^{5/3}\right]^{1/2}} \qquad 9.15$$

Where n is Manning friction factor, f is Darcy-Weisbach friction factor, g is acceleration due to gravity, R is catchment average hydraulic radius, D is D_{84} catchment average grainsize and a^1 and a^2 are coefficients.

In the Lymington Basin catchment average values are; channel depth = 1.19m, channel width=4.98m, D_{84}=19mm (Kitts, 2011; Millington, 2007; Sear et al., 2006). Using values of a_1=6.5, a_2=2.5 (Ferguson, 2010) an estimated Manning's n=0.048 ±0.003 is calculated.

9.6.1.2. Jarrett adjusted channel method

This method takes a base value for a channel calculated as the minimum value for Manning's n assuming a straight uniform channel, this can be derived either from a table of values (e.g. Chow, 1959), or based on average grainsize equations (e.g. Strickler, 1981).

$$n = (n_0 + n_1 + n_2 + n_3 + n_4)m \qquad 9.16$$

Where, n_0 is the base value for a straight uniform channel, n_1, n_2, n_3 and n_4 are additive values due to the effects of; cross-section irregularity, variations in the channel, relative effects of obstructions and type and density of vegetation respectively and m represents the degree of meandering (Jarrett, 1985).

The value for n_0 is based on Equation 9.17 from Strickler (1981):

$$n \approx 0.039{D_{84}}^{1/6} \qquad 9.17$$

Values for adjustment factors are based on field surveys and field photographs, giving a Manning's n estimation of n=0.058, with a range of $0.041<n<0.069$.

9.6.1.3. Reference tables

Using Chow (1959) tables of Manning roughness values for reference channels a channel type of a stoney bottom, with some pools and shoals and some weeds, which is closest to that of channels within the Lymington River basin gives a median value of n=0.05 with a range $0.45<n<0.06$.

With reference to the three approaches and in consultation with advice based on field photographs (Odoni, 2012; Sear, 2012)(Odoni, *pers comm.*, 16th May 2012; Sear, *pers comm.*, 1st June 2012) a value of n=0.05 was used for calibration of the catchment.

9.7. Model Parameters

The model therefore has the following parameters which have an impact on the equilibrium isochrone map (Odoni and Lane, *In Prep*):

Parameter	Value	Source
The unit discharge used to define the channel network	7.5 m^3s^{-1}	Iterating steady rainfall rate to obtain bankfull discharge at catchment outflow
The fixed baseflow contribution fraction of the hydrograph	1.18 m^3s^{-1}	Storm Hydrograph – NRFA – CEH Wallingford
Parameters associated with the estimation of effective precipitation (runoff percentage)	87.7%	Storm Hydrograph – NRFA – CEH Wallingford and Rainfall gauge data
The spill flow diffusion exponent	u=1	Diffusion to two neighbouring cells – See section 9.3.6
Parameters associated with hydraulic geometry relationship estimates	See Table 9.1	Field data
Manning's *n* values for channel and hillslopes	Channel n=0.05 Hillslope n=0.07	See section 9.5

Table 9.2 – OVERFLOW parameters

The parameters in Table 9.2 represent catchment characteristics or processes which cannot be directly measured at sufficient spatial scales. Values for these parameters are initially estimated through field data and observations and these estimates refined through model calibration. Although estimates of these parameters represent considerable uncertainty in the model, consideration of these parameters and conducting an uncertainty analysis can yield important insights from such a modelling approach.

The parameters associated with representing hydraulic resistance and runoff represent important simplifications in the modelling approach and therefore should form the focus of any uncertainty analysis performed on the model.

9.8. Calibration

Using the parameter values listed above a calibration for the March 2006 event has a Nash-Sutcliffe ≥0.96 and a mass balance agreement to within 2%, which is deemed a good agreement for this modelling approach (Odoni, *pers. comm*., 7th August, 2012).

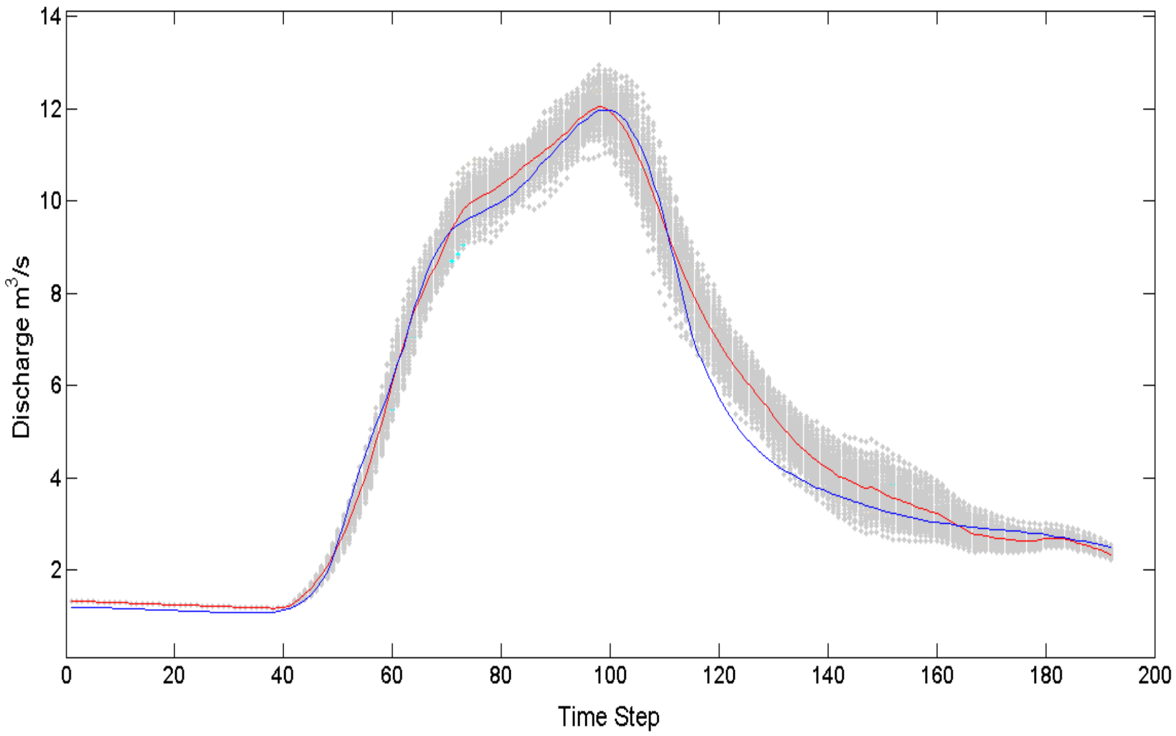


Figure 9.3 – Calibration curve for March 30th 2006 flood event. Blue line shows the observed discharge at Brockenhurst, Red line is the mean modelled hydrographs, light grey points shows the spread of 200 best simulations from the Monte-Carlo calibration with Nash-Sutcliffe ≥0.98. Time Step is 15 minutes

Figure 9.3 shows the calibration curve for the March 2006 event. The calibration routine involves randomly sampling time maps for each time step of the model, and then refining the selection of these time maps to those with the highest Nash-Sutcliffe indices through a process of iteration. This iteration is done until 200 calibrated solutions are produced each with a Nash-Sutcliffe index of ≥0.98, the spread of these solutions is shown in grey on Figure 9.3. The mean flow hydrograph from these 200 calibrations is shown as a red line in Figure 9.3, this mean flow hydrograph is used as the basis for comparisons between the calibrated output and the observed discharge (blue line in Figure 9.3). These same 200 calibrated hydrographs are used as the basis for any further modelling investigation of alternative landuse scenarios, where the mean of the output is compared to the mean of these base line cases. Such an approach allows a degree of model uncertainty to be incorporated in the model results (Odoni and Lane, 2010). The calibration routine needs to be modified for each application of OVERFLOW in order to manually optimise the Nash-Sutcliffe indices (Odoni, *pers. comm*. 20th November 2010). In this application the calibration routine was coded by Dr Odoni and run on computers at University of Bristol using parameters supplied by the author, including the final 20m DEM, channel network and hydraulic geometry. The calibration process is one of iteration where the output calibration hydrograph and inundation extents need to be ground truthed against observed events; this ground truthing was done at University of Southampton by the author.

Figure 9.3 shows the model predicts the observed hydrograph very well, specifically in the case of peak discharge and hydrograph shape. The observed discharge shows a rapid drop off in flow following the peak which the model does not simulate as well as the other parts of the hydrograph. This suggests that the model is slightly overestimating the travel times for the more distal part of the watershed, this over estimation could be due to the roughness parameters used, specifically the use of spatially homogenous roughness values averaged across the catchment. The upper hillslopes of the parts of the catchment most hydrologically remote from the outlet are mostly grass and heathland, the lower slopes and floodplain are dominated by complex riparian woodland (Sear et al., 2010; Tubbs, 2001). Thus the spatially averaged value of Manning's n=0.07 may be an overestimate for the upper hillslopes resulting in over estimates of wave travel time from these cells.

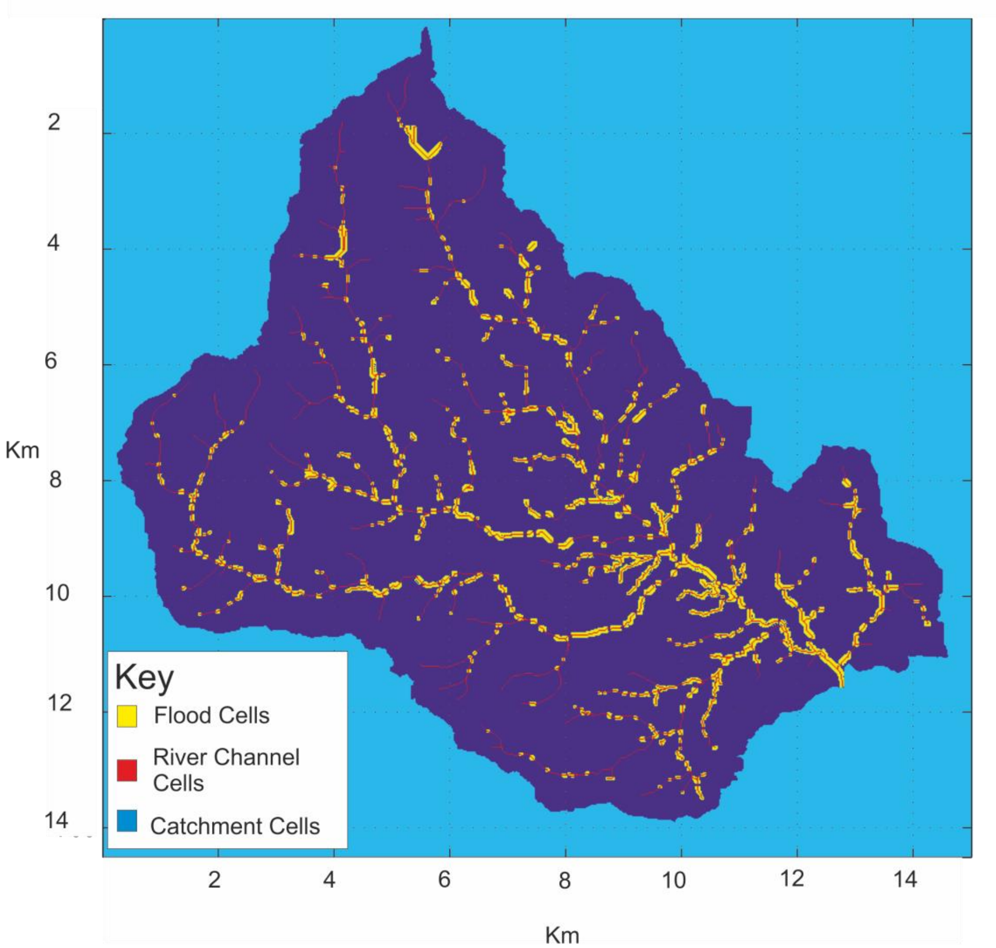


Figure 9.4 – OVERFLOW output showing the Lymington River at Brockenhurst catchment area in navy blue, the perennial channel network in red and overbank flood cells under a 8mm/day rainfall event in yellow.

Figure 9.4 shows the Lymington River catchment area at Brockenhurst along with the river network and areas of flooding under a constant 8mm/day rainfall rate. In order to check the output of OVERFLOW it useful to run several constant rainfall rates through the calibrated model in order to check the discharge estimates at the catchment outflow and the area of inundation predicted match generally to those reported from Environment Agency flood maps and local knowledge. Figure 9.5 shows a detailed look at a section of the catchment network around Brockenhurst and compares inundation output from OVERFLOW under a constant rainfall rate of 8mm/day against the EA flood map for the same area. Figure 9.5 demonstrates the OVERFLOW model is predicting inundation and overbank flow in broadly the same areas as the EA flood map

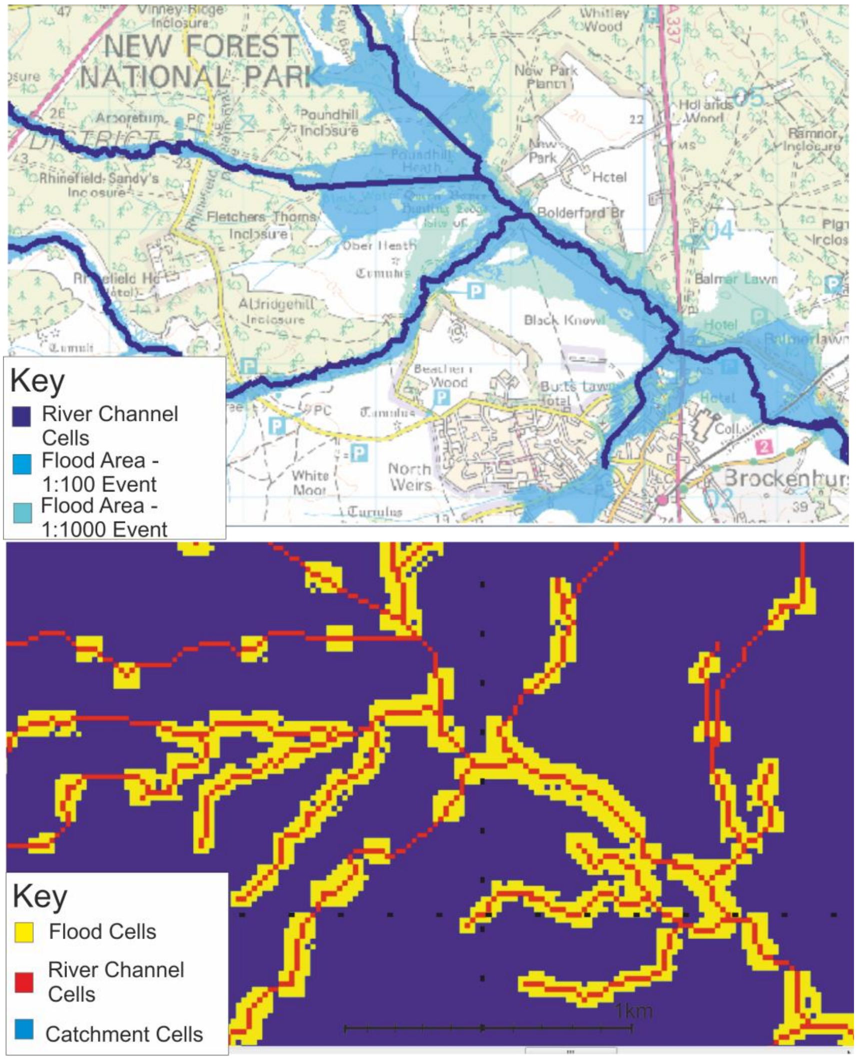


Figure 9.5 – Comparison of inundated areas between the Environment Agency (EA) Flood Map for the Lymington River (top) and flood cells within OVERFLOW under a steady 8mm/day rainfall rate (bottom). This figure illustrates that predicted patterns and areas of inundation are broadly similar for OVERFLOW and the EA Flood map. Note the EA flood map only displays main trunk channels whereas the OVERFLOW map shows the whole perennial channel network.

9.9. Example model investigation

After the calibration process an initial exploratory investigation was conducted to examine the effect of reach scale modifications to land use within the Lymington River study catchment. The investigation was based on a management scenario where a single reach in isolation is modified so that the floodplain forest cover is denser with young saplings and a moderately dense undercroft of bushes and dead wood, which has the additional effect of increasing the volume of dead wood in the river channel itself leading to more frequent logjams and obstructions. The changes in forest cover will increase the general resistance to flow across the floodplain, and greater obstructions will increase flow resistance in the channel.

Modified values for Manning's *n* for the channel were generated using the Jarrett (1985) method described above and assuming only the severity of obstructions (n_3) would change from minor to moderate yields an estimated Manning n=0.058. In reality we may expect additional large wood to drive geomorphic change such that the channel becomes more heterogeneous and channel variations (n_1, n_2) and sinuosity (m) increase, however the time scales and magnitude of such geomorphic changes are hard to quantify. With reference to Chow (1959) a value n≈0.06 seems reasonable as it corresponds to the maximum suggested value for the channel description used in calibration (0.045<n<0.06). The modified value for floodplain Manning's n was taken from Chow (1959) to correspond to the 'normal' value for "heavy stand of timber, a few downed trees, little undergrowth, flood stage below branches" which is n=0.10.

A Manning roughness 'map' was used where a value for Manning's n is defined for every grid cell, these values can then be varied individually or systematically to represent land use changes. In this modelling investigation "reaches" are defined as segments of river between two confluence nodes. In the derived drainage network this equates to N=296 reaches. The Manning roughness map was varied by changing the floodplain and channel Manning's n values to the modified values above for each reach in turn, generating N model runs and N output hydrographs. Figure 9.6 shows an example Manning's roughness map in which roughness values for floodplain and channel have been modified for 5 contiguous segments.

Model runtimes on a laptop with 6Gb of RAM and a 1.80 Ghz processor are approximately 8 minutes, allowing the n=296 run model investigation to complete in less than 40 hours.

Small changes to the peak flow of the output hydrograph compared to the calibration hydrograph were observed, but these were largely within the margin of error (<0.1% change). This is to be expected as the changes applied in this test investigation correspond to less than 2% of the catchment area and/or network stream length (and typically <0.1%) and are only minor changes to the hydraulic resistance.

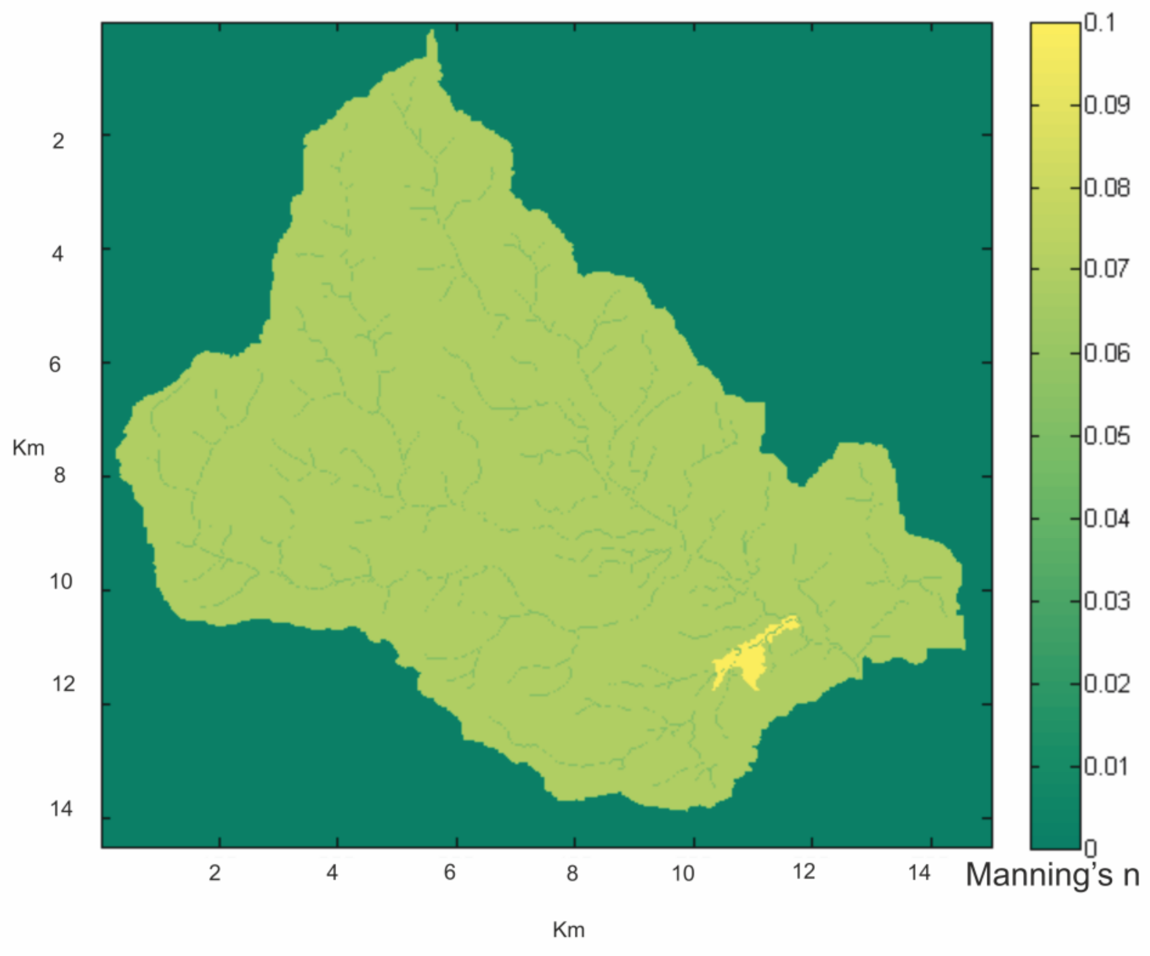


Figure 9.6 – Example 'Manning's Map'. This map shows default values of n=0.07 for hillslopes and n=0.05 for channels, along with 5 segments which have been modified to have elevated Manning's values of n=0.10 for floodplain and n=0.07 for channels

Figure 9.7 shows the results of an example scenario in which individual reaches are altered to raise the floodplain Manning's n from n=0.07 to n=0.10 and in channel Manning's n from n=0.05 to n=0.06. The individual reaches are coloured to show the percentage change in output hydrograph peak discharge (magnitude) compared to the baseline scenario for all of the reaches in the drainage network. Such spatial outputs allow the sensitivity of flood hydrology to land use changes to be evaluated at the catchment scale and can highlight spatial patterns in response. Although the relative magnitude of both changes and responses are small, there is a clear indication of a spatial pattern in the hydrograph responses. Increasing channel and overbank roughness in reaches hydraulically proximal to the catchment outlet tend to increase

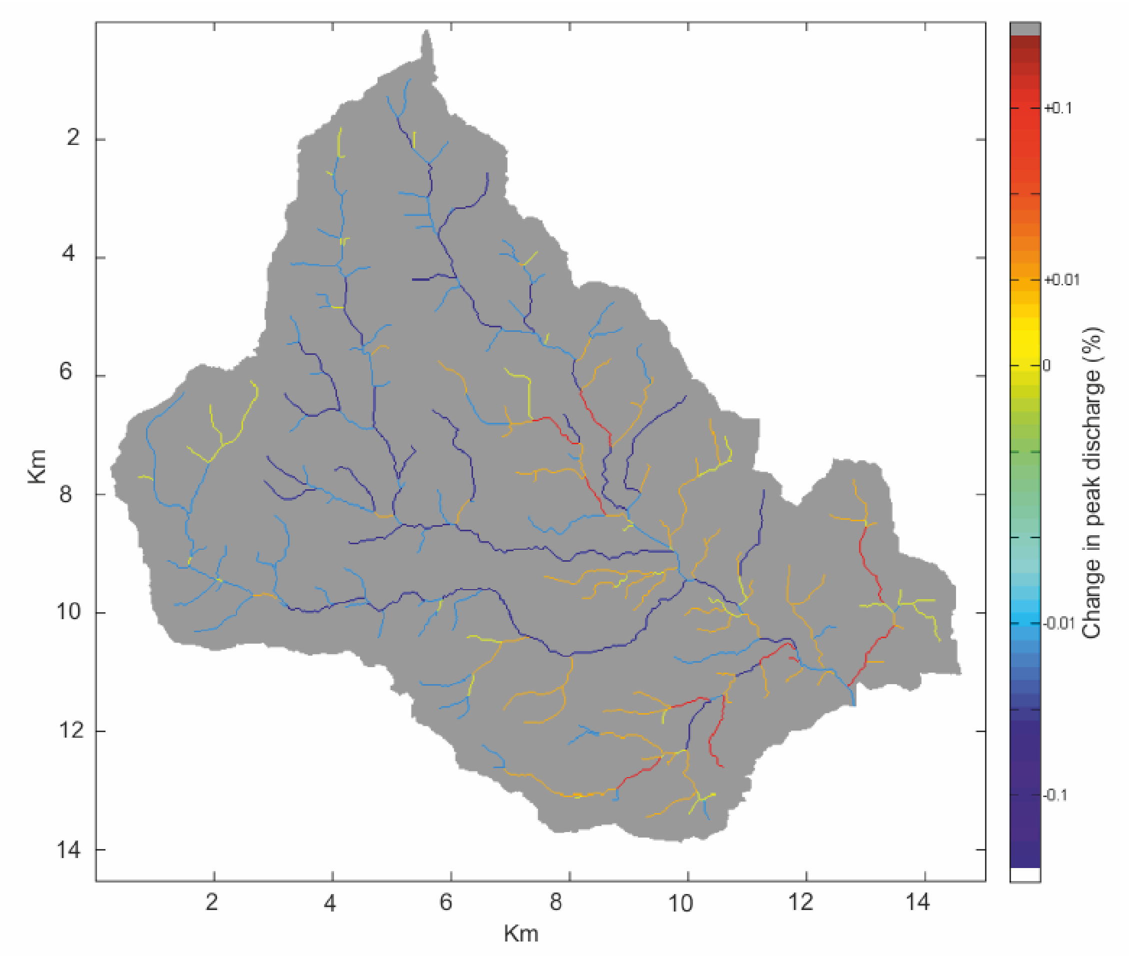


Figure 9.7 – Map of the Lymington River drainage network showing change in hydrograph peak flow in response to changes in reach-scale land cover (0.1-2% of catchment area), where floodplain Manning's n is changed from n=0.07 to n=0.10 and in channel Manning's n is changed from n=0.05 to n=0.06.

the peak flow or be neutral, the same changes made to main channels in the mid-catchment tend to decrease peak flow and the same changes to hydraulically remote reaches tend to be neutral or lead to small decreases in peak flow. Figure 9.8 shows the largest increase in peak discharge (model run 204) and largest decrease in peak discharge (model run 172).

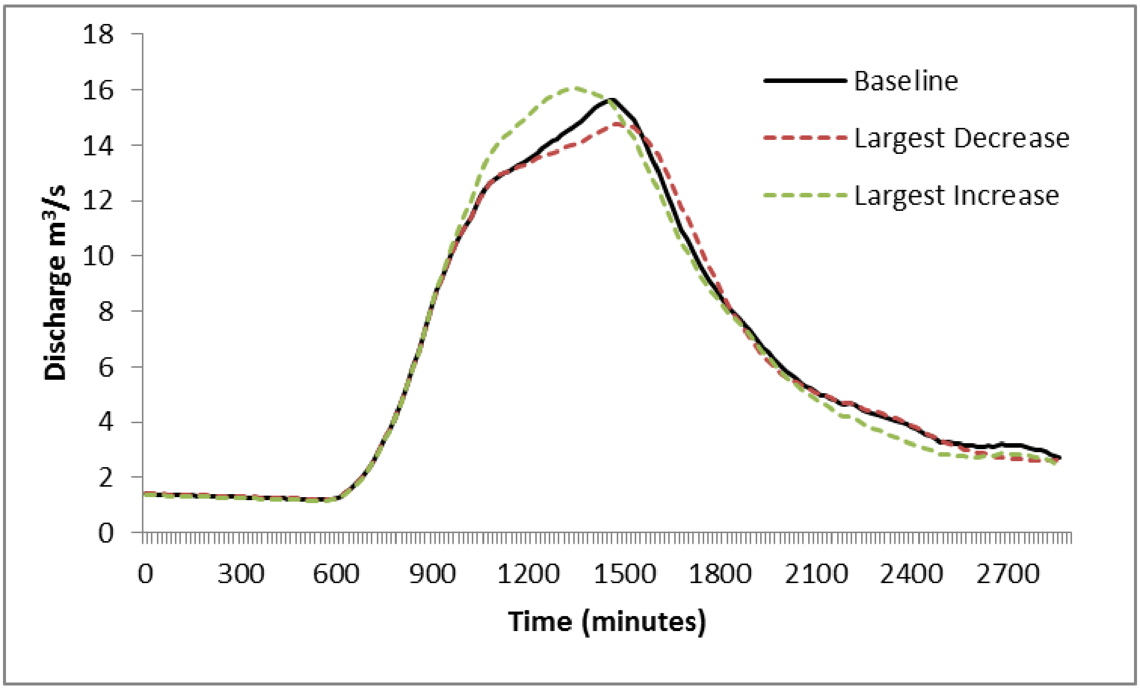


Figure 9.8 – Hydrograph showing the largest decrease and largest increases to peak discharge compared to baseline (calibration) hydrograph for example model application. Scenarios are changes in reach-scale land cover (0.1-2% of catchment area), where floodplain Manning's n is changed from n=0.07 to n=0.10 and in channel Manning's n is changed from n=0.05 to n=0.06

9.10. Discussion and Conclusions

The model described here demonstrates that heuristically informative data can be generated using the reduced complexity approach of OVERFLOW. The spatially distributed unit hydrograph approach developed by Odoni and Lane (Odoni and Lane, 2010; Odoni and Lane, *In Prep*) allows for a substantial reduction in computational resources for catchment wide hydrology modelling of flood events and allows the effects of multiple spatially diffuse land use scenarios to be tested. Such an approach can form the initial exploratory modelling phase in a catchwide investigation into the effects of land use changes on flood hydrology, either with the aim of targeting areas which can be part of a holistic flood defence scheme or as an investigation into the effects of potential proposed land use change. Once areas of interest have been identified using OVERFLOW a more computationally expensive, physically based modelling approach, such as ISIS or Hecras could be used to provide quantitative predictions of flood behaviour for a limited number of areas.

Alternatively OVERFLOW can form the basis for a heuristic modelling exercise to compare the effects of spatially distributed land use where the goal is to study directionality and magnitude of change between different scenarios and where quantitative predictions are not of paramount importance. In this study OVERFLOW will be used to investigate changing land cover and changing in-stream wood loads resulting from river restoration programmes. The investigation is describe in detail in Chapter 10 and comprises a study of two different river restoration strategies; 1) the insertion of engineered logjams, 2) the restoration of floodplain forests. For 1) Manning's n values based on the hydraulic resistance calculated for logjams in the New Forest and reported in Chapter 7 are used to parameterised model scenarios. In 2) the conceptual model of floodplain forest succession following restoration described in Chapter 8 is used to derive model scenarios for floodplain and channel complexity at 25 years, 50 years and 100 years post-restoration. These investigations will be used to give insight into the use of engineered logjams and changing forest cover to alter flood hydrology.

10. Peak flow hydrology simulation in Lymington catchment using OVERFLOW. II: Modelling the effects of river restoration on flood hydrology

10.1. Introduction

Historically, flood mitigation and defence strategies have focused on two overlapping goals i) increasing the conveyance capacity of river channels, in order to decrease the likelihood of flooding occurring for a given rainfall event, and ii) constructing physical point defences to protect vulnerable areas of land from inundation. Climate change is likely to lead to more extreme weather (Arnell, 2003) with modelling studies predicting that the magnitude and frequency of flood events will increase (Hannaford and Marsh, 2007; IPCC, 2007; Robson, 2002). A flood which has a current return period of 50 years could become a 25 year flood within the next century (Kleinen and Petschel-Held, 2007), although the level of uncertainty in climate projections makes precise local predictions difficult (Prudhomme et al., 2003). Studies are already showing more frequent and higher magnitude floods in recent decades (Robson, 2002; Wheater, 2006). With increasing awareness that the likelihood of intense rainfall events will become more common in the future due to climate change (Evans et al., 2006a) a policy of increasing the extent and height of at-a-point flood defences is both unsustainable and undesirable (Johnson et al., 2007; Johnson and Priest, 2008). Attention is turning towards alternative ways of mitigating flood risk through diffuse land use methods (e.g. Defra, 2005a; Defra, 2007; Johnson and Priest, 2008). Small scale studies have shown that spatially distributed land use changes can reduce runoff speed and volumes, and act to reduce downstream flood peaks and increase flood wave travel times (Acreman et al., 2003; Liu et al., 2004; Sholtes and Doyle, 2011; Thomas and Nisbet, 2007; Thomas and Nisbet, 2012). The extent to which local effects translate into measurable and important changes to flood hydrology at the catchment scale remains uncertain (Parrott et al., 2009; Pattison and Lane, 2012a).

There is an existing programme of riparian land use change being undertaken in the European Union as a result of the Water Framework Directive (WFD); river managers are increasingly using river restoration and river rehabilitation as a mechanism to meet the aims of the WFD to ensure rivers have "good ecological status" (Kallis and Butler, 2001). There are two possible mechanisms by which riparian land use is likely to change at local levels; either through specifically targeted flood risk mitigation, or through programmes of river restoration, which may also include flood risk mitigation as an ancillary aim, furthermore efforts to promote and conserve European forests will see increased afforestation (Robinson et al., 2003). Given future changes to catchment land management there is pressing need to understand the directionality and magnitude of changes to catchment flood hydrology resulting from local land use change. Reach scale modelling studies have indicated that restoring channel planform for engineered channels can reduce peak flow by 10-15% (Acreman et al., 2003), floodplain buffer strips can reduce peak flow by 3.8% (Ghavasieh et al., 2006) and large wood jams can delay flood peaks (Gregory et al., 1985; Sear et al., 2006; Thomas and Nisbet, 2007; Thomas and Nisbet, 2012).

Given the uncertainty in applying diffuse land use flood mitigation at a catchment scale (Parrott et al., 2009; Pattison and Lane, 2012a) and the logistical problems of setting up study catchments, initial exploration of the effects of catchment scale diffuse land use on flood hydrology must be achieved through numerical modelling. This study seeks to apply a numerical modelling approach to answer some key questions about the potential changes to flood hydrology resulting from river restoration. Conducting catchment wide modelling investigations with multiple diffuse land use scenarios is problematic using conventional hydrological models (see Chapter 9 and Appendix B). This is due to: the need for model response to be spatially explicit, ruling out some modelling approaches (Liu et al., 2003; Olivera and Maidment, 1999), a frequent lack of sufficient parameterisation data at required scales for catchment modelling and computational challenges given the number of potential model scenarios to be tested, which for even a small catchment and a limited number of scenarios can easily exceed one thousand model runs (see Chapter 9.2).

OVERFLOW (see Chapter 9) represents an alternative approach to conventional hydrological models by using a spatially distributed unit hydrograph method to simplify many of the physical processes involved in flood wave propagation in order to reduce model complexity and allow for rapid model run times of around 10-12 minutes. Furthermore, the simplified representations of land cover and hydraulic geometry allows multiple land use scenarios to be tested with relative ease. OVERFLOW uses a map of the study catchment with a Manning's n hydraulic resistance value for each grid cell as a model input. The 'Manning map' can be easily manipulated to vary the hydraulic resistance at any part of the catchment to represent different land uses. Channel hydraulic resistance values can also be altered within the Manning map to represent changes in channel characteristics.

The combination of rapid model run times and the ease of setting up multiple land use scenarios for model investigations makes OVERFLOW very useful as an exploratory tool for catchment modelling. It should be noted that although previous studies utilising OVERFLOW have produced promising results (Byers, 2011; Odoni and Lane, 2010) the model is designed only as an initial exploratory tool. The purpose of using an exploratory modelling approach is to identify potential areas or parts of a catchment which result in changes to flood hydrology which are of interest; in this case those which result in substantial reductions to the peak discharge at the catchment outflow. In cases where OVERFLOW is applied as part of a detailed flood risk management assessment for a specific catchment the small number of land use scenarios which have been identified as of interest during the exploratory modelling can then be modelled using more physically based models than OVERFLOW (such as ISIS Tuflow) in order to validate the output of OVERFLOW and to deliver quantitative predictions of flood behaviour. In this study the aim is to explore general relationships between the extent, location and magnitude of land use change within a catchment and therefore such explicit, site specific analyses are beyond the scope of this work.

River restoration techniques can be divided into broad categories representing the types of changes made to the channel and riparian zone (Table 10.1). The restoration categories were examined to determine which had the greatest potential to affect runoff and flood wave travel time, along with which techniques have the widest implementation (Table 10.1). Of these restoration techniques, changing riparian land use to woodland, and increasing the amount of in-channel large wood were identified as having a large potential effect on flood hydrology and also widely applied during

Restoration Technique	Example	Potential Flood hydrology effects	Implementation	Reference [a]
Bank stabilisation	Use of rip-rap, willow spilling etc to prevent bank erosion	Small potential change in bank hydraulic resistance	Widely used	(Maynord, 1991)
Flow modification	Insertion of dams, bunds etc to regulate flow through the catchment	Attenuate flood wave across the area the structure is installed in.	Used rarely, typically linked to flood mitigation schemes	(Nicholson et al., 2012)
Fish passage	Modifying weirs, locks, etc to allow migrating fish to pass obstruction	Negligible	Widely used	(Rajaratnam et al., 1988)
Insertion of logjams or boulders	Increasing complexity of river channel by inserting objects to deflect and pond flow	Increasing in channel hydraulic resistance. Increase frequency of overbank flows and attenuate flood wave	Commonly used	(Brooks et al., 2004; Gregory et al., 1985; Thomas and Nisbet, 2012)
Insertion/promotion of large wood input	Changing river management to stop removing naturally occurring large wood	Increasing in channel hydraulic resistance. Increase frequency of overbank flows and attenuate flood wave	Commonly used	(Shields Jr and Gippel, 1995)
Removal of non-native vegetation	Cutting of invasive riparian plant species such as Japanese knotweed, Himalayan balsam	Possible short-term decrease in overbank hydraulic resistance	Commonly used – only applicable where vegetation exists	(Bal and Meire, 2009; Tabacchi et al., 2000)
Planting of riparian	Adding buffer strips to attenuate	Increase in overbank hydraulic	Occasionally used	(Peterken and

woodland buffer strips	runoff derived problems such as sediment delivery to river	resistance, decrease in both runoff volumes and speed		Hughes, 1995; Tabacchi et al., 2000)
Re-meandering	Altering channel planform to include more bends	Decrease in bed slope, decrease in stream power.	Occasionally used*	(Liu et al., 2004)
Planting of riparian forests		Increase in overbank hydraulic resistance, decrease in both runoff volumes and speed	Rarely used	(Archer, 1989; Piegay, 1997; Tabacchi et al., 2000)
Raising bed level and/or narrowing channel		Decrease in channel capacity, increase in frequency of overbank flows	Occasionally used	(Bathurst, 1985; Hey, 1979)
Species stocking	Adding salmonids to a river	Negligible	Occasionally used	(Montgomery et al., 1996b)
Dam/weir removal		Decrease in flood wave travel time.	Rarely used – site specific, structure must already exist	

Table 10.1- Restoration techniques and the potential flood hydrology impacts of their use. The final implementation column is an estimate of how frequently these techniques are used in river management, not solely as part of river restoration projects. * - although generally this technique is only occasionally used it has seen widespread usage in the New Forest in recent years (Oakley, *pers. comm.*, 6th July, 2011). [a] – references in this column are for the hydraulic effects of restoration types, references on the implementation of restoration types are from (Bernhardt et al., 2005; Forestry Commission, 2011; River Restoration Centre, 2013)

river restoration. These two techniques thus represent the focus of the modelling study. Other techniques which are likely to have a measureable effect on flood hydrology are the removal of dams and weirs, however these are site specific, not spatially diffuse and therefore can be modelled using conventional hydrological models (e.g. Conly and Martz, 1998; Fjeldstad et al., 2012). Techniques such as re-meandering and installation of ponded flood storage and bunds can potentially mitigate flood risk; however they have not been included in this modelling study due to their relative low usage compared to the ubiquity of large wood based restoration (Bernhardt et al., 2005; River Restoration Centre, 2013) and in order to retain a tight focus to the modelling work and keep the computational time of model runs manageable.

10.2. Aims

The aim of this modelling exercise is to investigate the effects on downstream flood hydrology of plausible forest and channel management scenarios which may arise as a result of river restoration schemes. Two broad types of restoration scheme were chosen as scenarios; those using the insertion of Engineered Logjams (ELJ's) with little or no changes to existing land management plans (Bernhardt et al., 2005; Brooks, 2006; Thomas and Nisbet, 2012), and those using a philosophy of re-naturalisation or forest regeneration where riparian forests are allowed to mature and develop a complex understorey of standing and lying deadwood (Nislow et al., 2002). In practice, the first scenario is unsustainable without the second, and thus restoration guidance often advocates combining the two – the latter taking longer to develop (Collins et al., 2012).

The specific aims of the modelling investigation are:

- Determine if reach scale changes to channel or floodplain hydraulic resistance are observable in the catchment flood hydrograph.
- Establish if there is a spatial sensitivity in catchment hydrograph response to local changes to land use and investigate the sensitivity of response to reach characteristics.
- Constrain the magnitude of change in catchment hydrograph peak discharge for land use changes at scales from reach to sub-catchment.

- Determine if there is a relationship between the spatial extent of restoration and the magnitude of change to flood peak discharges.

10.3. Scenarios

A major challenge for river restoration design is the use and management of in-stream large wood within projects where there is engagement with diverse sets of stakeholders. In-stream large wood has a negative public perception (Chin et al., 2008; Chin et al., 2012; Piegay et al., 2005), and the use of large wood in river restoration and failure to remove natural large wood from river channels by managers is often resisted by stakeholder groups (Piegay et al., 2005, Oakley, *pers. comm.*, 6th July, 2011). As a result of public resistance river restoration projects are often designed to have either; no in-stream large wood, only heavily stabilised and anchored large wood (Brooks et al., 2006; Lewis, 2010), and/or an agreement to periodically remove additional large wood naturally ending up the channel (Oakley, *pers. comm*., 6th July, 2011). As a result of these practises there is a pressing need to understand the difference in flood hydrology response to restoration schemes where in-channel large wood is limited through management, whilst riparian forest management is changed to allow regeneration.

10.3.1. Engineered logjam scenarios

In practise ELJ schemes are typically limited in extent, usually covering lengths of river of 1km or less (Bernhardt et al., 2005), the implementation of such schemes is extremely variable with few specific design guidelines in either the academic or grey literature (see chapter 5), although ELJ's filling the channel and as far as possibly mimicking natural processes are often specified as effective for meeting restoration aims (Abbe and Montgomery, 1996; Thomas and Nisbet, 2012). Previous research has shown channel spanning jams can be divided in complete and active types (Gregory et al., 1993; Gregory et al., 1985; Gurnell and Gregory, 1995), where active jams cause a step in the water profile at low flows, and will thus have a greater hydraulic resistance than complete jams. Although in the short term it is possible to design and construct a logjam to be either complete or active, research has shown channel spanning jams will shift between active and complete types on an annual and a seasonal basis as a function of the packing of interstitial space with organic debris such as leaf packs and fine wood (Sear et al., 2010). It would therefore be unrealistic to model a river reach with ELJ's as having only active or complete logjams.

Field studies reported in Chapter 6 and in Kitts (2011) show that at near to bankfull discharge a complete logjam has a Manning roughness of n≈0.16 and an active logjam a Manning roughness of n≈0.27. Field data from Chapter 5 shows the ratio active:complete logjams in the Highland Water is 8:17; assuming this ratio of logjams and averaging the field data for Manning's n for these two logjam types gives a mean Manning's n=0.196 ±0.08. This mean value of n=0.20 (2sf) was used to represent all modelled ELJ's in this study. An alternative to using one mean value for all logjams in the study would have been to randomly distribute logjam resistance values within the range of standard deviation. Such a method would have advantages in terms of the robustness of experimental design, but would have necessitated multiple additional model runs to randomly sample the distribution. There remains however an inherent uncertainty in setting or choosing any hydraulic roughness coefficients. Although incorporating variable values within standard deviation addresses some of these uncertainties it is an approach which is computational intensive. It was decided that such an approach was not proportionate and would have necessitated compromises in the scope of the investigation given that computational and user resources were limited. Furthermore each modelled logjam represents around 0.02% of the channel network, it is therefore unlikely at small scales that changes in hydraulic resistance within standard deviation will have a large magnitude effect on model outputs. In order to address uncertainty and limitations of this averaging approach for ELJ roughness coefficients an uncertainty analysis is performed to explore the sensitivity of the model output to the Manning's n value used (see section 10.6).

Although specific guidance for installation of ELJ's is sparse, there is literature advocating a downstream spacing of 7-10 channel widths between ELJ's to mimic natural processes (Linstead and Gurnell, 1998; Thomas and Nisbet, 2012), which is used in practice (Oakley, *pers. comm.*, 6th July 2011; Thomas and Nisbet, 2012). Although data exist on the spacing of logjams within New Forest streams (Gregory et al., 1985; Gurnell et al., 1995; Sear et al., 2010) (chapter 5/paper), these are not completely natural due to a legacy of past management which affects logjam spacing, furthermore the objective of this study is to examine effects of ELJ's so using guidelines on ELJ usage and insertion, rather than field data is appropriate. Using the drainage network generated for OVERFLOW an overlay map was created pinpointing potential logjam locations based on local channel width maps to generate the cell spacing between logjams representing 7-10 channel widths, this results in cells potentially containing ELJ's which are closer together in narrow channels, and spaced further apart in wider channels (Figure 10.1).

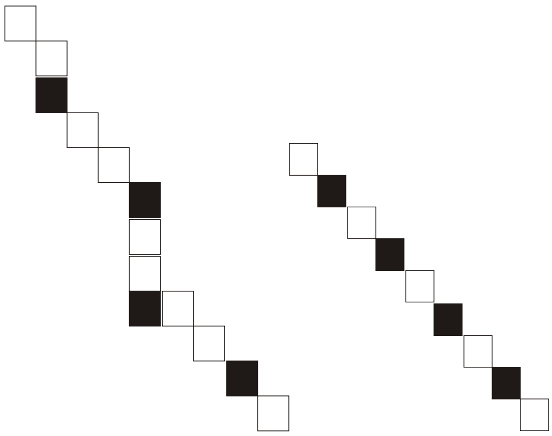


Figure 10.1 - Black cells represent the position of ELJ's spaced 7-10 channel widths apart present in two streams of differing bankfull width; the left is a wider channel. In smaller streams (right) the cells with logjams in are closer together than those in the wider channel (left). In the OVERFLOW model, these represent cells given a Manning's roughness value of 0.196.

10.3.2. Forest Regeneration Scenarios

Forest regeneration scenarios represent a change in both riparian land management and river management. As a forest ages and moves through stages of succession the patterns of density of trees, size of tree stems and volumes of deadwood will change, in the case of riparian forests the volumes of large wood delivered to the channel will also change through time (see Chapter 8). The hydraulic resistance of a forested floodplain and its associated channel will therefore also be expected to change through time as the forest matures. Modelling scenarios of changes to flood hydrology following riparian forest regeneration need to be based on forecasts of how the riparian forest changes over time in respect of the number and size of live trees and the volumes of deadwood both on the floodplain and in the river channel, and how those changes will be reflected in changes in hydraulic resistance.

The methods used for predicting the composition of a regenerating riparian forest are explained in Chapter 8. Predictions for forest composition are achieved through two methods; the first is with reference to literature values for dead wood biomass in forest stands of different ages (e.g. Christensen et al., 2005); the second is via use of a forest growth model (NE-CWD) to simulate how the age composition of a forest changes through time (Nislow, 2010).

Predictions of how a regenerating forest changes over time (Chapter 8), were used to derive a conceptual model of riparian beech forest composition in respect of living and deadwood for 25 years, 50 years and 100 years after the establishment of riparian forest restoration (Figure 8.6). These three temporal scenarios are used as the basis of riparian forest restoration modelling scenarios (Table 10.2 – scenarios 2, 3 and 4). The 25 year scenario was selected as the first point in the forest regeneration at which the number of pieces and biomass of large wood begins to increase in the system (see Figure 8.6), before this point any changes in forest composition are likely to have negligible effects on hydraulic roughness as there are virtually no changes in quantities of large deadwood (Bretz Guby and Dobbertin, 1996; Laser et al., 2009; Chapter 8, Figure 8.6). The 100 year scenario was selected as the point at which the number of pieces and biomass of large wood reach a maximum value and the forest achieves a state of dynamic equilibrium (Figure 8.6). The quantity of large wood is near a maximum at this point and the effects of large wood and forest cover on hydraulic roughness will also be near maximum values. The 100 year scenario therefore acts to constrain the magnitude of potential large wood effects. The 50 year scenario was selected as a median point within the forest regeneration (Figure 8.6).

Three additional modelling scenarios are used for 25 years, 50 years and 100 years post restoration, with the existing in channel large wood loadings maintained for all three scenarios (Table 10.2 – scenarios 5, 6 and 7); i.e. the hydraulic resistance of floodplain forest is changed for the scenarios whilst leaving the in channel hydraulic resistance unchanged at the calibration values. These three scenarios explore the effects of river restoration where in-stream large wood loads are continually managed to maintain low values.

Scenario Number	Scenario	Channel Hydraulic Resistance	Floodplain Hydraulic Resistance
1	Engineered Logjams (Spot Dams)	n=0.20 at alternating grid cells (Figure 10.1)	n=0.07 (calibration value)
2	25 year forest growth	n=0.06	n=0.10
3	50 year forest growth	n=0.075	n=0.12
4	100 year forest growth	n=0.10	n=0.15
5	25 year forest growth (forest only)	n=0.05 (calibration value)	n=0.10
6	50 year forest growth (forest only)	n=0.05 (calibration value)	n=0.12
7	100 year forest growth (forest only)	n=0.05 (calibration value)	n=0.15

Table 10.2 - summary of modelling scenarios and associated Manning 'n' values.

10.4. Determination of hydraulic roughness values

In chapter 9 the rationale behind the use of spatially uniform, flow invariant values for Manning's n in OVERFLOW is explained, along with the importance of selecting appropriate values for this parameter. Although it is possible to measure hydraulic resistance associated with logjams in the field and to partition the roughness contributions from large wood (Curran and Wohl, 2003; Kitts, 2011; Shields Jr and Gippel, 1995)(see Chapter 7) it is not possible to directly measure hydraulic resistance associated with riparian forest succession in the New Forest due to an absence of suitable analogues. Given the uncertainties in setting Manning's n values, the goal is to choose values which are both realistic and representative, particularly given modelling scenarios in this study involve forecasting land cover and in-channel wood loads and then translating these conceptual land cover scenarios into values for hydraulic resistance.

Hydraulic resistance values for ELJs are taken from field measurement reported in Chapter 7. In order to select appropriate values for Manning's n for floodplain and channel resistance for riparian forest restoration scenarios reference was made to two techniques; a general approach based on adjustment for channel characteristics (Jarrett, 1985) and selecting values based on broad channel characteristics from reference tables (Chow, 1959). Manning's n values selected for modelling scenarios and comparison with values derived from these techniques are displayed in Table 10.3.

A variable power equation (VPE) calculation for plane bed roughness (n=0.048 ±0.003) was reported in section 9.5.1.1. as part of model calibration. Although VPE has a sound theoretical basis, it is only designed to be applied to channels without obstructions or form roughness (Ferguson, 2007; Ferguson, 2010). In the case of the modelling scenarios in Table 10.2 there are large wood obstructions in the channel so the VPE method is therefore inappropriate to forecast changes in hydraulic resistance associated with these increased large wood loads.

10.4.1. Jarrett adjusted channel method

In this method of calculating hydraulic resistance the total hydraulic resistance is partitioned into contributing factors which are either calculated or selected from a table and then combined to give a range for the overall resistance.

$$n = (n_0 + n_1 + n_2 + n_3 + n_4)m \qquad 10.1$$

Where, n_0 is the base value from channel based average grainsize equations for a straight uniform channel (Strickler, 1981), n_1, n_2, n_3 and n_4 are additive values due to the effects of; cross-section irregularity, variations in the channel, relative effects of obstructions and type and density of vegetation respectively and m represents the degree of meandering (see Appendix C for reproduction of original table of values). Of these values it would be expected that only n_3 will be directly affected by a greater loading of dead wood to the channel. Although greater wood loadings would be likely

Scenario	Description	Manning's n used	Jarrett (1985)		Chow (1959)	
			Min	Max	n	Description
25 year Channel	In-stream large wood loading broadly same as pre-restoration, higher loading of small wood	0.060	0.040	0.069	0.045<n<0.060	Main channel, clean, winding, some pools and shoals, some weeds, lots of stones
50 year Channel	In-stream large wood loadings elevated, particular in very large size categories (>50cm dbh)	0.075	0.069	0.095	0.050>n>0.080	Main channel, sluggish reaches, weedy, deep pools
100 year Channel	In-stream large wood loadings at maximum values	0.110	0.092	0.130	0.075>n>0.150	Channel, very weedy reaches, deep pools, floodways; heavy stands of timber and underbrush
25 year Forest	Floodplain consists of dense stands of narrow stemmed, young trees, little deadwood	0.100			0.100>n>0.120	Dense willows
50 year Forest	Floodplain consists of mixed stands of semi-mature and young trees with abundant small deadwood	0.120			0.100>n>0.160	Floodplain heavy stand of timber, a few down trees, little undergrowth, flood stage at branches
100 year Forest	Floodplain is structurally at its most complex with large mature trees, large fallen logs and abundant deadwood.	0.150			0.100>n>0.160	Floodplain heavy stand of timber, a few down trees, little undergrowth, flood stage at branches

Table 10.3 – Values of Manning's n selected for riparian forest restoration scenarios for floodplain and channel. These are compared with values derived from methods described by Jarrett and Broad (1985) and Chow (1959), explanation of calculated values is provided in the text.

to lead to greater geomorphological heterogeneity and thus changes over time in the values of n_1, n_2 and n_4 compared to present day conditions, predicting the degree of these changes is a highly complex morphological problem beyond the scope of this study. The scenario values for n therefore only take account of direct changes in the degree of channel obstructions related to changes in wood load, and not ancillary channel changes resulting from these wood loads. As the roughness values are being applied on a spatially uniform basis the values for "obstructions" in the adjusted channel method does not represent the roughness of individual jams, rather than percentage of channel within the reach which will be blocked by logjams. The initial value calculated (Chapter 9) for the channel n=0.058 with a range $0.040<n<0.069$, using a value for n_3 corresponding to "effects of obstructions: minor" for which less than 15% of the cross-sectional area of a channel is blocked with obstructions, this is consistent with measured wood loadings of 29.16 m^3/ha (Chapter 5), which are quite low compared to the typical values of 110 m^3/ha for deciduous forests quoted by Gurnell at al (2002).

For a 25 year forest regeneration scenario the wood loading will only be slightly elevated for the channel compared to current values. Field data of wood loadings against logjam frequency (Chapter 5) indicates that small increases in wood loading would not be expected to substantially increase the frequency of logjams (obstructions) compared to current catchment averaged frequencies. However with an increase in the number of young trees and thus input of organic debris such as leaves there will be increased packing of pore spaces in existing logjams (Sear et al., 2010). The value for in channel hydraulic resistance was therefore assumed to be based on a n_3 value of "effects of obstructions: minor". This gives the same range of $0.040>n>0.069$ as the calibration values, however given the expected increase in roughness due to an increase in leaf packing an upper end value of n=0.06 was selected for this scenario, as compared to the lower end value of 0.05 used for calibration.

For the 50 year forest regeneration scenario the wood loading will be higher than present with a potentially total loading to the stream (equal to loading on the floodplain) of nearly 100m^3/ha (Figure 8.2). However the large pieces of wood shown to be effective at acting as foci for logjam formation (Chapter 5) will still be relatively rare with a loading of large stable wood (>30cm dbh) around 25m^3/ha, or roughly one log every 40-50m of channel length. This is due to the forest being insufficiently mature to have grown trees large enough to generate substantial amounts of this large deadwood (Figure 8.2). It would, therefore, be expected that the frequency of logjams would be slightly, but not substantially higher than in the present channel. A value of n_3=0.020-0.030 corresponding to "effect of obstructions: appreciable", corresponding to between 15% and 50% of the channel cross-section being blocked by obstructions was used giving a range of $0.069<n<0.095$ for total n. A value of n=0.075 was selected reflecting that wood loading values have only just begun to increase sharply at 50 years post regeneration.

For the 100 year forest regeneration scenario the in channel wood loading values will have begun to asymptotically approach a maximum value, reflecting that at this point the channel will have close to the maximum wood load component of hydraulic resistance. A value of n_3=0.040-0.060 for "effect of obstructions: severe" corresponding to obstructions occupying more than 50% of the cross-section and inter-obstacle spacing being short enough to cause turbulence across the cross-section was used giving a range of $0.092<n<0.130$ for total n. A value of n=0.110 was selected reflecting that the hydraulic roughness effects of wood have reached a maximum

10.4.2. Reference table method

Any Manning's n values for projected in-stream large wood loads based solely on values selected from reference tables such as Chow (1959) are likely to be highly speculative due to the general descriptions of complex natural channels and the broad range of values applied to each category. Furthermore in Chow (1959) specific reference is not made to quantity of obstructions within the channel which represent the major variable in the study scenarios. The limitations with using reference tables make them unsuitable for a priori estimates of Manning's n for complex natural channels, however, the reference tables are useful to check that estimates derived from other methods (such as the Jarrett adjusted channel method above) are within the bounds of the respective descriptive category. For these reasons descriptions from Chow (1959) have been included in Table 10.3 showing the selected values correspond to broadly applicable description of the conceptualised channels at 25 years, 50 years and 100 years post riparian forest restoration.

10.5. Modelling strategy

For each of the seven modelling scenarios (Table 10.2) a similar strategy is followed which is summarised in Table 10.4. Short sections of restoration are frequently the norm where restoration is applied on an opportunistic basis (Bernhardt et al., 2005; River Restoration Centre, 2013), whereas more extensive sub-catchment and catchment scale restoration efforts are increasingly advocated by scientists as a more sustainable and successful method of restoration (Beechie et al., 2010; Nisbet et al., 2011). To model the potential effects of realistic restoration projects on flood hydrology it is therefore necessary to apply scenarios from the reach to the sub-catchment scale. The modelling strategy is based on simulating changes to land use at the individual and sequential reach scale, as well as at the sub-catchment scale and at headwaters across the entire catchment in order to compare different restoration approaches. Single runs (Table 10.4) are discrete segments of river of short length, sequential segments are sets of 5 sequential lengths of river channel which have been shown individually to either all reduce or all increase peak flood magnitude, sub-catchments are where altered hydraulic resistance values are applied across areas of the main catchment, headwaters are model runs in which every stream of a given Shreve stream order across the catchment has altered hydraulic resistance values applied.

Model Runs	Number of model runs	Stream length (km)		% Catchment area		Reference Code
		Min	Max	Min	Max	
Single Runs	298	0.025	4.2	0.01	3.00	Z1-Z298
Sequential Segments	60	1	10	4	12	Y1-Y60
Sub-Catchments	42	1	77	5	67	X1-X42
Headwaters	34	1	129	0.6	78	W1-W34

Table 10.4– summary of modelling strategy showing all model runs.

The catchment network is divided into discrete segments (Frissell et al., 1986), which are defined as a length of channel between confluence nodes (Figure 10.2), and as a result can be of varying lengths. These segments (n=298) form the basis of the modelling strategy where changes to hydraulic resistance values are applied to either individual segments or combinations of segments.

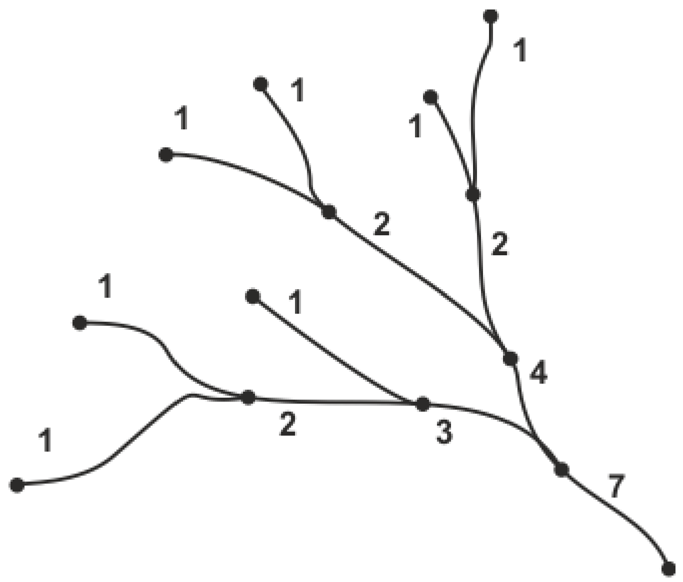


Figure 10.2 - cartoon showing how the drainage network is divided up into discrete reaches based on confluence nodes. Black circles show the start and end of each segment/reach and the Shreve stream order is annotated next to each segment.

10.5.1. Single Runs

All 298 segments in the study catchment are modelled with changes to hydraulic resistance (Table 10.3) applied to a single segment at a time (model runs Z1-Z298). These model runs represent the smallest scale of opportunistic river restoration such as installation of a small number of ELJs, in the case of scenario 1, or planting of riparian buffer strips, in the case of scenarios 2-7. The objective of these model runs is to establish if small reach scale modifications can be detected in changes to the catchment flood hydrograph, these runs also represent the bulk of river restoration projects which are typically applied to river lengths of less than 1km (Bernhardt et al., 2005).

10.5.2. Sequential segments

Individual model runs were analysed and divided into those reaches that have a 'positive' effect on the peak discharge (decrease of ≤-0.1% in peak magnitude), a 'negative' effect on the peak discharge (increase of ≥0.1% in peak magnitude) and 'neutral' (a change of between >-0.1% and <0.1%). An additional series of model runs (Y1-Y60) were comprised of 5 segments which are either all 'negative'/ 'neutral' or all 'positive'/'neutral'. Pilot studies indicate a threshold of ±0.1% for single segment model runs results in approximately a third of model segments being rated as positive, negative and neutral respectively; higher thresholds result in too few segments classified as positive/negative to allow sequential segments with the same directionality to be determined. The objective of these runs is to examine flood hydrology effects of changing hydraulic resistance values at scales up to 10km of river channel length; these model runs together with the single segments represent reach scale river restoration. By selecting sequential segments with the same directionality, the magnitude of peak hydrograph change can be constrained for this scale of restoration in a computationally efficient way; running every possible combination of 5 segments would result in over 2×10^{12} model runs, which is far beyond the scope of available computing resources.

10.5.3. Sub-Catchments

The effects of making changes at the sub-catchment scale are explored with model runs (X1-X42) in which the changes to hydraulic resistance are applied to entire sub-catchments, representing a change of between approximately 5% and 67% of the total catchment area. Additionally a single model run is conducted with the hydraulic resistance changes applied to the entire catchment. These sub-catchment runs allow the directionality and magnitude of changes to peak discharge, time over discharge and time to peak to be analysed relative to the proportion of the catchment which has hydraulic resistance changes applied to it.

10.5.4. Headwaters

Larger scale spatially diffuse effects are explored by running combinations of Shreve stream orders, to represent restoration being applied to headwater reaches across an entire catchment (model runs W1-W34). A total of 34 runs are conducted with all stream orders from 1-10 altered individually, along with combinations of two different and three different stream orders. The objective of these model runs is to test previous findings that afforestation of headwaters can be most effective in reducing flood peak flows (Thomas and Nisbet, 2007).

10.6. Uncertainty Analysis

OVERFLOW includes several parameters which are determined empirically, or are applied in the model at greater spatial resolution than calibration data can be collected; as a result there is a degree of uncertainty in the output from the models.

The key areas of uncertainty in calibrating OVERFLOW are the values used for hillslope/floodplain Manning's n, channel Manning's n, channel depth, channel width and the empirical backwatering functionality in the model code determined from the source DEM. A full uncertainty analysis would involve allowing all these parameters to vary independently and to conduct new calibration sets for them. This would result in 2^5, or 32 new calibration sets. For each of these calibration sets there will be 200 hydrographs; 6400 hydrographs in total. Each of the model run scenarios would then need to be run through these alternative calibration hydrographs to determine the precise degree of uncertainty in the model output. Such a full exploration of model uncertainty however would require around 1.4×10^7 model runs, or 7000 years of computational time, even using the entire Iridis Super-computer cluster at University of Southampton, which due to competing demands is unfeasible, this would take nearly 8 months to run all models. Although such an exploration would ideally be done, at this scale it is clearly beyond the scope of any project.

As well as being used in the calibration process, estimated values for Manning's n were also used for all model scenarios simulating different land cover types. The scenarios can be separated into two broad types; ELJ spot dam scenarios and spatially averaged scenarios, as all the forest regeneration based scenarios involve applying a spatially averaged Manning's n value to the restored areas for floodplain and channel cells. A full uncertainty analysis would therefore need to explore the variability in model output when varying values used for Manning's n for modified reaches.

This modelling study is primarily a conceptual exercise to demonstrate a method of modelling spatially diffuse land use change at the catchment scale, and as per the model description (see Chapter 9 and Appendix B) is designed to be used as an initial exploratory tool. Using an exploratory modelling strategy the individual model runs are not treated as providing explicit quantitative predictions of flood hydrology, rather the modelling exercise as a whole attempts to define what is plausible in terms of directionality and magnitude of changes to flood hydrology (Bankes, 1993). In an exploratory modelling approach concepts of model validation and sensitivity analysis can be seen as nonsequiturs which are more relevant to predicative, consolidative modelling, in exploratory modelling issues of quality rest with the validity of the analytic strategy and the accuracy of the input data (prior knowledge) used by the model to generate new knowledge and insights (Bankes, 1993).

In terms of answering the modelling question the greatest source of uncertainty is not how well the calibrated model matches to verifiable ground data, but the confidence in the accuracy of outputs from modelling scenarios relative to the calibrated baseline, i.e. when a series of changes are made to catchment land-use the objective is to discern directionality of hydrograph response relative to the baseline and the relative magnitude of response between a series of alternative scenarios. The new information generated in the modelling approach is implicit in the information used to generate it (Bankes, 1993), in this case the empirical values used for Manning's n roughness coefficients and values of hydraulic geometry. In that respect the values used for modified Manning's n for the restoration scenarios are the source of greatest uncertainty in this modelling exercise as they represent the "prior knowledge" (Bankes, 1993) input into the model in order to generate new information.

A full uncertainty analysis of modified Manning's n values would involve allowing the hillslope and channel values to vary independently to a higher or a lower value to give

2^3 variations per model run (Figure 10.3) and to allow the ELJ spot dam value for Manning's n to vary singly to give 2 variations of a higher and a lower Manning's n

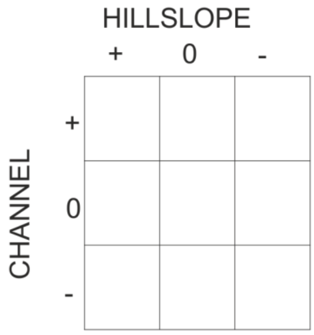


Figure 10.3 – Experimental design for uncertainty analysis; Manning's n values for hillslope and channel are allowed to vary independently to a higher and lower value to give 2^3 alternative model outputs for each model run.

value per model run. Such an approach would involve approximately 14,500 additional model runs, which equates to 6 months computational time. There are methods for reducing such an uncertainty analysis (e.g. McKay et al., 1979), however such methods are computationally and time expensive and beyond the scope of this project.

As an alternative method of analysing the uncertainty associated with model output a restricted uncertainty analysis was conducted. A two stage, three level experimental design (Figure 10.3) was used to vary independently the modified Manning's n values for the floodplain and channel to a lower and a higher value. A separate one stage, four level experimental design (Figure 10.4) was used to vary the value of Manning's n for ELJ spot dams to two lower values and two higher values. These experimental designs were applied to six scenarios from the main modelling study; two that show increases in peak discharge, two neutral and two which showed decreases in peak discharge. This uncertainty analysis allows a first pass at attempting to quantify the level of uncertainty in the model output; given the uncertainty in modified Manning's n used in model scenarios.

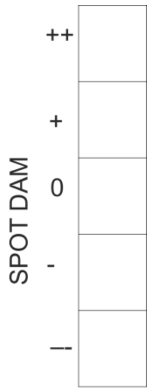


Figure 10.4 – Experimental design for uncertainty analysis of ELJ spot dams, Manning's n value for ELJ spot dams is allowed to vary to four alternative values; two higher than used in the main study and two lower than the main study.

10.7. Results

The results are reported relative to changes in the hydrograph at the catchment outlet, which is an urban location (Brockenhurst, Hampshire). It is assumed that lowering the peak discharge at this point is beneficial to land owners and other stakeholders and therefore a short hand notation is used describing a reduction in peak discharge as a "positive" change in response to the land use change scenario and an increase in peak discharge as a "negative" change, scenarios in which there is no change in peak discharge are described as "neutral".

10.7.1. Uncertainty Analysis

In order to provide context for the modelling results it is first necessary to constrain the level of uncertainty in modelling output. Figure 10.5 shows the degree of uncertainty in model output for ELJ spot dam scenarios and Figure 10.6 shows the degree of uncertainty in model output for riparian forest regeneration scenarios. The two axes show the variability in percentage change to peak discharge magnitude compared to the baseline calibration scenario (x-axis) and the variability in percentage time over peak compared to the baseline calibration scenario (y-axis). The reference for time over peak is the approximate bankfull discharge at the catchment outlet, i.e. this is a measure of the duration of overbank flooding at the catchment outflow. A tight clustering of points for a given scenario in both axes shows that modelling output is insensitive to values of Manning's n used for either hill slope/floodplain or channels.

The average uncertainty for ELJ spot dam scenarios (Figure 10.5) for peak discharge magnitude is ±0.23% and the average uncertainty for all forest restoration scenarios (Figure 10.6) is ±0.50%. Within these uncertainties in magnitude however there is no uncertainty in directionality. For all tested model runs across both ELJ spot dams and forest regeneration all "neutral" runs (those which show no change in flood peak magnitude or duration compared to the baseline calibration scenario) remained within ±0.1% of the baseline, all runs predicted to increase peak discharge remained >0.1% of the baseline and all runs predicted to decrease peak discharge remained <-0.1% of the baseline. Uncertainty is shown to be greatest in respect of the magnitude of reduction in peak discharge for runs which are predicted to decrease peak discharge (Figures 10.5 and 10.6).

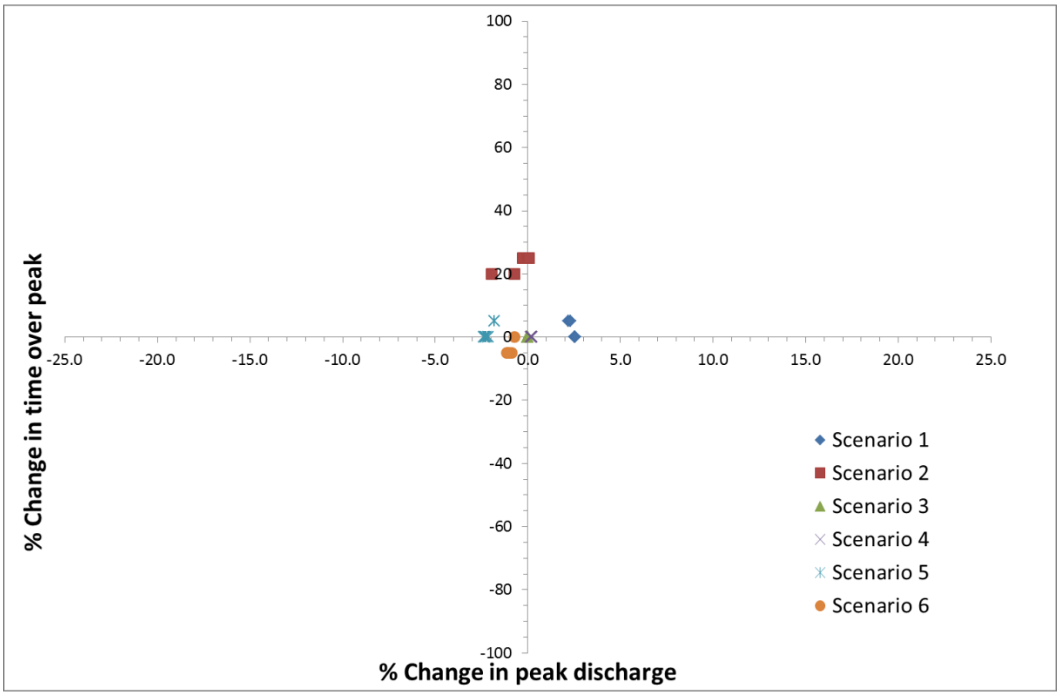


Figure 10.5 – phase space diagram for ELJ spot dam scenarios showing the uncertainty in model output given uncertainty in the Manning's n value used for spot dams. A tight clustering of points shows that modelling output is insensitive to values of Manning's n used for ELJs. This shows the average uncertainty in modelling output for change in peak discharge is ±0.23%.

Figure 10.6 shows there is low uncertainty for model output which predict no change to flood hydrology (centre) or increases in flood peak height and duration (top right quadrant). However, there is greater uncertainty with model results predicting decreases in flood peak height (bottom left quadrant), although this uncertainty is only in terms of magnitude, rather than directionality – i.e. the scenario always decreases the flood peak height irrespective of the manning's n value used in the uncertainty analysis, however the magnitude of this decrease is subject to uncertainty. The level of uncertainty in peak discharge reduction increases as the predicted magnitude of reduction increases.

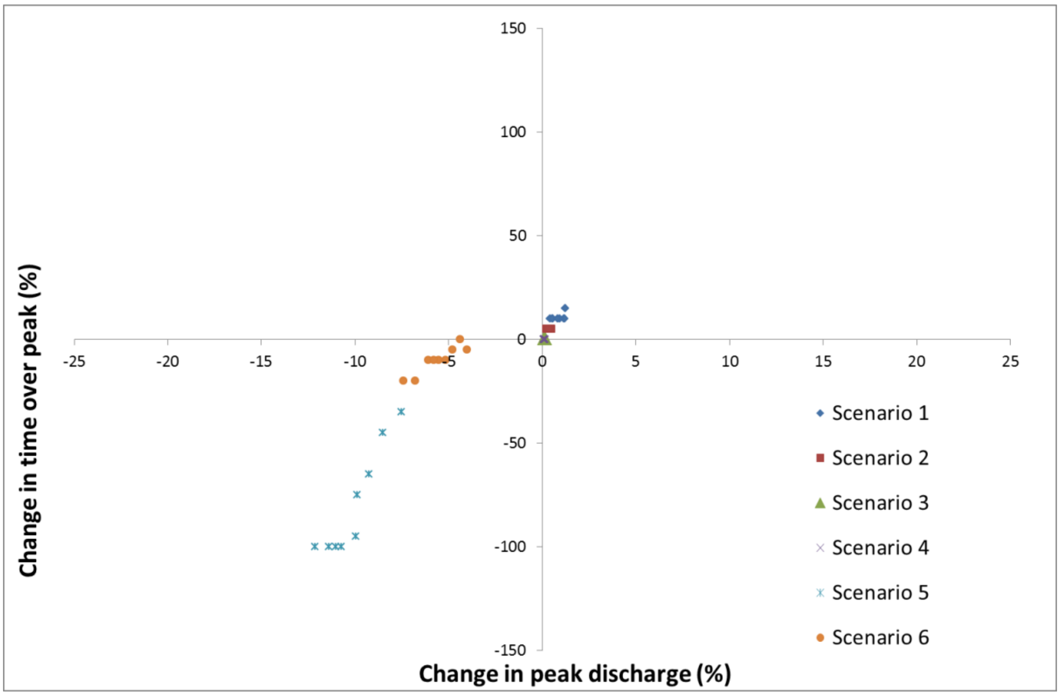


Figure 10.6 – phase space diagram for forest regeneration scenarios showing uncertainty in model output given uncertainty in the Manning's n used for hillslopes and channels. The average uncertainty in modelling output for change in peak discharge is ±0.50%.

10.7.2. ELJ Spot Dam scenarios

The magnitude of change to peak discharge is relatively small when ELJ spot dams are applied to a single reach with the majority of reaches showing a change in peak discharge less than the model uncertainty of 0.23% (see Appendix D for detailed reproduction of results). Figure 10.7 shows a catchment map displaying the directionality of change in peak flood magnitude for each reach/segment in the channel network when engineered logjams are applied to it individually. Figure 10.7 shows the largest reductions to peak discharge (reaches coloured dark blue in Figure 10.7) result from ELJ spot dams inserted into main channels with a Shreve stream order of greater than ten. First and second order reaches with ELJ spot dams are mostly neutral or result in a slight decrease to the peak discharge. Despite these general patterns there is a great deal of spatial variability in the response of peak magnitude to ELJ spot dam insertion.

Note that many of the responses in Figure 10.7 are less than the model uncertainty of 0.23%. However uncertainty analysis (Section 10.7.1) shows there is a high degree of confidence in directionality of response for changes in peak magnitude predicted as greater than 0.1% or less than -0.1%, i.e. for reaches coloured dark blue and red in Figure 10.7 the magnitude of change to peak discharge remains uncertain, but that these reaches will lead to decreases or increases respectively is within the bounds of uncertainty.

Figure 10.8 shows the percentage change in peak discharge for all ELJ spot dam scenarios plotted as a function of the total proportion of channel network altered in each scenario. This shows there is a trend for scenarios with a higher proportion of the network restored to display larger magnitude changes in peak discharge, although the directionality of this change is variable. Restoration with engineered logjams can result in changes of ±6% in peak discharge, but there do not appear to be any simple relationships to predict effect magnitude and direction given a specific type of restoration scenario.

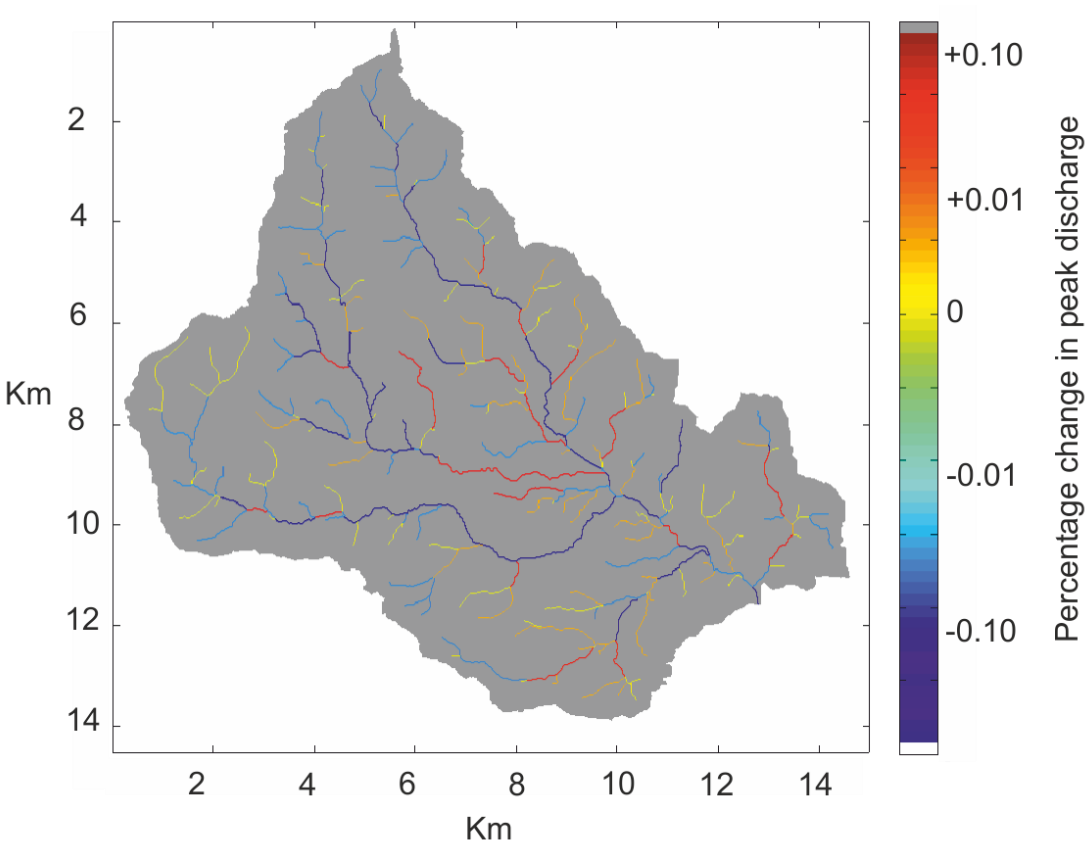


Figure 10.7 –map of the study catchment showing the directionality of change to flood peak magnitude with ELJ spot dam insertion into study reaches.

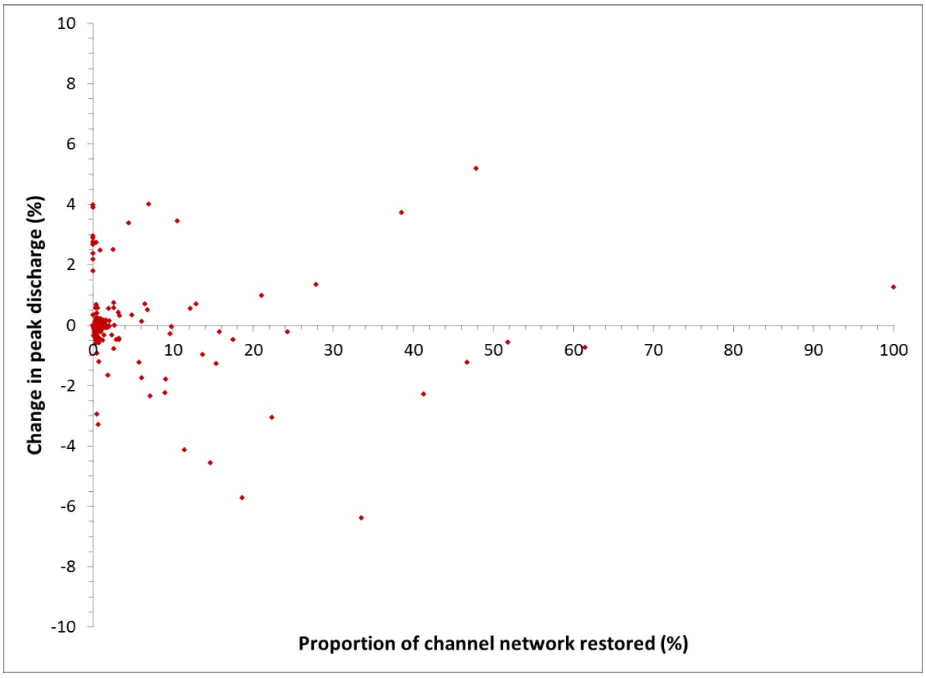


Figure 10.8 – plot of all ELJ spot dam model runs showing the relationship between change in peak discharge compared to baseline scenario and the percentage of the total catchment stream network which had ELJ spot dam applied to it (restored).

Figure 10.9 shows a quadrant analysis of the changes to the height and duration of flood peak for all ELJ spot dam scenarios. A “flood” is defined as when approximate bankfull discharge is exceeded at the catchment outflow, the x-axis in Figure 10.9 shows the change in the length of time that discharge is exceeding bankfull discharge at the outflow compared to the calibration scenario. Upper right and lower left quadrants of Figure 10.9 represent a broad shrinkage or extension to the flood hydrograph peak, without dramatically changing the shape of the hydrograph; i.e. both the duration and peak discharge either increase or decrease. The top left quadrant represents floods which have become shorter in duration, but higher in peak discharge, so are essentially becoming more “flashy”, the bottom right quadrant are floods which are lower in peak magnitude, but persist for longer. Although resultant hydrographs

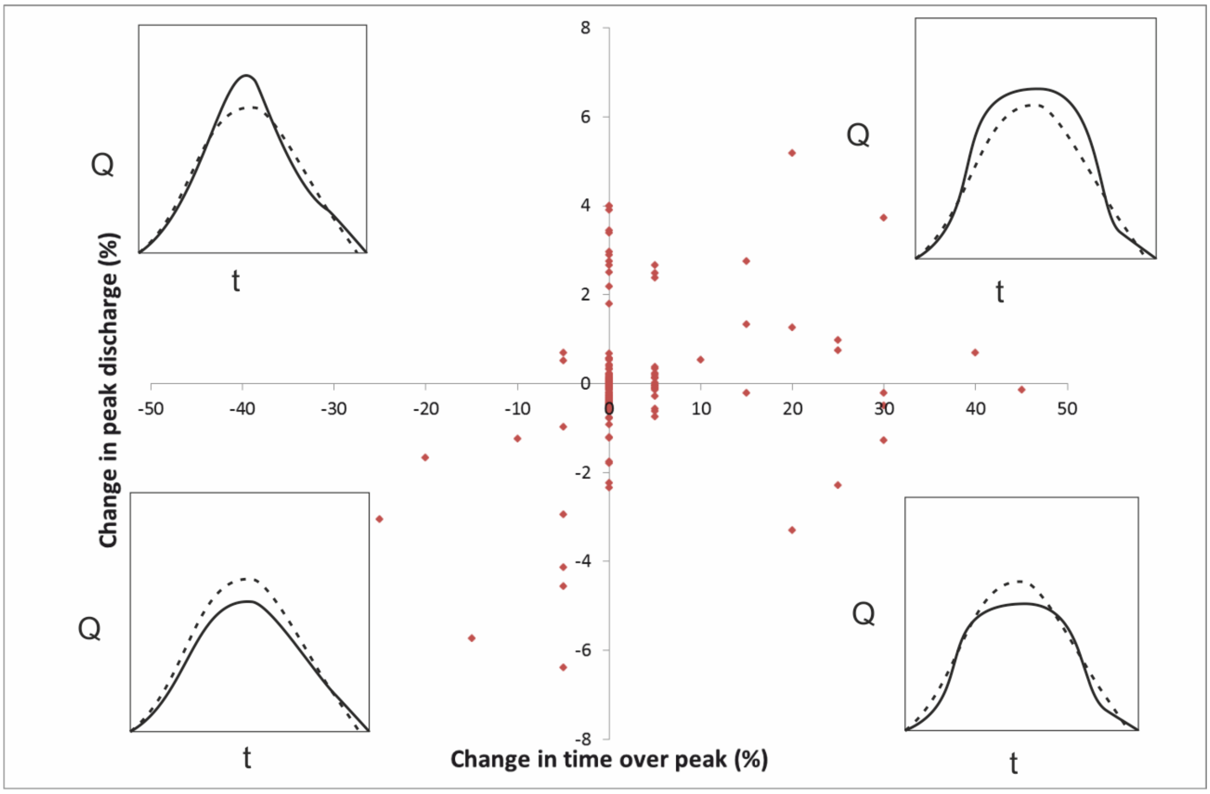


Figure 10.9 – quadrant analysis of change to flood hydrograph for all ELJ spot dam scenarios, with hydrograph cartoons superimposed in each quadrant. Hashed line in each cartoon is the baseline hydrograph and solid line is the resulting hydrograph.

are mainly distributed in the bottom left and top right quadrants, there is a large amount of scatter reflecting the unpredictable response of flood hydrology to engineered logjam insertion.

10.7.3. Forest Regeneration Scenarios

10.7.3.1. Forest Only

Figure 10.10 shows the spatial distribution of directionality in change to the flood peak discharge for the three forest regeneration scenarios of 25 years, 50 years and 100 years post forest restoration. Maps in Figure 10.10 show where forest restoration is applied to segments close to the catchment outflow (bottom right of maps) this tend to lead to increases in peak discharge (orange/red), whereas where forest restoration is applied to segments distant from the outflow this tend to lead to decreases in peak discharge (dark blue). As the model moves from 25 years (Figure 10.10A, n=0.100) to 100 years (Figure 10.10C, n=0.150) post forest restoration the complexity of floodplain forest cover increases and the spatial pattern is reinforced. Most reaches show a greater magnitude of change in peak discharge with increasing hydraulic resistance, and only a small fraction (<1%) of reaches change directionality.

Figure 10.10 indicates both the spatial position of restoration and the complexity of floodplain forest cover are important in determining peak discharge response. Note model uncertainty for forest model runs is 0.5%, however despite this uncertainty in magnitude the uncertainty analysis shows no change in directionality for predictions of greater than ±0.1%, i.e. for red and dark blue coloured reaches there is uncertainty around the magnitude of change in peak discharge but a high degree of confidence these reaches do represent increases in peak discharge and decreases in peak discharge respectively. Detailed results for individual segment restoration are reported in Appendix D.

Figure 10.11 shows the relationships between the change to flood peak magnitude for model runs of individual reaches and the characteristics of those reaches. There is a weak trend for increasing magnitude of change with both an increasing reach channel length (Figure 10.11D) and reach catchment area (Figure 10.11C), suggesting a weak relationship between the basic number of cells for which hydraulic resistance is modified and the change in peak discharge. The greatest magnitude of change, either increase or decrease is found at the shallowest slopes (Figure 10.11A), suggesting that for steep reaches the stream power negates any changes to hydraulic resistance. There is no identifiable trend between magnitude of flood peak change and either Shreve stream order or distance to catchment outflow. Together Figures 10.14 and 10.15 illustrate the variability in flood peak response given variations in the reach for which forest regeneration is modelled.

Figure 10.12 shows how flood peak magnitude varies with proportion of channel network which has forest regeneration modelled. This figure shows the majority of model runs lead to a decrease in peak discharge. Figure 10.12 shows a three stage relationship between proportion of channel network restored and change in peak discharge; between 0-22% of network restored the change in peak discharge is highly variable in both magnitude and directionality, with the majority of model runs showing only modest (<1%) changes in peak discharge. For scenarios with between 22-50% of the channel network restored all model runs lead to decreases in peak discharge, with increasing area leading to greater decreases in peak discharge. These large decreases in peak discharge are likely due to de-synchronisation of sub-catchment contributions from the main flood hydrograph. For scenarios in which over 50% of the channel network is restored all model runs lead to a decrease in peak discharge. These decreases are due to attenuation of the main flood wave and are of lower magnitude than runs in which 22-50% of channel network is restored. The figure implicitly indicates restoration which de-synchronises sub-catchments has a greater effect on reducing peak discharge than more extensive restoration which attenuates the entire flood peak.

Figure 10.12 shows as the forest matures from 25 years post restoration (Forest n=0.100) through to 100 years post restoration (Forest n=0.150) in almost all cases greater hydraulic resistance results in a larger magnitude reduction in peak discharge.

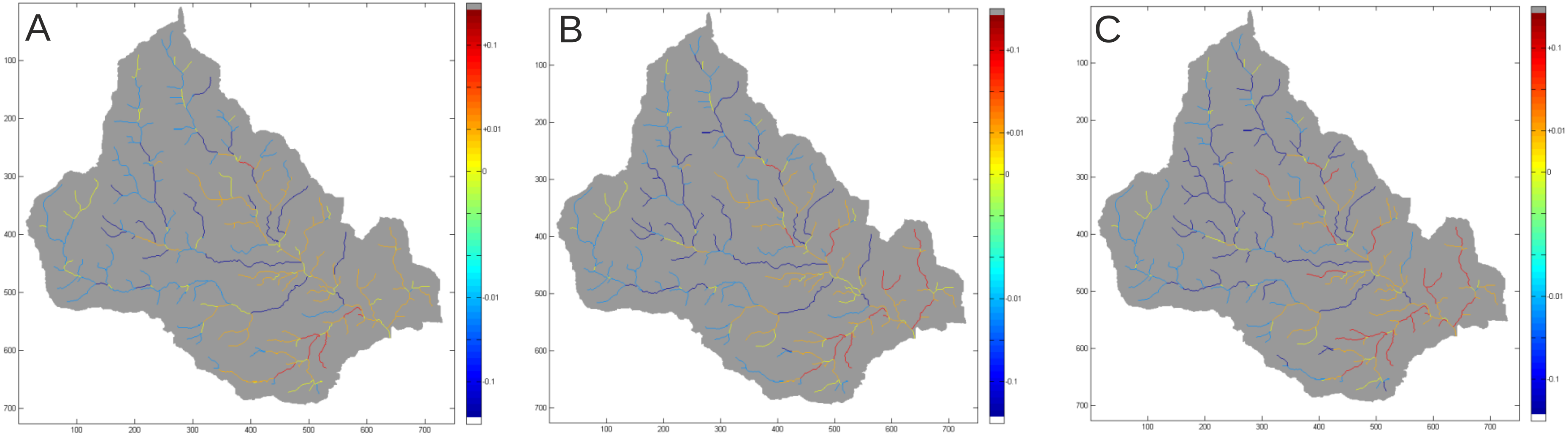


Figure 10.10 - maps showing the directionality of change to flood peak magnitude for application to a single reach.

A) 25 years post forest regeneration (change to overbank Manning n=0.100) for study reaches, B) 50 years post forest regeneration (overbank Manning n=0.120) for study reaches, C) 100 years post forest regeneration (overbank Manning n=0.150) for study reaches. Reaches coloured yellow are neutral and show no change to peak discharge, orange and red are increases of >0.01% and >0.1% to peak discharge respectively, light blue and dark blue are decreases of <0.01% and <0.1% respectively.

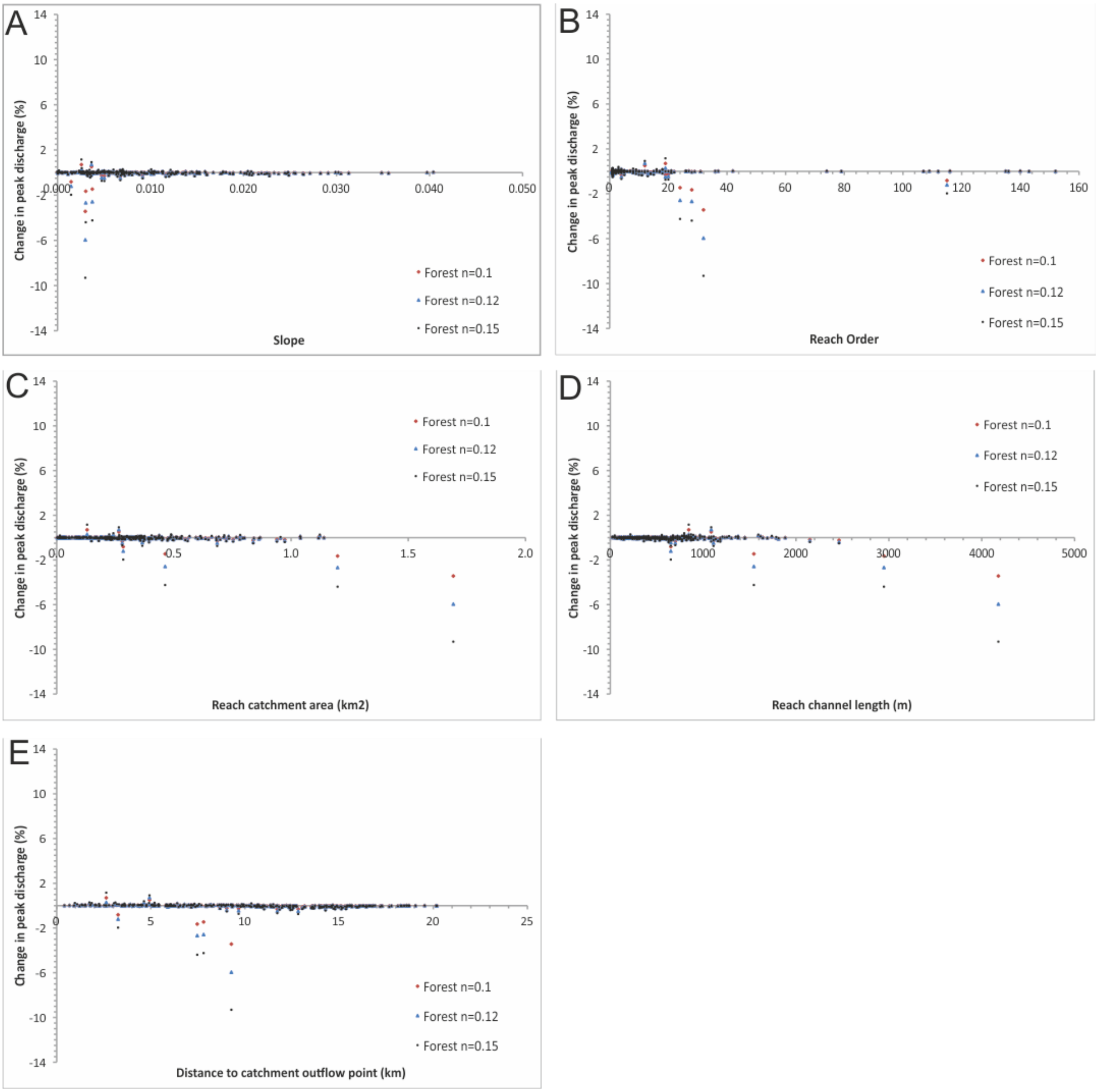


Figure 10.11 – charts showing the relationship between reach characteristics and change in flood peak discharge following a change in overbank hydraulic resistance to simulate forest regeneration for a single reach;

A – Study reach slope, B – Shreve stream order of study reach, C – catchment area draining to the study reach, D – the length of the simulated study reach, E – the cumulative channel length from the reach to the catchment outflow, a proxy measurement for how hydrologically proximal the reach is to the outflow.

Figure 10.17 displays all of the model runs for the three forest regeneration scenarios as a dimensionless quadrant analysis showing how the output hydrograph varies in magnitude and time over peak duration. A "flood" is defined as discharge exceeding approximate bankfull discharge at the catchment outflow, the x-axis in Figure 10.13 shows the change in the length of time that discharge is exceeding bankfull discharge at the outflow compared to the calibration scenario. The quadrant analysis shows the vast majority of model runs (>95%) are located in the lower left quadrant (peak discharge and time over peak reduced) or the upper right quadrant (peak discharge and time over peak increase) and thus result in a broadly linear scaling of the flood hydrograph peak with the peaks retaining a similar shape to the calibration hydrograph. Very few model runs result in a flood which becomes more "flashy" with an increased peak, but shorter duration (upper left quadrant), nor conversely a flood which is smaller in peak discharge but persists over peak for a longer duration than the calibration hydrograph (bottom right quadrant).

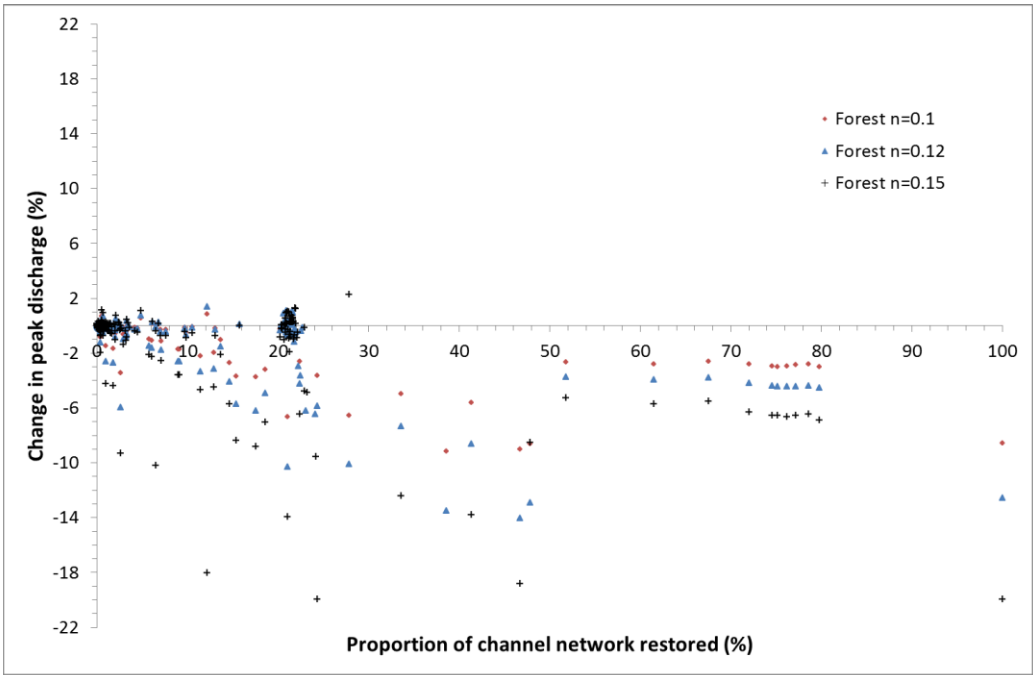


Figure 10.12 – showing the relationship between change in peak discharge and the percentage of the total catchment stream network which has floodplain forest applied to it, for each of the three values of Manning's n.

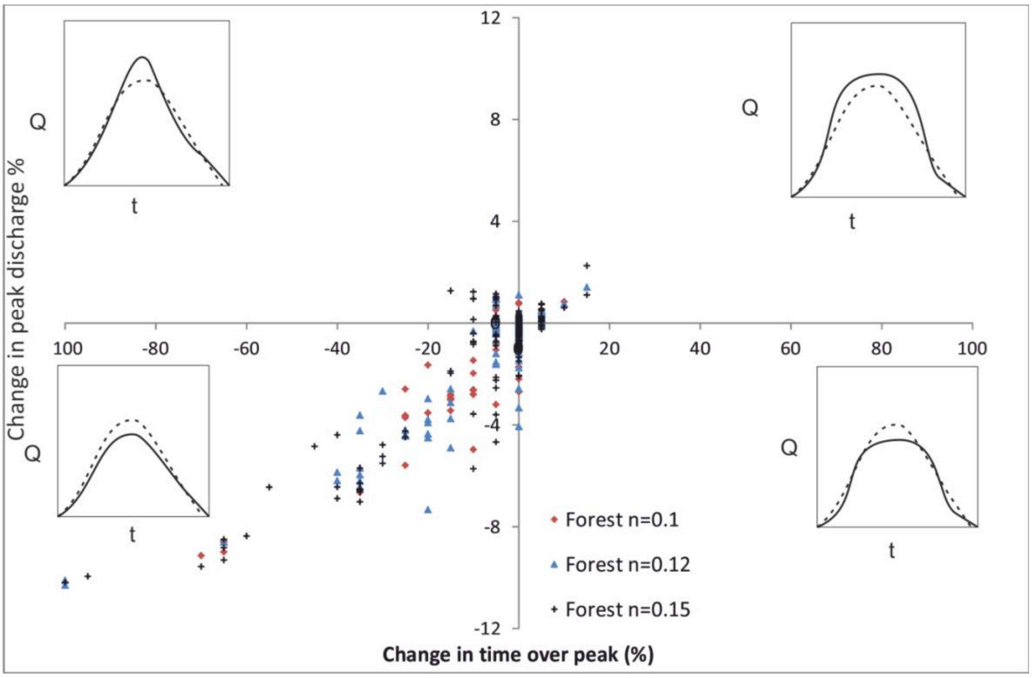


Figure 10.13 – quadrant analysis of change to flood hydrograph for all floodplain forest scenarios, with hydrograph cartoons superimposed in each quadrant. Hashed line in each cartoon is the baseline hydrograph and solid line is the resulting hydrograph.

10.7.3.2. Forest and Channel

Figure 10.14 shows the spatial distribution of directionality in peak discharge changes following forest and channel restoration to individual reaches/segments. The three maps in Figure 10.14 correspond to three scenarios for 25 years, 50 years and 100 years post-restoration. The three maps show similarities in their broad spatial pattern; segments close to the catchment outflow (bottom right in Figure 10.14) tend to led to increases in peak discharge (orange/red), whereas segments which are distant from the outflow tend to lead to decreases in peak discharge (dark blue). As the model moves from 25 years to 100 years post regeneration (Figure 10.14A through Figure 10.14C) the spatial pattern is reinforced, with many more segments in the middle and upper parts of the catchment showing reductions in peak magnitude (coloured dark blue). The number of 'neutral' reaches decreases from Figure 10.14A to Figure 10.14C, with almost all reaches either increasing or decreasing the peak discharge in Figure 10.14C (forest n=0.15, channel n=0.1).

Note model uncertainty for forest model runs is 0.5%, however uncertainty analysis shows no change in directionality for predictions of greater than ±0.1%, i.e. for red and dark blue coloured reaches there is uncertainty around the magnitude of change in peak discharge but a high degree of confidence these reaches do represent increases in peak discharge and decreases in peak discharge respectively.

Unlike Figure 10.10 for the purely forest regeneration Figure 10.14 shows a number of reaches which switch directionality as the hydraulic resistance increases in the 100 year post-restoration scenario, this is most noticeable near the catchment outflow, where around ten reaches shift from a reduction in peak discharge at 50 years (Figure 10.14B) to an increase in peak discharge at 100 years (Figure 10.14C). Figure 10.14 shows a broad pattern of restoration to trunk channels with a Shreve stream order of ~40-150 reducing the peak discharge across the three age scenarios; this is the opposite pattern to the forest-only regeneration scenarios (Figure 10.10) where the majority of these reaches increase peak discharge when restored.

Figure 10.15 show the percentage change in peak discharge for all forest and channel restoration scenarios. This figure shows the majority of model runs lead to a decrease in peak discharge although for smaller areas of restoration (<20%) there is a great deal of variability. The magnitude of change to peak discharge increases with increasing hydraulic resistance in almost all cases, however the relationship between the proportion of the network restored and the magnitude of change is not linear. Figure 10.15 shows a three stage relationship between proportion of channel network restored and change in peak discharge. Between 0-22% of network restored the change in peak discharge is highly variable in both magnitude and directionality, with the majority of model runs showing only modest (<2%) changes in peak discharge, between 22-50% of network restored all model runs lead to decreases in peak discharge, with larger proportions of the channel network leading to greater decreases in peak discharge, this is likely due to de-synchronisation of sub-catchment contributions from the main flood hydrograph. Model runs with over 50% of the catchment network restored all show a decrease in peak discharge, however this is typically more modest than runs with 25%-50% of the network restored; approximately a 3-7% reduction in peak discharge.

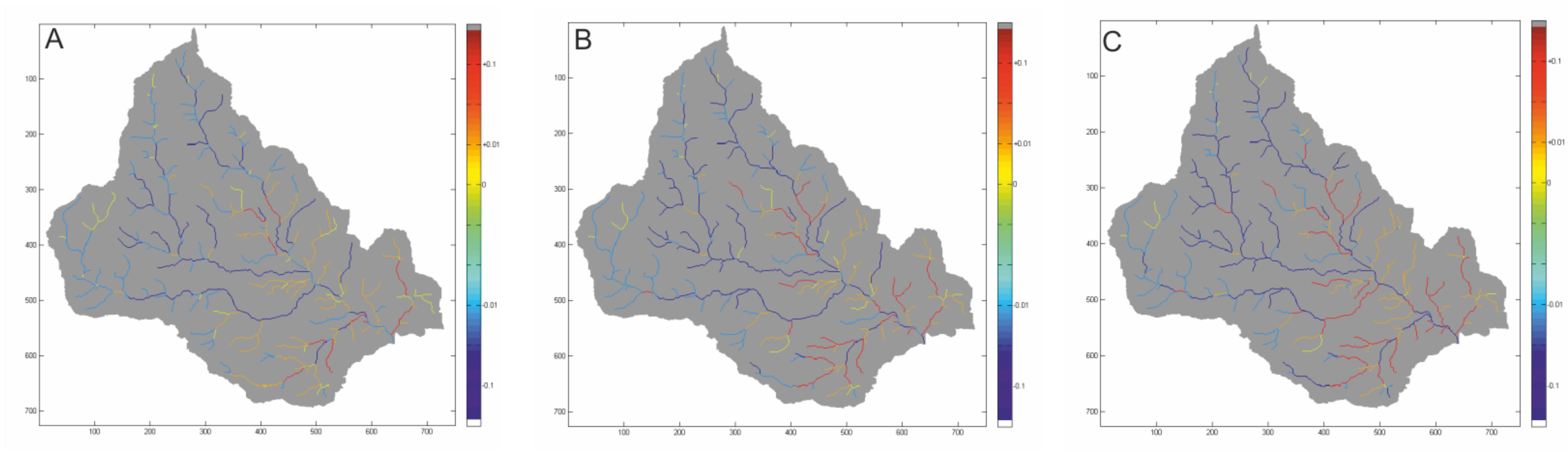


Figure 10.14 maps of the study catchment showing directionality of change to flood peak magnitude with changes to hydraulic resistance for both channel and floodplain applied to a single segment.

A) 25 years post forest regeneration (change to overbank Manning n=0.100 & channel Manning n=0.06 for study reaches), B) 50 years post forest regeneration (overbank Manning n=0.120 & channel Manning n=0.075), C) 100 years post forest regeneration (overbank Manning n=0.150 & channel Manning n=0.100). Reaches coloured yellow show no change to peak discharge, orange and red are increases of >0.01% and >0.1% to peak discharge respectively, light blue and dark blue are decreases of <0.01% and <0.1% respectively.

Figure 10.15 implies restoration which de-synchronises sub-catchments (22-50% of catchment channel network) has a greater effect on reducing peak discharge than more extensive restoration which attenuates the entire flood peak.

Figure 10.16 shows a quadrant analysis of modelled flood peaks for all model runs conducted for all three forest and channel restoration scenarios. As with the forest only restoration model runs (Figure 10.13) this shows the majority of modelled flood peaks are a broad linear scaling of the calibration hydrograph peak. The model runs result in a hydrograph of broadly the same shape with a similar relationship between peak discharge and time over peak compared to the baseline. Unlike Figure 10.13 for the forest only scenarios there are a small number of model runs which display a smaller magnitude peak which persists over threshold for longer than the calibration hydrograph; i.e. the flood peak is lower but lasts longer, there are also a number of model runs which result in such a large reduction in peak discharge (>10%) that the hydrograph is never over the threshold and thus shows a 100% reduction in time over peak.

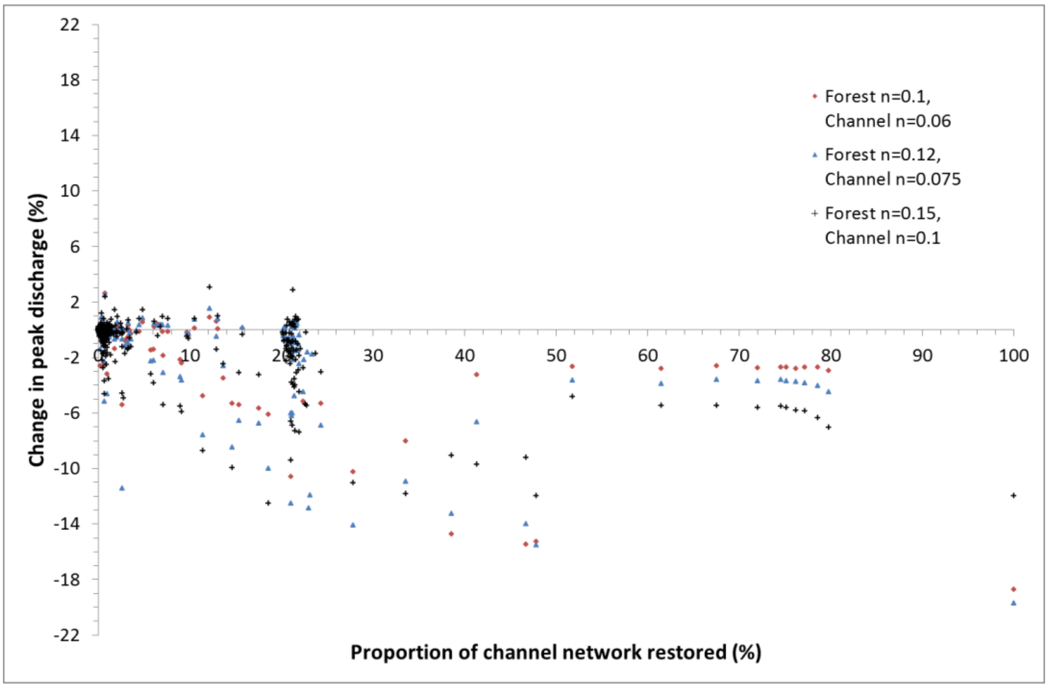


Figure 10.15 –showing the relationship between change in peak discharge and the percentage of the total catchment stream network which has floodplain forest applied to it, for each of the three values of Manning's n.

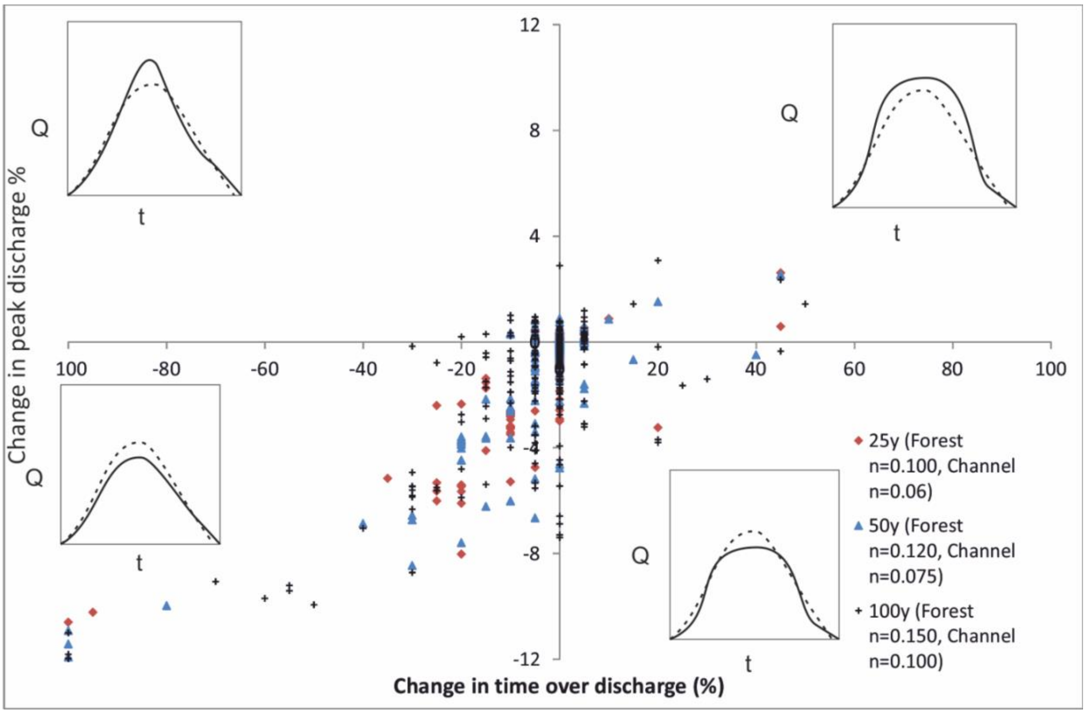


Figure 10.16 – quadrant analysis of change to flood hydrograph for all floodplain forest scenarios, with hydrograph cartoons superimposed in each quadrant. Hashed line in each cartoon is the baseline hydrograph and solid line is the resulting hydrograph.

10.8. Discussion

10.8.1. Key findings

Results from this modelling study demonstrate that spatially distributed restoration of land cover and/or channel wood loadings does affect catchment flood hydrology. Restoration scenarios applied at the reach scale are observable in changes to the catchment scale flood hydrograph, although the majority of reach locations show output responses within the likely margin of model uncertainty. The largest changes in peak discharge occur with restoration scenarios applied to between 20% and 50% of the catchment channel network.

The variability in output hydrograph response to restoration is largest for ELJ spot dams, meaning the catchment response to the insertion of large wood structures alone is hard to predict. For scenarios of riparian forest regeneration, with and without continued channel management, variability in response is much lower than for ELJ spot dams with three distinct groups of model response; 1) up to ~22% of the channel network restored shows a variability in hydrograph response, although few model runs display large changes (>2%) in peak discharge, and many changes are less than model uncertainty (0.5%), 2) between ~23% and 50% of the channel network restored shows large decreases in peak discharge, with the decrease increasing linearly with the proportion of network restored, 3) above 50% the peak discharge is reduced, but this reduction is more modest compared to (2). These three groups can be explained as being due primarily to (1) increasing flood wave travel time through short sections of channel, resulting in modest and variable catchment level hydrograph response, (2) the decoupling of individual sub-catchments from the main flood wave resulting in substantial reductions to peak discharge, (3) with multiple sub-catchments restored decoupling is no longer as effective as (2) as increasing restoration "re-couples" sub-catchments, thus hydrograph response is primarily due to attenuation of the flood wave travelling through restored sections and slowing hillslope run off reaching the channel network.

10.8.2. Uncertainty Analysis

Figure 10.5 for uncertainty in ELJ spot dam model response shows there is no uncertainty in the directionality of model outputs compared to the baseline calibration, although the magnitude of change in peak discharge is subject to uncertainty. Overall uncertainty for ELJ spot dam model runs is ±0.23% in peak discharge; model runs which show a neutral response to ELJ spot dam insertion (between +0.1% and -0.1% change to peak baseline discharge) there is virtually no uncertainty in the model output with less than a 0.01% change in peak discharge with variable Manning's n. This level of uncertainty is greater than the model response for many of the single segment model runs.

Figure 10.6 for uncertainty in forest and channel restoration scenarios shows there is no uncertainty in the directionality of model run output compared to the baseline calibration, as with ELJ spot dams. The overall uncertainty in model output is ±0.5% in peak discharge. For model runs predicting a large decrease in peak discharge (>5%

reduction over baseline) there is a greater uncertainty in magnitude of response; scenario 5 in Figure 10.6 shows for a model run predicating a ~10% reduction there is an individual uncertainty of ±2.5% in magnitude of peak discharge change, although such a large uncertainty is not ideal, it is not a major limitation in model interpretation as there can be high degree of confidence these model runs are leading to large reductions in peak discharge with just the precise magnitude subject to uncertainty.

10.8.3. ELJ Spot Dams

Changes in catchment scale flood hydrology can be observed with ELJ spot dams inserted in single reaches (Figure 10.7, Appendix D), although the majority of model runs show a peak discharge change of less than the model uncertainty (±0.23%). Results show the magnitude of peak discharge change is not proportional to the length or area of catchment to which the restoration is applied, this suggests the location and reach characteristics of where ELJ spot dams are applied is more important that the number of dams inserted. The spatial pattern of flood hydrograph response to reach scale insertion of ELJ spot dams (Figure 10.7) is not clear cut; broadly the greatest reductions in peak discharge occur for main channel reaches in the mid and lower catchment (Shreve stream order greater than 10), headwater streams distant from the catchment outflow point are more likely to be neutral or reduce peak discharge than watercourses nearer the catchment outflow. However, within this broad picture there is high variability.

The pattern of changing sensitivity of peak discharge to reach scale ELJ spot dam insertion can be partly explained by slope (see Appendix D). Reaches with high slopes show little change in peak discharge, such reaches have high stream power on account of the slope and thus the increased hydraulic resistance provided by logjams is outweighed by the increased slope in headwater sections, thus tends towards a kinematic type flood wave which is less susceptible to attenuation by hydraulic resistance (Sholtes and Doyle, 2011). Using the Manning equation:

$$v = \frac{1}{n} R^{2/3} S_o^{1/2} \qquad 10.2$$

the increase in velocity due to an increase of 0.05 in channel Manning's n is the equivalent to the reduction in velocity due to an increase of 0.01 in slope. In this study the increased Manning's n used for ELJ spot dams averaged across the mean cell spacing of dam insertion (3 cells) is n≈0.1, therefore the range of Manning's n between modified and unmodified sections is ≈0.05, the range in slope however is >0.04 indicating at higher slopes the increased hydraulic resistance will not have much, if any, effect on slowing flood wave travel time. Previous studies have also suggested gentle channel slopes result in greatest flood peak attenuation (Sholtes and Doyle, 2011; Wolff and Burges, 1994), the flood wave attenuates rather than translating down the catchment as a kinematic wave with momentum. Sholtes & Doyle (2011) found a slope threshold of approximately 0.001m/m for a transition to a kinematic flood wave; results from this study indicate a higher slope threshold of ~0.005m/m below which single segment restoration can lead to changes in peak magnitude greater than model uncertainty (±0.23% for ELJ spot dams and ±0.5% for forest restoration) (Figure 10.11 and Appendix D) where restoration applied to slopes steeper than 0.005m/m there is little effect on changing peak discharge.

As the proportion of channel network with ELJ spot dams applied to it increases, the variability in flood peak discharge response increases (Figure 10.8); model runs with <5% of the network restored tend to lead to small changes in peak discharge, between 10% and 50% of the network restored the peak discharge shows a response range of -6% to +6%. One possible explanation for this variability is the pattern of floodplain flow paths OVERFLOW calculates for model runs. The Lymington River catchment used for this modelling study has experienced channel modifications in the past (Sear et al., 1998; Tubbs, 2001) and some modified channel paths are not reflected in the topography; when flow accumulation and flow routing algorithms are run for the source topography this generates a channel network which differs from the actual channel network. In parts of the catchment shallow paeleochannels exist on the floodplain (Sear et al., 2010), as a result of previous river straightening (Tubbs, 2001), and perhaps more importantly in parts of the catchment it appears as if channels have been diverted into adjacent sub-catchments by cutting new channel paths. For all these scenarios the overbank Manning's n is spatially uniform (n=0.07), so it is possible in some model runs the ELJ spot dams are forcing more water out of bank, which then forms floodplain flow paths into parts of a paeleochannel network in which the overland flow paths are shorter than the actual modified channel network flow path. These new overbank flow paths, coupled with a relatively low overbank Manning's n=0.07 results in a faster travel time for the overbank proportion of the flood wave than for the actual channel network, which in turn results in an elevation of the peak discharge at the catchment outlet.

10.8.4. Forest Regeneration

The forest regeneration scenarios, both with and without continued channel management, show reach scale changes in riparian land use are detectable in the output flood hydrograph peak discharge, although for reach scale model runs the majority of scenarios either predict little change in peak discharge or small changes which are less than the model uncertainty of 0.5%. Reductions in peak discharge from five sequential reaches are greater than the sum of reductions from individual contributing reaches suggesting the attenuating effects of forest cover on a flood wave are not linearly proportional to the length or area of channel network restored. For larger areas of restoration the attenuation of sub-catchment contribution to the flood hydrograph may be sufficiently delayed to result in it becoming de-synchronised from the main flood wave and thus substantially reducing the overall peak discharge.

The magnitude of change to peak discharge resulting from reach scale restoration depends on the location of the restoration, the extent of the restoration and the slope of the reach. The catchment maps showing the directionality of change (Figures 10.9, 10.16, and 10.20) indicate a strong trend towards reaches in middle and upper catchment tending to result in reductions to the peak discharge. Conversely reaches in sub-catchments with connections to the main channel near to the catchment outflow tend to increase the peak discharge. The spatial pattern of responses can be explained by considering each reach as contributing a proportion of water to the main hydrograph (See Figures 10.24, 10.25 and 10.26). Reaches near to the catchment outflow will contribute water to the early part of the hydrograph, whereas in a sufficiently long rainfall event to approach equilibrium runoff the most distal reaches will be contributing runoff that will arrive at the catchment outflow at, or near the peak discharge (Saghafian and Julien, 1995). Flood wave attenuation through a reach may result in either extension of the contribution hydrograph, where the discharge is spread over a longer timeframe, translation of the hydrograph where the peak magnitude of contribution remains largely the same, but is slowed down so that the contributing peak arrives later, or a combination of the two.

For hydrologically proximal reaches and sub-catchments the contributing flood wave will arrive at the main channel quicker than contributions from the majority of the catchment and therefore this contribution forms part of the rising limb of the main hydrograph and its contributing peak has already passed before the main hydrograph peaks (Figure 10.18). If the contribution from hydrologically proximal reaches is sufficiently translated so the contributing peak arrives at the main channel such that it is now synchronised with the main catchment flood wave this will increase the peak discharge and exacerbate flooding (Figure 10.18). Conversely for hydrologically distal reaches and sub-catchments where runoff normally forms part of the main catchment flood peak discharge, if this contribution is either extended, or translated the main peak discharge will be reduced by the same proportion as the local attenuation (Figure 10.19).

Extension and translation of contributing hydrographs has the largest effect on changing catchment peak discharge when applied at a sub-catchment scale. Model runs representing entire sub-catchment restoration equate to between 17% and 47% of the catchment channel network; the change to peak discharge for these model runs can be as much as 19% for forest regeneration with channel management (Figure 10.12) and 17% for forest regeneration with no channel management (Figure 10.15). For these runs the contribution from the respective sub-catchment has been completely de-coupled from the main peak discharge and results in the substantial reductions observed, these figures are in line with a previous study suggesting up to 20% of variability in peak flood discharge could be due to synchronisation of sub-catchment flood waves (Lane, 2003).

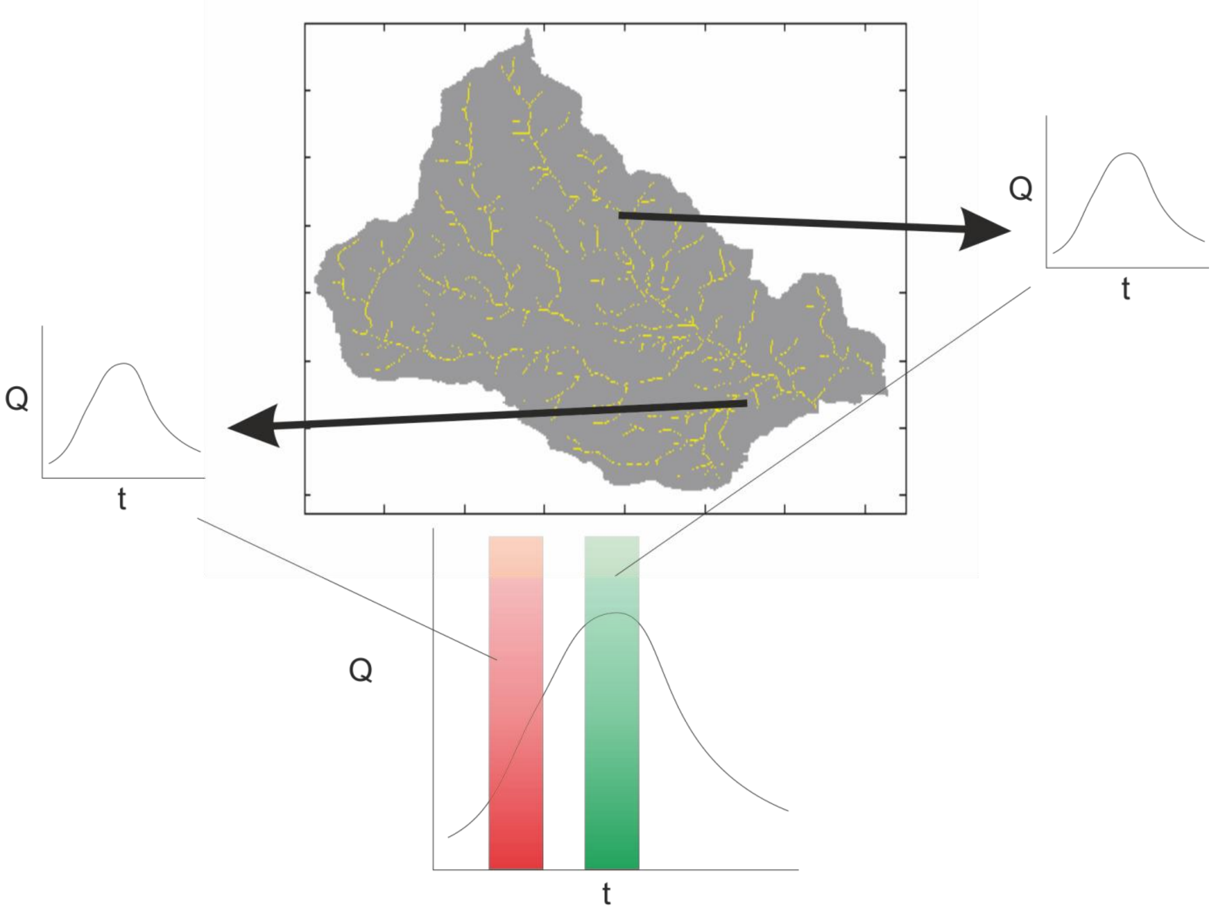


Figure 10.17 – Illustration of effects of changing flood wave travel time in different parts of a catchment.

Each part of the catchment has a runoff contribution which forms part of the catchment outflow storm hydrograph. Due to the travel time taken for sub-catchment contributions to reach the catchment outflow these contributions are manifested in different parts of the storm hydrograph. In this example the runoff from a sub-catchment near to the outflow quickly reaches the catchment outflow and runoff forms part of the rising limb of the hydrograph in the area shaded red. Conversely runoff from a hydrologically distant part of the catchment arrives at the catchment outflow slowly and runoff forms part of the peak discharge of the main catchment hydrograph in the area shaded green.

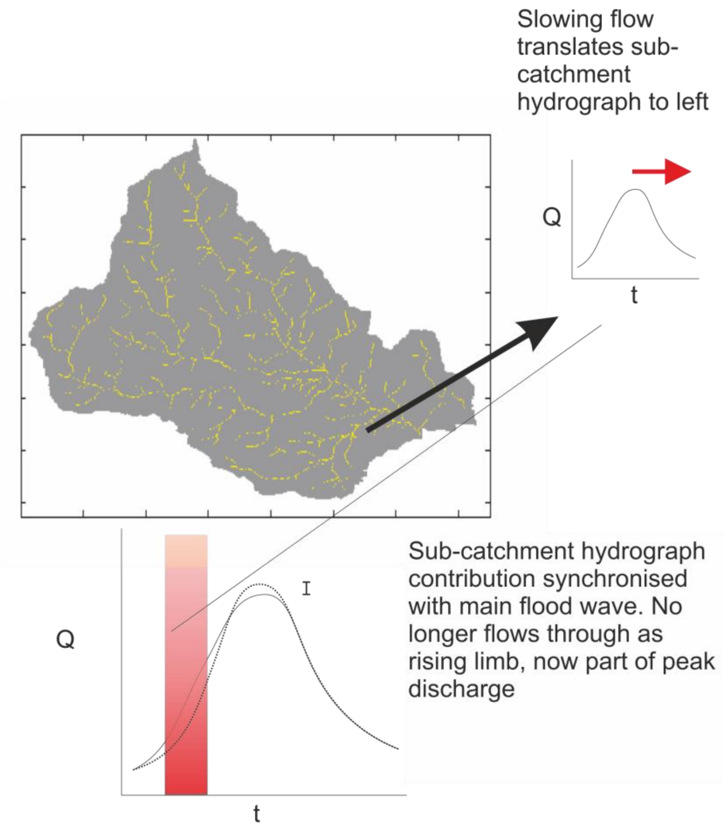


Figure 10.18 – showing the effects on the main catchment outflow of slowing the runoff contribution from a sub-catchment in the lower portion of the catchment.

Flow out of the sub-catchment is slowed, translating the sub-catchment hydrograph to the right. The translation of the sub-catchment contribution means runoff no longer arrives at the catchment outflow in the red shaded area of the main hydrograph, but instead arrives later. The later arrival of runoff synchronises the sub-catchment contribution with the main flood wave and raises peak discharge. The resultant outflow hydrograph is shown as a hashed line, with the original hydrograph as a solid line.

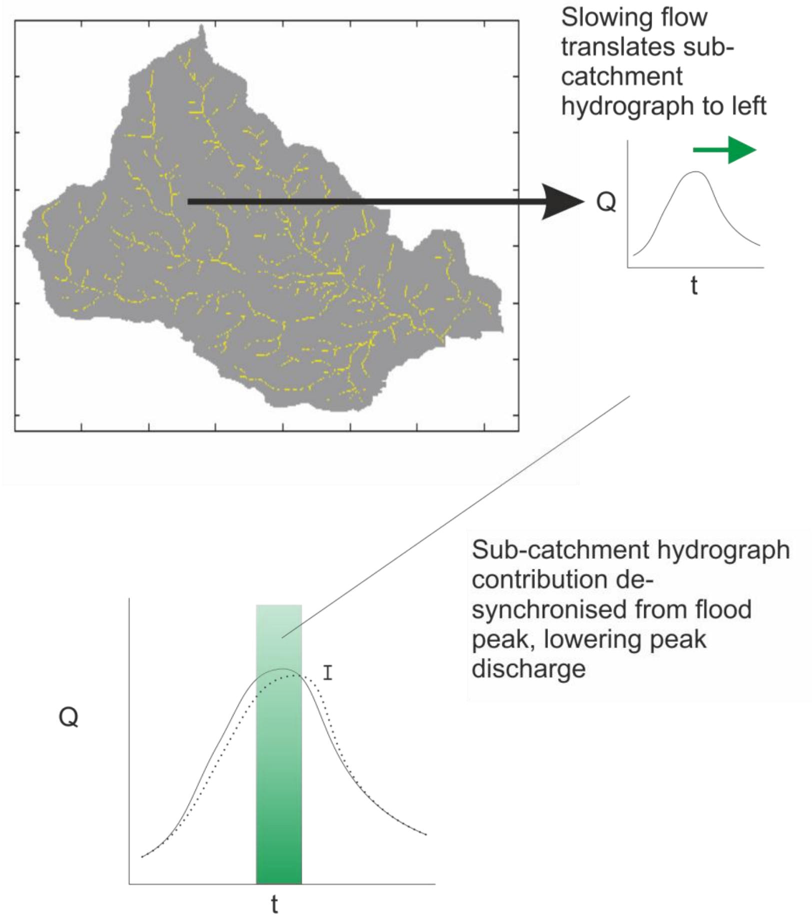


Figure 10.19 – showing the effects on the main catchment outflow of slowing the runoff contribution from a sub-catchment in a hydrologically distant part of the catchment.

Flow out of the sub-catchment is slowed, translating the sub-catchment hydrograph to the right. The translation of the sub-catchment contribution means runoff no longer arrives at the catchment outflow in the green shaded area of the main hydrograph, corresponding the time of peak discharge, but instead arrives later during the falling limb of the hydrograph. The later arrival of runoff de-synchronises the sub-catchment contribution from the main flood wave resulting in reduced peak discharge. The resultant outflow hydrograph is shown as a hashed line, with the original hydrograph as a solid line.

The scenarios used for forest regeneration model runs were based on estimates of the hydraulic resistance resulting from forest succession after 25 years, 50 years and 100 years. Figure 10.12 shows that after only 25 years forest growth, with continued channel management maintaining in-stream large wood at current loadings can have an effect on flood peak magnitudes. Figure 10.12 also shows resistance of Forest n=0.1 applied at a large sub-catchment level (22%-47% of channel network restored) will result in a decrease in peak discharge of ~6%. As forest succession proceeds and the riparian forest becomes more mature and more complex the attenuation of flood peak discharge at the sub-catchment level becomes more pronounced. The pattern of increasing flood wave attenuation with increasing hydraulic resistance is to be expected from the Manning equation, where increasing resistance will lead to lower overbank flow velocities and thus greater travel times for overbank flow, with increased travel times leading to greater extension and translation of sub-catchment hydrograph (Wolff and Burges, 1994).

In scenarios with channel management (i.e. the channel hydraulic resistance remains unchanged from calibration scenario), the spatially invariant in-channel hydraulic resistance means the in channel portion of the flow in a given channel cell remains constant across all model runs. In short the scenarios can only affect flood hydrology if applied in areas which already experience overbank flows. Changes in floodplain hydraulic resistance may result in small changes to individual channel cell discharges between model scenarios and/or additional overbank cells, but these changes are likely to be modest. This helps explain the spatial pattern of flood peak response; where overbank flow is already predicted in the calibration scenario the increase in overbank hydraulic resistance is effective in attenuating the flood wave and reducing peak discharge. Where overbank flows do not already exist the effects of increasing floodplain hydraulic resistance are restricted to attenuating a small proportion of the overall runoff and thus have negligible effects on attenuating outflow peak discharge.

Where channel management is not continued and the loadings of large wood in the channel are allowed to increase as the riparian forest matures the flood hydrology response is slightly different to those scenarios with forest regeneration but continued low levels of in channel large wood loadings. Figure 10.15 shows for the 25 year scenario at the large sub-catchment level (22%-47% of channel network restored) will result in a mean peak discharge reduction of ~10%, compared to ~6% for just forest regeneration alone. For the 50 year scenarios the mean peak discharge reduction is ~12% for forest and channel restoration, compared to ~9% for forest regeneration alone. In the 100 year scenarios the mean reduction in peak discharge is slightly lower at ~10% for forest and channel restoration compared to ~13% for forest regeneration alone, although there is much less variability in response with a range of 6% compared to a range of 17% for forest regeneration alone. In these scenarios both the floodplain and in channel hydraulic resistance are increased through the three succession scenarios. These forest and channel restoration scenarios result in a greater proportion of overbank channel cells and floodplain spill cells due to the reduced channel capacity of channel cells with simulated higher wood loads compared to the calibration model. The combination of higher channel resistance forcing more overbank flow and higher floodplain resistance slowing this overbank flow across the floodplain could be expected to substantially increase flood wave travel times and attenuate peak discharge. The increase in overbank flows extends the spatial effectiveness of this form of restoration in attenuating flood waves; it is not only effective in areas where overbank flows already occur, but can mediate overbank flows in any reach with a sufficiently low slope (less than 0.008m/m, Appendix D).

The findings in this study are important in that they help unravel the complex interplay of river restoration and land cover management at a catchment scale. Results indicate mature, unmanaged deciduous forests can have an important effect in attenuating peak flows, these findings are an important contribution as although previous work has demonstrated the potential of logjams to slow flood wave travel time and reduce flood peaks in small catchments (e.g. Gregory et al., 1985; Thomas and Nisbet, 2007; Thomas and Nisbet, 2012) the effects of forests on flooding remains a hotly debated issue (Van Dijk et al., 2009).

10.8.5. Implications for river restoration

Assuming that the goal of river restoration is to minimise, rather than exacerbate flood risk it is possible to highlight restoration scenarios which are “better” or “worse” in this regard.

Applying engineered logjams to channel reaches of 1-5km of stream length gives highly variable changes to flood peak discharge of ±≤4%. These scenarios can be thought of as “worst” for flood risk management due to the difficulty in predicting the magnitude and directionality of flood response. This finding has very important implications for river restoration and flood risk management. Findings illustrate that the local attenuation effects of logjams (e.g. Gregory et al., 1985; Sholtes and Doyle, 2011; Thomas and Nisbet, 2012) are not always beneficial in reducing peak discharges at the catchment scale. When applied to third and fourth order streams in the middle and lower portions of a catchment engineered logjams are likely to increase flood peak magnitude at downstream locations.

The “best” scenarios over a 25 year time period are restored floodplain forests with associated increases in in-stream large wood loadings, applied to 10-15% of the catchment in headwater sub-catchments. Such scenarios can lead to 5-6% reductions in flood peak magnitude. These scenarios, although representing forest restoration of up to 15 km^2 are of a realistic scale for implementation and are effective in the medium term. Where floodplain forest restoration with associated increases in in-stream wood loadings is extended to 25-35% of the catchment, representing restoration applied to an entire tributary sub-catchment, reduction in peak discharge can be 10-15%. Although this represents a substantial scale of restoration the magnitude of reduction to flood peak discharges necessitates serious consideration is given to such scenarios by land and river managers.

In most cases as the restored forest matures to 50 and 100 years old the magnitude of reduction in peak discharge increases. Reductions to peak flood magnitude of 6-8% are modelled for scenarios representing 50 years of forest growth for 10-15% of the catchment area, compared to a reduction in peak magnitude of 5-6% after 25 years.

There is much interest in incorporating river restoration into an integrated catchment management approach, which includes reducing flood risk (Defra, 2005a). The most widespread form of river restoration is the insertion of engineered logjams (Bernhardt et al., 2005), however the findings of this study suggest these will at best have modest, or no effects on reducing flood peaks downstream, and at worst will increase downstream flood peak magnitude. If a goal of integrated catchment management is to reduce long-term flood risk this study indicates the most promising approach is the use of floodplain forest restoration at the sub-catchment scale, however examples of this type of restoration remain relatively rare (Broadmeadow and Nisbet, 2009; Collins et al., 2012). In order to effectively incorporate river restoration into flood risk management there will need to be a re-evaluation of river restoration techniques and widespread understanding that local effects on flood hydrology can have counter-intuitive effects at the catchment scale.

10.8.6. Limitations

OVERFLOW, and tools like it, which allow spatial relationships and sensitivities to be tested in a computationally efficient way can be a valuable tool in planning river restoration projects. Where there is a limited budget and flood vulnerability is an important consideration OVERFLOW can identify promising sites for delivering flood mitigation as part of river restoration works. Furthermore OVERFLOW can identify areas where restoration works are unsuitable due to a risk of exacerbating existing flood risk. Ideally future use of OVERFLOW for river restoration planning would include a range of calibration flows representing floods from the annual to the 1-in-100 year events. In its existing form OVERFLOW is heavily dependent on expert user knowledge and a bespoke calibration routine for each catchment it is applied to, which is a major limitation making it currently unsuitable for widespread commercial use. General limitations of OVERFLOW are detailed in Chapter 9. A key limitation for the specific application of modelling forest restoration is the inability to directly account for changes in interception loss and increased infiltration into soil with increasingly complex forest cover, which has been hypothesised as a major impact of forests on flood attenuation (McCulloch and Robinson, 1993; Robinson et al., 2003; Van Dijk et al., 2009). Absence of such a function within OVERFLOW may result in underestimates of flood attenuation with increased forest cover.

These findings are for a single, fairly modest flood event. Previous modelling studies at the reach scale have reported that shorter return period flood events are more sensitive to attenuation due to altered hydraulic resistance (Anderson et al., 2006; Sholtes and Doyle, 2011; Woltemade, 1994), and that forests are ineffective in reducing peak flows of large magnitudes (McCulloch and Robinson, 1993). The findings of this study need to be replicated for higher magnitude and longer duration flood events to test the sensitivity of flood peak attenuation to peak magnitude.

10.9. Conclusions

The computationally intensive nature of conventional hydrological models means they are unsuitable for conducting catchment wide modelling exercises involving spatially diffuse land-use scenarios. OVERFLOW has proved to be a useful and insightful method of attempting a first pass at investigating catchment scale land cover/river restoration and flood hydrology interactions. Although not detailed enough to provide spatially explicit, quantitative predictions of flood hydrology given specific land cover scenarios and rainfall events, modelling with OVERFLOW allows broad spatial patterns in behaviour to be identified and indicates which parts of a catchment are most sensitive to changes in land cover and channel restoration.

This study has identified that although increasing large wood loads and floodplain complexity at a reach scale is capable of attenuating local flood waves the effects of this local attenuation at the catchment scale has a high degree of spatial sensitivity and is not always ‘positive’, with respect to reducing downstream flood peaks. Generally, implementing either engineered logjams or floodplain forest regeneration in reaches or sub-catchments which are hydrologically proximal to the main catchment outflow tends to increase outflow flood peaks due to delaying water that would ordinarily pass through the outflow before the main flood wave. Implementing forest regeneration in the middle and upper reaches of a catchment tends to either reduce downstream flood peaks, or have a neutral effect. Where a rainfall event over a small catchment is only of a few hours duration, storm water flowing through middle and upper reaches of a catchment will typically contribute to peak discharge at the catchment outflow. Conversely storm water from tributaries close to the outflow will contribute only to the rising limb of the hydrograph. Delaying water flowing from the middle and upper parts of the catchment will therefore tend to reduce peak discharge at the catchment outflow and lead to a longer tail to the hydrograph.

The most widespread form of river restoration is the insertion of engineered logjams at a reach scale. Results from this study indicated engineered logjams applied to 1-5km of channel length can lead to changes in flood peak magnitude of up to ±4%. Where floodplain land cover is relatively simple and has a low hydraulic resistance, insertion of engineered logjams should not be expected to produce substantial attenuation effects as the overbank flow is unlikely to be substantially slower than that confined to the channel. This finding has important implications for river restoration as it indicates the insertion of engineered logjams cannot be counted on to provide reductions in flood risk at the catchment scale, despite locally attenuating flood waves.

The best river restoration scenarios (e.g. Figure 10.20), balancing practicality of implementation and substantial reductions in flood risk, are the restoration of flood plain forests with associated increases in wood loadings to the channel. Where floodplain forest restoration is conducted at a sub-catchment scale with 10-15% of the catchment area restored (Figure 10.15), reductions of 5-6% in peak discharge can be seen at the catchment outflow after 25 years. Although small changes in flood peak discharge can be observed for all floodplain forest restoration scenarios the largest reductions are for restoration applied at a sub-catchment scale representing around 20-35% of the catchment channel network. For these scenarios reductions in peak magnitude up to 20% are observed due to de-synchronisation of the sub-catchment hydrograph, ensuring the peak of the sub-catchment runoff contribution arrives at the catchment outflow after the main peak has passed.

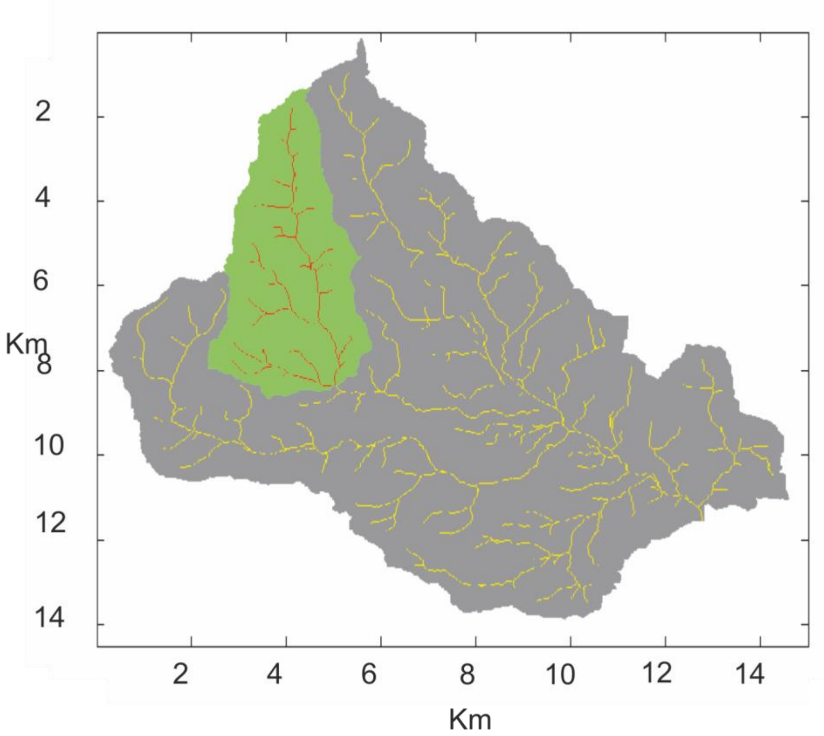


Figure 10.20 – showing a forest restoration scenario in which 14.6% of the total channel network has been restored. The river reaches highlighted in red have had restoration applied to them and the areas in green are the catchment area of restored channels. This scenario leads to a 5.3% reduction in peak discharge after 25 years.

The use of river restoration as part of an integrated flood risk management programme is likely to be ineffective and unpredictable if restricted to insertion of engineered logjams with no accompanying increase in the complexity of floodplain land cover, such as provided by mature riparian forest. Substantial benefits in attenuating downstream flood peaks can be seen by allowing floodplain forest succession at a small sub-catchment scale of 10-15% of the catchment at timescales of as little as 25 years, with increasing flood attenuation through succession up to 100 years. Attenuation of flood peaks is greatest and most predictable with no management of in channel large wood, but benefits are still substantial with mature forest and managed in channel large wood.

These findings should be tested to see if they can be replicated with alternative rainfall and flood events within this study catchment as well as in alternative study locations.

11. Conclusions

The integration of river restoration programmes with flood risk management has been shown to be effective in reducing the peak discharge of moderate flood events at the catchment scale where restoration of floodplain forests is conducted. The insertion of engineered logjams into a channel at a reach scale, typical of works conducted widely as part of river restoration programmes, leads to only small changes in catchment outflow peak discharges. Changes are also of unpredictable magnitude and directionality; particular caution should be exercised in using engineered logjams in the lower portion of catchment networks close to vulnerable urban locations due to the possibility of slowing the flood wave from sub-catchments and synchronising the sub-catchment peak discharge with the main flood wave, thus raising overall peak discharge and potentially exacerbating flood risk. The use of floodplain forest restoration over 20-50% of a catchment on a sub-catchment basis is capable of substantial flood peak attenuation, reducing peak discharge by over 10% within 25 years post-restoration; up to reductions in peak discharge of 20%, 100 years post restoration.

Individual pieces of wood in a small river have been shown to be highly mobile, however the likelihood of mobility and transport distance are correlated with the piece length ($p=0.068$, Odds Ratio=1.56). A piece length of $L^*=1.5$ was found to be a threshold with only 27% of pieces of wood greater than 1.5 channel widths ($n=22$) found to move during a 33 month study compared to 76% of pieces moving across all sizes of log ($n=162$). Where wood is mobilised, logjams downstream have been shown to be effective trapping points. The use of large wood in engineered logjams should take into account the size of wood used relative to the channel dimensions and consider using very large pieces of wood greater than 1.5 channel widths in length to anchor the structure and minimise the risk of wood being transported downstream and causing damage to infrastructure.

11.1. Detailed section conclusions

11.1.1. Logjam survey conclusions

A survey of logjams in the Highland Water, UK, shows that channel and catchment characteristics are not strongly linked to the distribution and frequency of logjam locations. Logjams in this river appear to form predominantly around stochastically distributed wind-thrown pieces of large wood and location is not determined by channel dimensions, sinuosity or distance downstream. The characteristics of the key structural piece which anchors a logjam have been shown to be important in the type of logjam that is formed around it. Key piece size and orientation are identified as the most important factors. Ramp orientated key pieces are strongly associated with channel spanning jams and upright key pieces preferentially form smaller, partial logjams.

By analysing logjam variables it has been shown that the logjam classification system developed by Gregory et al., (1985) has a weak relationship with logjam size ($p<0.001$, $r^2=9.58$), with partial logjams shown to be smaller than other types ($z=4.98-6.15$). The Gregory et al., (1985) logjam classification system shows a strong relationship with logjam function, with channel spanning logjams of active and complete types having much higher odds of having geomorphological features (Odds Ratio=x122.53 and OR=x17.61 respectively) and habitat creation (OR=x166.63 and OR=x7.72 respectively) associated with them than other logjam types. Conversely no statistically significant relationships ($\alpha=0.05$) were found between logjam size and function. These findings indicate Gregory et al., (1985) logjam types are a better predictor of logjam function than logjam size within small forested channel systems.

With these findings river rehabilitation guidelines for more effective habitat creation using logjams in small channels can be proposed. Large Ramp orientated key pieces are identified as being more likely than other types of key piece to create a stable logjam which generates habitat diversity (58% of such key pieces). Such pieces of wood are therefore suitable for 'seeding' a river channel as potential key structural pieces in logjams and trapping points for mobile wood in the channel. Where engineered logjams are inserted into the channel to increase biodiversity or abundance of a large species, these logjams should ideally be channel spanning and induce a step in the water profile, this will maximise the chances of the logjam creating habitat diversity.

Currently there is only sparse information on using logjams to achieve specific ecohydromorphic aims. The conclusions from this study begin to make links between the type and orientation of large wood and logjams and their functions within river channels and thus are valuable for designing subsequent river restoration guidelines.

11.1.2. Large wood mobility conclusions

Large wood in small forest rivers such as the Highland Water is highly mobile with over 75% of tagged pieces moving during a two and a half year study, furthermore transport distances were found to be up to a maximum recorded 5.6km with 5% of pieces (n=162) found to have moved in excess of 0.5km.

The most important factor explaining large wood mobility was found to be dimensionless piece length relative to the channel width, with a dimensionless length over 1.5 leading to progressively lower likelihood of mobility. The piece dimensionless diameter (relative to channel depth), branching complexity, species type and location were found to be important in only some reaches. Although multivariate analyses produced statistically significant models using wood piece characteristics as predictors, the majority of variance in transport distance and likelihood of mobility remains unaccounted for by the variables investigated in this study.

By tracking individual pieces of wood over multiple seasons it has been possible to demonstrate that logjams formed around a stable key piece of large wood can persist for several years and through multiple bankfull discharge events. Furthermore although such logjams may have the same function and structure and may ostensibly appear the same, the component racked pieces of wood are often reworked and are transported out of one logjam and subsequently trapped at another logjam downstream. This observation indicates that in systems where logjams persist at locations over several years there may still be substantial reworking of logjam structure and component pieces of wood.

11.1.3. Hydraulic resistance of logjams conclusions

Hydraulic resistance of logjams during bankfull flow events in a lowland forest channel were calculated from field measurements giving Darcy-Weisbach friction factor (f) ranging from 0.09 to 19.8. With other data published in the literature these values fit into a continuum of increasing hydraulic resistance relative to increasing slope; values calculated are higher than those found in low gradient sand-bed rivers with low slopes (f=0.1-0.6) and lower than found in steep step-pool channels (f=5-380). Data on logjam hydraulic resistance derived from field measurements are rare in the literature and the friction factors calculated are important for parameterising flood modelling studies.

A roughness partitioning methodology shows large wood to be the dominant contributor to hydraulic resistance within this lowland forest river. Wood resistance accounts for 75-98% of total resistance in the presence of logjams causing spill over the front edge of a logjam, or underflow where water is forced under the logjam towards the bed and a median 65% of total resistance overall across all logjams measured. Where spill was only present under certain discharges, not a constant feature of a logjam, mean friction factor was calculated as f=0.3 in the absence of spill and f=3.6 where at least one piece of wood was causing a small spill step. Where underflow was only present under certain discharges mean friction factor was calculated as f=0.6 in the absence of underflow and f=8.7 where flow was forced under a bridging log.

11.1.4. Forest modelling conclusions

Using results from a numerical model of riparian forest growth, broadly validated with data from the literature, a conceptual model of riparian forest growth for one hundred years post forest restoration was developed.

The conceptual model was used to identify three phases of forest development which will have different effects upon hydraulic resistance. The conceptual model was used to inform the selection of hydraulic resistance coefficients representative of in-channel and floodplain land cover complexity for forests of different ages and has wider applicability for application to similar studies.

Using the conceptual model it is possible to conclude that young, early succession stands are dominated by abundant thin stemmed trees with negligible deadwood either on the floodplain or in the river, due to low mortality rates. In later successional phases initial pioneer trees mature to larger trees as an even aged cohort and mortality increases driven primarily by competition. The total number of trees per unit area decreases through this phase compared to early succession. As trees die off the amount of deadwood increases on the floodplain, however smaller deadwood within the river channel is transported away and thus deadwood volumes within the channel increases more slowly than that on the floodplain. In later, mature successional phases, the forest reaches a dynamic equilibrium of live wood biomass, floodplain deadwood and in-stream deadwood. In mature forests there will be a mixture of tree ages and live wood biomass will remain broadly constant through gap-phase regeneration of seedlings upon the death of a large tree. The input rate of new deadwood to the floodplain and the river channel is balanced by the decay rate of existing deadwood.

11.1.5. Overflow hydrological modelling conclusions

Results from numerical modelling indicate that reach-based river restoration, either through riparian forest regeneration or insertion of engineered logjams, is capable of attenuating local flood waves and this can be observed in the catchment outflow hydrograph with changes of between -6.0% and +2.0% in peak discharge. However, the response at the catchment outflow is typically less than the model uncertainty (±0.5%) and displays a high degree of spatial sensitivity. The response at the catchment outflow to local flood wave attenuation is not always “positive” with respect to reducing peak discharge, and can increase peak discharge by up to 2.0%. An increase in peak discharge is observed where restoration slows a sub-catchment flood wave such that the peak discharge from the sub-catchment becomes synchronised with the main catchment flood wave, raising overall peak discharge. Restoration conducted at the multi-reach (1-5km channel length/1-5% of catchment area) up to sub-catchment scale (20-50% of catchment area) show potentially large changes in peak discharge at the catchment outflow, with reductions up to 20% modelled.

Restoration scenarios using forest regeneration showed larger magnitude changes in peak discharge (-19% to +5%) compared to insertion of engineered logjams alone (±7%), furthermore, as forests age the magnitude of change to peak discharge increases. Complex interactions between flood wave contributions from sub-catchments result in a spatial pattern of restoration location and outflow hydrograph response. Restoration scenarios in reaches or sub-catchments hydrologically proximal to the catchment outflow tend to increase outflow peak discharge, or have no noticeable response. Where scenarios increase outflow peak discharge this is attributed to synchronisation of sub-catchment flood waves, such that water arrives from multiple contributing sub-catchments at the same time, increasing peak flow in the main channel. Restoration scenarios applied to the middle and upper parts of the catchment tend to reduce outflow peak discharge.

Insertion of engineered logjams alone produced a variable response, with less clear spatial trends than for forest restoration. OVERFLOW allows the dynamic evolution of floodplain channels during overbank flow; due to a relatively low hydraulic resistance of the floodplain in the absence of floodplain forests, flow forced out of the bank into floodplain channels by additional logjams is subjected to broadly similar hydraulic resistance in floodplain channels (n=0.07) compared to the calibration channel resistance (n=0.05). Similar hydraulic resistance values for channel and non-vegetated floodplains means overbank flow in floodplain channels is unlikely to be substantially slower than that confined to the channel, despite shallower flow depths when compared to flow in vegetated floodplain forest channels (n=0.10-0.15).

Catchment scale integrated flood risk management can include river restoration; however results are likely to be unpredictable and ineffective if restoration is restricted to the use of engineered logjams with no change in floodplain vegetation complexity. Where restoration of floodplain forests is used substantial benefits in attenuating flood peak magnitude (reductions of up to 5%) can be seen in as little as 25 years forest growth, at scales of just 10-15% of the catchment re-forested, with increasing flood peak attenuation as the forest matures (reductions of up to 10% after 100 years).

11.2. Limitations

11.2.1. Logjam survey limitations

A major limitation in a logjam survey methodology is generating data at a single point in time for features which display a high degree of temporal variability. Wood in a river is redistributed during flood events and the porosity of logjams changes seasonally with the delivery of leaves to the stream in autumn, packing of interstitial space and subsequent breakdown of leaf packs. In order to address this potential variability several resurveys could have been undertaken to establish how logjams within the study area change seasonally.

Logjam classification systems remain highly subjective and in the case of the Gregory et al., (1985) system classification can change with rising and falling stage. Furthermore the key piece position classification used (Table 5.1) can become subjective in river channels which do not fit an idealised box-shaped or trapezoidal channel cross-section; for example a piece of wood at a meander bend with one end on the top of the outer bank and the other resting on a gravel point bar could be classified as ramp or bridge depending on river stage. These limitations were addressed by using consistent criteria in classification, based on a pilot study, but care is needed in extrapolating conclusions based on classification systems to other river types and to data collected in other studies.

11.2.2. Large wood mobility limitations

The primary limitations in any study using tagged logs are the coarse temporal resolution of data and difficulties in relocating tagged pieces. Pieces of wood in rivers move in a staggered conveyor belt motion down reaches and often become dislodged from one logjam to be trapped by the next logjam downstream. Furthermore a moderate event may prime a piece of wood for mobilisation by reorienting it relative to the flow, or by eroding sediment which is anchoring the piece, before a subsequent larger event mobilises it a substantial distance. By only capturing location every 6 months these subtle inter-event processes and the potential importance of location in mobilisation can be missed. Of logs that moved, 70% were recovered; this is a comparable fraction to other published studies using similar methodologies which range from 26-89% recovery rates, but still represents a large unknown proportion of mobility. Using radio tags, or another remote method of tracking mobile wood, would eliminate many of the problems of these two limitations, albeit at greater experimental cost and greater data processing time due to a greater volume of data collected at a higher temporal resolution.

11.2.3. Hydraulic roughness limitations

Collecting field data to calculate hydraulic roughness in the field proved problematic given that hydraulic roughness is predicted to vary with discharge and that roughness values for near to bankfull were needed in order to parameterise future hydraulic modelling. Data collection was only possible for five logjams, over three high discharge events; data collection could have been supplemented using a scaled flume approach.

Previous studies have highlighted the subjective nature of partitioning hydraulic roughness, particularly in a tendency to overestimate any unmeasured roughness fractions; in this study the roughness contribution due to large wood. Limitations in hydraulic roughness partitioning were addressed by cautious, comparative use of calculated values to primarily contrast the importance of large wood roughness to total hydraulic resistance between logjams and between events.

11.2.4. Conceptual forest growth model limitations

The conceptual model of riparian forest growth is limited in that it is derived from a variety of data sources from different continents; it therefore must be applied cautiously to any particular locations, such as the UK. Tree species represented within the NE-CWD model are different from tree species in the Southern UK; however NE-CWD treats tree species as having the same growth, death and decay rates within genus or sub-genus. Given that UK tree species of interest are the same genus as trees included in NE-CWD this difference in species is not seen as a major limitation within the context of the original model.

The NE-CWD growth model was designed to simulate forest growth in the North-eastern states of the USA and so quantitative predictions are unlikely to be correct for other geographical locations. However parameters which have been shown to be affected by climate, such as tree growth, mortality and decay have also been shown in the literature to display inter-regional and inter-plot order of magnitude variability. Such variability indicates that regional climatic variables are unlikely to be the primary driver of growth, death and decay rates in trees. Given that forest growth models are by necessity an averaging of predictions across a wide range of variability and given the similarities in the climate of North-eastern USA and Southern UK, it was considered that results from NE-CWD could be used cautiously as conceptual data on how a riparian forest develops over time.

In light of the limitations discussed above it was determined that NE-CWD provided a useful insight into the process of riparian forest development, over and above individual values for biomass and deadwood reported in the literature. With cautious use and interpretation it was determined that output from NE-CWD could make a valuable input into the development of a conceptual model.

The conceptual model needs to be critically tested against unmanaged broadleaf riparian forest plots of varying ages in order to establish the accuracy of forest composition projections and the development of forest composition over time.

11.2.5. OVERFLOW hydrological modelling limitations

The principle limitation of modelling using OVERFLOW is the uncertainty in quantitative predictions of changes in flood peak magnitude, which are calculated as ±0.5%. This limitation means OVERFLOW output needs to be treated cautiously, particularly in relation to absolute changes in flood peak magnitude. An initial uncertainty analysis indicated that with hydraulic resistance values used for model calibration, uncertainty in input values leads to an uncertainty in magnitude of flood peak response (±0.23% for engineered logjam scenarios and ±0.50% for forest restoration scenarios) but there is no uncertainty in directionality of flood peak response. It is therefore possible to say what the response of the catchment hydrograph flood peak is to a particular modelling scenario, but the magnitude of flood peak response is subject to the above degree of uncertainty. Ideally scenarios could be identified using OVERFLOW shown to be “best” at reducing flood peaks and then a limited number of these scenarios modelling using a more computationally intensive program such as ISIS, however this follow-up work was beyond the scope of the project due to time constraints.

A further limitation of modelling flood response to land cover change, or channel hydraulic resistance with any modelling approach is that the flood response is likely to vary with different magnitude events, and also with different types of events (short, intense rainfall as opposed to prolonged low intensity rainfall). Therefore it is only possible to say with a degree of confidence what the flood response is to modelling scenarios, given the specific rainfall event modelled. In order to explore the effect of land cover changes in general it would be necessary to repeat the modelling exercise using a range of flood magnitudes. Such an approach was not taken for two reasons; the first is the amount of work necessary in order to calibrate OVERFLOW to a particular rainfall-discharge event. Although using a model in which the bulk of work is "front loaded" as part of a lengthy calibration routine allows a wide range of land cover scenarios to be explored with relative ease, it does mean setting up OVERFLOW for a range of events requires each event to be calibrated as a separate model; this was beyond the scope of the study. Secondly due to a problem with a weather station no rainfall data was collected for the New Forest from 2006 to 2013, so there were only a limited number of flood events which could be used to calibrate OVERFLOW. Between the end of significant restoration works in the New Forest in 2005 and the end of the data series in 2006 there were only a handful of suitable, gauged events and the largest of these was used as the calibration event for the modelling in this study.

11.3. Recommendations for future study

Information from the review of large wood in chapter 4.4.5 on logjam classification systems and data presented in chapter 5 on logjam function, demonstrates there is a need for a method of classifying logjams linking their form and function. At present there is not a scale invariant method applicable to all river systems and this hampers attempts at comparing and contrasting the influence of logjams on fluvial processes in different types of rivers. Despite numerous studies of logjams in the literature, development of holistic theories is hampered by a lack of common metrics. As noted by Wohl et al., (2010), there is a need for more data on logjam form and function reported in either common metrics or with dimensionless units to enable the development of classification systems.

Knowledge of long term log mobility in streams is very sparse and there is a great need for more, long-term data sets. The tagging study presented in chapter 6 has been set up in such a way that existing tagged logs can be resurveyed over coming years to develop a long term data set of mobility for this river. Ideally this study should be continued and should involve tagging new pieces of wood which enter the study reaches, as has been done so far. In this way not only will a valuable long term data set on mobility be developed, but estimates can be made on residence times of large wood as well as annual loading rates.

There remains a need for more direct field measurements of hydraulic resistance in the presence of logjams. The work in this thesis along with data from Kitts (2011) forms the basis of a data set of hydraulic resistance of logjams within the Highland Water. Field measurements would need to be taken across a small number of logjams for a variety of discharges up to and exceeding bankfull discharge to examine hydraulic resistance in the presence of flow bypass. This would enable more sophisticated flood hydrology models to be developed which can be parameterised with depth dependent roughness coefficients. Furthermore, measurements are needed across a greater variety of logjams in order to quantify the variance in hydraulic resistance between logjams of similar size and type. Given the logistical requirements in such an operation it would be useful to explore flume or computational fluid dynamics approaches to simulating natural logjams.

The results from hydrological modelling presented in this thesis begin to illustrate the effects upon flood hydrology of spatially distributed land cover changes arising from river restoration. The results need to be extended to a range of rainfall event magnitudes. By modelling a range of event magnitudes it will be possible to show whether land cover effects on flood hydrology are more or less effective for larger flood events. There is also scope to investigate land use effects at larger spatial scales by modelling using OVERFLOW on catchments >100km^2. Larger scale modelling will be able to explore whether sub-catchment de-synchronisation effects demonstrated here are scale invariant or are only observed in smaller catchments. Finally a comprehensive hydrological study should be undertaken using a more physically based hydrological model to investigate a small number of scenarios identified through OVERFLOW to provide quantitative estimates of the effects of model scenarios on flood hydrology.

Appendix A – Literature values for Beech Wood biomass

Forest Reserve Name	Country	Forest Type	Minimum Forest Age	Living Wood Biomass (m3/ha)	Log Biomass (M3/ha)	Snag Biomass (m3/ha)	Total Deadwood Biomass (m3/ha)	Deadwood:live wood ratio (%)	Reference
Dobra	Austria	F5a	60	582			45	8	Mayer & Reimoser, 1978
Rothwald	Austria	F5b	77	547	92	164	256	54	Mayer & Neumann, 1981
Zoinenwould	Belgium	F5a	7	794	19	123	142	17	De Keersmaeker et al, 2002
Boubin	Czech Republic	F5b	138	772	74	185	258	30	Vrska et al, 2001c
Milesice	Czech Republic	F5b	48	567	52	101	153	24	Vrska et al, 2001b
Mionsi	Czech Republic	F5a	61	590	63	108	172	26	Vrska et al, 2000b
Polom	Czech Republic	F5b	40	593	49	104	152	23	Vrska et al, 2000a
Razula	Czech Republic	F5b	62	592	89	199	287	35	Vrska et al, 2001a
Salajka	Czech Republic	F5b	38	473	89	159	248	47	Vrska, 1998

Stozec	Czech Republic	F5b	0	663			63	9	Prusa, 1982, 1985
V Kluci	Czech Republic	F5a	47	681	54	169	223	30	Odehnalova, 2001
Zakova hora	Czech Republic	F5b	62	580	33	114	147	23	Vrska et al, 1999
Zofin	Czech Republic	F5b	137	666	54	87	141	19	Prusa, 1982, 1985
Knagerne	Denmark	F5a	13	449	31	56	87	20	Christensen and Hahn, unpublished
Mons Klinteskov	Denmark	F5a	66	201	24	48	73	37	Christensen and Hahn, unpublished
Strodam Reservatet	Denmark	F5a	34	490	38	101	139	29	Christensen and Hahn, unpublished
Suserup Skov	Denmark	F5a	75	674	9	154	163	25	Christensen and Hahn, unpublished
Velling	Denmark	F5a	11	489	31	68	99	21	Christensen and Hahn, unpublished
La Massane	France	F5a	78		8	25	33		Garrigue and Magdalou, 2000
La Tillaie	France	F5a	147	260	55	165	220	85	Wijdeven, 2003
Bw Birkenkopf	Germany	F5a	2	333	2	8	10	3	Labudda, 1999b
Bw Feldseewald	Germany	F5b	19	423	39	23	62	14	Labudda, 2000
Bw Grubenhau	Germany	F5a	27	604	22	47	69	11	Labudda, 1999a
Bw Napf	Germany	F5b	26	483	82	32	114	23	Hanke, 1998
Bw Pfannenberg	Germany	F5a	8	469	17	51	69	14	Seiler, 2001
Bw Sommerberg	Germany	F5a	1	333	5	16	22	6	Wotke and Bucking, 1999
Bw Zweribach	Germany	F5b	29	538	22	43	65	12	Keller and Riedel, 2000

Eisgraben	Germany	F5a	0	774	39	142	181	23	Kobel, 1999
Fauler Ort	Germany	F5a	62	481	104	156	260	47	Winter, Unpublished
Franzhorn	Germany	F5a	25	584	4	15	19	3	Meyer, Unpublished
Gitschger	Germany	F5a	0	640	42	96	138	21	Kobel, 1999
Grosser Stauenberg	Germany	F5a	27	545	4	24	28	5	Meyer, Unpublished
Hainich	Germany	F5a	0	567	22	42	64	11	Beneke and Manning, 2003
Heiligen Hallen	Germany	F5a	61	506	74	211	284	48	Winter, Unpublished
Hoher Knuck	Germany	F5a	13	576	16	81	97	16	Kobel, 1999
Hoxfels	Germany	F5a	28	360	46	10	56	15	Heupel, 2002
Hunstollen	Germany	F5a	26	576	5	16	21	4	Meyer, 1999
Kalkberg	Germany	F5a	13	681	10	28	38	5	Kobel, 1999
Koningsbuche	Germany	F5a	24	611	18	62	79	13	Meyer, 1999
Limker Strang	Germany	F5a	27	496	7	18	25	5	Meyer, 1999
Lohn	Germany	F5a	24	458	1	41	42	9	Meyer, 1999
Lussberg	Germany	F5a	27	542	2	8	9	3	Meyer, 1999
Niddahange	Germany	F5a	34	542	5	44	49	7	Hocke, 1996
Niddahange 2	Germany	F5a	34	599	6	35	41	5	Hocke, 1996
Platzer Kuppe	Germany	F5a	13	595	24	35	58	10	Kolbel, 1999
Serrahn	Germany	F5a	22	458	45	113	158	29	Winter, Unpublished
Stoberhai	Germany	F5a	30	622	21	36	57	9	Meyer, Unpublished
Swarzwihrberg	Germany	F5a	13	876	13	61	75	8	Kolbel, 1999
Vilm	Germany	F5a	61	561	44	109	153	27	Schmaltz and Lange, 1999

Volgelherd	Germany	F5a	24	439	3	24	27	6	Meyer, 1999
Volgelherd 2	Germany	F5a	27	478	3	41	44	9	Meyer, Unpublished
Waldhaus	Germany	F5a	13	780	6	118	124	16	Kolbel, 1999
Alsohegy	Hungary	F5a	23	284	17	23	40	14	Odor & Standovar, unpublished
Oserdo	Hungary	F5a	25	765	23	152	175	21	Odor & Standovar, unpublished
Kekes	Hungary	F5a	15	454	14	92	106	22	Odor & Standovar, unpublished
Dassenberg	Netherlands	F5a	10	402	18	43	61	16	Van Hees et al, 2004
Gortel	Netherlands	F5a	10	507	8	56	65	13	Van Hees et al, 2004
Pijpebrandje	Netherlands	F5a	25	457	11	32	43	10	Van Hees et al, 2004
Weversbergen	Netherlands	F5a	9	469	1	46	48	11	Van Hees et al, 2004
BarbisGoraNP	Poland	F5b	42	537	99	168	267	50	Jaworski and Paluch, 2001
Bieszczady	Poland	F5a	25	596	34	148	182	31	Jaworski et al, 2002
Gorce NP	Poland	F5b	10	683	71	99	169	24	Jaworski and Skrzyszewski, 1995
Swietokrzyski NP	Poland	F5b	68	362	144	152	296	78	Jaworski et al, 1999
Badlin	Slovakia	F5b	84	627	42	228	271	46	Saniga, 1999, Saniga and Schutz, 2001b
Dobrec	Slovakia	F5b	85	741	66	190	256	41	Saniga and Schutz, 2001b
Havesova	Slovakia	F5a	35	736	32	70	103	17	Saniga and Schutz, 2001a
Kyjov	Slovakia	F5a	19	465	47	115	162	42	Korpel, 1995, Saniga and Schutz, 2001a
Rastun	Slovakia	F5a	0	527	28	31	58	13	Korpel, 1992, 1997
Rozok	Slovakia	F5a	35	816	28	96	124	18	Saniga and Schutz, 2001a
Sitno	Slovakia	F5a	0	594	24	62	86	17	Korpel, 1997
Stuzica 4	Slovakia	F5a	26	569	51	40	91	19	Korpel, 1997
Stuzica 5	Slovakia	F5a	26	647	50	40	90	17	Korpel, 1997

Bukov vrh	Slovenia	F5b	15	525	65		92	18	Kovac, 1999
Krokar	Slovenia	F5b	0	634			69	11	Papez et al, 1997
Pecka	Slovenia	F5b	46	687	283	269	552	83	Debeljak, 1999
Rajhenavski Rog	Slovenia	F5b	91	813	119	16	134	17	Hartman, 1987
Strmec	Slovenia	F5b	88	660			166	25	Rozenbergar et al, 2003
Neunkirch	Switzerland	F5a	49	470	6	51	58	13	M. Dobbertin pers comm
Cozzo Ferriero	Italy	F5a		1383.3	37.076		71.3		Lombardi et al, 2010
Monte Sacro	Italy	F5a		469.3	31.108		70.7		Lombardi et al, 2010
Val Cervara	Italy	F5a		363.6	87.23		143		Lombardi et al, 2010
Monte de Mezzo	Italy	F5a		702.5	3.71		26.5		Lombardi et al, 2010
Monti Cimini	Italy	F5a		783.8	15.504		32.3		Lombardi et al, 2010
Fonte Novello	Italy	F5a		1030.3	18.669		88.9		Lombardi et al, 2010
Sasso Fratino	Italy	F5a		1189.1	48.975		65.3		Lombardi et al, 2010
Gargano Pavari	Italy	F5a		666.3	18.795		89.5		Lombardi et al, 2010
Fosso Cecita	Italy	Pine		583.9	0.16		2		Lombardi et al, 2010
Abeti Soparni	Italy	Fir		569.8	4.78		95.6		Lombardi et al, 2010
Collemelluccio	Italy	Fir		557.8	2.958		17.4		Lombardi et al, 2010
Buckholt Wood	UK	F5a	24		3	3	6	51	Mountford, 2003
Dendles Wood	UK	F5a	33		61	109	170		Mountford et al, 2001
Denny Inclosure	UK	F5a	41		78	195	274		Mountford et al, 1999
Lady Park Wood	UK	F5a	49		28	53	81		Green and Peterken, 1997
Noar Hill Hanger	UK	F5a	13		40	300	340		Mountford, in press
Ridge Hanger	UK	F5a	14		1	264	265		Mountford and Ball, in press

The Mens	UK	F5a	31	28	85	113	Mountford and Peterken, 2001
Toy's Hill	UK	F5a	12	30	456	486	Mountford and Peterken, 2000
	Italy					40	Lombardi et al, 2011
	Italy					20	Lombardi et al, 2011
	Italy					22	Lombardi et al, 2011
	Slovenia					19	Lombardi et al, 2011
	Slovenia					145	Lombardi et al, 2011
	Switzerland					22.7	Bretz Guby and Dobbertin, 1996
	Switzerland					13.4	Bretz Guby and Dobbertin, 1996
	UK					12	Kirby et al, 1993
	UK					23	Kirby et al, 1993
	Switzerland					4	Lemba, 1996
	Finland					32	Siitonen, 1994
	Sweden					7	Albrecht, 1991
	Poland					94	Kirby et al, 1991
	Germany					50	Albrecht, 1991
	Germany					200	Albrecht, 1991
	Germany					15	Detsch et al, 1994
	Germany					54	Detsch et al, 1994
	Spain					24.99	Gill, 2010
	Spain					14.42	Gill, 2010
	Spain					44.33	Gill, 2010
	Spain					12.36	Gill, 2010

Spain	9.82	Gill, 2010
Spain	9.63	Gill, 2010
Spain	3.41	Gill, 2010
Spain	16.99	Gill, 2010

Appendix B – Draft OVERFLOW paper

The following paper is in preparation by Nick Odoni and Stuart Lane as a paper introducing the model OVERFLOW into the scientific literature. It has been referenced in the thesis text where relevant.

(Odoni and Lane, in prep).

OVERLOW 1: development of a spatially-distributed unit hydrograph method for testing diffuse land management interventions

Abstract:

OVERFLOW has been developed as an exploratory model to demonstrate the effects of different channel and catchment intervention measures on flooding in rural areas caused by major rainfall events. The results generated by OVERFLOW are intended to be used as guidance to both practitioners and non-experts as to what types of measures - and particularly what spatial arrangements of them - should be studied in detail using the conventional, more strongly physically-based models commonly applied to flooding problems. These intervention measures include, with respect to channels, changes in channel depth, width, sinuosity, roughness, and associated floodplain roughness, the latter two implemented in OVERFLOW by simple adjustments to Manning's '*n*' values. Channel roughness changes are also used to simulate the incorporation of flow delay structures, such as large woody debris dams, in the streams. Similarly, the wider catchment interventions in OVERFLOW can potentially include changes in planting and cropping, building of ponds, digging or blocking of drains and ditches, planting of hedgerows, and the incorporation of low level bunds as temporary water storage zones across the floodplain or as smaller storage interventions in areas nearer the channel heads. The model uses simplified representations of land surface cover and channel (hydraulic) geometry in order to allow rapid inclusion in the model of any number and combination of the interventions potentially of interest in a particular catchment. This facility in using the model enables the significance of the spatial arrangements of those interventions, as a means to reduce flood risk, to be demonstrated and thus the more optimal solutions, both of intervention type and their spatial arrangement, to be identified. The underlying model

simplifications of the hydrology and hydraulics are achieved by calculating flows according to a sequence of 'time maps', which are themselves first calculated from a set of hypothesised rainfall rates covering the range of those observed during a known rainfall event or events that have caused a major flood. In calibration of the model, the time maps are used in a temporal order that follows broadly the applied gauged rainfall, and the order is further adjusted to achieve high agreement between the observed and modelled discharges at one or more reference points of interest in the channel network. Once calibrated, the same time map order is then applied to the catchment when using the model to explore how different intervention arrangements affect the flood hydrograph. Each intervention case result is compared with the base modelled result in order to assess the efficacy or otherwise of the intervention case as a potential flood reduction solution. Evaporation, losses to groundwater and baseflow effects are also taken into account in a highly simplified way in the model set up

and calibration. OVERFLOW is presently intended to be applied to flood related problems affecting small to medium-sized catchments (up to 150 km^2), at spatial resolutions 10-50 m^2, although work is in hand to adapt the model for application to larger catchments. Here we present a complete model description and the calibration of the model for the catchment of Pickering Beck, North Yorkshire, for two major flood events (2007 and 2000). Application of the model to simulate these floods in Pickering Beck and the results generated by exploring different combinations for flood intervention measures are explained in part 2.

INTRODUCTION

There is considerable interest in the possibility that diffuse interventions in river catchments might provide an alternative methodology for reducing downstream flood risk. Field measurements at quite small scales (between 1 m^2 and 1 km^2) have confirmed that land management can impact locally upon the amount of runoff generation and its speed of transfer over the land surface (e.g. Heathwaite *et al.*, 1990; Marshall *et al.*, 2009). Small scale interventions have been shown to have considerable local benefits in reducing peak river flows during extreme events and such interventions have included: (1) small ponds (e.g. Wilkinson *et al.*, 2008); (2) localised tree planting and restrictions on livestock grazing (e.g. Marshall *et al.*, 2009); (3) field-scale land use change such as replacing arable cover with grass land (e.g. Boardman *et al.*, 2003).

However, there remains considerable uncertainty over the extent to which such small scale benefits might scale up to have larger scale impacts (O'Connell *et al.*, 2007). Further, if many interventions are introduced, each having beneficial local impacts upon the reduction of peak river flows, the interventions change the relative timings of sub-catchment responses and, possibly, increase peak river flows downstream. Indeed, statistical analysis has shown that in larger river basins (> 10 km_2), the relative timing of tributary peaks with respect to the main channel may explain between 10 and 20% of the variance in downstream peak river flow magnitudes (e.g. Lane, 2003). Even measures that are locally beneficial may be problematic at larger scales and need to be properly evaluated in a catchment-scale framework.

Unfortunately, this presents a major challenge for hydrological modelling. First, if an intervention produces with downstream impacts that depend upon the response of other parts of the catchment, the analysis needs to be spatially explicit. This obviates the use of effective runoff modelling tools such as those that use width functions or instantaneous unit hydrographs (Olivera and Maidment, 1999; Liu *et al.*, 2003). Second, there may be many possible types of interventions that might be considered in many possible different locations. Consider the following: (1) N possible types of interventions; (2) m possible stream reaches, where a reach is defined as either an order 1 stream or any subsequent stream segment bounded by an upstream and a downstream confluence; and (3) l possible intensities of intervention (e.g. densities of riparian woodland planting). Assessing all possible combinations of intervention would require $(l \times N)_m$ simulations which given that m might be expected to be > 100, represents an impossible computational problem unless either: (1) methods for reducing the number of simulations; or (2) computational simplicities; can be found. Third, many of the interventions have small-scale local impacts that can be sensitively dependent upon the detailed local characteristics of the system such as soil depth or channel width and depth. These are not necessarily measurable over large spatial scales with a sufficient spatial resolution and may need to be estimated or inferred from other variables. They may therefore be highly uncertain and this has implications for model complexity: there is little to be gained from having a model whose complexity necessitates data that are not available or a data precision that is not achievable.

Given these three constraints, this paper seeks to develop a reduced complexity hydrological model that could allow the testing of multiple, distributed interventions in river catchments with the aim of reducing downstream flood risk. Its companion paper (Odoni *et al.*, in review) explores the application of this model to flood risk reduction using channel and riparian interventions.

Model Conceptualisation: a spatially-distributed unit hydrograph treatment with time dependent flow path evolution

The process by which complexity was reduced is an integral part of the model reported. Model development was undertaken as part of a wider project concerned with undertaking flood risk research in a collaborative collective where both academic natural and social scientists worked alongside local people to develop new flood risk reducing strategies (Whatmore, 2009; Odoni and Lane, 2010; Lane *et al.*, 2011). The particular focus of this project was rural areas in England which, under England's Department of the Environment, Food and Rural Affairs prioritisation policies, were unlikely to receive conventional flood defence investment. Thus, the model conceptualisation, and its associated complexity, was grounded in both: (1) those interventions that the collective decided might be feasible in the study catchments and which they felt merited further testing; and (2) the characteristics of hydrological response for the chosen catchment as perceived by local people and as shown in available rainfall and river discharge data for the study catchments. The conceptualisation in this paper focuses upon two types of intervention identified as feasible: (1) installation of woody debris dams within higher order streams to reconnect those streams with their floodplains; and (2) floodplain woodland expansion; both expected to lead to greater hydrological attenuation. Both interventions were expected: (1) to have benefits that were sensitive to where they were located in the catchment; (2) to require potentially many interventions to have a significant impact; and (3) to change the interactions, through relative timing effects, of runoff generation from different parts of the catchment. Thus, the model had to be both spatially-distributed and time dependent but also, to allow testing of different combinations of interventions in different locations, computationally efficient. Although these are the focus of this paper, there are other interventions that could readily be incorporated into the same model framework (e.g. stream remeandering).

The particular hydrological focus was upon the more extreme flood events measured in river catchments (typically with return periods greater than 50 years) in which the catchments have a high level of saturation as revealed by standard percentage runoff coefficients of greater than 70%. In turn this allowed for a major reduction in model complexity by focusing upon the response of rapid runoff generation routes and their contribution to flow peaks in particularly extreme events. The decision was taken to base the modelling upon a spatially-distributed unit hydrograph approach (e.g. Maidment, 1993; Maidment *et al.*¸1996; Olivera and Maidment, 1999; Saghafian *et al.*, 2002; Liu *et al.* 2003; Du *et al.*, 2009) that uses the time to equilibrium (t_e) approach pioneered by Saghafian and Julien (1995). Initial applications of the spatially-distributed unit hydrograph method assumed: (1) a single continuous and time-invariant flow path (e.g. Maidment *et al.*, 1996); (2) a linear system response in which at higher flows, travel times are independent of the amount of runoff being routed (e.g. Kull and Feldman, 1998; Olivera and Maidment, 1999); and (3) independence of response where two locations share elements of the same flow path (e.g. Maidment *et al.*, 1996). Despite these assumptions, these early applications were found to reproduce measured hydrographs extremely effectively (e.g. Maidment *et al.*, 1996). Given the interventions we explore, these assumptions needed to be relaxed. For instance, a woody debris dam is designed to switch the local flow path from channel to floodplain once local bank heights are exceeded. Similarly, work that has followed Maidment *et al.* (1996) has shown that it is possible to introduce travel time treatments that change with the amount of runoff being generated and delivered from upstream contributing areas (Saghafian *et al.*, 2002). Thus, our conceptual model builds upon the work of Saghafian and Julian (1995) and Saghafian *et al.* (2002) by allowing for time dependent evolution of travel times but introduces an additional modification by which the flow path followed by water is also allowed to evolve as a function of time, so as to capture time-dependent, spatially collocated transitions from channel to overbank flow.

The time to equilibrium is defined as the time required for maximum potential runoff to be reached for a catchment under a constant rainfall intensity and which varies as a function of both intensity and catchment geometry and physical properties (see Saghafian and Julien (1995) for review). Saghafian and Julien (1995) develop a formulation for the time to equilibrium that allows for distributed rainfall and runoff generation as the basis of estimating 'travel time' maps, such as maps of equal travel times called isochrones (Saghafian and Julien, 1995). Saghafian and Julien (1995) show that a (flood) wave travel time (*tw*) formulated for a kinematic wave approximation can be described by:

$$t_w = \int_{x_1}^{x_2} \frac{\gamma b_1}{(\beta - 1)\alpha^{1-\gamma}} \left(\frac{a_1}{Q_e}\right)^{\gamma} \mathrm{d}x$$

with

$$\gamma = \frac{\beta - 1}{\beta + b_1 - 1}$$

[1]

where x_i is the distance between points i = 1 and i = 2; α and β are parameters that depend on form of the resistance law used; a, b = constants dependent upon local channel cross-section geometry; and Q_e = upslope discharge delivered to the upstream point; x = distance along flow path. With a Manning resistance equation, this can then be formulated for the treatment of both overland flow and channel flow (Saghafian and Julien, 1995). Saghafian *et al.* (2002) moved away from the linear routing assumption by calculating travel time maps using [1] with a Manning resistance formulation for a number, *N*, of different rainfall intensities or, in effect, maps of runoff generation to produce *N* isochrone maps. These isochrone maps were then convolved to produce *N* incremental hydrographs, each of which were delayed by the time corresponding to each map, and then superimposed to obtain the final hydrograph.

Our model is based upon the Saghafian *et al.* (2002) approach. To address our need to understand the effects of riparian zone interventions such as floodplain forest, we modify this approach to allow for the explicit effects of flow from channels onto floodplains, as conditioned by both the local flow magnitude, channel geometry and channel and floodplain resistance.

Detailed model description

Determination of channel network

The DEM used in the analysis is initially pit-filled following the Planchon and Darboux (2003) method. Flow paths are then calculated using two methods according to whether a grid cell is labelled as a hillslope cell or a channel cell. In the case of a hillslope cell, the flow is routed using Quinn *et al.'s* (1991) FD8 algorithm, with a diffusion exponent of 3. In the case of channel cells, we use the steepest downslope flow path (i.e. D8) algorithm.

Clearly, we have no *a priori* definition of what is a hillslope cell and what is a channel cell. Thus, we approach the problem using a single, steady state, extreme, effective rainfall. This involves routing the runoff generated by an extreme rainfall event using the FD8 algorithm. We then apply a unit discharge threshold to the FD8 accumulated rainfall to identify the onset of channel routing across the landscape and apply the D8 routing to all cells downstream. We use an effective rainfall that gives a discharge at the catchment outlet that corresponds to bankfull, with a return period of *c.* 2 years.

For this, and for all subsequent calculations, following other applications of the spatially distributed unit hydrograph approach (e.g. Maidment *et al.*, 1996; Saghafian *et al.* 20 02), we set the effective rainfall as a runoff, based upon the excess of net rainfall, after evapotranspiration and interception losses, over local infiltration rate. Thus, the effective rainfall rate is defined as the rainfall rate minus some assumed percentage loss, the runoff percentage. The runoff percentage is assumed to be spatially uniform and reflecting our observations that during the flood event simulated, the catchment was close to saturation. We explain below how we introduce some temporal variability into the runoff percentage.

Isochrones

Isochrones are determined for each member of a set of rainfall rates. We subtract the assumed percentage loss from each rainfall rate to get a runoff rate. This runoff rate is routed either by the FD8 method or the D8 method according to our definition of the channel network.

Modification of flow path routing for extreme rainfall rates

Thus far, much of what we have described has formed the basis of published work. The novelty in this paper is that we recognise that flow paths can evolve as a function of rainfall rate. There are two modifications: (1) to allow for headward extension of channelized flow within an event when a unit discharge threshold is exceeded; and (2) to allow for flow path routing across floodplains rather than entirely within the river channel.

The first modification is undertaken relatively easily by considering the first calculation of the FD8 and D8 routed discharges based upon the channel network and then allowing the D8 routing to extend headwards in situations where an estimated unit discharge exceeds the threshold for channelized flow. Note that this adjusts the flow path (from FD8 to D8) but does not extend headwards the physical expression of the river network as an incised channel. In other words, we continue to calculate cell velocities and travel times using the full cell width. We argue that this reflects the distinction between the more dynamic hydrological expansion (and contraction) of a channel network, which we are addressing here, and the much slower geomorphological adjustment, which we assume is constant over the scale of a flood event. The second modification is more complex but also more important because it is required so that we can explore diffuse land management interventions that increase the amount of flow locally on river floodplains. It is based upon a simplified representation of overbank flow mechanisms, reflecting the data uncertainties associated with catchment-scale modelling, especially in relation to channel geometry estimation (see below). Our goal is to correct the travel times estimated assuming channel flow and D8 routing for situations where some flow is routed across a floodplain.

In the first step of the analysis, we consider the initial D8-based Q estimates (by definition, channel cells) for each rainfall rate. These can be converted into estimated flow depths (d_{chij}) using the Manning equation and estimated channel width (see below). We then calculate the estimated water surface elevation

(z_{wij}) as $z_{wij} = z_{ij} + d_{chij}$ [2]

where z_{ij} is the elevation of the DEM cell that the river is passing through. In theory, we should only have to compare z_{wij} with adjacent non-channel cells to see if there could be flux of water into those non-channel cells. However, almost invariably except for rivers with widths much greater than the DEM resolution, the channel occupies only a proportion of the DEM cell and z_{ij} is a spatial average of elevations associated with that cell such that:

$$z_{chij} < z_{ij} < z_{fij} \qquad [3]$$

where f indicates a floodplain cell. In process terms, this means that the z_{wij} from [2] may be higher than it should be and lead to a flux of water into the floodplain too readily. Thus, we introduce a correction term k which scales the estimated channel depth:

$$z_{wij} = z_{ij} + k\, d_{chij} \qquad [4]$$

We would expect the proportion of a cell that is floodplain to be greater for narrower channels which, by implication, have steeper slopes. Hence, we define k as

$$k = (1 + k_c S)^{-1} \qquad [5]$$

noting that k_c is an adjustable parameter which could be parameterised using high resolution remotely sensed data but which, in the absence of such data in this study, is treated as an adjustable parameter whose effects on prediction uncertainty are explored. Note that this scaling effect should also show a DEM resolution dependence. We then label a channel cell (i,j) as an overbank channel cell if:

$_{wij}Z > {}_{ij}Z$ [6]

Once [6] has been applied to all cells containing a channel, and channel overbank cells identified, we identify the set of cells surrounding each channel overbank cells as floodplain spill cells by:

$_{ij}z + k\ {}_{chij}d > z_{i+x,j+y}$ for $x = -1{:}+1;\ y = -1{:}+1;$ excluding $x = y = 0$ [7]

In the second stage of the analysis, we repeat the flow path, flow routing and flow accumulation but with modification for overbank channel and floodplain spill cells (taken together, these are labelled in combination as flood cells). This stage seeks to identify the flowpaths followed by water across the floodplain associated with the flow resulting from each rainfall rate. Commonly, floodplain flow routing is based upon the analysis of water surface gradients. The most simplified forms of floodplain routing treat the flow as a diffusion wave (e.g. Bates and de Roo, 2000; Horritt and Bates, 2001; Bradbrook *et al.*, 2004; Yu and Lane, 2006). Water is spread iteratively across the floodplain based upon flux apportionment but routing is only allowed to any two of the orthogonal cardinal flow directions in any one time step. Our aim is not to represent the progressive spreading of water within the floodplain during an event but to characterise the routes followed by water across the floodplain at a series of local flow discharges as defined by each rainfall rate. Thus, we do this by modifying the flow paths for flood cells in a way that makes them more diffusive under the assumption that, for floodplain flow, water surface gradient and momentum effects should reduce further the dependence of flow path upon topographic steering.

By definition, up until this point, a floodplain spill cell cannot be a channel cell, and so has the FD8 routing with the default diffusion exponent of u = 3. Given routing to eight possible cardinal directions from cell (i,j), flow is partitioned into fractions, f, defined as:

$f_{i+x,j+y} = [\ {}_{i\,x\,j\,yij}s_{\ +\ ,\ +}]u\ (\Sigma x\ \Sigma y\ [\ {}_{i\,x\,j\,yij}s_{\ +\ ,\ +}]u)\text{-}1$ for $x = -1{:}+1;\ y = -1{:}+1;$ excluding $x = y = 0$; and $_{i\,x\,j\,yij}s_{\ +\ ,\ +} < 0$

[8]

In [8], u is effectively a measure of the sensitivity of routing of flow to topography: as u tends to infinity, the routing tends towards a channel type D8 routing, with water following the line of steepest descent. As u tends to zero, routing becomes progressively less sensitive to topography and progressively more diffusive, until at u = 0, routing is independent of topography. Hence, we introduce u as an adjustable parameter, with u < 3. Comparison with conventional floodplain diffusion wave routing algorithms is not straightforward as these only allow routing to any two cells at any one time period. However, if we take u = 1, then we become close to approximating the slope dependence used in diffusion wave models. By applying this treatment to flood cells, we have to make a distinction between a channel overbank cell and a floodplain spill cell. A channel overbank cell is assumed to have two types of routing: D8 for the linear proportion of discharge that corresponds to in-channel flow:

$$ {}_{wij}Z \leq {}_{ij}Z \qquad [9] $$

and FD8 with u = 1 for that proportion that corresponds to out of channel or overbank flow:

$$ {}_{wij}Z \geq {}_{ij}Z \; . \qquad [10] $$

All floodplain spill cells have FD8 routing with u = 1 but [8] is modified so that we do not allow any diffusion from a floodplain spill cell into a channel overbank cell. We then iteratively identify additional floodplain spill cells, defined as those that are adjacent to existing floodplain spill cells except that: (1) we route water from the floodplain spill cell back into the main channel where a floodplain spill cell finds itself next to a channel cell that is not labelled as a channel overbank cell; and (2) where the discharge becomes very low (set here as the critical discharge threshold for channel flow) and topographic influences are likely to become greater, we return to u =3 in the FD8 algorithm. When water is returned to a channel cell, we check using [6] whether or not the new accumulated flow can still be accommodated within the channel, or whether this channel cell should also now itself become an overbank cell. In theory, this process could be iterated many times, but we found that just one or two iterations led to typically stable flow paths for a given rainfall rate.

Following from the flow path modifications, the cell travel time calculations also need some adjustment. For floodplain spill cells we use the lowest neighbouring cell rather than the steepest flow path cell, although these are sometimes the same. We introduce this modification because, particularly for floodplain spill cells next to channel cells, the steepest flow path route is often immediately straight back into the channel. For overbank channel cells, we have to combine two controls on routing: (1) the within-channel proportion which we assume to follow the steepest path as defined by D8; and (2) the overbank proportion which should be routed to the lowest neighbouring cell. These routes may or may not be the same for the cell in question, and there is also still the problem that the flow velocities should be different, that of the in-bank channel flow being in likelihood much faster than that of the shallow water over the bank side areas. We handle these two routes in combination. The different flow components, within bank and overbank, are calculated. The flow depth for the channel component is always assumed to be the channel depth, the width of the flow the channel width, and the length of travel the distance in the steepest path direction. Also, the resistance to the flow, the Manning's *n* value, is that for the channel. For the overbank proportion, the flow depth is inferred from the full width of the channel overbank cell and the velocity is calculated applying this to lowest neighbour flow path, this time using the Manning's *n* value appropriate for the bank side. To simplify matters here, the bankside *n* value is assumed to be the same as that of the adjoining floodplain cells, but the model includes the possibility to set it to a different value if desired (for example, if modelling a narrow woodland buffer strip immediately alongside the channel and separating it from a pasture floodplain). Once the cell passage times for each flow type have been calculated, they are weighted according to the values for each type of flow, and an overall weighted average cell flow passage time is then calculated for the overbank channel cell using:

$$t_w = \frac{V_{inbk} t_{inbk} + V_{obk} t_{obk}}{V_{inbk} + V_{obk}} \quad [11]$$

where t_w is the weighted cell passage time, t_{inbk} and t_{obk} are respectively the cell passage times for the within bank and the overbank components of the cell flow, and V_{inbk} and V_{obk} are respectively the values corresponding to the within bank and overbank flows found from the revised flow map.

Sampling the isochrones maps associated with each rainfall rate

With each rainfall rate modified for headward extension and overbank flow, the final step is to sample the isochrones maps so as to produce an estimated outlet hydrograph. We view this as a calibration problem, not least given the very poor coverage of rainfall recording devices typical in river catchments. However, if there is a downstream flood problem, it is common for there to be a river flow record, and so we focus on using the flow record to infer the isochrones maps that produce the measured flow. Each isochrone map is convolved with the observed rainfall, corrected for percentage runoff to provides a vector describing when the runoff from each (*i,j*) location in the model will reach the catchment outlet. This is repeated for all observed rainfalls. The question then becomes which isochrone map to use for each observed rainfall and we deal with this problem in a Monte Carlo calibration framework by randomly and then strategically sampling from all possible isochrone maps for each time step in the model. In the initial phase, we randomly sample an isochrones map for each time period to produce a hydrograph and this is repeated 200 times. We then calculate, for each time step, those isochrone maps which produce the best Nash Sutcliffe index of model efficiency for the entire flood hydrograph. In the second and subsequent phases, we repeat this process, but updating the set of isochrones maps that can be sampled as each time step, in light of the Nash Sutcliffe index results. We repeat this until we have 200 simulations all of which have a Nash Sutcliffe index >0.98. Given the dependence of our work upon calibration, we independently assess model performance with respect to internal flow field information.

Model application

Runoff percentage estimation, adjustments for baseflow and wetting up period

The focus of the spatially distributed unit hydrograph approach is upon modelling the rapid transfer of effective runoff. The effective runoff is a product of both the rainfall rate and the percentage runoff. With a discharge series inferred from the stage record, and a preliminary estimate of catchment-average rainfall, it was possible to obtain a first approximation of the estimated percentage runoff for each event being modelled through a mass balance calculation. We do this using an event-specific method, described here for the example of the 25th to 26th June 2007 event. First, we consider the total volume of water making up the flood event and the declining limb of the hydrograph, in this case between from 12 a.m. on 25th June to 12 a.m. on 29th June 2007, as compared with the rain input over the same period. For this period, the total rain volume, using the mean rainfall from the two gauges and a catchment area of 67.4 km^2 to Ropery Bridge, is *c.* 4.79×10^6 m^3 and the total discharge volume is *c.* 2.57×10^6 m^3. The latter includes a volume attributable to the initial baseflow which, after correction, leaves a total of is *c.* 2.42 $\times$ 10^6 m^3, or a percentage runoff of about 50%. However, the calculation is complicated by the problem of how to deal with Haugh Howl and Gundale Slack (area 10.7 km^2, just under 16% of the catchment, which, according to local observations, contributed almost nothing to the June 2007 flood because much of the water was drawn into the local limestone and leaves the catchment through Costa Beck. Similarly, the subcatchment of Levisham Beck (area 11 km^2 to Levisham Mill, also around 16% of the catchment) is partly affected by losses into the limestone, although a field visit some months after the flood indicated that the dry channels feeding into Levisham Beck had probably experience overland flow during the event.

To deal with these effects, we make two assumptions. First, for Haugh Howl and Gundale Slack we assume a small and constant baseflow of 0.4 cumecs and remove these catchments from contributing rainfall to the mass balance calculation. Second, we retain Levisham Beck, but give this a different weighting in the mass balance calculation. Thus, and for instance, after Haugh Howl and Gundale Slack are removed from the analyses, the mass balance can be achieved with a runoff coefficient of 25% set for Levisham Beck and 65% for the remaining 45.7 km^2 of Pickering Beck. Given the uncertainties in this mass balance calculation associated with the rainfall interpolation and to a lesser extent the flow gauge, we take these runoff coefficients as important calibration parameters.

We introduce one further modification. The calculations of runoff percentage are based upon the entire event. However, it is logical to expect that the runoff percentage varies as a function of the rainfall rate within the event. Thus, we introduce a weak non-linearity into runoff percentage such that it increases weakly with rainfall rate:

$$P_{eff} = A_{mb}(\mathrm{a}P_{mb}+\mathrm{b}P_{mb}^2) + A_{lb}(\mathrm{c}P_{lb}+\mathrm{d}P_{lb}^2) \qquad [12]$$

where P_{eff} is the precipitation that becomes runoff, mb is the main beck, lb is Levisham Beck and a, b, c and d are constants that meet the criterion set by the mass balance calculation. Given that there are multiple ways in which the observed P_{eff}, as well as the possibility of error in P_{eff} itself, we treat the problem as a calibration and uncertainty issue, except that we set 0 > c > a > 1 and 0 > d > b > 1. In each case, the first term in [12] is applied to rainfall over the main Beck and the second term to rainfall over Levisham Beck, both for the calculation of the equilibrium time maps, and when the time maps are sampled to simulate the hydrograph.

Finally, we also have to provide the model with a fixed baseflow value, which on the basis of analysis, we set at 0.4 m^3s^{-1}.

Digital topographic data

The DEM used in these calculations is of 20 m resolution, and formed by resampling from the Ordnance Survey's source 5 m 'NEXTMAP' of Great Britain data series.

Estimates of hydraulic geometry

As noted above, a major challenge for testing multiple, diffuse land management interventions is knowledge of the spatial patterns of river geometry especially given the fact that channel width and depth will effect the ease with which the river connects with its floodplain and hence the magnitude of flow at which water switches from moving within-channel to across the floodplain. We base the hydraulic geometry estimation on the assumption that the perennial channel estimated for flows with return periods of *c.* 2 years corresponds with the bankfull discharge which is likely to be a formative flow in channel geometry terms. We then apply the discharges estimated for each cell in the perennial network to the empirical relations of the functional form derived by Leopold and Maddock (1953). Firstly, for the channel widths, we use the equation:

$$w = 1.09Q^{0.5} \quad [13]$$

and then, for the channel depths (bank heights), we use:

$$d = 0.528Q^{0.344} \quad [14]$$

where w is the width of the perennial channel and d its depth. The values for the exponents and coefficients in [13] and [14] are estimated, and set so as to achieve realistic values (compared with values from direct observation) for both depth and width across the whole catchment. In many cases, the predictions of w from [13] are smaller in magnitude than the cell width and in such cases it is the channel width rather than the cell width that is used in channel calculations, with any residual cell width distributed to equally to floodplains assumed to exist either side if the channel.

We also need to specify a threshold between channel and hillslope cells. We set this as a unit discharge of 0.003 m^2s^{-1}.

Flow resistance

Application of [1] with a Manning formulation requires cell specific estimates of Manning's n values. We based these on the division between hillslopes and the perennial channel network. For simplicity, we set fixed values of n following Chow (1958). These are 0.060 for hillslopes corresponding to an open woodland or mixed low scrub and grassland land cover type; and 0.035 for for channels corresponding to a largely unobstructed channel with. Again, we take flow resistance parameters as ones that need exploring using calibration and uncertainty analysis.

The model has the following parameters: (1) the unit discharge that defines the channel network; (2) the diffusion exponents for both hillslope and spill flows; (3) the parameters in the hydraulic geometry relationships; (4) the Manning's n values for both hillslopes and channels; (5) the slope coefficient, k_{bk}, used to calculate bank height elevation in the spill flow procedure; (6) parameters associated with estimation of the runoff percentage; and 7) the fixed baseflow contribution. Each of these has an impact on any equilibrium isochrone map.

END

Appendix C – Hydraulic resistance reference tables

Manning's n for Channels (Chow, 1959).

Type of Channel and Description	Minimum	Normal	Maximum
Natural streams - minor streams (top width at floodstage < 100 ft)			
1. Main Channels			
a. clean, straight, full stage, no rifts or deep pools	0.025	0.030	0.033
b. same as above, but more stones and weeds	0.030	0.035	0.040
c. clean, winding, some pools and shoals	0.033	0.040	0.045
d. same as above, but some weeds and stones	0.035	0.045	0.050
e. same as above, lower stages, more ineffective slopes and sections	0.040	0.048	0.055
f. same as "d" with more stones	0.045	0.050	0.060
g. sluggish reaches, weedy, deep pools	0.050	0.070	0.080
h. very weedy reaches, deep pools, or floodways with heavy stand of timber and underbrush	0.075	0.100	0.150
2. Mountain streams, no vegetation in channel, banks usually steep, trees and brush along banks submerged at high stages			
a. bottom: gravels, cobbles, and few boulders	0.030	0.040	0.050
b. bottom: cobbles with large boulders	0.040	0.050	0.070
3. Floodplains			
a. Pasture, no brush			
1.short grass	0.025	0.030	0.035
2. high grass	0.030	0.035	0.050
b. Cultivated areas			
1. no crop	0.020	0.030	0.040
2. mature row crops	0.025	0.035	0.045
3. mature field crops	0.030	0.040	0.050
c. Brush			
1. scattered brush, heavy weeds	0.035	0.050	0.070
2. light brush and trees, in winter	0.035	0.050	0.060
3. light brush and trees, in summer	0.040	0.060	0.080
4. medium to dense brush, in winter	0.045	0.070	0.110
5. medium to dense brush, in summer	0.070	0.100	0.160
d. Trees			
1. dense willows, summer, straight	0.110	0.150	0.200
2. cleared land with tree stumps, no sprouts	0.030	0.040	0.050
3. same as above, but with heavy growth of sprouts	0.050	0.060	0.080
4. heavy stand of timber, a few down trees, little undergrowth, flood stage below branches	0.080	0.100	0.120
5. same as 4. with flood stage reaching branches	0.100	0.120	0.160
4. Excavated or Dredged Channels			
a. Earth, straight, and uniform			
1. clean, recently completed	0.016	0.018	0.020

2. clean, after weathering	0.018	0.022	0.025
3. gravel, uniform section, clean	0.022	0.025	0.030
4. with short grass, few weeds	0.022	0.027	0.033
b. Earth winding and sluggish			
1. no vegetation	0.023	0.025	0.030
2. grass, some weeds	0.025	0.030	0.033
3. dense weeds or aquatic plants in deep channels	0.030	0.035	0.040
4. earth bottom and rubble sides	0.028	0.030	0.035
5. stony bottom and weedy banks	0.025	0.035	0.040
6. cobble bottom and clean sides	0.030	0.040	0.050
c. Dragline-excavated or dredged			
1. no vegetation	0.025	0.028	0.033
2. light brush on banks	0.035	0.050	0.060
d. Rock cuts			
1. smooth and uniform	0.025	0.035	0.040
2. jagged and irregular	0.035	0.040	0.050
e. Channels not maintained, weeds and brush uncut			
1. dense weeds, high as flow depth	0.050	0.080	0.120
2. clean bottom, brush on sides	0.040	0.050	0.080
3. same as above, highest stage of flow	0.045	0.070	0.110
4. dense brush, high stage	0.080	0.100	0.140
5. Lined or Constructed Channels			
a. Cement			
1. neat surface	0.010	0.011	0.013
2. mortar	0.011	0.013	0.015
b. Wood			
1. planed, untreated	0.010	0.012	0.014
2. planed, creosoted	0.011	0.012	0.015
3. unplaned	0.011	0.013	0.015
4. plank with battens	0.012	0.015	0.018
5. lined with roofing paper	0.010	0.014	0.017
c. Concrete			
1. trowel finish	0.011	0.013	0.015
2. float finish	0.013	0.015	0.016
3. finished, with gravel on bottom	0.015	0.017	0.020
4. unfinished	0.014	0.017	0.020
5. gunite, good section	0.016	0.019	0.023
6. gunite, wavy section	0.018	0.022	0.025
7. on good excavated rock	0.017	0.020	
8. on irregular excavated rock	0.022	0.027	
d. Concrete bottom float finish with sides of:			
1. dressed stone in mortar	0.015	0.017	0.020
2. random stone in mortar	0.017	0.020	0.024
3. cement rubble masonry, plastered	0.016	0.020	0.024
4. cement rubble masonry	0.020	0.025	0.030
5. dry rubble or riprap	0.020	0.030	0.035
e. Gravel bottom with sides of:			

1. formed concrete	0.017	0.020	0.025
2. random stone mortar	0.020	0.023	0.026
3. dry rubble or riprap	0.023	0.033	0.036
f. Brick			
1. glazed	0.011	0.013	0.015
2. in cement mortar	0.012	0.015	0.018
g. Masonry			
1. cemented rubble	0.017	0.025	0.030
2. dry rubble	0.023	0.032	0.035
h. Dressed ashlar/stone paving	0.013	0.015	0.017
i. Asphalt			
1. smooth	0.013	0.013	
2. rough	0.016	0.016	
j. Vegetal lining	0.030		0.500

Determination of roughness coefficients for streams in Colorado (Jarrett, 1985)

Channel conditions		n value adjustment [1]	Example
Cross-section irregularities, n_1.	Smooth	0.000	Compares to the smoothest channel attainable in a given bed material.
	Minor	0.001-0.005	Compares to carefully dredged channels in good condition but having slightly eroded or scoured side slopes.
	Moderate	0.006-0.010	Compares to dredged channels having moderate to considerable bed roughness and moderately sloughed or eroded side slopes.
	Severe	0.011-0.020	Badly sloughed or scalloped banks of natural streams; badly eroded or sloughed sides of canals or drainage channels; unshaped, jagged, and irregular surfaces of channels in rock.
Channel variations, n_2 (Do not re-evaluate channel variation in the hydraulic computations).	Gradual	0.000	Size and shape of channel cross sections change gradually.
	Alternating occasionally.	0.001-0.005	Large and small cross sections alternate occasionally, or the main flow occasionally shifts from side to side owing to changes in cross-sectional shape.
	Alternating frequently.	0.010-0.015	Large and small cross sections alternate frequently, or the main flow frequently shifts from side to side owing to changes in cross-sectional shape.
Effect of obstructions, n_3.	Negligible	0.000-0.004	A few scattered obstructions, which include debris deposits, stumps, exposed roots, logs, piers, or isolated boulders, that

			occupy less than 5 percent of the cross-sectional area.
	Minor	0.005-0.015	Obstructions occupy less than 15 percent of the cross-sectional area, and the spacing between obstructions is such that the sphere of influence around one obstruction does not extend to the sphere of influence around another obstruction. Smaller adjustments are used for curved smooth-surfaced objects than are used for sharp-edged angular objects.
	Appreciable	0.020-0.030	Obstructions occupy from 15 to 50 percent of the cross-sectional area, or the space between obstructions is small enough to cause the effects of several obstructions to be additive, thereby blocking an equivalent part of a cross section.
	Severe	0.040-0.060	Obstructions occupy more than 50 percent of the cross-sectional area, or the space between obstructions is small enough to cause turbulence across most of the cross section.
Channel vegetation, n_4.	Small	0.002-0.010	Dense growths of flexible turf grass, such as Bermuda, or weeds growing where the average depth of flow is at least two times the height of the vegetation; supple tree seedlings such as willow, cottonwood, arrowweed, or saltcedar growing where the average depth of flow is at least three times the height of the vegetation.
	Medium	0.010-0.025	Turf grass growing where the average depth of flow is from one to two times the height of the vegetation; moderately dense

			stemmy grass, weeds, or tree seedlings growing where the average depth of flow is from two to three times the height of the vegetation; brushy, moderately dense vegetation, similar to 1- to 2-year-old willow trees in the dormant season, growing along the banks and no significant vegetation along the channel bottoms where the hydraulic radius exceeds 2 feet.
	Large	0.025-0.050	Turf grass growing where the average depth of flow is about equal to the height of vegetation; 8- to 10-year-old willow or cottonwood trees intergrown with some weeds and brush (none of the vegetation in foliage) where the hydraulic radius exceeds 2 feet; bushy willows about 1 year old intergrown with some weeds along side slopes (all vegetation in full foliage) and no significant vegetation along channel bottoms where the hydraulic radius is greater than 2 feet.
	Very large	0.050-0.100	Turf grass growing where the average depth of flow is less than half the height of the vegetation; bushy willow trees about 1 year old intergrown with weeds alongside slopes (all vegetation in full foliage) or dense cattails growing along channel bottom; trees intergrown with weeds and brush (all vegetation in full foliage).
Degree of meandering, *m* (Adjustment	Minor	1	Ratio of the channel length to valley length is 1.0 to 1.2.

values apply to flow confined in the channel and do not apply where down valley flow crosses meanders.)	Appreciable	1.15	Ratio of the channel length to valley length is 1.2 to 1.5.
	Severe	1.30	Ratio of the channel length to valley length is greater than 1.5.

[1] *Adjustment for cross-section irregularities, channel variations, effect of obstructions, and channel vegetation are added to the base n value (tables 2 or 5 or the prediction equations) before multiplying by the adjustment for degree of meandering.*

Appendix D - Supplemental OVERFLOW results and figures

This appendix contains additional results and figures not presented in Chapter 10.

Engineered logjams (spotdams) results

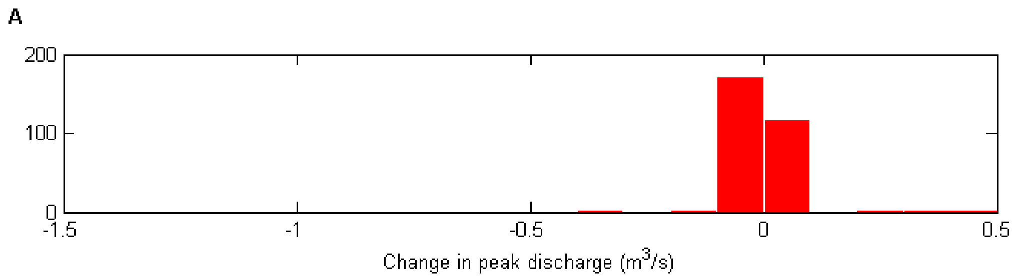


Figure D.1 – histogram showing the distribution of change to flood peak discharge following simulated insertion of logjams in a single reach. This shows the majority of runs are between ±0.15% of the baseline calibration peak discharge and are within the 0.23% model uncertainty. These small responses reflects that very small scale insertion of Engineered Logjams is unlikely to have a large effect on catchment hydrology.

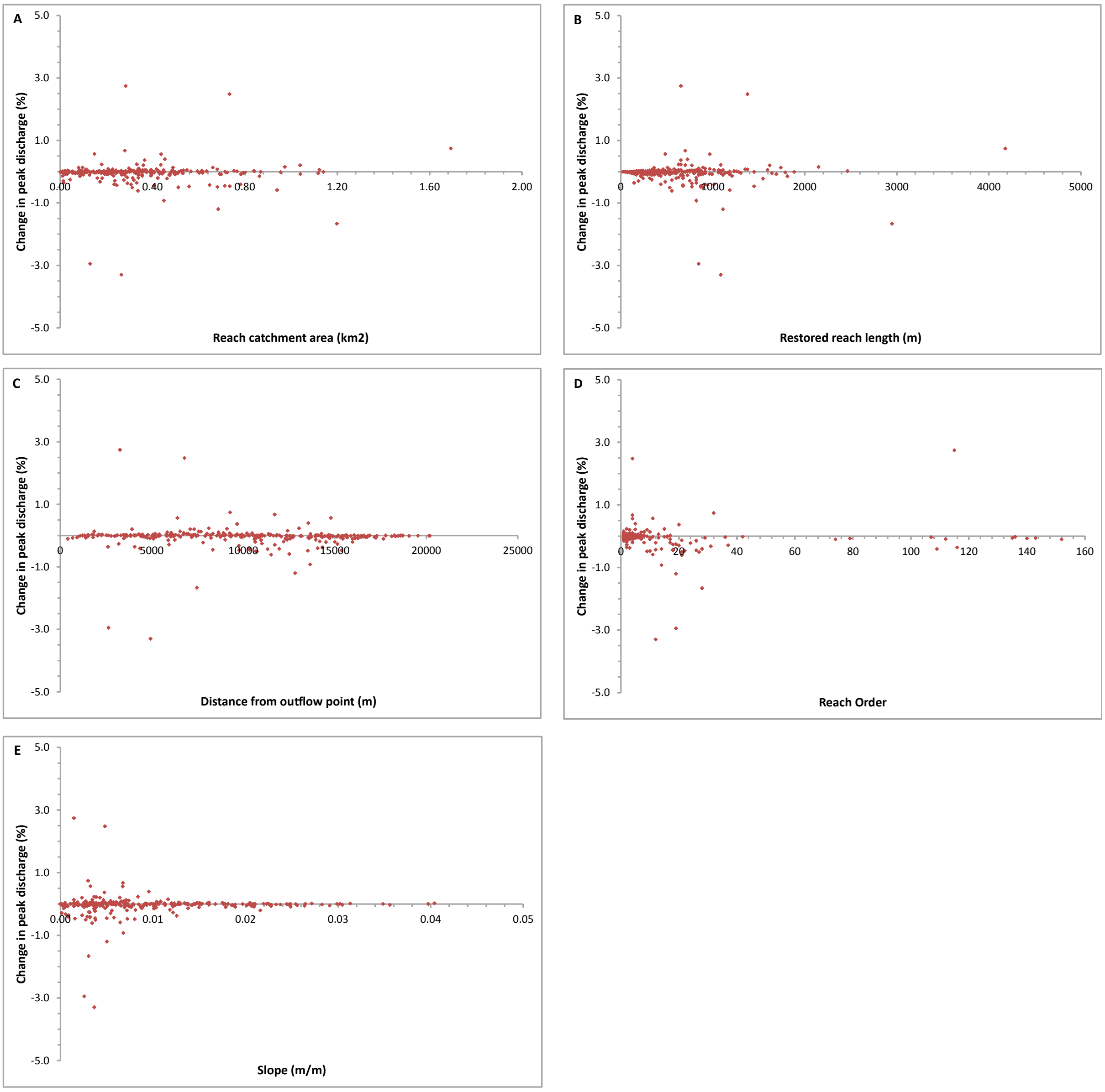


Figure D.2 – charts showing the relationship between reach characteristics and change in flood peak discharge following simulated insertion of ELJ spot dams in a single reach. A – catchment area draining to the study reach, B – the length of the simulated study reach, C – the cumulative channel length from the reach to the catchment outflow; a proxy measurement for how hydrologically proximal the reach is to the outflow, D – Shreve stream order of study reach, E – Study reach slope. There are no strong relationships of reach characteristics explaining hydrograph response. There is a possible weak relationship for headwater segments near to the catchment outflow to show larger changes in peak discharge magnitude, both increases and decreases. There are no strong trends between percentage change in peak magnitude for a reach and the reach: slope, length, catchment area, Shreve stream order or distance from catchment outflow.

Forest regeneration results

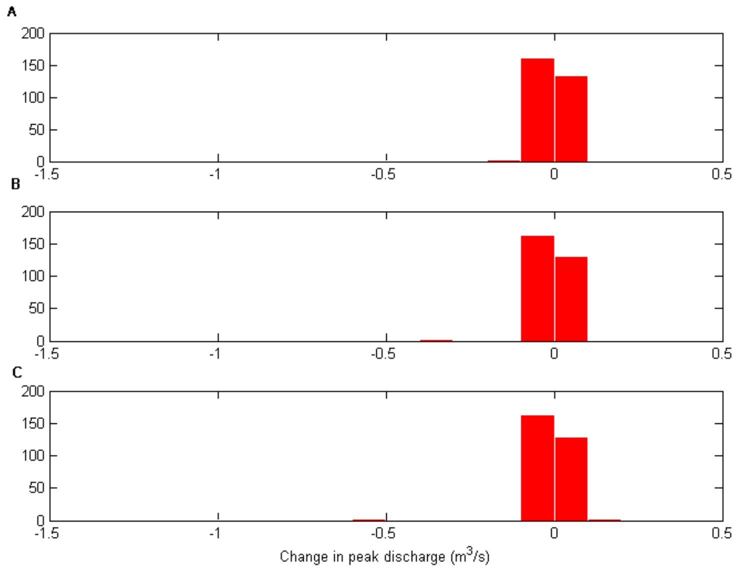


Figure D.3 – histograms showing the change in flood peak magnitude for single reaches with floodplain forest restoration only (no change to channel roughness). A) 25 years post riparian forest regeneration (Manning n=0.10), B) 50 years post riparian forest regeneration (Manning n=0.12), C) 100 years post riparian forest regeneration (Manning n=0.15). Note the model uncertainty for forest runs is 0.5%.

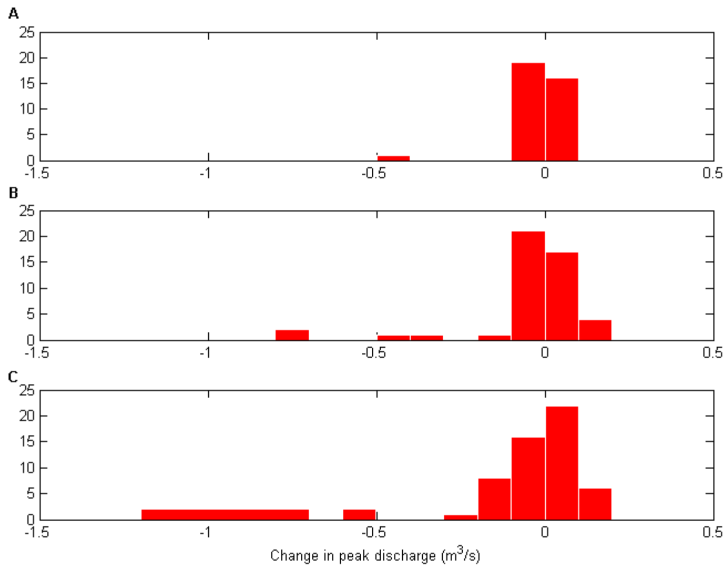


Figure D.4 – histograms showing the distribution of change to flood peak magnitude for sets of five sequential reaches with changes to forest cover applied only (no change to channel roughness). A) 25 years post riparian forest regeneration (Manning n=0.10), B) 50 years post riparian forest regeneration (Manning n=0.12), C) 100 years post riparian forest regeneration (Manning n=0.15). Note the model uncertainty for forest runs is 0.5%.

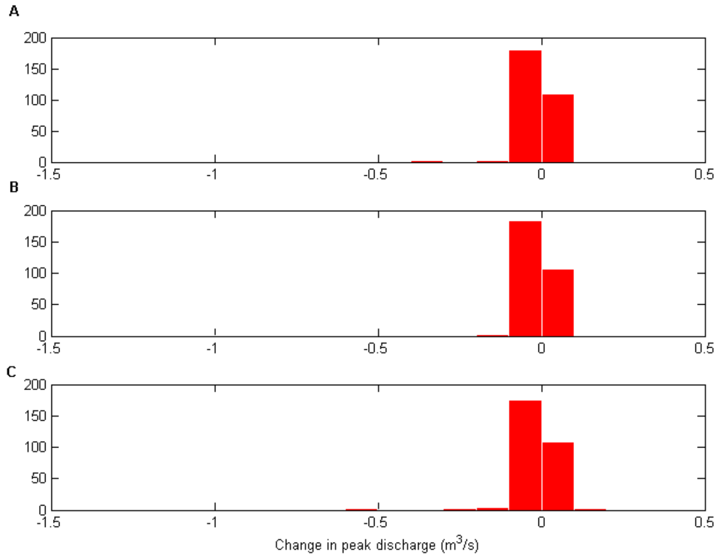


Figure D.5 – histograms showing change in flood peak magnitude for single reaches with riparian forest regeneration and accompanying increase in channel large wood loads applied. A) 25 years post forest regeneration (Forest Manning n=0.100, Channel Manning n=0.06) B) 50 years (Forest Manning n=0.120, Channel Manning n=0.075) C) 100 years (Forest Manning n=0.150, Channel Manning n=0.100). These figures show the majority of single segment runs result in only minor changes in peak discharge magnitude.

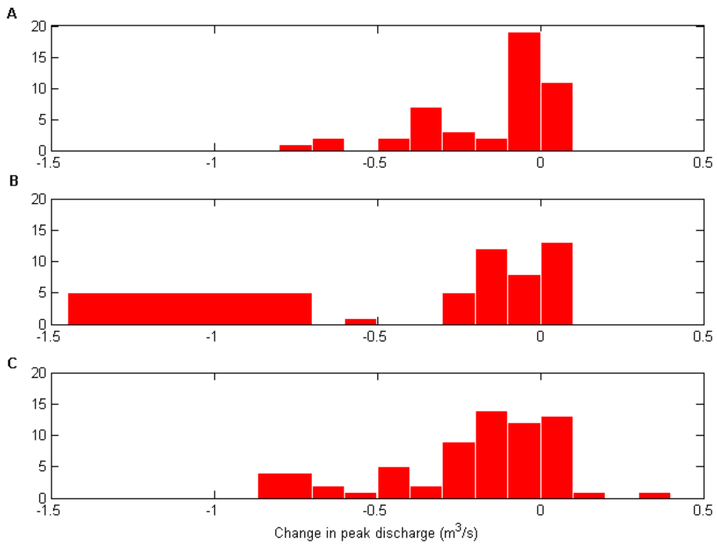


Figure D.6 – histograms showing change in flood peak magnitude for five sequential reaches with riparian forest regeneration and increase in channel large wood loads applied. A) 25 years post forest regeneration (Forest Manning n=0.100, Channel Manning n=0.06) B) 50 years (Forest Manning n=0.120, Channel Manning n=0.075)

C) 100 years (Forest Manning n=0.150, Channel Manning n=0.100). Note the model uncertainty for forest runs is 0.5% (approximately 0.2 m^3/s).

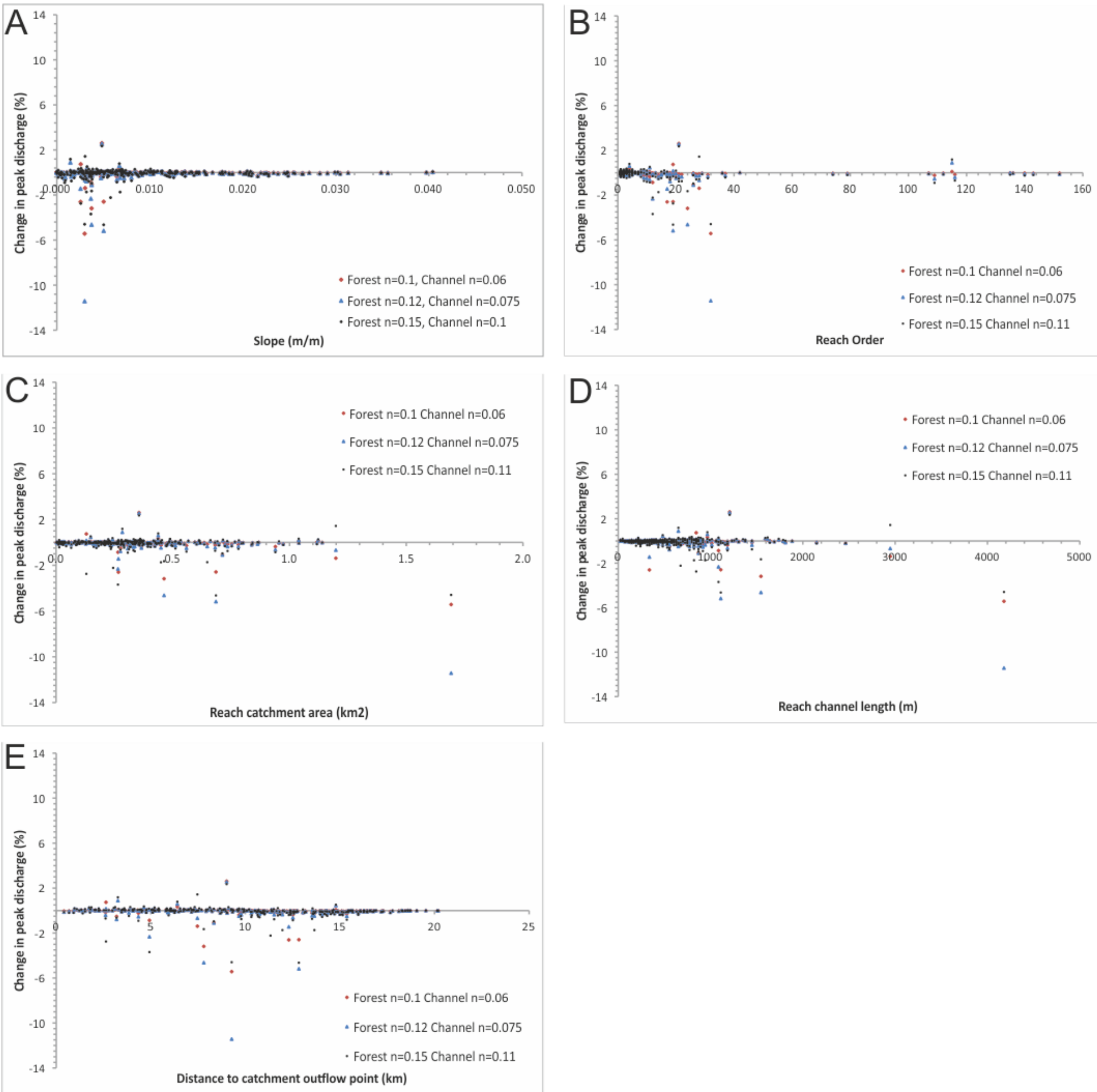


Figure D.7 – charts showing the relationship between reach characteristics and the change in flood peak discharge following changes in overbank and in channel hydraulic resistance to simulate forest regeneration and accompanying elevated in channel large wood loads for a single reach; A – Study reach slope, B – Shreve stream order of study reach, C – catchment area draining to the study reach, D – the length of the simulated study reach, E – the cumulative channel length from the reach to the catchment outflow, a proxy measurement for how hydrologically proximal the reach is to the outflow.

References

Abbe, T.B., Brooks, A.P., 2011. Geomorphic, Engineering, and Ecological Considerations When Using Wood in River Restoration. In: A. Simon, S.J. Bennett, J.M. Castro (Eds.), Stream Restoration in Dynamic Fluvial Systems: Scientific Approaches, Analyses and Tools. American Geophysical Union, Washington DC, pp. 419-451.

Abbe, T.B., Brooks, A.P., Montgomery, D.R., 2003. Wood in river rehabilitation and management. Ecology and Management of Wood in World Rivers, 37, 367-389.

Abbe, T.B., Montgomery, D.R., 1996. Large woody debris jams, channel hydraulics and habitat formation in large rivers. Regulated Rivers Research & Management, 12(23), 201-221.

Abbe, T.B., Montgomery, D.R., 2003. Patterns and processes of wood debris accumulation in the Queets river basin, Washington. Geomorphology, 51(1-3), 81-107.

Abdul, A.S., Gillham, R.W., 1989. Field studies of the effects of the capillary fringe on streamflow generation. Journal of Hydrology, 112(1), 1-18.

Acreman, M.C., Riddington, R., Booker, D.J., 2003. Hydrological impacts of floodplain restoration: a case study of the River Cherwell, UK. Hydrology and Earth System Sciences Discussions, 7(1), 75-85.

Agee, J.K., 2003. Historical range of variability in eastern Cascades forests, Washington, USA. Landscape Ecology, 18(8), 725-740.

Agresti, A., 2002. Categorical Data Analysis. John Wiley & Sons, New Jersey.

Agresti, A., Finlay, B., 2007. Statistical methods for the social sciences. Pearson.

Alaback, P.B., 1982. Dynamics of understory biomass in Sitka spruce-western hemlock forests of southeast Alaska. Ecology, 63(6), 1932-1948.

Alden, H.A., 1997. Softwoods of North America, 102. US Department of Agriculture, Forest Service, Forest Products Laboratory.

Alvarado, R.H., 1979. Amphibians. In: G.M.O. Maloiy (Ed.), Comparative Physiology of Osmoregulation in Animals. Academic Press: London., pp. 261-303.

Anderson, B.G., Rutherfurd, I.D., Western, A.W., 2006. An analysis of the influence of riparian vegetation on the propagation of flood waves. Environmental Modelling & Software, 21(9), 1290-1296.

Andréassian, V., 2004. Waters and forests: from historical controversy to scientific debate. Journal of Hydrology, 291(1), 1-27.

Archer, D.R., 1989. Flood wave attenuation due to channel and floodplain storage and effects on flood frequency. In: K.J. Beven, P.A. Carling (Eds.), Floods: Hydrological, Sedimentological and Geomorphological Implications. John Wiley & Sons, New York, pp. 37-46.

Archer, D.R., 2003. Scale effects on the hydrological impact of upland afforestation and drainage using indices of flow variability: the River Irthing, England. Hydrology and Earth System Sciences Discussions, 7(3), 325-338.

Archer, D.R., Climent-Soler, D., Holman, I., 2010. Changes in discharge rise and fall rates applied to impact assessment of catchment land use.

Arlettaz, R., Lugon, A., Sierro, A., Werner, P., Kéry, M., Oggier, P.A., 2011. River bed restoration boosts habitat mosaics and the demography of two rare non-aquatic vertebrates. Biological Conservation, 144, 2126-2132.
Arnell, N.W., 2003. Relative effects of multi-decadal climatic variability and changes in the mean and variability of climate due to global warming: future streamflows in Britain. Journal of Hydrology, 270(3), 195-213.
Assani, A.A., Petit, F., 1995. Log-jam effects on bed-load mobility from experiments conducted in a small gravel-bed forest ditch. Catena, 25(1-4), 117-126.
Auzet, A.V., Boiffin, J., Papy, F., Ludwig, B., Maucorps, J., 1993. Rill erosion as a function of the characteristics of cultivated catchments in the North of France. Catena, 20(1), 41-62.
Azuma, K., Ikeda, K., Kagi, N., Yanagi, U., Hasegawa, K., Osawa, H., 2013. Effects of water-damaged homes after flooding: health status of the residents and the environmental risk factors. International Journal of Environmental Health Research, In Press.
Bagnold, R.A., 1966. An approach to the sediment transport problem from general physics, USGS, Washington DC, USA.
Bahuguna, D., Mitchell, S.J., Miquelajauregui, Y., 2010. Windthrow and recruitment of large woody debris in riparian stands. Forest Ecology and Management, 259(10), 2048-2055.
Bal, K., Meire, P., 2009. The influence of macrophyte cutting on the hydraulic resistance of lowland rivers. Journal of Aquatic Plant Management, 47, 65-68.
Ball, T., 2008. Management approaches to floodplain restoration and stakeholder engagement in the UK: a survey. Ecohydrology & Hydrobiology, 8(2), 273-280.
Bankes, S., 1993. Exploratory Modeling for Policy Analysis. Operations Research, 41(3), 435-449.
Bates, P.D., De Roo, A.P.J., 2000. A simple raster-based model for flood inundation simulation. Journal of Hydrology, 236(1-2), 54-77.
Bathurst, J.C., 1985. Flow resistance estimation in mountain rivers. Journal of Hydraulic Engineering, 111(4), 625-643.
Bathurst, J.C., 2002. At-a-site variation and minimum flow resistance for mountain rivers. Journal of Hydrology, 269(1), 11-26.
Beadle, L.C., 1969. Osmotic regulation and the adaptation of freshwater animals to inland saline waters. Verh. Int. Ver. Limnol, 17, 421-429.
Beechie, T.J., Liermann, M., Pollock, M.M., Baker, S., Davies, J., 2006. Channel pattern and river-floodplain dynamics in forested mountain river systems. Geomorphology, 78(1-2), 124-141.
Beechie, T.J., Pess, G., Kennard, P., Bilby, R.E., Bolton, S., 2000. Modeling recovery rates and pathways for woody debris recruitment in northwestern Washington streams. North American Journal of Fisheries Management, 20(2), 436-452.
Beechie, T.J., Sear, D.A., Olden, J.D., Pess, G.R., Buffington, J.M., Moir, H., Roni, P., Pollock, M.M., 2010. Process-based principles for restoring river ecosystems. Bioscience, 60(3), 209-222.
Bell, B., 1970. The Oldest Records of the Nile Floods. The Geographical Journal, 136(4), 569-573.
Benda, L.E., Sias, J.C., 2003. A quantitative framework for evaluating the mass balance of in-stream organic debris. Forest Ecology and Management, 172(1), 1-16.

Bendix, J., Cowell, C.M., 2010. Fire, floods and woody debris: Interactions between biotic and geomorphic processes. Geomorphology, 116(3-4), 297-304.
Benke, A.C., Wallace, J.B., 1990. Wood dynamics in coastal plain blackwater streams. Canadian Journal of Fisheries and Aquatic Sciences CJFSDX, 47(1).
Benke, A.C., Wallace, J.B., 2003. Influence of wood on invertebrate communities in streams and rivers. In: S.V. Gregory, K.L. Boyer, A.M. Gurnell (Eds.), The ecology and management of wood in world rivers. American Fisheries Society, Symposium 37, Bethesda, Maryland, pp. 149-177.
Berg, N., Carlson, A., Azuma, D., 1998. Function and dynamics of woody debris in stream reaches in the central Sierra Nevada, California. Canadian Journal of Fisheries and Aquatic Sciences, 55(8), 1807-1820.
Bergkamp, G., Cammeraat, L.H., Martinez-Fernandez, J., 1996. Water movement and vegetation patterns on shrubland and an abandoned field in two desertification-threatened areas in Spain. Earth Surface Processes and Landforms, 21(12), 1073-1090.
Bernhardt, E.S., Palmer, M.A., Allan, J.D., Alexander, G., Barnas, K., Brooks, S., Carr, J., Clayton, S., Dahm, C.N., Follstad-Shah, J., 2005. Ecology: synthesizing US river restoration efforts. Science, 308(5722), 636.
Bertoldi, W., Drake, N.A., Gurnell, A.M., 2011. Interactions between river flows and colonizing vegetation on a braided river: exploring spatial and temporal dynamics in riparian vegetation cover using satellite data. Earth Surface Processes and Landforms, 36(11), 1474-1486.
Bertoldi, W., Gurnell, A.M., Surian, N., Tockner, K., Zanoni, L., Ziliani, L., Zolezzi, G., 2009. Understanding reference processes: linkages between river flows, sediment dynamics and vegetated landforms along the Tagliamento River, Italy. River Research and Applications, 25(5), 501-516.
Bertoldi, W., Gurnell, A.M., Welber, M., 2013. Wood recruitment and retention: The fate of eroded trees on a braided river explored using a combination of field and remotely-sensed data sources. Geomorphology, 180–181, 146-155.
Berz, G., 2000. Flood disasters: lessons from the past—worries for the future. Proceedings of the ICE-Water and Maritime Engineering, 142(1), 3-8.
Beschta, R.L., 1979. Debris removal and its effects on sedimentation in an Oregon Coast Range stream. Northwest Science, 53(1), 71-77.
Beven, K.J., 1993. Riverine flooding in a warmer Britain. Geographical Journal, 157-161.
Bilby, R.E., 1984. Removal of woody debris may affect stream channel stability. Journal of Forestry, 82(10), 609-613.
Bilby, R.E., 2003. Decomposition and nutrient dynamics of wood in streams and rivers. In: S.V. Gregory, K.L. Boyer, A.M. Gurnell (Eds.), The Ecology and Management of Wood in World Rivers. American Fisheries Society, Bethesda, Maryland, pp. 135-147.
Bilby, R.E., Likens, G.E., 1980. Importance of organic debris dams in the structure and function of stream ecosystems. Ecology, 1107-1113.
Bilby, R.E., Ward, J.W., 1991. Characteristics and function of large woody debris in streams draining old-growth, clear-cut, and second-growth forests in southwestern Washington. Canadian Journal of Fisheries and Aquatic Sciences, 48(12), 2499-2508.
Bisson, P.A., Bilby, R.E., Bryant, M.D., Dolloff, C.A., Grette, G.B., House, R.A., Murphy, M.L., Koski, K.V., Sedell, J.R., 1987. Large Woody Debris in Forested Streams in

the Pacific Northwest: Past, Present, and Future. In: E.O. Salo, T.W. Cundy (Eds.), Proceedings of the Symposium on Forest Hydrology and Watershed Management. International Association of Hydrologic Sciences, Wallingford, Oxfordshire, pp. 143-190.

Blöschl, G., Ardoin-Bardin, S., Bonell, M., Dorninger, M., Goodrich, D., Gutknecht, D., Matamoros, D., Merz, B., Shand, P., Szolgay, J., 2007. At what scales do climate variability and land cover change impact on flooding and low flows? Hydrological Processes, 21(9), 1241-1247.

Blöschl, G., Nester, T., Komma, J., Parajka, J., Perdigão, R.A.P., 2013. The June 2013 flood in the Upper Danube basin, and comparisons with the 2002, 1954 and 1899 floods. Hydrology and Earth System Sciences Discussions, 10, 9533-9573.

Boardman, J., 1995. Damage to property by runoff from agricultural land, South Downs, southern England, 1976-93. Geographical Journal, 177-191.

Bocchiola, D., Rulli, M.C., Rosso, R., 2006a. Flume experiments on wood entrainment in rivers. Advances in water resources, 29(8), 1182-1195.

Bocchiola, D., Rulli, M.C., Rosso, R., 2006b. Transport of large woody debris in the presence of obstacles. Geomorphology, 76(1), 166-178.

Boddy, L., Swift, M.J., 1983. Wood Decomposition in an Abandoned Beech and Oak Coppiced Woodland in SE England: I. Patterns of Wood-Litter Fall. Holarctic ecology, 320-332.

Boer, M., Puigdefábregas, J., 2005. Effects of spatially structured vegetation patterns on hillslope erosion in a semiarid Mediterranean environment: a simulation study. Earth Surface Processes and Landforms, 30(2), 149-167.

Boix-Fayos, C., Calvo-Cases, A., Imeson, A.C., Soriano-Soto, M.D., Tiemessen, I.R., 1998. Spatial and short-term temporal variations in runoff, soil aggregation and other soil properties along a Mediterranean climatological gradient. Catena, 33(2), 123-138.

Bond, T.A., Unpublished. Forest tree regeneration potential across restored and unrestored river reaches in the New Forest. B.Sc, University of Southampton, Southampton, UK, 51 pp.

Bormann, F.H., Likens, G.E., 1979. Catastrophic disturbance and the steady state in northern hardwood forests. American Scientist, 67, 660-669.

Botkin, D.B., Janak, J.F., Wallis, J.R., 1972. Some Ecological Consequences of a Computer Model of Forest Growth. Journal of Ecology, 60(3), 849-872.

Braccia, A., Batzer, D.P., 2008. Breakdown and invertebrate colonization of dead wood in wetland, upland, and river habitats. Canadian Journal of Forest Research, 38(10), 2697-2704.

Bracken, L.J., Croke, J., 2007. The concept of hydrological connectivity and its contribution to understanding runoff-dominated geomorphic systems. Hydrological processes, 21(13), 1749-1763.

Bradbrook, K.F., Lane, S.N., Waller, S.G., Bates, P.D., 2004. Two dimensional diffusion wave modelling of flood inundation using a simplified channel representation. International Journal of River Basin Management, 2(3), 211-223.

Bradford, J.M., Ferris, J.E., Remley, P.A., 1987a. Interrill soil erosion processes: I. Effect of surface sealing on infiltration, runoff, and soil splash detachment. Soil Science Society of America Journal, 51(6), 1566-1571.

Bradford, J.M., Ferris, J.E., Remley, P.A., 1987b. Interrill soil erosion processes: II. Relationship of splash detachment to soil properties. Soil Science Society of America Journal, 51(6), 1571-1575.
Bragg, D.C., 2000. Simulating catastrophic and individualistic large woody debris recruitment for a small riparian system. Ecology, 81(5), 1383-1394.
Braudrick, C.A., Grant, G.E., 2001. Transport and deposition of large woody debris in streams: a flume experiment. Geomorphology, 41(4), 263-283.
Braudrick, C.A., Grant, G.E., Ishikawa, Y., Ikeda, H., 1997. Dynamics of wood transport in streams: a flume experiment. Earth Surface Processes and Landforms, 22(7), 669-683.
Bretz Guby, N.A., Dobbertin, M., 1996. Quantitative estimates of coarse woody debris and standing dead trees in selected Swiss forests. Global Ecology and Biogeography Letters, 327-341.
Brierley, G.J., Fryirs, K., 2009. Don't Fight the Site: Three Geomorphic Considerations in Catchment-Scale River Rehabilitation Planning. Environmental Management, 43(6), 1201-1218.
Broadmeadow, S., 2012. UK forest growth models. Forest Research, Forestry Commission.
Broadmeadow, S., Nisbet, T.R., 2004. The effects of riparian forest management on the freshwater environment: a literature review of best management practice. Hydrology and Earth System Sciences, 8(3), 286-305.
Broadmeadow, S., Nisbet, T.R., 2009. Opportunity Mapping for Woodland to Reduce Flooding in the Yorkshire & the Humber Region, Forest Research.
Brookes, A., Sear, D.A., 1996. Geomorphological principles for restoring channels. In: A. Brookes, F.D. Shields Jr (Eds.), River Channel Restoration: guiding principles for sustainable projects. Wiley, pp. 75–101.
Brooks, A.P., Abbe, T.B., Cohen, T., Marsh, N., Mika, S., Boulton, A., Broderick, T., Borg, D., Rutherfurd, I.D., 2006. Design guideline for the reintroduction of wood into Australian streams. Land & Water Australia, Canberra, Australia.
Brooks, A.P., Gehrke, P.C., Jansen, J.D., Abbe, T.B., 2004. Experimental reintroduction of woody debris on the Williams River, NSW: Geomorphic and ecological responses. River Research and Applications, 20(5), 513-536.
Brooks, R.T., Nislow, K.H., Lowe, W.H., Wilson, M.K., King, D.I., 2012. Forest succession and terrestrial–aquatic biodiversity in small forested watersheds: a review of principles, relationships and implications for management. Forestry, 85(3), 315-328.
Brown, A.E., Zhang, L., McMahon, T.A., Western, A.W., Vertessy, R.A., 2005. A review of paired catchment studies for determining changes in water yield resulting from alterations in vegetation. Journal of Hydrology, 310(1), 28-61.
Brown, A.G., 1997. Alluvial geoarchaeology: floodplain archaeology and environmental change. Cambridge University Press, Cambridge.
Brown, J.D., Damery, S.L., 2002. Managing flood risk in the UK: towards an integration of social and technical perspectives. Transactions of the Institute of British Geographers, 27(4), 412-426.
Brummer, C.J., Abbe, T.B., Sampson, J.R., Montgomery, D.R., 2006. Influence of vertical channel change associated with wood accumulations on delineating channel migration zones, Washington, USA. Geomorphology, 80(3), 295-309.

Brzeziecki, B., Kienast, F., 1994. Classifying the Life-History Strategies of Trees on the Basis of the Grimian Model. Forest Ecology and Management, 69(1-3), 167-187.
Buffington, J.M., Montgomery, D.R., 1999. Effects of hydraulic roughness on surface textures of gravel-bed rivers. Water Resources Research, 35(11), 3507-3521.
Buffington, J.M., Montgomery, D.R., Greenberg, H.M., 2004. Basin-scale availability of salmonid spawning gravel as influenced by channel type and hydraulic roughness in mountain catchments. Canadian Journal of Fisheries and Aquatic Sciences, 61(11), 2085-2096.
Bull, L.J., Kirkby, M.J., Shannon, J., Hooke, J.M., 2000. The impact of rainstorms on floods in ephemeral channels in southeast Spain. Catena, 38(3), 191-209.
Burrel, B.C., Davar, K., Hughes, R., 2007. A review of flood management considering the impacts of climate change. Water International, 32(3), 342-359.
Burt, T.P., 1997. The hydrological role of floodplains within the drainage basin system. In: H. N.E., B. T.P., G. K., P. G. (Eds.), Buffer zones: their processes and potential in water protection. Quest Environmental, pp. 21-32.
Burt, T.P., Haycock, N.E., 1996. Linking floodplains to rivers. In: M.G. Anderson, D.E. Walling, P.D. Bates (Eds.), Floodplain Processes. Wiley, Chichester, pp. 461-492.
Burt, T.P., Pinay, G., 2005. Linking hydrology and biogeochemistry in complex landscapes. Progress in Physical Geography, 29(3), 297-316.
Busing, R.T., Solomon, A.M., 2004. A comparison of forest survey data with forest dynamics simulators FORCLIM and ZELIG along climatic gradients in the Pacific Northwest. US Department of the Interior, US Geological Survey.
Butler, D.R., 2012. Characteristics of Beaver Ponds on Deltas in a Mountain Environment. Earth Surface Processes and Landforms, 37(8), 876-882.
Byers, E.E., 2011. The use of catchment-scale riparian intervention measures in downstream flood hazard mitigation. Master of Science by Research, Durham University, Durham, 220 pp.
Caissie, D., Jolicoeur, S., Bouchard, M., Poncet, E., 2002. Comparison of streamflow between pre and post timber harvesting in Catamaran Brook (Canada). Journal of Hydrology, 258(1), 232-248.
Cammeraat, L.H., Imeson, A.C., 1999. The evolution and significance of soil–vegetation patterns following land abandonment and fire in Spain. Catena, 37(1), 107-127.
Carroll, Z., Bird, S., Emmett, B., Reynolds, B., Sinclair, F., 2004. Can tree shelterbelts on agricultural land reduce flood risk? Soil Use and Management, 20(3), 357-359.
Caruso, B.S., Downs, P.W., 2007. Rehabilitation and Flood Management Planning in a Steep, Boulder-Bedded Stream. Environmental Management, 40(2), 256-271.
Charlton, F.G., Brown, P.M., Benson, R.W., 1978. The hydraulic geometry of some gravel rivers in Britain, Hydraulics Research Station, Wallingford, England.
Chen, X., Wei, X., Scherer, R., Luider, C., Darlington, W., 2006. A watershed scale assessment of in-stream large woody debris patterns in the southern interior of British Columbia. Forest Ecology and Management, 229(1-3), 50-62.
Chin, A., Daniels, M.D., Urban, M.A., Piegay, H., Gregory, K.J., Bigler, W., Butt, A.Z., Grable, J.L., Gregory, S.V., Lafrenz, M., 2008. Perceptions of wood in rivers and challenges for stream restoration in the United States. Environmental Management, 41(6), 893-903.
Chin, A., Laurencio, L.R., Daniels, M.D., Wohl, E.E., Urban, M.A., Boyer, K.L., Butt, A.Z., Piegay, H., Gregory, K.J., 2012. The significance of perceptions and

feedbacks for effectively managing wood in rivers. River Research and Applications, 30(1), 98-111.
Chow, V.T., 1959. Open-channel hydraulics. McGraw-Hill, New York, US.
Christensen, J.H., Christensen, O.B., 2003. Severe summertime flooding in Europe. Nature, 421(6925), 805-806.
Christensen, M., Hahn, K., Mountford, E.P., Ódor, P., Standovár, T., Rozenbergar, D., Diaci, J., Wijdeven, S.M.J., Meyer, P., Winter, S., 2005. Dead wood in European beech (Fagus sylvatica) forest reserves. Forest Ecology and Management, 210(1-3), 267-282.
Christensen, M., Hahn, K., Mountford, E.P., Wijdeven, S.M.J., Manning, D.B., Standovar, T., Odor, P., Rozenbergar, D., 2003. NAT-MAN Working Report 9, Study on deadwood in European beech forest reserves., European Union, Brussels.
Church, M., 1992. Channel morphology and typology. In: P. Callow, G.E. Petts (Eds.), The rivers handbook vol. 1: hydrological and ecological principles Blackwell Science, pp. 126–143.
Clarke, S.J., Bruce Burgess, L., Wharton, G., 2003. Linking form and function: towards an eco hydromorphic approach to sustainable river restoration. Aquatic Conservation: Marine and Freshwater Ecosystems, 13(5), 439-450.
Cline, S.P., Berg, A.B., Wight, H.M., 1980. Snag characteristics and dynamics in Douglas-fir forests, western Oregon. The Journal of Wildlife Management, 44(4), 773-786.
Collins, B.D., Montgomery, D.R., 2002. Forest development, wood jams, and restoration of floodplain rivers in the Puget Lowland, Washington. Restoration Ecology, 10(2), 237-247.
Collins, B.D., Montgomery, D.R., Fetherston, K.L., Abbe, T.B., 2012. The floodplain large-wood cycle hypothesis: A mechanism for the physical and biotic structuring of temperate forested alluvial valleys in the North Pacific coastal ecoregion. Geomorphology, 139-140, 460-470.
Collins, B.D., Montgomery, D.R., Haas, A.D., 2002. Historical changes in the distribution and functions of large wood in Puget Lowland rivers. Canadian Journal of Fisheries and Aquatic Sciences, 59(1), 66-76.
Conly, F.M., Martz, L.W., 1998. Modelling channel profile response to weir removal: South Saskatchewan River at Saskatoon. Canadian Water Resources Journal, 23(1), 31-44.
Cooper, J.R., Gilliam, J.W., Daniels, R.B., Robarge, W.P., 1987. Riparian areas as filters for agricultural sediment. Soil Science Society of America Journal, 51(2), 416-420.
Costa, J.E., 1987. A comparison of the largest rainfall-runoff floods in the United States with those of the People's Republic of China and the world. Journal of Hydrology, 96(1), 101-115.
Crockford, R.H., Richardson, D.P., 2000. Partitioning of rainfall into throughfall, stemflow and interception: effect of forest type, ground cover and climate. Hydrological Processes, 14(16-17), 2903-2920.
Curran, J.C., 2010. Mobility of large woody debris (LWD) jams in a low gradient channel. Geomorphology, 116(3-4), 320-329.
Curran, J.H., Wohl, E.E., 2003. Large woody debris and flow resistance in step-pool channels, Cascade Range, Washington. Geomorphology, 51(1-3), 141-157.

Dahlström, N., 2005. Function and dynamics of woody debris in boreal forest streams. PhD, UMEÅ UNIVERSITY, Umeå, 21 pp.
Dahlström, N., Jönsson, K., Nilsson, C., 2005. Long-term dynamics of large woody debris in a managed boreal forest stream. Forest Ecology and Management, 210(1-3), 363-373.
Dalley, S., 1989. Myths from mesopotamia: Creation, the flood, gilgamesh, and others. Oxford University Press.
Daniels, M.D., 2006. Distribution and dynamics of large woody debris and organic matter in a low-energy meandering stream. Geomorphology, 77(3-4), 286-298.
Daniels, M.D., Rhoads, B.L., 2004. Effect of large woody debris configuration on three-dimensional flow structure in two low-energy meander bends at varying stages. Water Resources Research, 40(11).
Daniels, R.B., Gilliam, J.W., 1996. Sediment and chemical load reduction by grass and riparian filters. Soil Science Society of America Journal, 60(1), 246-251.
Darby, S.E., Thorne, C.R., 1996. Predicting stage-discharge curves in channels with bank vegetation. Journal of Hydraulic Engineering, 122(10), 583-586.
David, G.C.L., Wohl, E.E., Yochum, S.E., Bledsoe, B.P., 2011. Comparative analysis of bed resistance partitioning in high-gradient streams. Water Resources Research, 47(7).
Davies-Colley, R.J., 1997. Stream channels are narrower in pasture than in forest. New Zealand Journal of Marine and Freshwater Research, 31(5), 599-608.
Davies, N.S., Gibling, M.R., 2011. Evolution of fixed-channel alluvial plains in response to Carboniferous vegetation. Nature Geoscience, 4(9), 629-633.
Davies, N.S., Gibling, M.R., 2012. Early Cambrian metazoans in fluvial environments, evidence of the non-marine Cambrian radiation: COMMENT. Geology, 40(6), e270-e270.
Davies, N.S., Gibling, M.R., Rygel, M.C., 2011. Alluvial facies evolution during the Palaeozoic greening of the continents: case studies, conceptual models and modern analogues. Sedimentology, 58(1), 220-258.
Davis, S.R., Brown, A.G., Dinnin, M.H., 2007. Floodplain connectivity, disturbance and change: a palaeoentomological investigation of floodplain ecology from south west England. Journal of Animal Ecology, 76(2), 276-288.
De Ploey, J., 1984. Hydraulics of runoff and loess loam deposition. Earth Surface Processes and Landforms, 9(6), 533-539.
Decamps, H., Fortune, M., Gazelle, F., Pautou, G., 1988. Historical influence of man on the riparian dynamics of a fluvial landscape. Landscape Ecology, 1(3), 163-173.
Defra, 2005a. Making Space for Water: Taking Forward a New Government Strategy for Flood and Coastal Erosion Risk Management in England: Delivery Plan, Department for Food and Rural Affairs, London.
Defra, 2005b. Review of Impacts of Rural Land Use and Management on Flood Generation: Impact Study Report, Department for Environment, Food and Rural Affairs, London.
Defra, 2007. Making Space for Water: Taking Forward a New Government Strategy for Flood and Coastal Erosion Risk Management Encouraging and Incentivising Increased Resilience to Flooding, Department for Food and Rural Affairs, London.
Dolloff, C.A., Warren, M.L., 2003. Fish relationships with large wood in small streams. Ecology and Management of Wood in World Rivers, 37, 179-193.

Downs, P.W., Gregory, K.J., Brookes, A., 1991. How integrated is river basin management? Environmental Management, 15(3), 299-309.
Downs, P.W., Simon, A., 2001. Fluvial geomorphological analysis of the recruitment of large woody debris in the Yalobusha River network, Central Mississippi, USA. Geomorphology, 37(1-2), 65-91.
Du, J., Xie, H., Hu, Y., Xu, Y., Xu, C., 2009. Development and testing of a new storm runoff routing approach based on time variant spatially distributed travel time method. Journal of Hydrology, 369(1-2), 44-54.
Dunkerley, D.L., 1999. Banded chenopod shrublands of arid Australia: modelling responses to interannual rainfall variability with cellular automata. Ecological modelling, 121(2), 127-138.
Dury, G.H., 1981. An Introduction to Environmental Systems. Heinemann.
Dust, D., Wohl, E.E., 2012. Characterization of the hydraulics at natural step crests in step-pool streams via weir flow concepts. Water Resources Research, 48(9), W09542.
EA, 2009a. Flooding in England: A National Assessment of Flood Risk, Environment Agency, Bristol.
EA, 2009b. Investing for the future, Flood and coastal risk management in England, A long-term investment strategy, Environment Agency, Bristol.
Ehrman, T.P., Lamberti, G.A., 1992. Hydraulic and particulate matter retention in a 3rd-order Indiana stream. Journal of the North American Benthological Society, 11(4), 341-349.
Einstein, H.A., Barbarossa, N.L., 1952. River channel roughness. Transactions of the American Society of Civil Engineers(117), 1121-1146.
Escarameia, M., Karanxha, A., Tagg, A., 2007. Quantifying the flood resilience properties of walls in typical UK dwellings. Building Services Engineering Research and Technology, 28(3), 249-263.
EU, 2000. Directive 2000/60/EC of the European Parliament and of the Council establishing a framework for the Community action in the field of water policy (EU Water Framework Directive), 2000/60/EC. European Union, EU.
Evans, E., Hall, J., Penning-Rowsell, E., Sayers, P., Thorne, C., Watkinson, A., 2006a. Future flood risk management in the UK, PROCEEDINGS-INSTITUTION OF CIVIL ENGINEERS WATER MANAGEMENT, pp. 53.
Evans, E.P., Hall, J., Penning-Rowsell, E.C., Sayers, P., Thorne, C.R., Watkinson, A., 2006b. Future flood risk management in the UK. Proceedings of the ICE-Water Management, 159(1), 53-61.
Faustini, J.M., Jones, J.A., 2003. Influence of large woody debris on channel morphology and dynamics in steep, boulder-rich mountain streams, western Cascades, Oregon. Geomorphology, 51(1-3), 187-205.
Ferguson, R.I., 2007. Flow resistance equations for gravel-and boulder-bed streams. Water resources research., 43, W05427.
Ferguson, R.I., 2010. Time to abandon the Manning equation? Earth Surface Processes and Landforms, 35(15), 1873-1876.
Fetherston, K.L., 2005. Pattern and process in mountain river valley forests. Ph.D. Thesis, University of Washington, Seattle, WA, USA.
Fetherston, K.L., Naiman, R.J., Bilby, R.E., 1995. Large woody debris, physical process, and riparian forest development in montane river networks of the Pacific Northwest. Geomorphology, 13(1-4), 133-144.

Fierke, M.K., Kauffman, J.B., 2005. Structural dynamics of riparian forests along a black cottonwood successional gradient. Forest Ecology and Management, 215(1), 149-162.

Fischenich, C., Morrow Jr, J., 1999. Streambank Habitat Enhancement with Large Woody Debris, U.S. Army Engineer Research and Development Center, Vicksburg, MS.

Fisher, G.B., Magilligan, F.J., Kaste, J.M., Nislow, K.H., 2010. Constraining the timescales of sediment sequestration associated with large woody debris using cosmogenic ^{7}Be. Journal of geophysical research, 115(F1), 1-19.

Fitzjohn, C., Ternan, J.L., Williams, A.G., 1998. Soil moisture variability in a semi-arid gully catchment: implications for runoff and erosion control. Catena, 32(1), 55-70.

Fjeldstad, H.P., Barlaup, B.T., Stickler, M., Gabrielsen, S.E., Alfredsen, K., 2012. Removal of weirs and the influence on physical habitat for salmonids in a Norwegian River. River Research and Applications, 28(6), 753-763.

Fonda, R.W., 1974. Forest succession in relation to river terrace development in Olympic National Park, Washington. Ecology, 927-942.

Forestry Commission, 2011. Restoration of SSSI sites in the New Forest (unpublished dataset). In: S. Oakley, S. Weymouth (Eds.). Forestry Commission, Queen's House, Lyndhurst, UK.

Francis, R.A., Petts, G.E., Gurnell, A.M., 2008. Wood as a driver of past landscape change along river corridors. Earth Surface Processes and Landforms, 33(10), 1622-1626.

Frangi, J.L., Richter, L.L., Barrera, M.D., Aloggia, M., 1997. Decomposition of Nothofagus fallen woody debris in forests of Tierra del Fuego, Argentina. Canadian Journal of Forest Research, 27(7), 1095-1102.

Franklin, J.F., Cromack Jr, K., Denison, W., McKee, A., Maser, C., Sedell, J.R., Swanson, F.J., Juday, G., 1981. Ecological characteristics of old-growth Douglas-fir forests. 118, U.S.D.A. Forest Service, Pacific Northwest Forest and Range Experimental Station, Corvallis, Oregon, USA.

Franklin, J.F., Shugart, H.H., Harmon, M.E., 1987. Tree death as an ecological process. Bioscience, 37(8), 550-556.

Freer, J., McDonnell, J.J., Beven, K.J., Peters, N.E., Burns, D.A., Hooper, R.P., Aulenbach, B., Kendall, C., 2002. The role of bedrock topography on subsurface storm flow. Water Resources Research, 38(12), 1269.

Freeze, R.A., 1974. Streamflow generation. Reviews of Geophysics, 12(4), 627-647.

Frelich, L.E., Graumlich, L.J., 1994. Age-class distribution and spatial patterns in an old-growth hemlock-hardwood forest. Canadian Journal of Forest Research, 24, 1939-1947.

Frissell, C., Nawa, R., 1992. Incidence and causes of physical failure of artificial habitat structures in streams of western Oregon and Washington. North American Journal of Fisheries Management, 12(1), 182-197.

Frissell, C.A., Liss, W.J., Warren, C.E., Hurley, M.D., 1986. A hierarchical framework for stream habitat classification: viewing streams in a watershed context. Environmental management, 10(2), 199-214.

Fritsch, J.-M., 1990. Les effets du défrichement de la forét amazonienne et de la mise en culture sur l'hydrologie des petits bassins versants. Ph.D., Université des Sciences et Techniques du Languedoc, Montpellier, 392 pp.

Gallart, F., Llorens, P., 2003. Catchment management under environmental change: impact of land cover change on water resources. Water International, 28(3), 334-340.
George, A.R., 2003. The Babylonian Gilgamesh epic: introduction, critical edition and cuneiform texts, 1. Oxford University Press.
Ghavasieh, A.R., Poulard, C., Paquier, A., 2006. Effect of roughened strips on flood propagation: Assessment on representative virtual cases and validation. Journal of Hydrology, 318(1), 121-137.
Gibling, M.R., Davies, N.S., 2012. Palaeozoic landscapes shaped by plant evolution. Nature Geoscience, 5(2), 99-105.
Gilvear, D.J., 1999. Fluvial geomorphology and river engineering: future roles utilizing a fluvial hydrosystems framework. Geomorphology, 31(1-4), 229-245.
Golding, B., Clark, P., May, B., 2005. The Boscastle flood: Meteorological analysis of the conditions leading to flooding on 16 August 2004. Weather, 60(8), 230-235.
Gomi, T., Sidle, R.C., Bryant, M.D., Woodsmith, R.D., 2001. The characteristics of woody debris and sediment distribution in headwater streams, southeastern Alaska. Canadian Journal of Forest Research, 31(8), 1386-1399.
Google, 2012. Google Earth (version 6.2) [software]. Google Inc.
Govers, G., 1991. Rill erosion on arable land in central Belgium: rates, controls and predictability. Catena, 18(2), 133-155.
Govers, G., Takken, I., Helming, K., 2000. Soil roughness and overland flow. Agronomie, 20(2), 131-146.
Graham, R.L.L., 1981. Biomass dynamics of dead Douglas-fir and western hemlock boles in mid-elevation forests of the Cascade Range. Ph.D., Oregon State University, Oregon, USA.
Grant, G.E., Lewis, S.L., Swanson, F.J., Cissel, J.H., McDonnell, J.J., 2008. Effects of forest practices on peak flows and consequent channel response: a state-of-science report for western Oregon and Washington. 1437927130, US Department of Agriculture, Forest Service, Pacific Northwest Research Station.
Grant, M.J., Edwards, M.E., 2008. Conserving idealized landscapes: past history, public perception and future management in the New Forest (UK). Vegetation History and Archaeobotany, 17(5), 551-562.
Gregory, K.J., 1992. Vegetation and river channel process interactions. In: P.J. Boon, P. Calow, G.E. Petts (Eds.), River conservation and management. Wiley, Chichester.
Gregory, K.J., Davis, R.J., Tooth, S., 1993. Spatial distribution of coarse woody debris dams in the Lymington Basin, Hampshire, UK. Geomorphology, 6(3), 207-224.
Gregory, K.J., Gurnell, A.M., Hill, C.T., 1985. The permanence of debris dams related to river channel processes. Hydrological Sciences Journal-Journal Des Sciences Hydrologiques, 30(3), 371-381.
Gregory, S.V., Swanson, F.J., McKee, W.A., Cummins, K.W., 1991. An ecosystem perspective of riparian zones. Bioscience, 41(8), 540-551.
Grette, G.B., 1985. The Role of Large Organic Debris in Juvenile Salmonid Rearing Habitat in Small Streams. M.Sc, University of Washington, Seattle, Washington, 210 pp.
Griffiths, G.A., 1981. Flow resistance in coarse gravel bed rivers. Journal of the Hydraulics Division, 107(7), 899-918.

Gurnell, A.M., 1998. The hydrogeomorphological effects of beaver dam-building activity. Progress in Physical Geography, 22(2), 167-189.

Gurnell, A.M., 2003. Wood storage and mobility. In: S.V. Gregory, K.L. Boyer, A.M. Gurnell (Eds.), Ecology and Management of Wood in World Rivers. American Fisheries Society Symposium. American Fisheries Society, Bethesda, pp. 75-91.

Gurnell, A.M., 2014. Plants as river system engineers. Earth Surface Processes and Landforms, 39(1), 4-25.

Gurnell, A.M., Gregory, K.J., 1995. Interactions between Seminatural Vegetation and Hydrogeomorphological Processes. Geomorphology, 13(1-4), 49-69.

Gurnell, A.M., Gregory, K.J., Petts, G.E., 1995. The role of coarse woody debris in forest aquatic habitats: Implications for management. Aquatic Conservation: Marine and Freshwater Ecosystems, 5(2), 143-166.

Gurnell, A.M., Petts, G.E., Harris, N., Ward, J.V., Tockner, K., Edwards, P.J., Kollmann, J., 2000. Large wood retention in river channels: the case of the Fiume Tagliamento, Italy. Earth Surface Processes and Landforms, 25(3), 255-275.

Gurnell, A.M., Piegay, H., Swanson, F.J., Gregory, S.V., 2002. Large wood and fluvial processes. Freshwater Biology, 47(4), 601-619.

Gurnell, A.M., Sweet, R., 1998. The distribution of large woody debris accumulations and pools in relation to woodland stream management in a small, low-gradient stream. Earth Surface Processes and Landforms, 23(12), 1101-1121.

Gurnell, A.M., Tockner, K., Edwards, P.J., Petts, G.E., 2005. Effects of deposited wood on biocomplexity of river corridors. Frontiers in Ecology and the Environment, 3(7), 377-382.

Haga, H., Kumagai, T., Otsuki, K., Ogawa, S., 2002. Transport and retention of coarse woody debris in mountain streams: An in situ field experiment of log transport and a field survey of coarse woody debris distribution. Water Resources Research, 38(8), 1-16.

Halcrow, 2001. ISIS flow user manual, Vol. 1. HR Wallingford.

Hannaford, J., Marsh, T., 2007. High-flow and flood trends in a network of undisturbed catchments in the UK. International Journal of Climatology, 28(10), 1325-1338.

Hanson, J.S., Malanson, G.P., Armstrong, M.P., 1990. Landscape fragmentation and dispersal in a model of riparian forest dynamics. Ecological modelling, 49(3), 277-296.

Harmon, M.E., 1982. Decomposition of standing dead trees in the southern Appalachian Mountains. Oecologia, 52(2), 214-215.

Harmon, M.E., Franklin, J.F., Swanson, F.J., Sollins, P., Gregory, S.V., Lattin, J.D., Anderson, N.H., Cline, S.P., Aumen, N.G., Sedell, J.R., Lienkaemper, G.W., Cromack, K., Cummins, K.W., 1986. Ecology of coarse woody debris in temperate ecosystems. Advances in ecological research. Academic Press.

Harper, D.M., Ebrahimnezhad, M., Taylor, E., Dickinson, S., Decamp, O., Verniers, G., Balbi, T., 1999. A catchment-scale approach to the physical restoration of lowland UK rivers. Aquatic Conservation: Marine and Freshwater Ecosystems, 9(1), 141-157.

Harr, R.D., 1986. Effects of clearcutting on rain-on-snow runoff in western Oregon: A new look at old studies. Water Resources Research, 22(7), 1095-1100.

Hart, B.T., Bailey, P., Edwards, R., Hortle, K., James, K., McMahon, A., Meredith, C., Swadling, K., 1991. A review of the salt sensitivity of the Australian freshwater biota. Hydrobiologia, 210(1), 105-144.

Hasegawa, H., Yoshimura, T., 2003. Application of dual-frequency GPS receivers for static surveying under tree canopies. Journal of Forest Research, 8(2), 103-110.
Hawk, G.M., Zobel, D.B., 1974. Forest succession on alluvial land forms of the McKenzie River valley, Oregon. Northwest Science, 48(4), 245-265.
Heathwaite, A., Burt, T., Trudgill, S., 1990. Land-use controls on sediment production in a lowland catchment, south-west England. In: J. Boardman, I. Foster, J. Dearing (Eds.), Soil erosion on agricultural land. Proceedings of a workshop sponsored by the British Geomorphological Research Group, . John Wiley & Sons Ltd., Coventry, UK, pp. 69-86.
HEC-RAS, 2010. HEC-RAS River analysis system. User's Manual, Ver. 4.1. US Army Corps of Engineers, Davis, CA.
Hedman, C.W., Van Lear, D.H., Swank, W.T., 1996. In-stream large woody debris loading and riparian forest seral stage associations in the southern Appalachian Mountains. Canadian Journal of Forest Research, 26(7), 1218-1227.
Helming, K., Römkens, M.J.M., Prasad, S.N., 1998. Surface roughness related processes of runoff and soil loss: a flume study. Soil Science Society of America Journal, 62(1), 243-250.
Henderson, F.M., 1989. Open channel flow. Macmillan series in civil engineering. Macmillan.
Herget, J., 2000. Holocene development of the River Lippe valley, Germany: a case study of anthropogenic influence. Earth Surface Processes and Landforms, 25(3), 293-305.
Herrera Inc, 2006. Conceptual Design Guidelines: Application of Engineered Logjams, Scottish Environmental Protection Agency, Seattle, Washington.
Hey, R.D., 1979. Flow resistance in gravel-bed rivers. Journal of the Hydraulics Division, 105(4), 365-379.
Hey, R.D., 1988. Bar form resistance in gravel-bed rivers. Journal of Hydraulic Engineering, 114(12), 1498-1508.
Hibbert, A.R., 1967. Forest treatment effects on water yield. In: W.E. Sopper, H.W. Lull (Eds.), Forest Hydrology, Proceedings of a National Science Foundation Advanced Science Seminar. Pergamon Press, Oxford, pp. 527-543.
Higler, L.W.G., 1993. The riparian community of north west European lowland streams. Freshwater Biology, 29(2), 229-241.
Hilt, D.E., Teck, R.M., 1989. NE-TWIGS: An individual-tree growth and yield projection system for the northeastern United States. The Compiler, 7(2), 10-16.
Hogan, A.E., Nicholson, J.C., 1987. Sperm motility of sooty grunter, Hephaestus fuliginosus (Macleay), and jungle perch, Kuhlia rupestris (Lacepede), in different salinities. Marine and Freshwater Research, 38(4), 523-528.
Hogan, D.L., 1987. The influence of large organic debris on channel recovery in the Queen Charlotte Islands, British Columbia, Canada. IN: Erosion and Sedimentation in the Pacific Rim. IAHS Publication(165).
Homer, 1961. The Odyssey William Heinemann Ltd, London, pp. 427.
Homer, 2008. The Iliad. Oxford Paperbacks, Oxford, pp. 496.
Horritt, M.S., Bates, P.D., 2001. Effects of spatial resolution on a raster based model of flood flow. Journal of Hydrology, 253(1-4), 239-249.
Hosmer, D.W., Lemeshow, S., 2000. Applied Logistic Regression. John Wiley & Sons.
Howe, J., White, I., 2001. Flooding: are we ignoring the real problem and solution? Regional Studies, 35(4), 368-370.

Huber, M.O., Eastaugh, C.S., Gschwantner, T., Hasenauer, H., Kindermann, G., Ledermann, T., Lexer, M.J., Rammer, W., Schörghuber, S., Sterba, H., 2013. Comparing simulations of three conceptually different forest models with National Forest Inventory data. Environmental Modelling & Software, 40, 88-97.

Huff, M.H., 1984. Post-fire succession in the Olympic Mountains, Washington: Forest vegetation, fuels and avifauna. Ph.D., University of Washington, Seattle, USA.

Hughes, F.M.R., Colston, A., Mountford, J.O., 2005. Restoring riparian ecosystems: the challenge of accommodating variability and designing restoration trajectories. Ecology and Society, 10(1), 12.

Hundecha, Y., Bárdossy, A., 2004. Modeling of the effect of land use changes on the runoff generation of a river basin through parameter regionalization of a watershed model. Journal of Hydrology, 292(1), 281-295.

Hyatt, T.L., Naiman, R.J., 2001. The residence time of large woody debris in the Queets River, Washington, USA. Ecological Applications, 11(1), 191-202.

Hygelund, B., Manga, M., 2003. Field measurements of drag coefficients for model large woody debris. Geomorphology, 51(1), 175-185.

Imeson, A.C., Verstraten, J.M., 1988. Rills on badland slopes: a physico-chemically controlled phenomenon. In: A.C. Imeson, M. Sala (Eds.), Geomorphic Processes in Environments with Strong Seasonal Contrasts, Vol. I: Hillslope Processes. Catena Supplement 12. Catena Verlag, Reiskirchen, pp. 139-150.

IPCC, 2007. Climate change 2007: The Physical Science Basis, Summary for Policy Makers, Working Group I Contribution to the

Intergovernmental Panel on Climate Change, Fourth Assessment Report, Geneva.

Jarrett, R.D., 1985. Determination of roughness coefficients for streams in Colorado, US Department of the Interior, Lakewood, Colorado, US.

Jeffries, R., Darby, S.E., Sear, D.A., 2003. The influence of vegetation and organic debris on flood-plain sediment dynamics: case study of a low-order stream in the New Forest, England. Geomorphology, 51(1-3), 61-80.

Johnson, C., Penning-Rowsell, E., Tapsell, S., 2007. Aspiration and reality: flood policy, economic damages and the appraisal process. Area, 39(2), 214-223.

Johnson, C.L., Priest, S.J., 2008. Flood risk management in England: A changing landscape of risk responsibility? International Journal of Water Resources Development, 24(4), 513-525.

Johnson, S.L., Swanson, F.J., Grant, G.E., Wondzell, G.M., 2000. Riparian forest disturbances by a mountain flood-the influence of floated wood. Hydrological Processes, 14(16-17), 3031-3050.

Jones, C.G., Lawton, J.H., Shachak, M., 1994. Organisms as ecosystem engineers. Oikos, 373-386.

Jones, E.B.D., Helfman, G.S., Harper, J.O., Bolstad, P.V., 1999. Effects of Riparian Forest Removal on Fish Assemblages in Southern Appalachian Streams. Conservation Biology, 13(6), 1454-1465.

Jones, T.A., Daniels, L.D., 2008. Dynamics of large woody debris in small streams disturbed by the 2001 Dogrib fire in the Alberta foothills. Forest Ecology and Management, 256(10), 1751-1759.

Jones, T.A., Daniels, L.D., Powell, S.R., 2011. Abundance and function of large woody debris in small, headwater streams in the Rocky Mountain foothills of Alberta, Canada. River Research and Applications, 27(3), 297-311.

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(1), 110-127.

Kallis, G., Butler, D., 2001. The EU water framework directive: measures and implications. Water policy, 3(2), 125-142.

Kasprak, A., Magilligan, F.J., Nislow, K.H., Snyder, N.P., 2012. A Lidar-derived evaluation of watershed-scale large woody debris sources and recruitment mechanisms: Coastal Maine, USA. River Research and Applications, 28(9), 1462-1476.

Keeton, W.S., Kraft, C.E., Warren, D.R., 2007. Mature and old-growth riparian forests: structure, dynamics, and effects on Adirondack stream habitats. Ecological Applications, 17(3), 852-868.

Keim, F., Skaugset, E., Bateman, S., 2000. Dynamics of coarse woody debris placed in three Oregon streams. Forest Science, 46(1), 13-22.

Keller, E.A., MacDonald, A., Tally, T., Merritt, N.J., 1995. Effects of large organic debris on channel morphology and sediment storage in selected tributaries of Redwood Creek. In: K.M. Nolan, H.M. Kelsey, D.C. Marron (Eds.), Geomorphic Processes and Aquatic Habitat in the Redwood Creek Drainage Basin. U.S. Geological Survey Profesional Paper. USGS, Washington DC, pp. 29.

Keller, E.A., Swanson, F.J., 1979. Effects of large organic material on channel form and fluvial processes. Earth surface processes, 4(4), 361-380.

Keller, E.A., Tally, T., 1982. Effects of large organic debris on channel form and fluvial processes in the coastal redwood environment. In: D.D. Rhodes, G.P. Williams (Eds.), Adjustments of the fluvial system. Geomorphology Symposia series. George Allen Unwin, London, pp. 169–197.

Kelman, I., 2001. The autumn 2000 floods in England and flood management. Weather, 56(10), 346-360.

Kemp, J.L., Harper, D.M., Crosa, G.A., 1999. Use of 'functional habitats' to link ecology with morphology and hydrology in river rehabilitation. Aquatic Conservation: Marine and Freshwater Ecosystems, 9(1), 159-178.

King, D.I., Nislow, K.H., Brooks, R.T., DeGraaf, R.M., Yamasaki, M., 2011. Early-successional forest ecosystems: far from "forgotten". Frontiers in Ecology and the Environment, 9(6), 319-320.

King, J.S., 2013. Eucalyptus and Water Use in South Africa. International Journal of Forestry Research, 2013.

Kirkby, M.J., Bracken, L., Reaney, S., 2002. The influence of land use, soils and topography on the delivery of hillslope runoff to channels in SE Spain. Earth Surface Processes and Landforms, 27(13), 1459-1473.

Kitts, D.R., 2011. The Hydraulic and Hydrological Performance of Large Wood Accumulations in a Low-order Forest Stream. Ph.D, University of Southampton, Southampton, 367 pp.

Kleinen, T., Petschel-Held, G., 2007. Integrated assessment of changes in flooding probabilities due to climate change. Climatic Change, 81(3), 283-312.

Klopf, M., Hasenauer, H., 2012. Adapting the tree growth model MOSES for Sitka spruce in Great Britain, Forest models for research and decision support in sustainable forest management, Pierroton, Bordeaux, France.

Klostermaier, K.K., 2007. A survey of Hinduism. SUNY Press, Albany, NY, USA.

Kondolf, G.M., 1998. Lessons learned from river restoration projects in California. Aquatic Conservation: Marine and Freshwater Ecosystems, 8(1), 39-52.
Kondolf, G.M., Boulton, A.J., O'Daniel, S., Poole, G.C., Rahel, F.J., Stanley, E.H., Wohl, E.E., Bång, A., Carlstrom, J., Cristoni, C., 2006. Process-based ecological river restoration: visualizing three-dimensional connectivity and dynamic vectors to recover lost linkages. Ecology and Society, 11(2), 5.
Kraft, C.E., Schneider, R.L., Warren, D.R., 2002. Ice storm impacts on woody debris and debris dam formation in northeastern US streams. Canadian Journal of Fisheries and Aquatic Sciences, 59(10), 1677-1684.
Krejčí, L., Máčka, Z., 2012. Anthropogenic controls on large wood input, removal and mobility: examples from rivers in the Czech Republic. Area, 44(2), 226-236.
Kreutzweiser, D.P., Good, K.P., Sutton, T.M., 2005. Large woody debris characteristics and contributions to pool formation in forest streams of the Boreal Shield. Canadian Journal of Forest Research, 35(5), 1213-1223.
Krishnaswamy, J., Bonell, M., Venkatesh, B., Purandara, B.K., Lele, S., Kiran, M.C., Reddy, V., Badiger, S., Rakesh, K.N., 2012. The rain-runoff response of tropical humid forest ecosystems to use and reforestation in the Western Ghats of India. Journal of Hydrology, 472-473, 216-237.
Kupfer, J.A., Malanson, G.P., 1993. Observed and modeled directional change in riparian forest composition at a cutbank edge. Landscape Ecology, 8(3), 185-199.
Laarmann, D., Korjus, H., Sims, A., Stanturf, J.A., Kiviste, A., Köster, K., 2009. Analysis of forest naturalness and tree mortality patterns in Estonia. Forest Ecology and management, 258, S187-S195.
Lambert, F.H., Stott, P.A., Allen, M.R., Palmer, M.A., 2004. Detection and attribution of changes in 20th century land precipitation. Geophysical Research Letters, 31(10).
Lambert, R.L., Lang, G.E., Reiners, W.A., 1980. Loss of mass and chemical change in decaying boles of a subalpine balsam fir forest. Ecology, 1460-1473.
Lambert, W.W.G., Millard, A.A.R., Civil, M., 1999. Atra-Ḫasīs: the Babylonian story of the Flood. Eisenbrauns.
Lamond, J.E., Proverbs, D.G., 2009. Resilience to flooding: lessons from international comparison. Proceedings of the ICE-Urban Design and Planning, 162(2), 63-70.
Lane, P.N.J., Best, A.E., Hickel, K., Zhang, L., 2005. The response of flow duration curves to afforestation. Journal of Hydrology, 310(1), 253-265.
Lane, S., Morris, J., O'Connell, P., Quinn, P., 2007. 18 Managing the rural landscape. Future flooding and coastal erosion risks, 297.
Lane, S.N., 2003. More floods, less rain? Changing hydrology in a Yorkshire context. In: M. Atherden (Ed.), Global Warming in a Yorkshire Context, York.
Lane, S.N., 2008. Climate change and the summer 2007 floods in the UK. Geography, 93(2), 91-97.
Larson, M.G., Booth, D.B., Morley, S.A., 2001. Effectiveness of large woody debris in stream rehabilitation projects in urban basins. Ecological Engineering, 18(2), 211-226.
Lasanta, T., Garcıa-Ruiz, J.M., Pérez-Rontomé, C., Sancho-Marcén, C., 2000. Runoff and sediment yield in a semi-arid environment: the effect of land management after farmland abandonment. Catena, 38(4), 265-278.
Laser, M., Jordan, J., Nislow, K., 2009. Riparian forest and instream large wood characteristics, West Branch Sheepscot River, Maine, USA. Forest Ecology and Management, 257(7), 1558-1565.

Latterell, J.J., Naiman, R.J., 2007. Sources and dynamics of large logs in a temperate floodplain river. Ecological Applications, 17(4), 1127-1141.
Leopold, L.B., Maddock, T., 1953. The hydraulic geometry of stream channels and some physiographic implications, US Department of the Interior, Washington, US.
Leopold, L.B., Wolman, M.G., Miller, J.P., 1964. Fluvial processes in geomorphology. Dover Publications.
Lester, A.M., Beatty, I.D., Nislow, K.H., 2003. Upland and Riparian Northeastern Coarse Woody Debris (NE-CWD) Model: User's Guide, University of Massachusetts, Amherst, USA.
Lewis, J., Mori, S.R., Keppeler, E.T., Ziemer, R.R., 2001. Impacts of logging on storm peak flows, flow volumes and suspended sediment loads in Caspar Creek, California. Land use and watersheds: human influence on hydrology and geomorphology in urban and forest areas, 85-125.
Lewis, V., 2008. The Chalkstream Habitat Manual, Wild Trout Trust, http://www.wildtrout.org/content/wtt-publications.
Lewis, V., 2010. The Upland Rivers Habitat Manual, Wild Trout Trust, http://www.wildtrout.org/content/wtt-publications.
Lienkaemper, G.W., Swanson, F.J., 1987. Dynamics of large woody debris in streams in old-growth Douglas-fir forests. Canadian Journal of Forest Research, 17(2), 150-156.
Linstead, C., Gurnell, A.M., 1998. Large woody debris in British headwater rivers: physical habitat and management guidelines. R&D Technical Report W185, School of Geography and Environmental Sciences, University of Birmingham, Bristol, UK.
Liu, Y.B., Gebremeskel, S., De Smedt, F., Hoffmann, L., Pfister, L., 2003. A diffusive transport approach for flow routing in GIS-based flood modeling. Journal of Hydrology, 283(1-4), 91-106.
Liu, Y.B., Gebremeskel, S., De Smedt, F., Hoffmann, L., Pfister, L., 2004. Simulation of flood reduction by natural river rehabilitation using a distributed hydrological model. Hydrology and Earth System Sciences Discussions, 8(6), 1129-1140.
Liu, Z.J., Malanson, G.P., 1992. Long-term cyclic dynamics of simulated riparian forest stands. Forest Ecology and Management, 48(3–4), 217-231.
Lofroth, E., 1998. The dead wood cycle. In: J. Voller, S. Harrison (Eds.), Conservation biology principles for forested landscapes. UBC Press, Vancouver, BC, pp. 185-214.
Lombardi, F.L.F., Chirici, G., Marchetti, M.M.M., Tognetti, R.T.R., Lasserre, B., Corona, P., Barbati, A., Ferrari, B., Di Paolo, S., Giuliarelli, D., 2010. Deadwood in forest stands close to old-growthness under mediterranean conditions in the Italian peninsula. L'Italia Forestale e Montana, 6.
Lombardi, F.L.F., Klopcic, M., Di Martino, P., Tognetti, R.T.R., Chirici, G., Boncina, A., Marchetti, M.M.M., 2011. Comparison of forest stand structure and management of silver fir–European beech forests in the Central Apennines, Italy and in the Dinaric Mountains, Slovenia. Plant Biosystems, 1.
López-Moreno, J.I., Beguería, S., García-Ruiz, J.M., 2006. Trends in high flows in the central Spanish Pyrenees: response to climatic factors or to land-use change? Hydrological Sciences Journal, 51(6), 1039-1050.

Lorimer, C.G., 1977. The Presettlement Forest and Natural Disturbance Cycle of Northeastern Maine. Ecology, 58(1), 139-148.
Lorimer, C.G., White, A.S., 2003. Scale and frequency of natural disturbances in the northeastern US: implications for early successional forest habitats and regional age distributions. Forest Ecology and Management, 185(1-2), 41-64.
Lowrance, R., 1992. Groundwater nitrate and denitrification in a coastal plain riparian forest. Journal of Environmental Quality, 21(3), 401-405.
Lowrance, R., 1998. Riparian forest ecosystems as filters for nonpoint-source pollution. In: M.L. Pace, P.M. Groffmann (Eds.), Successes, limitations and frontiers in ecosystem science. Springer, pp. 113–141.
Lowrance, R., Altier, L.S., Newbold, J.D., Schnabel, R.R., Groffman, P.M., Denver, J.M., Correll, D.L., Gilliam, J.W., Robinson, J.L., Brinsfield, R.B., 1997. Water quality functions of riparian forest buffers in Chesapeake Bay watersheds. Environmental Management, 21(5), 687-712.
Ludwig, B., Boiffin, J., Auzet, A.-V., 1995. Hydrological structure and erosion damage caused by concentrated flow in cultivated catchments. Catena, 25(1), 227-252.
Ludwig, J.A., Tongway, D.J., Eager, R.W., Williams, R.J., Cook, G.D., 1999a. Fine-scale vegetation patches decline in size and cover with increasing rainfall in Australian savannas. Landscape Ecology, 14(6), 557-566.
Ludwig, J.A., Tongway, D.J., Marsden, S.G., 1999b. Stripes, strands or stipples: modelling the influence of three landscape banding patterns on resource capture and productivity in semi-arid woodlands, Australia. Catena, 37(1), 257-273.
Ludwig, J.A., Wiens, J.A., Tongway, D.J., 2000. A scaling rule for landscape patches and how it applies to conserving soil resources in savannas. Ecosystems, 3(1), 84-97.
Ludwig, J.A., Wilcox, B.P., Breshears, D.D., Tongway, D.J., Imeson, A.C., 2005. Vegetation patches and runoff-erosion as interacting ecohydrological processes in semiarid landscapes. Ecology, 86(2), 288-297.
Máčka, Z., Krejčí, L., 2010. Large woody debris mobility and accumulation by an extreme flood-an example from the Dyje River, Czech Republic, EGU General Assembly.
MacVicar, B., 2013. Local head loss coefficients of riffle-pools in gravel bed rivers. Journal of Hydraulic Engineering, 139(11), 1193-1198.
MacVicar, B., Piégay, H., 2012. Implementation and validation of video monitoring for wood budgeting in a wandering piedmont river, the Ain River (France). Earth Surface Processes and Landforms, 37(12), 1272-1289.
MacVicar, B.J., Piégay, H., Henderson, A., Comiti, F., Oberlin, C., Pecorari, E., 2009. Quantifying the temporal dynamics of wood in large rivers: field trials of wood surveying, dating, tracking, and monitoring techniques. Earth Surface Processes and Landforms, 34(15), 2031-2046.
Maidment, D., 1993. Developing a spatially distributed unit hydrograph by using GIS, HydroGIS 93: Application of Geographic Information Systems in Hydrology and Water Resources. IAHS, Vienna, pp. 181-192.
Maidment, D., Olivera, F., Calver, A., Eatherall, A., Fraczek, W., 1996. Unit hydrograph derived from a spatially distributed velocity field. Hydrological Processes, 10, 831-844.

Malanson, G.P., Kupfer, J.A., 1993. Simulated fate of leaf litter and large woody debris at a riparian cutbank. Canadian Journal of Forest Research, 23(4), 582-590.
Manga, M., Kirchner, J.W., 2000. Stress partitioning in streams by large woody debris. Water Resources Research, 36(8), 2373-2379.
Manners, R.B., Doyle, M.W., Small, M.J., 2007. Structure and hydraulics of natural woody debris jams. Water Resources Research, 43(6), DOI: 10.1029/2006WR004910.
Marcus, W.A., Marston, R.A., Colvard, C.R., 2002. Mapping the spatial and temporal distributions of woody debris in streams of the Greater Yellowstone Ecosystem, USA. Geomorphology, 44(3-4), 323-335.
Marsh, T., 2008. A hydrological overview of the summer 2007 floods in England and Wales. Weather, 63(9), 274-279.
Marshall, M., Francis, O., Frogbrook, Z., Jackson, B., McIntyre, N., Reynolds, B., Solloway, I., Wheater, H., Chell, J., 2009. The impact of upland land management on flooding: results from an improved pasture hillslope. Hydrological Processes, 23(3), 464-475.
Martin, D.J., Grotefendt, R.A., 2006. Stand mortality in buffer strips and the supply of woody debris to streams in Southeast Alaska. Canadian Journal of Forest Research, 37(1), 36-49.
Mattson, K.G., Swank, W.T., Waide, J.B., 1987. Decomposition of Woody Debris in a Regenerating, Clear-Cut Forest in the Southern Appalachians. Canadian Journal of Forest Research, 17(7), 712-721.
May, C.L., Gresswell, R.E., 2003. Large wood recruitment and redistribution in headwater streams in the southern Oregon Coast Range, USA. Canadian Journal of Forest Research, 33(8), 1352-1362.
Maynord, S.T., 1991. Flow resistance of riprap. Journal of Hydraulic Engineering, 117(6), 687-696.
McCarthy, J.J., Canziana, O.F., Leary, N.A., Dokken, D.J., White, K.S., 2001. Climate change 2001: impacts, adaptation, and vulnerability: contribution of Working Group II to the third assessment report of the Intergovernmental Panel on Climate Change. Cambridge University Press.
McCulloch, J.S.G., Robinson, M., 1993. History of forest hydrology. Journal of Hydrology, 150(2–4), 189-216.
McDade, M.H., Swanson, F.J., McKee, W.A., Franklin, J.F., Van Sickle, J., 1990. Source distances for coarse woody debris entering small streams in western Oregon and Washington. Canadian Journal of Forest Research, 20(3), 326-330.
McDonald, A., Lane, S.N., Haycock, N.E., Chalk, E.A., 2004. Rivers of dreams: on the gulf between theoretical and practical aspects of an upland river restoration. Transactions of the Institute of British Geographers, 29(3), 257-281.
McDonnell, J.J., 2003. Where does water go when it rains? Moving beyond the variable source area concept of rainfall-runoff response. Hydrological processes, 17(9), 1869-1875.
McDonnell, J.J., Sivapalan, M., Vaché, K., Dunn, S., Grant, G., Haggerty, R., Hinz, C., Hooper, R., Kirchner, J., Roderick, M.L., Selker, J., Weiler, M., 2007. Moving beyond heterogeneity and process complexity: A new vision for watershed hydrology. Water Resources Research, 43(7), W07301.

McKay, M.D., Beckman, R.J., Conover, W.J., 1979. Comparison of three methods for selecting values of input variables in the analysis of output from a computer code. Technometrics, 21(2), 239-245.

Meleason, M.A., 2001. A simulation model of wood dynamics in Pacific Northwest streams. Ph.D., Oregon State University, Corvallis, Oregon.

Meleason, M.A., Davies-Colley, R.J., Hall, G.M.J., 2007. Characterizing the variability of wood in streams: simulation modelling compared with multiple-reach surveys. Earth Surface Processes and Landforms, 32(8), 1164-1173.

Mendel, H.G., 1996. Hochwasser - Gedanken uber Ursachen und Vorsorge aus hydrologischer Sicht., 1022. Bundesanstalt fu¨r Gewasserkunde, Koblenz.

Met Office, 2013. The wet autumn of 2000. URL: http://www.metoffice.gov.uk/climate/uk/interesting/autumn2000.html. Accessed on: 09/08/2013.

Mikac, S., Klopf, M., Anić, I., Hasenauer, H., 2013. Using the tree growth model MOSES to assess the dynamics of Dinaric old-growth mixed beech–fir forest ecosystems. Plant Biosystems(ahead-of-print), 1-8.

Miller, J.D., Kjeldsen, T.R., Hannaford, J., Morris, D.G., 2013. A hydrological assessment of the November 2009 floods in Cumbria, UK. Hydrology Research, 44(1), 180-197.

Miller, J.D., Morris, D.G., Stewart, E.J., Gibson, H.S., 2012. The November 2009 floods in Cumbria, north-West England-an analysis of the rainfall and river flows in two catchments. In: F. Klijn, T. Schweckendiek (Eds.), Comprehensive Flood Risk Management: Research for Policy and Practice. CRC Press, Taylor & Francis Group, London, pp. 36.

Miller, S.W., Budy, P., Schmidt, J.C., 2010. Quantifying Macroinvertebrate Responses to In Stream Habitat Restoration: Applications of Meta Analysis to River Restoration. Restoration Ecology, 18(1), 8-19.

Millington, C.E., 2007. The Geomorphological dynamics of a restored forested floodplain. PhD, University of Southampton, Southampton, 328 pp.

Millington, C.E., Sear, D.A., 2007. Impacts of river restoration on small-wood dynamics in a low-gradient headwater stream. Earth Surface Processes and Landforms, 32(8), 1204-1218.

Montgomery, D.R., 2003. Wood in rivers: interactions with channel morphology and processes. Geomorphology, 51(1), 1.

Montgomery, D.R., Abbe, T.B., 2006. Influence of logjam-formed hard points on the formation of valley-bottom landforms in an old-growth forest valley, Queets River, Washington, USA. Quaternary Research, 65(1), 147-155.

Montgomery, D.R., Abbe, T.B., Buffington, J.M., Peterson, N.P., Schmidt, K.M., Stock, J.D., 1996a. Distribution of bedrock and alluvial channels in forested mountain drainage basins. Nature, 381(6583), 587-589.

Montgomery, D.R., Buffington, J.M., Peterson, N.P., Schuett-Hames, D., Quinn, T.P., 1996b. Stream-bed scour, egg burial depths, and the influence of salmonid spawning on bed surface mobility and embryo survival. Canadian Journal of Fisheries and Aquatic Sciences, 53(5), 1061-1070.

Montgomery, D.R., Buffington, J.M., Smith, R.D., Schmidt, K.M., Pess, G., 1995. Pool spacing in forest channels. Water Resources Research, 31(4), 1097-1105.

Montgomery, D.R., Collins, B.D., Buffington, J.M., Abbe, T.B., 2003. Geomorphic effects of wood in rivers. In: S.V. Gregory, K.L. Boyer, A.M. Gurnell (Eds.), The

ecology and management of wood in world rivers. American Fisheries Society Symposium, pp. 21-47.
Moore, D.J., Reed, R.H., Stewart, W.D.P., 1985. Responses of cyanobacteria to low level osmotic stress: implications for the use of buffers. Journal of General Microbiology, 131(6), 1267.
Morgan, R.P.C., 1988. Soil erosion and conservation. Soil Science, 145(6), 461.
Mossop, B., Bradford, M.J., 2004. Importance of large woody debris for juvenile chinook salmon habitat in small boreal forest streams in the upper Yukon River basin, Canada. Canadian Journal of Forest Research, 34(9), 1955-1966.
Mott, N., 2006. Managing Woody Debris in Rivers, Streams & Floodplains, Staffordshire Wildlife Trust, Stafford, UK.
Murphy, M.L., Koski, K.V., 1989. Input and depletion of woody debris in Alaska streams and implications for streamside management. North American Journal of Fisheries Management, 9(4), 427-436.
Nagayama, S., Kawaguchi, Y., Nakano, D., Nakamura, F., 2008. Methods for and fish responses to channel remeandering and large wood structure placement in the Shibetsu River Restoration Project in northern Japan. Landscape and Ecological Engineering, 4(1), 69-74.
Naiman, R.J., 1982. Characteristics of sediment and organic carbon export from pristine boreal forest watersheds. Canadian Journal of Fisheries and Aquatic Sciences, 39(12), 1699-1718.
Naiman, R.J., Décamps, H., 1997. The ecology of interfaces: riparian zones. Annual review of Ecology and Systematics, 621-658.
Naiman, R.J., Decamps, H., Pollock, M.M., 1993. The role of riparian corridors in maintaining regional biodiversity. Ecological applications, 3(2), 209-212.
Naiman, R.J., Fetherston, K.L., McKay, S.J., Chen, J., 1998. Riparian forests. In: R.J. Naiman, R.E. Bilby (Eds.), River ecology and management: lessons from the Pacific coastal ecoregion. . Springer-Verlag, New York, pp. 289-323.
Naiman, R.J., Johnston, C.A., Kelley, J.C., 1988. Alteration of North American streams by beaver. Bioscience, 38(11), 753-762.
Naiman, R.J., Melillo, J.M., Hobbie, J.E., 1986. Ecosystem Alteration of Boreal Forest Streams by Beaver (Castor Canadensis). Ecology, 67(5), 1254-1269.
Nakamura, F., Swanson, F.J., 1993. Effects of coarse woody debris on morphology and sediment storage of a mountain stream system in western Oregon. Earth Surface Processes and Landforms, 18, 43-43.
Nakamura, F., Swanson, F.J., Wondzell, S.M., 2000. Disturbance regimes of stream and riparian systems-a disturbance-cascade perspective. Hydrological Processes, 14(16-17), 2849-2860.
Nanson, G.C., Beach, H.F., 1977. Forest succession and sedimentation on a meandering-river floodplain, northeast British Columbia, Canada. Journal of Biogeography, 229-251.
Nash, J.E., Sutcliffe, J.V., 1970. River flow forecasting through conceptual models part I — A discussion of principles. Journal of Hydrology, 10(3), 282-290.
Newson, M.D., Pitlick, J., Sear, D.A., 2002. Running water: fluvial geomorphology and river restoration. In: M.R. Perrow, A.J. Davy (Eds.), Handbook of Ecological Restoration: Principles of restoration. Cambridge University Press, Cambridge, pp. 133-154.

Nicholson, A.R., Wilkinson, M.E., O'Donnell, G.M., Quinn, P.F., 2012. Runoff attenuation features: a sustainable flood mitigation strategy in the Belford catchment, UK. Area, 44(4), 463-469.
Nicolau, J.M., Solé-Benet, A., Puigdefábregas, J., Gutiérrez, L., 1996. Effects of soil and vegetation on runoff along a catena in semi-arid Spain. Geomorphology, 14(4), 297-309.
Niehoff, D., Fritsch, U., Bronstert, A., 2002. Land-use impacts on storm-runoff generation: scenarios of land-use change and simulation of hydrological response in a meso-scale catchment in SW-Germany. Journal of Hydrology, 267(1), 80-93.
Nierenberg, T.R., Hibbs, D.E., 2000. A characterization of unmanaged riparian areas in the central Coast Range of western Oregon. Forest Ecology and Management, 129(1), 195-206.
Nisbet, T.R., Marrington, S., Thomas, H., Broadmeadow, S., Valatin, G., 2011. DEFRA FCERM Multi-objective Flood Management Demonstration project. Project RMP5455: Slowing the flow at Pickering, Department for Environment, Food and Rural Affairs, London.
Nislow, K.H., 2005. Forest change and stream fish habitat: lessons from 'Olde'and New England. Journal of fish biology, 67(sB), 186-204.
Nislow, K.H., 2010. Riparian Management: Alternative Paradigms and Implications for Wild Salmon. In: P. Kemp (Ed.), Salmonid Fisheries: Freshwater Habitat Management. John Wiley & Sons, pp. 164-182.
Nislow, K.H., Lowe, W.H., 2006. Influences of logging history and riparian forest characteristics on macroinvertebrates and brook trout (Salvelinus fontinalis) in headwater streams (New Hampshire, USA). Freshwater Biology, 51(2), 388-397.
Nislow, K.H., Magilligan, F.J., Fassnacht, H., Bechtel, D., Ruesink, A., 2002. Effects of dam impoundment on the flood regime of natural floodplain communities in the Upper Connecticut River. JAWRA Journal of the American Water Resources Association, 38(6), 1533-1548.
Noguchi, S., Tsuboyama, Y., Sidle, R.C., Hosoda, I., 1999. Morphological Characteristics of Macropores and the Distribution of Preferential Flow Pathways in a Forested Slope Segment. Soil Science Society of America Journal, 63(5), 1413-1423.
Nuttle, T., Haefner, J.W., 2007. Design and validation of a spatially explicit simulation model for bottomland hardwood forests. Ecological Modelling, 200(1–2), 20-32.
O'Connell, P., Ewen, J., O'Donnell, G., Quinn, P., 2007. Is there a link between agricultural land-use management and flooding? Hydrology and Earth System Sciences Discussions, 11(1), 96-107.
Odoni, N.A., 2012. Advice on Manning's n roughness coefficient for Lymington River based on field photographs. In: S.J. Dixon (Ed.), Bristol, UK.
Odoni, N.A., Lane, S.N., 2010. Assessment of the impact of upstream land management measures on flood flows in Pickering Beck using OVERFLOW, Durham University, Durham, UK.
Odoni, N.A., Lane, S.N., *In Prep*. OVERFLOW: A reduced complexity hydrological model for investigating spatially distributed flood control measures.
Oleson, J.P., Brandon, C., Cramer, S.M., Cucitore, R., Gotti, E., Hohlfelder, R.L., 2004. The ROMACONS Project: a contribution to the historical and engineering

analysis of hydraulic concrete in roman maritime structures. International Journal of Nautical Archaeology, 33(2), 199-229.
Olivera, F., Maidment, D., 1999. Geographic Information Systems (GIS)-based spatially distributed model for runoff routing. Water Resources Research, 35(4), 115-1164.
Pabst, R.J., Spies, T.A., 1999. Structure and composition of unmanaged riparian forests in the coastal mountains of Oregon, USA. Canadian Journal of Forest Research, 29(10), 1557-1573.
Padhye, A.D., Ghate, H.V., 1992. Sodium chloride and potassium chloride tolerance of different stages of the frog, Microhyla ornata. Herpetological journal, 2(1), 18-23.
Parker, G.R., Peterson, A.W., 1980. Bar resistance of gravel-bed streams. Journal of the Hydraulics Division, 106(10), 1559-1575.
Parrott, A., Brooks, W., Harmar, O., Pygott, K., 2009. Role of rural land use management in flood and coastal risk management. Journal of Flood Risk Management, 2(4), 272-284.
Parry, G., 1966. Osmotic adaptation in fishes. Biological Reviews, 41(3), 392-440.
Pattison, I., Lane, S., 2012a. The link between land-use management and fluvial flood risk. Progress in Physical Geography, 36(1), 72-92.
Pattison, I., Lane, S.N., 2012b. The link between land-use management and fluvial flood risk A chaotic conception? Progress in Physical Geography, 36(1), 72-92.
Pattison, I., Lane, S.N., Hardy, R.J., Reaney, S., 2008. Sub-catchment peak flow magnitude and timing effects on downstream flood risk, BHS 10th National Hydrology Symposium, Exeter.
Pearlstine, L., McKellar, H., Kitchens, W., 1985. Modelling the impacts of a river diversion on bottomland forest communities in the Santee River floodplain, South Carolina. Ecological Modelling, 29(1), 283-302.
Peterjohn, W.T., Correll, D.L., 1984. Nutrient dynamics in an agricultural watershed: observations on the role of a riparian forest. Ecology, 65(5), 1466-1475.
Peterken, G.F., Hughes, F.M.R., 1995. Restoration of floodplain forests in Britain. Forestry, 68(3), 187-202.
Peterken, G.F., Spencer, J., Field, A., 1996. Maintaining the Ancient and Ornamental Woodlands of the New Forest, unpublished Consultation Document, Forestry Commission, Lyndhurst.
Peterson, N.P., 1982. Immigration of juvenile coho salmon (Oncorhynchus kisutch) into riverine ponds. Canadian Journal of Fisheries and Aquatic Sciences, 39(9), 1308-1310.
Pettit, N.E., Naiman, R.J., 2005. Flood-deposited wood debris and its contribution to heterogeneity and regeneration in a semi-arid riparian landscape. Oecologia, 145(3), 434-444.
Phillips, J.D., 2012. Log-jams and avulsions in the San Antonio River Delta, Texas. Earth Surface Processes and Landforms, 37(9), 936-950.
Phipps, R.L., 1979. Simulation of wetlands forest vegetation dynamics. Ecological Modelling, 7(4), 257-288.
Piegay, H., 1997. Interactions between floodplain forests and overbank flows: data from three piedmont rivers of southeastern France. Global Ecology and Biogeography Letters, 6, 187-196.
Piegay, H., 2003. Dynamics of wood in large rivers. In: S.V. Gregory, K.L. Boyer, A.M. Gurnell (Eds.), Ecology and Management of Wood in World Rivers. American

Fisheries Society Symposium. American Fisheries Society, Bethesda, pp. 109-133.
Piegay, H., Gregory, K.J., Bondarev, V., Chin, A., Dahlstrom, N., Elosegi, A., Gregory, S.V., Joshi, V., Mutz, M., Rinaldi, M., 2005. Public perception as a barrier to introducing wood in rivers for restoration purposes. Environmental Management, 36(5), 665-674.
Piégay, H., Gurnell, A.M., 1997. Large woody debris and river geomorphological pattern: examples from SE France and S. England. Geomorphology, 19(1-2), 99-116.
Piégay, H., Thévenet, A., Citterio, A., 1999. Input, storage and distribution of large woody debris along a mountain river continuum, the Drôme River, France. Catena, 35(1), 19-39.
Pinay, G., Ruffinoni, C., Wondzell, S.M., Gazelle, F., 1998. Change in groundwater nitrate concentration in a large river floodplain: denitrification, uptake, or mixing? Journal of the North American Benthological Society, 17(2), 179-189.
Pitlick, J., 1994. Relation between peak flows, precipitation, and physiography for five mountainous regions in the western USA. Journal of Hydrology, 158(3), 219-240.
Pitt, M., 2008. The Pitt Review: Lessons learned from the 2007 summer floods.
Posthumus, H., Hewett, C.J.M., Morris, J., Quinn, P.F., 2008. Agricultural land use and flood risk management: engaging with stakeholders in North Yorkshire. Agricultural Water Management, 95(7), 787-798.
Powell, S.R., Daniels, L.D., Jones, T.A., 2009. Temporal dynamics of large woody debris in small streams of the Alberta foothills, Canada. Canadian Journal of Forest Research, 39(6), 1159-1170.
Prestegaard, K.L., 1983. Bar resistance in gravel bed streams at bankfull stage. Water Resources Research, 19(2), 472-476.
Prudhomme, C., Jakob, D., Svensson, C., 2003. Uncertainty and climate change impact on the flood regime of small UK catchments. Journal of Hydrology, 277(1-2), 1-23.
Puigdefábregas, J., 2005. The role of vegetation patterns in structuring runoff and sediment fluxes in drylands. Earth Surface Processes and Landforms, 30(2), 133-147.
Rajaratnam, N., Katopodis, C., Lodewyk, S., 1988. Hydraulics of offset baffle culvert fishways. Canadian Journal of Civil Engineering, 15(6), 1043-1051.
Randle, T., 2000. Techncial manual for GROMIT (Growth model for individual trees), Forest Research, Forestry Commission, Alice Holt, Surrey, UK.
Raphael, M.G., Morrison, M.L., 1987. Notes: Decay and Dynamics of Snags in the Sierra Nevada, California. Forest Science, 33(3), 774-783.
Record, S.J., 1914. The Mechanical Properties of Wood Including a Discussion of the Factors Affecting the Mechanical Properties, and Methods of Timber Testing. Project Gunthenburg E-Books.
Reed, R.H., Richardson, D.L., Warr, S.R.C., Stewart, W.D.P., 1984. Carbohydrate accumulation and osmotic stress in cyanobacteria. Journal of General Microbiology, 130(1), 1.
Reich, M., Kershner, J.L., Wildman, R.C., 2003. Restoring streams with large wood: a synthesis, pp. 355-366.
Ricciardi, A., Rasmussen, J.B., 1999. Extinction rates of North American freshwater fauna. Conservation Biology, 13(5), 1220-1222.

Richmond, A.D., Fauseh, K.D., 1995. Characteristics and function of large woody debris in subalpine Rocky Mountain streams in northern Colorado. Canadian Journal of Fisheries and Aquatic Sciences, 52(8), 1789-1802.
Rickenmann, D., Recking, A., 2011. Evaluation of flow resistance in gravel-bed rivers through a large field data set. Water Resources Research, 47(7).
River Restoration Centre, 2013. National River Restoration Inventory. URL: http://www.therrc.co.uk/rrc_nrri.php, Accessed on 26/08/2013.
Robert, A., 2011. Flow resistance in alluvial channels. Progress in Physical Geography, 35(6), 765-781.
Robertson, K.M., Augspurger, C.K., 1999. Geomorphic processes and spatial patterns of primary forest succession on the Bogue Chitto River, USA. Journal of Ecology, 87(6), 1052-1063.
Robinson, M., 1986. Changes in catchment runoff following drainage and afforestation. Journal of Hydrology, 86(1), 71-84.
Robinson, M., Cognard-Plancq, A.L., Cosandey, C., David, J., Durand, P., Führer, H.W., Hall, R., Hendriques, M.O., Marc, V., McCarthy, R., McDonnell, M., Martin, C., Nisbet, T.R., O'Dea, P., Rodgers, M., Zollner, A., 2003. Studies of the impact of forests on peak flows and baseflows: a European perspective. Forest Ecology and Management, 186(1–3), 85-97.
Robinson, M., Moore, R.E., Nisbet, T.R., Blackie, J.R., 1998. From moorland to forest: the Coalburn catchment experiment. Institute of Hydrology.
Robison, E.G., Beschta, R.L., 1990. Coarse woody debris and channel morphology interactions for undisturbed streams in southeast Alaska, USA. Earth Surface Processes and Landforms, 15(2), 149-156.
Robson, A.J., 2002. Evidence for trends in UK flooding. Philosophical Transactions of the Royal Society of London. Series A: Mathematical, Physical and Engineering Sciences, 360(1796), 1327-1343.
Rosenfeld, J.S., Hatfield, T., 2006. Information needs for assessing critical habitat of freshwater fish. Canadian Journal of Fisheries and Aquatic Sciences, 63(3), 683-698.
Sacket, S.S., 1979. Natural fuel loadings in a ponderosa pine and mixed conifer forests of the Southwest, Rocky Mountain Forest and Range Experimental Station, Fort Collins, Colorado.
Saghafian, B., Julien, P.Y., 1995. TIme to equilibrium for spatially variable watersheds. Journal of Hydrology, 172(1-4), 231-245.
Saghafian, B., Julien, P.Y., Rajaie, H., 2002. Runoff hydrograph simulation based on time variable isochrone technique. Journal of Hydrology, 261(1-4), 193-203.
Salemi, L.F., Groppo, J.D., Trevisan, R., de Moraes, J.M., de Barros Ferraz, S.F., Villani, J.P., Duarte-Neto, P.J., Martinelli, L.A., 2013. Land use change in the Atlantic rainforest region: consequences for the hydrology of small catchments. Journal of Hydrology, 499, 100-109.
Sanchez, G., Puigdefabregas, J., 1994. Interactions of plant growth and sediment movement on slopes in a semi-arid environment. Geomorphology, 9(3), 243-260.
Sanzo, D., Hecnar, S.J., 2006. Effects of road de-icing salt (NaCl) on larval wood frogs (Rana sylvatica). Environmental Pollution, 140(2), 247-256.
Schelfaut, K., Pannemans, B., Van Der Craats, I., Krywkow, J., Mysiak, J., Cools, J., 2011. Bringing flood resilience into practice: the FREEMAN project. Environmental Science & Policy, 14(7), 825-833.

Schick, A.P., 1987. Hydrologic aspects of floods in extreme arid environments. In: V.R. Baker, R.C. Kochel, P.C. Patton (Eds.), Flood geomorphology. Wiley, New York, pp. 189-203.

Schutt, P., Schuck, H.J., Aas, G., Lang, U.M., 1994. Enzyklopadie der Holzgewachse. Handbuch und Atlas der Dendrologie. Landsberg.: Ecomed Verlag., 11, Landsberg am Lech, Germany.

Sear, D.A., 1994. River restoration and geomorphology. Aquatic Conservation: Marine and Freshwater Ecosystems, 4(2), 169-177.

Sear, D.A., 2012. Advice on Manning's n roughness coefficient for Lymington River based on field photographs and previous hydraulic modelling work. In: S.J. Dixon (Ed.), Southampton, UK.

Sear, D.A., Arnell, N.W., 2006. The application of palaeohydrology in river management. Catena, 66(1-2), 169-183.

Sear, D.A., Briggs, A.R., Brookes, A., 1998. A preliminary analysis of the morphological adjustment within and downstream of a lowland river subject to river restoration. Aquatic Conservation: Marine and Freshwater Ecosystems, 8(1), 167-183.

Sear, D.A., Kitts, D.R., Millington, C.E., 2006. New Forest LIFE-III Monitoring Report. The Geomorphic and Hydrological Response of New Forest streams to river restoration., University of Southampton.

Sear, D.A., Millington, C.E., Kitts, D.R., Jeffries, R., 2010. Logjam controls on channel: floodplain interactions in wooded catchments and their role in the formation of multi-channel patterns. Geomorphology, 116(3-4), 305-319.

Sear, D.A., Newson, M.D., Brookes, A., 1995. Sediment related river maintenance: The role of fluvial geomorphology. Earth Surface Processes and Landforms, 20(7), 629-647.

Sear, D.A., Wilcock, D., Robinson, M., Fisher, K., 2000. River channel modification in the UK. The hydrology of the United Kingdom: a study of change. Routledge, Oxford, UK.

Seehorn, M.E., 1992. Stream Habitat Improvement Handbook, USDA Forest Service, Atlanta, Georgia, USA.

Sheldon, F., Thoms, M.C., 2006. In-channel geomorphic complexity: The key to the dynamics of organic matter in large dryland rivers? Geomorphology, 77(3-4), 270-285.

Shields Jr, F.D., Alonso, C.V., 2012. Assessment of flow forces on large wood in rivers. Water Resources Research, 48(4).

Shields Jr, F.D., Gippel, C.J., 1995. Prediction of effects of woody debris removal on flow resistance. Journal of Hydraulic Engineering, 121(4), 341-354.

Shields Jr, F.D., Gray, D.H., 1992. Effects of woody vegetation on sandy levee integrity. JAWRA Journal of the American Water Resources Association, 28(5), 917-931.

Shields Jr, F.D., Knight, S.S., Stofleth, J.M., 2006. Large wood addition for aquatic habitat rehabilitation in an incised, sand-bed stream, Little Topashaw Creek, Mississippi. River Research and Applications, 22(7), 803-817.

Sholtes, J.S., Doyle, M.W., 2011. Effect of channel restoration on flood wave attenuation. Journal of Hydraulic Engineering, 137(2), 196-208.

Shreve, R.L., 1966. Statistical law of stream numbers. The Journal of Geology, 17-37.

Shugart, H.H., West, D.C., 1977. Development of an Appalachian deciduous forest succession model and its application to assessment of the impact of the chestnut blight. Journal of Environmental Management, 5, 161-179.
Sibley, A., 2010. Analysis of extreme rainfall and flooding in Cumbria 18–20 November 2009. Weather, 65(11), 287-292.
Singer, M.J., Le Bissonnais, Y., 1998. Importance of surface sealing in the erosion of some soils from a Mediterranean climate. Geomorphology, 24(1), 79-85.
Sobota, D.J., Gregory, S.V., Sickle, J.V., 2006. Riparian tree fall directionality and modeling large wood recruitment to streams. Canadian Journal of Forest Research, 36(5), 1243-1254.
Sokal, R., Rohlf, J., 1994. Biometry. W.H. Freeman.
Steel, E.A., Naiman, R.J., West, S.D., 1999. Use of woody debris piles by birds and small mammals in a riparian corridor. Northwest Science, 73(1).
Stevens, J.P., 2007. One Way Analysis of Variance. In: J.P. Stevens (Ed.), Intermediate Statistics: A Modern Approach. Routledge Academic, London, pp. 69-74.
Stewart, G.B., Bayliss, H.R., Showler, D.A., Sutherland, W.J., Pullin, A.S., 2009. Effectiveness of engineered in-stream structure mitigation measures to increase salmonid abundance: a systematic review. Ecological Applications, 19(4), 931-941.
Strickler, A., 1981. Contributions to the Question of a Velocity Formula and Roughness Data for Streams, Channels and Closed Pipelines. In: T. Roesgen, W.R. Brownlie (Eds.), W. M. Keck Laboratory of Hydraulics and Water Resources Translation. California Institute of Technology, Pasadena, CA, pp. 112.
Sullivan, A., Ternan, J.L., Williams, A.G., 2004. Land use change and hydrological response in the Camel catchment, Cornwall. Applied Geography, 24(2), 119-137.
Surian, N., Rinaldi, M., 2003. Morphological response to river engineering and management in alluvial channels in Italy. Geomorphology, 50(4), 307-326.
Sutton-Grier, A.E., Wright, J.P., Richardson, C.J., 2013. Different plant traits affect two pathways of riparian nitrogen removal in a restored freshwater wetland. Plant and Soil, 365(1-2), 41-57.
Svoboda, C.D., Russell, K., 2011. Flume Analysis of Engineered Large Wood Structures for Scour Development and Habitat, World Environmental and Water Resources Congress 2011: Bearing Knowledge for Sustainability. ASCE.
Swanson, F.J., Bryant, M.D., Lienkaemper, G.W., Sedell, J.R., 1984. Organic debris in small streams, Prince of Wales Island, Southeast Alaska, Pacific Northwest Forest and Range Experimental Station, USDA, Portland, Oregon.
Swanson, F.J., Lienkaemper, G.W., 1978. Physical consequences of large organic debris in Pacific Northwest streams. US Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experimnt Station.
Swanson, M.E., Franklin, J.F., Beschta, R.L., Crisafulli, C.M., DellaSala, D.A., Hutto, R.L., Lindenmayer, D.B., Swanson, F.J., 2010. The forgotten stage of forest succession: early-successional ecosystems on forest sites. Frontiers in Ecology and the Environment, 9(2), 117-125.
Sweka, J.A., Hartman, K.J., Niles, J.M., 2010. Long-Term Effects of Large Woody Debris Addition on Stream Habitat and Brook Trout Populations. Journal of Fish and Wildlife Management, 1(2), 146-151.

Swift, M.J., Healey, I.N., Hibberd, J.K., Sykes, J.M., Bampoe, V., Nesbitt, M.E., 1976. The decomposition of branch-wood in the canopy and floor of a mixed deciduous woodland. Oecologia, 26(2), 139-149.

Tabacchi, E., Lambs, L., Guilloy, H., Planty-Tabacchi, A.M., Muller, E., Decamps, H., 2000. Impacts of riparian vegetation on hydrological processes. Hydrological Processes, 14(16-17), 2959-2976.

Thomas, H., Nisbet, T.R., 2007. An assessment of the impact of floodplain woodland on flood flows. Water and Environment Journal, 21(2), 114-126.

Thomas, H., Nisbet, T.R., 2012. Modelling the hydraulic impact of reintroducing large woody debris into watercourses. Journal of Flood Risk Management, 5(2), 164-174.

Thompson, D.M., 1995. The effects of large organic debris on sediment processes and stream morphology in Vermont. Geomorphology, 11(3), 235-244.

Toews, D.A.A., Moore, M.K., 1982. The effects of streamside logging on large organic debris in Carnation Creek, Ministry of Forestry, Vancouver, British Columbia.

Tromp-van Meerveld, H.J., McDonnell, J.J., 2006. Threshold relations in subsurface stormflow: 1. A 147-storm analysis of the Panola hillslope. Water Resources Research, 42(2), W02410.

Tubbs, C.R., 2001. The New Forest: history, ecology and conservation. New Forest Ninth Centenary Trust, Lyndhurst, Hampshire, UK.

Tunstall, S.M., Penning-Rowsell, E.C., Tapsell, S.M., Eden, S.E., 2000. River restoration: Public attitudes and expectations. Journal of the Chartered Institution of Water and Environmental Management, 14(5), 363-370.

Uchida, T., Kosugi, K., Mizuyama, T., 2001. Effects of pipeflow on hydrological process and its relation to landslide: a review of pipeflow studies in forested headwater catchments. Hydrological Processes, 15(11), 2151-2174.

Valentin, C., d'Herbès, J.M., Poesen, J., 1999. Soil and water components of banded vegetation patterns. Catena, 37(1), 1-24.

van der Nat, D., Tockner, K., Edwards, P.J., Ward, J.V., 2003. Large wood dynamics of complex Alpine river floodplains. Journal of the North American Benthological Society, 22(1), 35-50.

Van Diggelen, R., Grootjans, A.P., Harris, J.A., 2001. Ecological restoration: state of the art or state of the science? Restoration Ecology, 9(2), 115-118.

Van Dijk, A.I.J.M., Van Noordwijk, M., Calder, I.R., Bruijnzeel, S.L.A., Schellekens, J., Chappell, N.A., 2009. Forest–flood relation still tenuous – comment on 'Global evidence that deforestation amplifies flood risk and severity in the developing world' by C. J. A. Bradshaw, N.S. Sodi, K. S.-H. Peh and B.W. Brook. Global Change Biology, 15(1), 110-115.

Van Pelt, R., O'Keefe, T.C., Latterell, J.J., Naiman, R.J., 2006. Riparian forest stand development along the Queets river in Olympic National Park, Washington. Ecological Monographs, 76(2), 277-298.

Van Sickle, J., Gregory, S.V., 1990. Modeling inputs of large woody debris to streams from falling trees. Canadian Journal of Forest Research, 20(10), 1593-1601.

Vis, M., Klijn, F., De Bruijn, K.M., Van Buuren, M., 2003. Resilience strategies for flood risk management in the Netherlands. International journal of river basin management, 1(1), 33-40.

Wallerstein, N.P., Alonso, C.V., Bennett, S.J., Thorne, C.R., 2002. Surface wave forces acting on submerged logs. Journal of Hydraulic Engineering, 128(3), 349-353.

Wallerstein, N.P., Thorne, C.R., 1997. Impacts of woody debris on fluvial processes and channel morphology in stable and unstable streams, Department of Geography, Nottingham University, United Kingdom.
Wallerstein, N.P., Thorne, C.R., 2004. Influence of large woody debris on morphological evolution of incised, sand-bed channels. Geomorphology, 57(1-2), 53-73.
Wallerstein, N.P., Thorne, C.R., Doyle, M.W., 1997. Spatial distribution and impact of large woody debris in northern Mississippi. In: S.S.Y. Wang (Ed.), Management of landscapes disturbed by channel incision: stabilization, rehabilitation, restoration: proceedings of the Conference on Management of Landscapes Disturbed by Channel Incision. Center for Computational Hydroscience and Engineering, The University of Mississippi, Oxford Campus, The University of Mississippi, pp. 145–150.
Wang, L., Duggin, J.A., Nie, D., 2012. Nitrate–nitrogen reduction by established tree and pasture buffer strips associated with a cattle feedlot effluent disposal area near Armidale, NSW Australia. Journal of environmental management, 99, 1-9.
Ward, J.V., Tockner, K., Uehlinger, U., Malard, F., 2001. Understanding natural patterns and processes in river corridors as the basis for effective river restoration. Regulated Rivers: Research & Management, 17(4-5), 311-323.
Ward, P.R.B., 1973. Prediction of mixing lengths for river flow gaging. Journal of the Hydraulics Division, 99(7), 1069-1081.
Warren, D.R., Kraft, C.E., 2008. Dynamics of large wood in an eastern US mountain stream. Forest Ecology and Management, 256(4), 808-814.
Warren, D.R., Kraft, C.E., Keeton, W.S., Nunery, J.S., Likens, G.E., 2009. Dynamics of wood recruitment in streams of the northeastern US. Forest Ecology and Management, 258(5), 804-813.
Weiler, M., McDonnell, J.J., Tromp-van Meerveld, H.J.I., Uchida, T., 2006. Subsurface stormflow. Encyclopedia of hydrological sciences. John Wiley & Sons.
Weill, S., Altissimo, M., Cassiani, G., Deiana, R., Marani, M., Putti, M., 2013. Saturated area dynamics and streamflow generation from coupled surface–subsurface simulations and field observations. Advances in Water Resources, 59, 196-208.
Werritty, A., 2006. Sustainable flood management: oxymoron or new paradigm? Area, 38(1), 16-23.
WFPB, 1997. Standard Methodology for Conducting Watershed Analysis Manual, Version 4.0. Forest Practices Watershed Analysis Manual. Washington Forest Practices Board, Washington.
Wharton, G., Gilvear, D.J., 2006. River restoration in the UK: meeting the dual needs of the European Union Water Framework Directive and flood defence. International Journal of River Basin Management, 4, 1-12.
Wheater, H.S., 2006. Flood hazard and management: a UK perspective. Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences, 364(1845), 2135-2145.
Wheaton, J.M., Darby, S.E., Sear, D.A., 2008. The scope of uncertainties in river restoration. In: S.E. Darby, D.A. Sear (Eds.), River Restoration: Managing the uncertainty in restoring physical habitat. John Wiley & Sons Ltd, pp. 21-39.
Wheaton, J.M., Pasternack, G.B., Merz, J.E., 2004. Spawning habitat rehabilitation-I. Conceptual approach and methods. International Journal of River Basin Management, 2(1), 3-20.

White, G.F., 1945. Human Adjustment to Floods. Ph.D., University of Chicago, Chicago.
Whiteway, S.L., Biron, P.M., Zimmermann, A., Venter, O., Grant, J.W.A., 2010. Do in-stream restoration structures enhance salmonid abundance? A meta-analysis. Canadian Journal of Fisheries and Aquatic Sciences, 67(5), 831-841.
Whiting, P.J., 2003. Flow Measurement and Characterization. In: G.M. Kondolf, H. Piégay (Eds.), Tools in Fluvial Geomorphology. John Wiley & Sons, Ltd, Chichester, pp. 323-346.
Wilby, R.L., Beven, K.J., Reynard, N.S., 2008. Climate change and fluvial flood risk in the UK: more of the same? Hydrological Processes, 22(14), 2511-2523.
Wilcock, P.R., 1996. Estimating local bed shear stress from velocity observations. Water Resources Research, 32(11), 3361-3366.
Wilcox, A.C., Nelson, J.M., Wohl, E.E., 2006. Flow resistance dynamics in step-pool channels: 2. Partitioning between grain, spill, and woody debris resistance. Water Resources Research, 42(5).
Wilkinson, M., Quinn, P., Welton, P., 2010. Runoff management during the September 2008 floods in the Belford catchment, Northumberland. Journal of Flood Risk Management, 3(4), 285-295.
Wilson, B.F., 1966. Development of the shoot system of Acer rubrum L, Harvard.
Wohl, E.E., 2011. Threshold-induced complex behavior of wood in mountain streams. Geology, 39(6), 587-590.
Wohl, E.E., Angermeier, P.L., Bledsoe, B., Kondolf, G.M., MacDonnell, L., Merritt, D.M., Palmer, M.A., Poff, N.L.R., Tarboton, D., 2005. River restoration. Water Resources Research, 41(10), W10301.
Wohl, E.E., Cadol, D., 2011. Neighborhood matters: Patterns and controls on wood distribution in old-growth forest streams of the Colorado Front Range, USA. Geomorphology, 125(1), 132-146.
Wohl, E.E., Cenderelli, D.A., Dwire, K.A., Ryan-Burkett, S.E., Young, M.K., Fausch, K.D., 2010. Large in-stream wood studies: a call for common metrics. Earth Surface Processes and Landforms, 35, 618-625.
Wohl, E.E., Goode, J.R., 2008. Wood dynamics in headwater streams of the Colorado Rocky Mountains. Water Resources Research, 44(9).
Wohl, E.E., Jaeger, K.L., 2009. A conceptual model for the longitudinal distribution of wood in mountain streams. Earth Surface Processes and Landforms, 34(3), 329-344.
Wohl, E.E., Madsen, S., MacDonald, L., 1997. Characteristics of log and clast bed-steps in step-pool streams of northwestern Montana, USA. Geomorphology, 20(1-2), 1-10.
Wolff, C.G., Burges, S.J., 1994. An analysis of the influence of river channel properties on flood frequency. Journal of Hydrology, 153(1), 317-337.
Woltemade, C., 1994. Form and Process: Fluvial Geomorphology and Flood-Flow Interaction, Grant River, Wisconsin. Annals of the Association of American Geographers, 84(3), 462-479.
Wykoff, W.R., Crookston, N.L., Stage, A.R., 1982. User's guide to the stand prognosis model. US Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station.
Yochum, S.E., Bledsoe, B.P., David, G.C.L., Wohl, E.E., 2012. Velocity prediction in high-gradient channels. Journal of Hydrology, 424, 84-98.

Young, M.K., 1994. Movement and characteristics of stream-borne coarse woody debris in adjacent burned and undisturbed watersheds in Wyoming. Canadian Journal of Forest Research, 24(9), 1933-1938.
Young, M.K., Mace, E.A., Ziegler, E.T., Sutherland, E.K., 2006. Characterizing and contrasting instream and riparian coarse wood in western Montana basins. Forest Ecology and Management, 226(1-3), 26-40.
Yu, D., Lane, S.N., 2006. Urban fluvial flood modelling using a two-dimensional diffusion-wave treatment, part 1: mesh resolution effects. Hydrological Processes, 20(7), 1541-1565.
Zanne, A.E., Lopez-Gonzalez, G., Coomes, D.A., Ilic, J., Jansen, S., Lewis, S.L., Miller, R.B., Swenson, N.G., Wiemann, M.C., Chave, J., 2009. Global wood density database. Dryad.
Zhang, X., Zwiers, F.W., Hegerl, G.C., Lambert, F.H., Gillett, N.P., Solomon, S., Stott, P.A., Nozawa, T., 2007. Detection of human influence on twentieth-century precipitation trends. Nature, 448(7152), 461-465.
Ziegler, A.D., 2012. Water management: Reduce urban flood vulnerability. Nature, 481(7380), 145-145.
Ziegler, S.S., 2002. Disturbance regimes of hemlock-dominated old-growth forests in northern New York, USA. Canadian Journal of Forest Research, 32(12), 2106-2115.
Zimmermann, A., 2010. Flow resistance in steep streams: An experimental study. Water Resources Research, 46(9), W09536.