Hydraulic, hydrological and ecological effects of Natural Flood Management: the case study of the Engineered Log Jam.

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Thesis Abstract

Large wood is an integral part of freshwater ecosystems that provides a plethora of ecosystem services and affects river hydraulics, channel morphology and catchment hydrology. However, large wood has been historically removed from rivers and channels. Moreover, rivers have been straightened, channelized and dredged. The main reason for these modifications was to provide a constant output of water for agriculture and farming, increase channel conductance, and decrease the local flood risks by lowering water levels and increasing water flow velocities locally. However, the importance of large wood for riverine ecosystems has been recently re-discovered.

The newly developed interest in large wood presence and the understanding of the benefits it provides on channel hydraulics, sediment retention and biodiversity made the re-introduction of large wood a sought-after restoration technique. More recently, in-channel large wood has been introduced in form of Engineered Log Jams (ELJs) as part of Natural Flood Management (NFM) projects. ELJs are deployed in channels as part of NFM projects with the aim of managing flood risk by increasing in-channel hydraulic resistance at the reach scale and impacting the peak flow magnitude and timing at the catchment scale.

This research was undertaken in a large rural catchment situated in South East England, the Evenlode catchment (431 km^2). In recent years, Thames Water and the Evenlode Catchment Partnership developed a 5 years NFM project (2016-2021) in a sub-catchment of the Evenlode: the Littlestock Brook (16.3 km^2). The Littlestock Brook NFM project has involved the design of several NFM measures, including the installation of 42 high water ELJs.

This thesis addressed the effects of high water ELJs in the Littlestock Brook catchment from both a flood management and an ecological perspective. Moreover, this work considered the different scales at which ELJs can exert an effect on channel and floodplain hydraulics, and on the hydrology of small and large catchments.

The effect of the installation of high water ELJs on channel conveyance at the reach scale was measured in the field by quantifying hydraulic resistance with Manning's *n* values. The data collected in the field showed Manning's *n* values between 0.09 and 0.6 for a short reach (30 m) impacted by one ELJ. The Manning's *n* values measured for the ELJ were in line with values found in the literature for naturally occurring large wood. The increase in channel roughness that follows ELJs installation was shown to cause a decrease in water flow velocity of up to 40% upstream of the ELJs for a 25-year flood event. Since the increase in channel roughness has the potential to lead to an increase in habitat heterogeneity at the local channel

scale, the response of macroinvertebrate communities to the presence of ELJs was also analysed. Macroinvertebrate communities at ELJs sites showed a higher taxonomical diversity compared to control sites. This higher taxonomical diversity was due to higher species richness. This higher diversity at ELJ sites might be correlated to an increase in habitat heterogeneity as a result of the presence of ELJs.

The effects of the widespread installation of ELJs in the Littlestock Brook catchment were quantified by using a hydrological-hydraulic modelling cascade. The effect of a wide installation of ELJs across the catchment was remarkable, with a reduction of 10-20%, depending on the number of ELJs deployed, of the peak flow height and a delay in the peak flow timing of 5-6 hours for a 1-year flood event. The effects of catchment-wide installation of ELJs were instead negligible for the 25-year flood event.

The sensitivity of a large catchment to the widespread change of the in-channel roughness was tested by simulating the hydrology of the Evenlode catchment with a hydrological model. The widespread change of the in-channel roughness gave a big reduction in peak flow height (12-17%) for the 25-year flood event. However, such a big change in channel roughness would be really difficult to achieve with river restoration measures that influence the roughness, such as ELJs.

In conclusion, high water ELJs represent an important NFM technique with the potential to restore innate hydraulic and morphological characteristics, impact small catchment hydrology, re-connect channels to floodplains and provide additional environmental co-benefits.

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1.1 Background

Flooding is one of the most common natural catastrophes, responsible for half of all natural catastrophe-related deaths and it impacts human activities at all scales, from local to continental (Di Baldassarre et al., 2013; Pauleit et al., 2005). Flooding can cause economic losses, loss of lives, damage to infrastructures, 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 and mental health illnesses for people who suffer flooding impacts (Burrel et al., 2007).

However, flooding is a natural process and is not always related to negative consequences. Despite the potential to cause devastating effects on human lives and infrastructure, it may also provide environmental benefits such as nutrient and organic matter transfer between the aquatic and terrestrial environments, which enriches the soils of the floodplain (Junk et al., 1989). Indeed, the seasonal floods in the Nile Delta have been the foundation of Egyptian agriculture for over five thousand years (Bell, 1970). Floodplain inundation is, therefore, a natural process, but floodplains have also been historically used for urban development due to the vicinity of water resources and the possibility to move people and commodities via navigation (Dadson et al., 2017).

In order to counteract and prevent the risks caused by flooding, human populations have always tried to manage the river network and the surrounding land to minimise the frequency of flooding and diminish the economic, environmental and health hazards generated by flooding. Furthermore, the current global changes in climate are also posing new hazards by shifting the patterns of precipitation that may exacerbate the issues coming from river flooding.

1.1.1 Flood frequency in Europe and the UK

Emerging evidence of the occurrence of increased flood trends associated with climate change has recently been highlighted in the published literature for Europe and the UK. For example, Blöschl et al. (2019) showed increasing flood trends in north-western Europe with an average local increase of 2.3% of the mean flood discharge per decade. In particular, in the northern UK, these patterns seem correlated to the rise in winter precipitation patterns and, to some extent, soil moisture, two of the main drivers of flooding in this area.

In recent years, the UK has indeed experienced a higher frequency of wet periods causing more regular flooding events of higher magnitude. For example, the UK experienced its wettest February in 2020, December in 2015, April and June in 2012 and November in 2009. This means that for five out of twelve months, the average monthly precipitation over the UK has been at its highest in little over a decade (Met Office, 2021). These patterns of increased precipitation resulted in substantial damage to infrastructures and private properties, and in loss of lives. For example, in the summer of 2007, prolonged heavy rainfall led to localised flash flooding in which 55,000 homes were flooded at an estimated cost of £3 billion with 13 deaths, making the event the costliest flood in the world during 2007 in terms of economic losses (Hannaford and Marsh, 2008; Pitt, 2008). During the autumn and winter of 2013 and 2014, the south of England faced the wettest period in 250 years triggered by pattern perturbations to the jet stream over the Pacific Ocean and North America (Slingo et al., 2014). And, finally, Storm Desmond in Cumbria in 2015 resulted in 340mm of rain falling over saturated soils originating floods that damaged road and railroad networks and caused two fatalities (Slingo et al., 2014).

The increasing flood discharges across Europe and the UK imply that the 100-year flood discharge five decades ago has now a return period that is shorter than 100 years. In north-western Europe, for example, the flood that had a 100-year recurrence period in 1960 has now typically become a 50- to 80-year flood discharge (Dadson et al., 2017).

Given these trends in flooding across Europe and the UK, it is of particular importance to understand the mechanisms leading to streamflow generation and how they can be manipulated to manage flood risk.

1.2 Streamflow and flood generation

Essential to addressing flood risk and the factors that may enhance or suppress fluvial flooding is an understanding of the mechanisms leading to streamflow generation. Streamflow is the fraction of precipitation reaching the drainage basin's outlet. Between the precipitation event and the actual streamflow discharge generation, water can follow different paths through the catchment via surface and subsurface processes (Figure 1.1).

Fluvial flooding results from the exceedance of the channel capacity caused by increased water levels (Dadson et al., 2017). Flood generation can either be caused by overland or sub-surface flow and depends on several factors that exhibit complex spatial and temporal patterns at small and large scales (Cammeraat and Imeson, 1999; Fitzjohn et al., 1998). Physical controlling factors include the so-called hydrological antecedent conditions (e.g. soil saturation, snow cover), meteorological conditions such as the amount, intensity and timing of precipitation, spatial and temporal variation of the meteorological event, and the runoff and streamflow generation processes together with the characteristics and structure of the river network (Nied et al., 2013). The combination of the above factors and their spatial distribution is essential in determining the potential flood risks arising from meteorological events (Merz and Blöschl, 2008). Wyzga et al. (2016), for example, highlighted how these flood-generating mechanisms might change across regions and across different catchments in the same area.

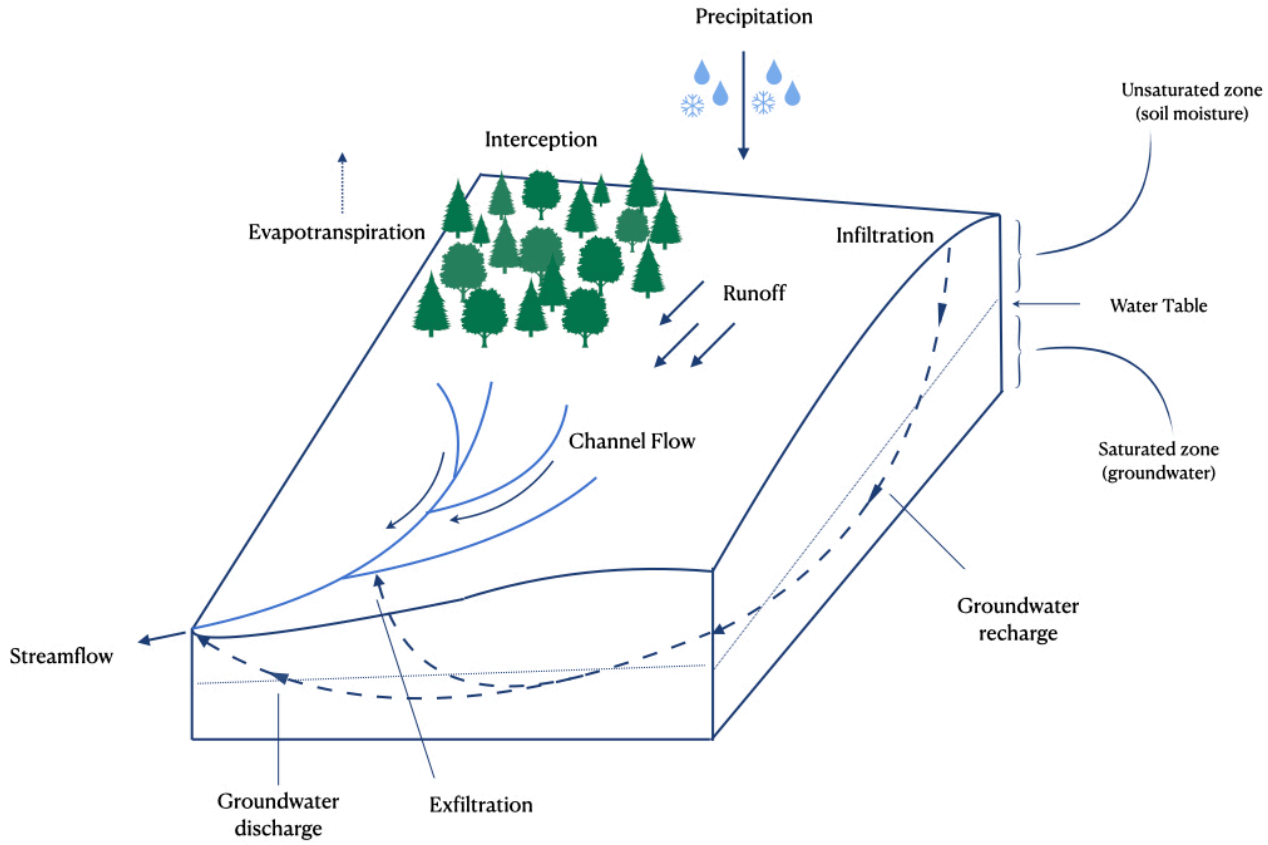


Figure 1.1 Schematic representation of hydrologic processes at the catchment scale. *The figure's top section represents the exchange of water between the catchment and the atmosphere. After precipitation, water can move above ground (solid lines) or via subsurface transfer of water following infiltration (dashed lines). The catchment's river network is represented in blue. The movement of water within the river network follows the hillslope's gradient towards the catchment outlet at the bottom left side of the figure.*

Land-use and different management strategies can also affect the flood generation processes. In-catchment vegetation, for example, has a significant role in most flood-generating processes by intercepting water from precipitation, drawing up water from the unsaturated zone, providing temporary storage and by releasing water to the atmosphere via evapotranspiration. The type and density of the vegetation can influence both the interception mediated by the leaf canopy and the water uptake from the soil (Robinson et al., 2003). Conifers, for example, have a higher water usage, resulting in lower moisture in the unsaturated zone, which may lead to higher infiltration during a storm (Bracken and Croke, 2007; Robinson

et al., 2003). The vegetation cover of a catchment and the related management practices can therefore exert an important role in flood generation (Marc and Robinson, 2007).

The different land cover types can also impact the hydraulic resistance to overland flow. As an example, vegetation cover within the catchment, compared to bare soil, will have higher resistance to runoff, thus decreasing the velocity of water transfer on the ground surface (Broadmeadow and Nisbet, 2009). Land cover types can also affect soil infiltration capabilities. Vegetated areas have indeed been shown to have better infiltration capabilities due to a better soil structure. In contrast, in farmed land, the compaction of the upper soil due to animal trampling has been shown to decrease infiltration and increase soil conductivity for the overland flow (O'Connell et al., 2007). Moreover, urban and semi-urban areas can affect runoff by decreasing infiltration due to impermeabilized soils. Runoff generated on relatively impermeable soils can then infiltrate into the soil when reaching vegetated areas (Boer and Puigdefábregas, 2005; Dunkerley and Booth, 1999).

Flood generation is controlled by several physical factors that exhibit complex spatial and temporal patterns at small and large scales. Many of these factors have been modified over centuries by human activities and impacts upon catchments. The mitigation of flood risk depends on the management of the physical controlling factors of flood generation.

1.3 Flood risk mitigation

Ropeik and Gray (2002) describe risk as "the probability that exposure to a hazard will lead to a negative consequence [….]". According to this definition, we can consider flood risk as the probability that exposure to flooding will lead to negative consequences for people, private properties and infrastructures, and the riverine ecosystem as a whole. Based on this definition, flood risk is the composite of three distinct factors: flood hazard, exposure, and

vulnerability (Lane, 2017). Modifying any of these three components may change the total flood risk arising from a determinate event.

The most common focus of flood risk strategies is to reduce the flood hazard by reducing the frequency of exceedance of the critical water level ($\bar{h}_c$) that triggers flooding at a specific location:

$$\bar{h} = Q\left(\frac{1}{w\bar{V}}\right) + z_b \qquad (1.1)$$

where $\bar{h}$ is the water level, Q is the discharge of the river, w is the width of the river, $\bar{V}$is the mean velocity, and z_b is the mean stream bed level (Lane, 2017). This equation emphasises how the potential reduction in flood hazard can be pursued through different measures targeted at different scales and for different flow and channel characteristics. For example, at the local scale, flood hazard may be reduced by measures that could increase some combination of river width and flow velocity (e.g. river straightening or other measures that increase channel conductance), or reduce stream bed level (e.g. river dredging) or, on the other hand, by an upstream intervention that may decrease the rate at which water is delivered to the flood risk zone, thus reducing local stream discharge (Q). Oftentimes, the reduction of discharge (Q) implies some intervention upstream in a perspective of catchment-based flood management, which is the manipulation of how water moves through a catchment.

Historically, flood and coastal risk management have involved the construction of engineered flood defences, such as levees and embankments, to mitigate risks from flooding (Howe and White, 2001). However, traditional flood protection methods of large engineered structures are economically expensive to construct, costly to maintain and linked to poor water quality, and may cause negative impacts on riverine ecosystems (Collentine and Futter, 2018; Thorne, 2014). For these reasons, policymakers and scientists are increasingly interested in

more natural techniques for landscape water retention and storage as a complementary flood risk management strategy (Nesshöver et al., 2017; Pitt, 2008).

The so-called Natural Flood Management (NFM) techniques (Section 1.4) indeed propose using either in-line or off-line nature-based structures for flood risk management. The former techniques temporarily store water while being directly connected to the river. The latter techniques are situated in the floodplain, thus disconnected from the channel, but connect to the channel when water needs to be retained. These NFM techniques rely less on infrastructure and more on the manipulation of hydrological and hydraulic characteristics of a catchment and its river network to manage discharge. These manipulations also aim to enhance or restore natural hydrological, hydraulic and morphological processes, often in river basins that human activities have heavily impacted. These techniques and methods have been recently summarised, and their evidence basis is outlined in a report by the Environment Agency (Burgess-Gamble et al., 2017) called Working with Natural Processes (WwNP). WwNP aims to protect, restore and emulate the natural functions of catchments, floodplains, rivers and the coast (Lavers and Charlesworth, 2016). WwNP can take many forms and be applied in urban and rural areas, rivers, estuaries and coasts. Globally, many different terms (Figure 1.2) refer to the techniques that are part of this type of flood and coastal risk management.

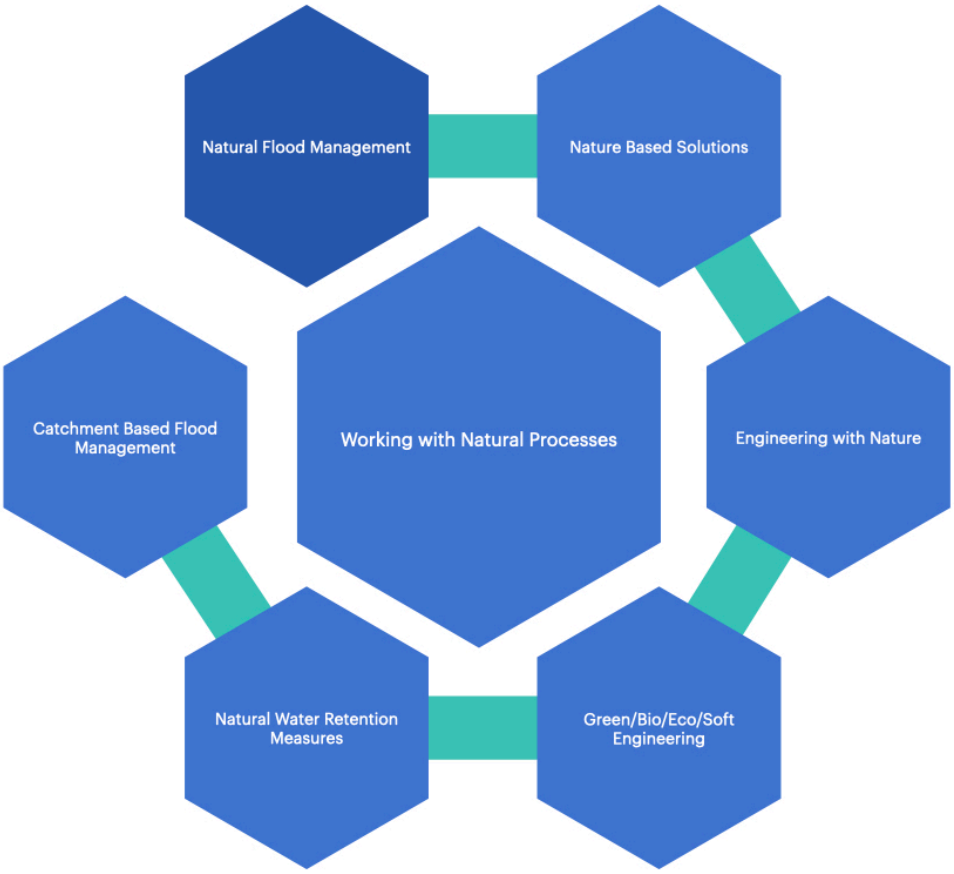


Figure 1.2 Alternative terms used for Natural Flood Management and Working with Natural Processes. *Adapted from Burgess-Gamble et al. (2017).*

1.4 Natural Flood Management

Natural Flood Management (NFM) is a suite of techniques that seeks to reduce flood hazards by restoring innate hydrological and morphological processes, features and characteristics while sustaining or enhancing environmental co-benefits such as biodiversity, increased soil and water quality and carbon sequestration (Forbes et al., 2015). NFM techniques include measures like retention ponds, woody dams, tree-belts, floodplain restoration and land-use change but exclude traditional engineered solutions like large dams, channels and levees that work against or disrupt, natural processes (Dadson et al., 2017).

As highlighted above (Section 1.3 - Equation 1.1), local flood hazard may be decreased by intervening either in the structure of the channel, by modifying the width and river-bed

elevation, or by intervening in those factors that may affect water velocity and channel discharge by applying either in-stream and off-stream interventions or by impacting catchment hydrological processes with management interventions and land use change. In particular, a reduction in discharge downstream can be achieved through the implementation of widespread interventions upstream that manipulate how water moves through the catchment and its drainage network. The management of flooding from an integrated catchment-wide approach is defined as catchment-based flood management (CBFM). Natural Flood Management is a subset of CBFM that relies less on traditional and spatially-intensive flood management infrastructures such as flood walls and more on nature-based and spatially-extensive infrastructures deployed over larger scales.

Catchment Based Flood Management (CBFM) in general, and NFM in particular aim to reduce flood risk by targeting the hydraulic and hydrological processes at both the catchment scale and at the local field and channel scale. The main principles that guide CBFM in general and NFM in particular, as defined by Lane (2017), are shown and schematised in Figure 1.3. The variability of discharge at any location within the river network of the catchment is a product of two counteracting processes; the attenuation of the discharge (slowing the flow) and additional inputs of discharge (increasing the flow). The attenuation of discharge is a natural process that leads to the attenuation of a single pulse of rain as it moves through the hillslope and then within the river channel as a result of catchment characteristics, processes associated with hillslope hydrology and then in-channel processes once it reaches the drainage network (e.g. turbulence, secondary flow, flow velocity differentiation). As a result of attenuation, and in the absence of other inputs, the peak flow should become progressively smaller as it moves through the catchment (Lane, 2017). However, other discharge inputs as a result of tributaries joining the main channel can result in an increase in total stream discharge (Figure 1.3 B).

Consequently, downstream peak discharge is a balance between attenuation and tributary inputs processes. NFM and CBFM recognise these processes and intervene by managing hillslope and channel processes to optimise the balance between attenuation and additional inputs of discharge.

NFM interventions, in particular, can be summarised in three groups based on the type of manipulation and the area of the catchment where these interventions occur. Each will be described in turn:

i) The first group of NFM interventions entails the manipulation of hillslopes to reduce rapid runoff generation and runoff travel times through the catchment. The broad assumption behind these interventions is that the landscape and land-use changes that occurred over the last decades (e.g. urbanisation, increase in stocking densities, deforestation) caused faster runoff rates compared to the innate state of the catchment. A partial reversal of these changes may, through time, attenuate the peak flow as a result of a reduction in runoff generation and longer runoff travel time in reaching the channel (Figure 1.3 C). The type of manipulations used to reduce rapid runoff generation is mainly related to changes in management practices for rural catchments. Boardman et al., (2003) have indeed shown how certain land use practices that leave a bare soil surface for longer periods of time or during the wettest months, may increase the susceptibility to rapid runoff generation. In this case, the replacement with grassland has been shown to reduce local flood risk by encouraging infiltration (Evans et al., 2003). Another management practice that could be used to reduce runoff generation is the reduction in livestock densities. The density reduction may reduce rapid runoff generation by decreasing livestock-induced soil compaction (Orr and Carling, 2006). The diminished trampling of livestock can, in turn, lead to an increase in biomass and this augments water loss to evapotranspiration in a positive feedback loop (Heatwaite et al., 1989).

Other more extensive land-cover changes, such as in-catchment afforestation across grasslands and arable land, could reduce overland flow generation thanks to water loss by evapotranspiration and increased infiltration through the three root networks (Marc and Robinson, 2007).

ii) The second group of NFM interventions involves the increase of the upstream storage of water during extreme flow events. This group of interventions is possibly more straightforward and may be introduced at various spatial scales. These measures include, for example, small features such as ponds, bunds and ditches or larger features like wetlands and big reservoirs. The basic principle underpinning this group of interventions is to create temporary storage of water during a rainstorm, thereby slowing the transfer of water to the channel network and downstream. The reduced rate of transfer of water could in turn attenuate the peak flow downstream where higher-value land, properties or infrastructures are present. It could also, depending on the stream network and catchment characteristics, de-synchronise inputs of water from different areas of the catchment and various tributaries, leading to downstream peak flow attenuation for the main channel. Wilkinson et al. (2010) showed how ponds, bunds and ditches can be effective in reducing peak flow height for small catchments (< 10 km^2).

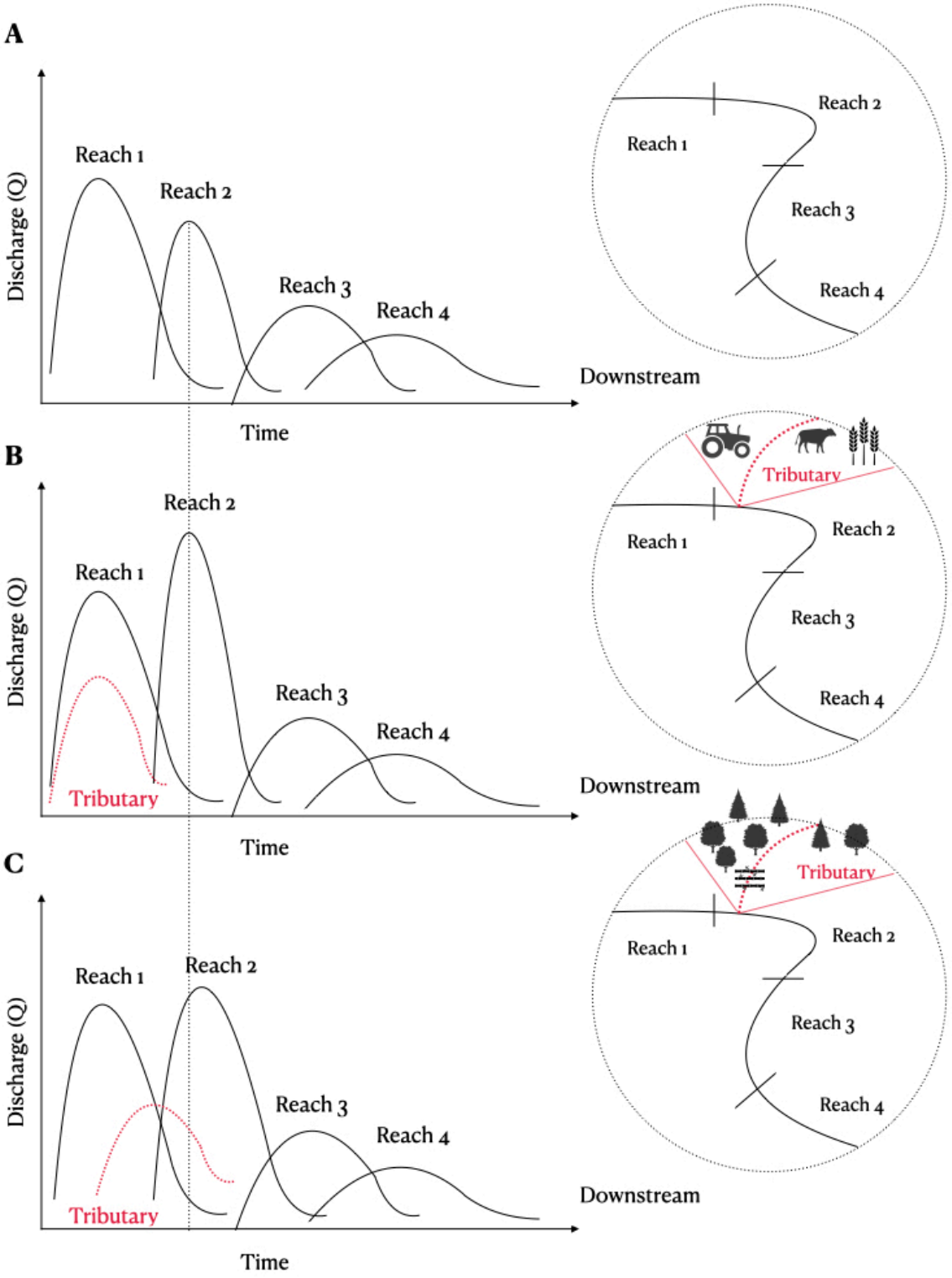

A
Discharge (Q)
Reach 1
Reach 2
Reach 3
Reach 4
Time
Downstream
Reach 1
Reach 2
Reach 3
Reach 4
B
Discharge (Q)
Reach 1
Reach 2
Reach 3
Reach 4
Tributary
Time
Downstream
Tributary
Reach 1
Reach 2
Reach 3
Reach 4
C
Discharge (Q)
Reach 1
Reach 2
Reach 3
Reach 4
Tributary
Time
Downstream
Tributary
Reach 1
Reach 2
Reach 3
Reach 4

Figure 1.3 NFM-led processes that can lead to attenuation by the decoupling of peak flow coming from a tributary.

Figure A (top) illustrates a schematic catchment with four reaches and the discharge for each is plotted as a function of time passed from the onset of the rainfall event. Figure B shows a fictional example of the same catchment with an added tributary. The additional discharge input coming from the tributary will affect the peak flow of the nearest reach downstream (Reach 2) with an additive effect due to synchronicity with the peak flow of Reach 1. Figure C shows a potential decoupling mechanism following a management intervention in the tributary catchment. As shown in the circle on the right side, interventions on both the land-use of the catchment, with afforestation, and in the channel, with the implementation of in-line structures to slow the flow (Engineered Log Jams) cause a delay in timing and a reduction in the height of the peak flow coming from the tributary catchment. As a result, the peak flow of Reach 1 and the peak flow of the tributary catchment will be de-synchronised. This de-synchronisation, together with the attenuation of the peak from the tributary catchment, will reduce the height of the peak flow of Reach 2 and delay its timing. In the fictional situation described, reaches 3 and 4 are not affected by the additional input of the tributary as they are supposed to be far downstream, and the effect of the tributary is therefore attenuated by distance. Adapted from Lane (2017).

iii) The third group of interventions incorporates measures that increase attenuation by reducing flow conveyance. Conveyance can be reduced by intervening in the hillslope-channel coupling, managing riparian conveyance or by intervening in channel morphology and structure (Lane, 2017). In particular, a decrease in connectivity between runoff source areas and the drainage network might increase attenuation. For instance, this could be realised by blocking drains that provide a rapid connection of runoff to the river network (Lane and Milledge, 2013). Moreover, the rate of conveyance of water also depends upon the river and floodplain hydraulic resistance, usually defined as roughness. By increasing river roughness, the water levels into the channel increase and water velocity in the river decreases. This could in turn slow the delivery of water downstream thus providing an attenuation of the peak flow (Figure 1.3 C) and increase the frequency and extent of floodplain inundation. Floodplain inundation can provide further attenuation as floodplain flows are normally shallow and thus slower (Lane, 2017) and this overland flow may be additionally slowed down by the floodplain's roughness. The application of these NFM measures may thus reduce flooding downstream by reducing flow velocities in the channel network, thus increasing travel times, and by increasing floodplain inundation upstream. These interventions include the installation of engineered log jams, leaky dams or the restoration of channel meanders (Dixon et al., 2016).

The application of the different types of NFM interventions will depend upon different factors: catchment characteristics (e.g. hydrological regime, runoff generation mechanism, land use), the extent of the area of intervention and its location within a catchment, the funding for the project, and the additional environmental and social benefits that should be targeted by the NFM project. Over the past decade, NFM has been increasing in popularity in Europe and the UK with strong government policy and financial support. Since 2017, for example, the UK Government has invested £15 million into 60 NFM pilot schemes across England. These projects have been evenly split between catchment-scale projects led by risk management

authorities, and community-scale projects led by local community groups, charities, and catchment partnerships. In addition, DEFRA and the Environment Agency recently published a £5.2 billion investment plan for flood and coastal erosion risk management for 2021 to 2027, including funding for NFM projects (DEFRA, 2018)

Given the extent of the global and national interest in Nature-Based Solutions such as NFM, it has never been timelier and more important to investigate the nuances of NFM and analyse their potential to meet legislative targets and deliver flood risk reduction. However, because NFM interventions usually occur alongside other drivers of flooding, such as the spatial and temporal variability in precipitation and overland flow generation, assessing the effectiveness of NFM at different scales is particularly challenging. An overview of the available scientific evidence on the effectiveness of NFM interventions is presented in Section 1.5 below.

1.5 Status of Natural Flood Management Science

1.5.1 NFM Effectiveness in reducing downstream flood risk

The land cover of the UK has dramatically changed due to human activities in the recent past. Moreover, rivers and channels have been highly modified by channelization, dredging and river clearing. This widespread modification of river basins and of their river networks has undoubtedly led to changes in hydrological processes at various scales. NFM aims at restoring some of the innate hydrological processes and characteristics of river basins to lower flood risk while delivering additional environmental benefits. The focus of this section is to synthesise whether the different types of manipulation introduced in Section 1.4 can reduce downstream flood risk. The current scientific evidence for NFM interventions is shown in Table 1.1.

There is now clear evidence that appropriately chosen land-use interventions can reduce runoff generation and thus local peak water flows after moderate rainfall events (De Roo et al.,

2003; Oudin et al., 2008). However, the evidence available for the large catchment scale does not suggest that realistic land-use changes will greatly impact peak flows. Long-term monitoring of hydrological trends for large catchments (>100 km^2) is needed to extrapolate the findings of individual small-scale studies. Moreover, more data and evidence are needed for different land use classes given their intrinsic variability. For example, the effects of afforestation have shown a strong dependence on forest type, density and age. However, only a few catchment scale studies have tried to quantify these variations (Lane et al., 2017).

The increase of water storage in the catchment using NFM structures may intervene at various levels, from widespread small-scale impoundments to the creation of larger and more spatially intensive wetlands and reservoirs. All of these measures temporarily store more water in the upstream part of the catchment, to release it slowly at a rate dependent on their characteristics, volume capacity and flood conditions. The effectiveness of these storage features depends on whether the stored water would have contributed to the flood peak and on the timing of its release. Large hard engineering structures that can mechanically control the release of water are more efficient in this perspective. Moreover, small storage features may fill up quickly and have no further effect in a large flood or for subsequent rainfall events. Small NFM water storage features are effective at the small scale (10 km^2) in reducing runoff (Wilkinson et al., 2010) but more evidence is needed from different spatial scales. Moreover, further studies on these types of water storage structures are needed since the implementation of these features has been shown to provide contradicting results in terms of performance (Dadson et al., 2017). NFM storage features such as ponds and bunds may also have higher maintenance costs when compared to other NFM structures due to the need of removing sediment in order to maintain their volume capacity.

Table 1.1 NFM measures to reduce downstream flood risk.

NFM intervention	Class of Manipulation	Effectiveness in reducing downstream flood risk
Land-cover changes	Reducing rapid runoff generation	Land cover changes effectively reduce flood risk at a small scale (< 20 km^2) but not for the most extreme floods (De Roo et al., 2003; Oudin et al., 2008). Afforestation has been shown to reduce total runoff but, during storm events, rainfall reduces evapotranspiration so decreasing runoff reduction. For smaller storms, flow peaks per unit area were smaller in a forested catchment when compared to a mostly grassland catchment. During high flood flows, however, no significant difference was found between flood peaks in the two different catchments with different land covers (Marc and Robinson, 2007).
Soil and Land Management	Reducing rapid runoff generation	Arable land-use practices: By replacing land-use practices that keep the soil surface exposed for extended periods of time with grasslands, the potential for flooding in the surrounding areas was reduced (Boardman et al. 2003). A study conducted in a small catchment located in the South Downs region of South-East England found that prior to any management measures, a rainfall of 231mm generated runoff from 55% of the catchment. However, after implementing management measures such as set-aside, the same amount of rainfall only generated runoff from 20% of the catchment (Evans and Boardman, 2003). Livestock land-use practices: Data from plot scale experiments showed a 28-78% reduction in the run-off between tree-planted plots and grazed control plots (Marshall et al., 2009). In a modelling study, soil compaction due to higher livestock density was shown to increase the flood peak height in a 36 km^2 catchment (O'Donnell et al., 2007). Tillage practices: Silgram et al. (2010) showed that changes in tillage practices at the plot scale were reducing runoff generation from 5.5-15.8 % to 0.2-1.7% of rainfall volume. Soil structural degradation linked to land management practices was indicated to cause a large increase in runoff consequently impacting peak river flow (Holman et al., 2003).

Buffer strips and riparian woodland	Reducing rapid runoff generation/reducing runoff conveyance to the channel	Buffer-strips have been shown to have an effect in the reduction of runoff (Borin et al., 2010). The effectiveness of buffer-strips depends on their width and vegetation density and their effects are limited to the local scale (Gao et al., 2016). The planting of riparian woodland (0.7% of the catchment area), together with the construction of 100 ELJs structures, was shown to reduce a 1% AEP event by 8% (Odoni and Lane, 2010). Ghavasieh et al. (2006) found that riparian woodland strips along a 20 km reach could reduce peak discharge by 3.8%, while Anderson et al. (2006) predicted a reduction in peak discharge by 12% over a 50km reach. Other studies have shown more negligible effects of riparian woodland planting with a 1-2% reduction in peak flow (McIntyre and Thorne, 2013).
Wetlands	Storage of flow	Restoring floodplain wetlands may reduce or delay flood peaks (Acreman et al., 2003) and may attenuate high frequency, low return period floods (Acreman and Holden, 2013).
Water storage areas	Storage of flow	A pilot runoff attenuation feature with a capacity of 800-100 m^3 was shown to held runoff for approximately 8 hours thus delaying the travel time of the peak flow from 20 minutes to 35 minutes (Wilkinson et al., 2010). In Holnicote (Somerset), modelled and observed data showed that, in a 40 km^2 catchment, 25 000 m^3 of storage decreased peak flow by 10% during the December 2013 floods. This reduction may be larger for floods with smaller return periods: the peak flow of a modelled 1 in 5-year flood event was reduced by 25% (National Trust, 2015). A modelling study identified that, in a 56 km^2 catchment, a 100,000 m^3 of storage volume could reduce the discharge of the 1 in 100-year flood by 11.5% (Burgess-Gamble et al., 2017). A 10.9% reduction in peak flow discharge for a 1 in 100-year flood event was modelled for a smaller catchment (8 km^2) in North England.
River restoration	Reducing in-channel conveyance	Modelling evidence showed a 10-15% reduction in the peak flow achieved by restoring a 5 km channel reach (Acreman et al., 2003). A monitoring study in a 25 km^2 catchment in the New Forest found that river restoration led to a 21% reduction in flood peak and a 33% increase in peak travel time for a 2-year recurrence event (Sear et al., 2006). Restoration reduced water velocities for a 1 in 100-year flood by 41% (Keestra et al., 2012).

		Restoring meanders in a 1 km reach in a 17 km^2 catchment, reduced flood peaks by less than 1%, for 2-year to 50-year return period (Sholtes and Doyle, 2011). River restoration in headwaters of a 400 km^2 catchment reduced peak flow by 14% (Liu et al., 2004)
Large Wood	Reducing in-channel conveyance	Large wood will be discussed in Section 1.6 and in the introductions to other chapters.

1.5.2 Environmental co-benefits of Natural Flood Management

In addition to proposed flood reduction, it is suggested that well-designed NFM projects may provide additional environmental benefits to nature and people (Lavers and Charlesworth, 2016). These co-benefits may include but are not limited to, the creation and restoration of both riverine and wetlands aquatic habitats, the mitigation of various sources of diffuse pollution and the mitigation of soil erosion impact on in-stream ecology. For example, in-catchment afforestation measures and riparian vegetation planting may be beneficial for flood risk reduction while providing environmental co-benefits as additional carbon sinks within the catchment. Moreover, in-channel measures such as re-meandering or Engineered Log Jams can restore innate hydraulic and morphological characteristics of the channel thus providing additional environmental co-benefits in terms of habitat restoration and diversification.

NFM techniques have also been proven to have the potential to modify water quality through the uptake of nutrient pollutants and by retaining sediment. For example, restoring wetlands in the floodplain can be effective in reducing and delaying flood peaks (Acreman et al., 2003). Additionally, wetland restoration was shown to be effective in removing ammonia, nitrogen and phosphorus by plant uptake and increased sediment times (Cameron et al., 2003). These benefits provided by multiple NFM measures could help to reduce the need for water treatment downstream, to sustain or enhance the productivity of agricultural soils, preserve and enhance ecosystems and biodiversity and increase recreational value for local communities. A more natural environment could also represent new sources of income for farmers and landowners moving from intensive farming to a more environmentally sustainable business model through a higher recreational value of their land and eco-tourism. The UK government has indeed recently proposed new incentives for farmers under the Agriculture Act (2020) in order to replace EU subsidies for farming. The DEFRA plan that followed '*The Path to Sustainable Farming: An Agricultural Transition Plan 2021 to 2024'*

set out plans for a range of schemes including initiatives to increase biodiversity, restore landscapes, promote animal welfare and increase productivity through investment in new equipment and technology (DEFRA, 2020).

Many NFM interventions seek to directly restore or enhance the natural processes within the riverine ecosystem, thus enhancing the potential to sustain a broader range of habitats. This can in turn support more species increasing local biodiversity. Wetland creation and in-stream channel interventions such as re-meandering and Engineered Log Jams are typical figurative examples of how new habitats may be created or restored. Freshwater habitat restoration, biodiversity increase and ecosystem functioning are particularly important since the EU Water Framework Directive required member states to reach at least a good ecological status for their water bodies (Kallis and Butler, 2001) but in 2016, 86% of UK rivers and other water bodies failed to reach good ecological status.

NFM interventions could provide additional environmental co-benefits that could help in meeting the targets set out by the government in its 25 years Environmental Plan (DEFRA, 2018) while decreasing the risks coming from flooding. In-channel large wood re-introduction, in the form of Engineered Log Jams, could restore natural sediment processes and improve water quality by accelerating nitrate fluxes (Grabowski et al., 2019; Gurnell et al., 2019). Moreover, the re-connection of rivers to the floodplain with the creation of wetlands may be an effective solution to manage high nutrient loads as these could be exploited by the vegetation to grow (Cameron et al., 2003).

One of the most widely recommended NFM technique that can provide effects on water flow regulation and additional environmental co-benefits is the re-introduction of in-channel large wood. According to Burgess-Gamble (2017) large wood re-introduction is one of the NFM techniques with the biggest potential to provide additional benefits in terms of water quality, habitat provision and biodiversity increase.

The re-introduction of large wood in rivers in the form of high water Engineered Log Jams will be the main subject of this thesis. The physical and ecological effects of natural

occurring large wood and artificially introduced Engineered Log Jams, together with their potential impact on flood dynamics will be discussed in detail in Section 1.6.

Despite the global interest in Natural Flood Management and the increase in the available scientific evidence on its effectiveness from both a flood risk and an environmental perspective, the uptake of NFM projects remains low. The longstanding barriers to the implementation of NFM are discussed in the following Section (1.5.3).

1.5.3 Challenges and barriers to the implementation of Natural Flood Management

Despite the support in science and policy and high-profile initiatives globally, the delivery of NFM projects has been progressing slowly (Klijn et al., 2018). This is common for Nature-based Solutions, especially those that involve large-scale landscape interventions and thus require the cooperation of multiple stakeholders (Seddon et al., 2019). Indeed, the planning and delivery of NFM projects require the joint approval and support of landowners, statutory agencies, local authorities, local communities and the general public. Moreover, large-scale landscape interventions, such as NFM, require collaborative governance, targeted funding and policy instruments to be effectively implemented (Benson et al., 2014; Bark et al., 2021). NFM pilot projects have been delivered across the UK in the last decade, but NFM is far from being commonly implemented and inserted into catchment management plans.

The barriers to NFM uptake and implementation are diverse and complex. They can be categorised into three macro-categories: socio-political barriers, biophysical barriers and barriers that are both socio-political and biophysical (O'Donnell et al., 2017).

The main socio-political barriers are the reluctance to support novel approaches to flood risk management (O'Donnell et al., 2017; Wingfield et al., 2019), the lack of clear scientific evidence on the effectiveness of NFM (Dadson et al., 2017; Wingfield et al., 2019; Bark et al., 2021; Thaler et al., 2023), the absence of an effective NFM governance (Holstead et al.,

2015), the lack of perceived control over flooding (Waylen et al., 2017), the scarcity of funding and other financial constraints for landowners (Wells et al., 2020), the cultural influences on land managers and practitioners (Holstead et al., 2015; Thaler et al., 2023), and the interactions between different stakeholders groups (Wells et al., 2019; Bark et al., 2021). Biophysical barriers include the barriers related to catchment characteristics (Thaler et al., 2023), the availability of space for NFM implementation, and the future land use and climate (O'Donnell et al., 2017). Other notable barriers that are both socio-political and biophysical regard the maintenance of NFM measures and the identification, quantification and monetisation of the multiple benefits of the NFM measures (O'Donnell et al., 2017).

Most of the barriers to implementation are interconnected and can form feedback loops that further hinder the uptake of NFM (Thaler et al., 2023). For example, the lack of clear scientific evidence of the effectiveness of NFM on both flood risk and environmental co-benefits is often cited as one of the main barriers to the uptake of NFM. Consequently, the low uptake of NFM projects reduces the chance to collect empirical data to fully understand NFM effectiveness thus complicating the expansion of the scientific evidence base. The lack of clear scientific evidence will further impact the availability of funding for NFM, especially as a proportion of the total funding available for flood management (Figure 1.4). The effectiveness of traditional flood management measures can indeed be calculated via traditional appraisal techniques and is thus easier to justify their uptake and calculate their cost-benefit ratio. Most of the barriers to a wider implementation of NFM also depend on its intrinsic characteristics. In particular, Natural Flood Management is spatially extensive as compared to traditional flood management which is spatially intensive. Consequently, NFM requires a catchment-wide framework to achieve noticeable levels of risk reduction. As a result of this requirement, more stakeholders need to be involved in the planning and delivery of NFM projects, with the potential to create more conflicts. For example, upstream-downstream conflicts might arise as NFM is often applied in the upper part of the catchment

while it serves to reduce flood risk in the downstream part of the catchment. As a result, landowners and communities in the upper part of the catchment might need to sustain the cost and risks of implementing NFM measures while not primarily gaining any benefit from the NFM measures in terms of flood risk reduction (Waylen et al., 2018). This can also have important implications in the allocation of funding as the landowners who decide to develop NFM measures on their land should be adequately compensated for re-purposing their land for NFM and for the resulting economic losses.

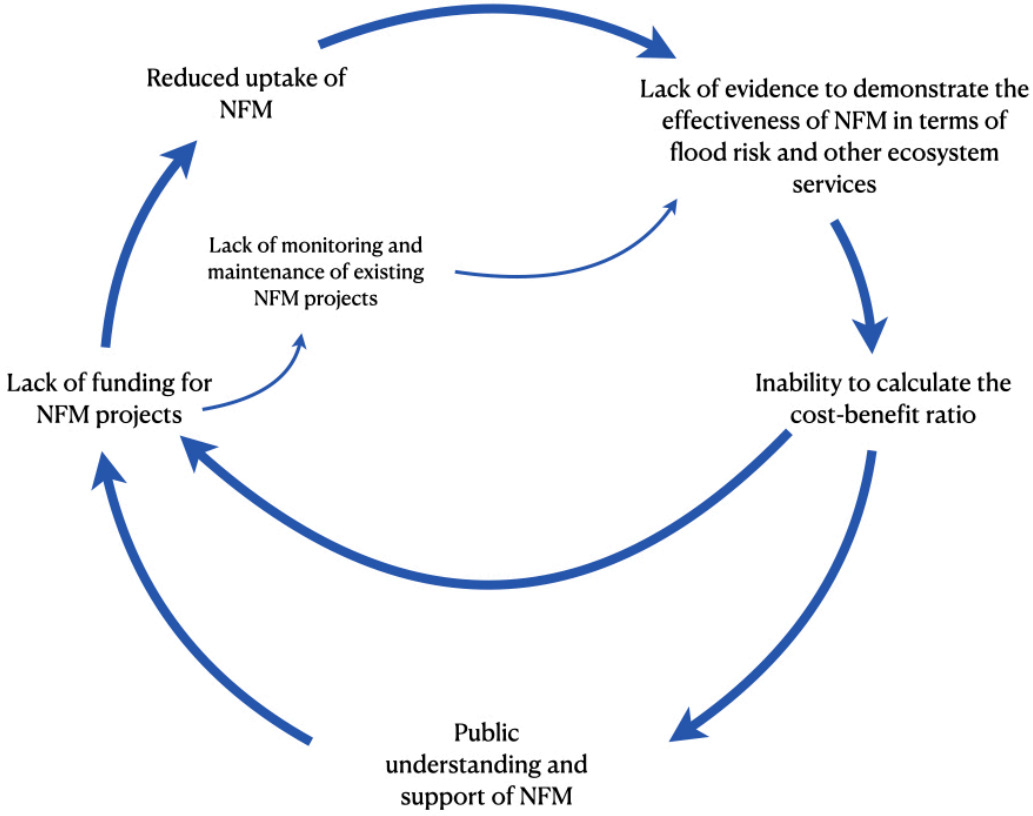


Figure 1.4 The feedback loop created among barriers to the implementation of NFM*. The feedback loop is created in the first instance by a lack of evidence and effectiveness of NFM in decreasing flood risk and delivering environmental co-benefits. The feedback loop further affects the capacity of NFM to attract the support of the general public and stakeholders, thus the lack of funding and reduced uptake of NFM. Adapted from Wells et al. (2020).*

Moreover, in the UK, the responsibility for flood risk management lies with multiple organisations and it is thus important for the multiple stakeholders and organisations to interact effectively for the successful implementation of NFM projects. However, as

highlighted by multiple studies (O'Donnell et al., 2017; Wells et al., 2019), barriers to the implementation identified by one stakeholder group (e.g. landowners) may be different to the ones highlighted by a different group (e.g. practitioners, statutory agencies) and this may act as a barrier for successful interactions between different groups (Wells et al., 2020). Since NFM is an extensive flood management approach and is often applied in the context of catchment wide flood management, the collaboration between different stakeholders is absolutely crucial. For this reason, the role of catchment partnerships in the context of the Catchment Based Approach (CaBA) is of particular importance. CaBA was established by DEFRA to adopt a collaborative approach to water management across England. River catchment partnerships were established following the regulations set out in the CaBA Policy Framework in each Water Framework Directive catchment across England, helping in meeting the targets set out by the Government's 25-year Environment Plan. Partnerships are hosted by a non-Government Organisation (NGO) and supported by a catchment coordinator from the Environment Agency. The key role of the hosts is to bring together a diverse range of partner organisation to collectively pool and review evidence and plan and deliver interventions that realise environmental benefits for the catchment (Collins et al., 2020).

Each catchment partnership undertakes an integrated management of land and water, addressing the whole catchment, and delivers crosscutting practical interventions on the ground. These interventions will in turn result in multiple benefits including improvements to water quality, enhanced biodiversity, reduced flood risk, resilience to climate change, more resource efficient and sustainable businesses and, health and wellbeing benefits for local communities as they engage with and take ownership of their local river environment. Due to the diversity of the partnerships, the technical expertise therein and the development of catchment plans – underpinned by robust data and evidence – CaBA partnerships have proven adept at securing project funds from multiple sources: to date, CaBA has leveraged, on average, more than 8 times the initial core investment.

The partnerships also have strong convening power, providing a catalyst to bring together a range of sectoral interests and to address land and water management in a strategic manner, at the whole catchment scale. The involvement of catchment partnership in the process of planning and delivery of NFM projects may thus be particularly functional to the successful implementation of these measures. In particular, catchment partnerships may act as super partes actors with the sole aim to deliver the NFM measures within the catchment area. In doing this, the partnership could involve all stakeholders and try to mitigate the conflicts that might arise between different parts and counteract the different interests by aiming at delivering NFM solutions on the ground with the aim of taking maximum advantage of the individual measures in terms of cost-effectiveness. In the context of Natural Flood Management, the role of catchment partnerships is particularly relevant to bring together local knowledge and expertise and to involve all important actors and stakeholders in the design and delivery of NFM projects.

1.6 In-channel large wood (LW) and Engineered Log Jams (ELJs)

From historical times to present, in-channel large wood (LW) has been recognised as a pivotal component of naturally functioning river ecosystems (Abbe and Montgomery, 1996; Wohl, 2017). LW can be described as single or multiple pieces of wood longer than 1 meter and more than 10 cm in diameter. The term can refer to entire trees, roots, trunks, logs, branches and other large pieces of wood that can accumulate within a river system (Piégay and Gurnell, 1997; Wohl et al., 2019). Large wood has a natural tendency to accumulate at retainment locations within the channel and form accumulations of debris acting as dams, therefore, slowing the flow and rising water levels locally (Thevenet et al., 1998).

In the literature, several different terms are used to define structures formed by natural accumulations of wood in streams and rivers or as artificial nature-based

intervention: large woody debris, large wood and leaky dams, woody dams, and log jams. The arbitrary definition of large wood cited above is commonly used to describe in-stream wood naturally accumulated and with heterogeneous structures. The term Large Wood (LW) will therefore be applied in this thesis when referring to naturally occurring wood accumulations and in a general context to refer to characteristics shared by all in-stream woody structures. Otherwise, the term Engineered Log Jam (ELJ) will be used throughout this thesis when referring to artificially displaced wood for flood protection purposes.

1.6.1 Physical effects of large wood

The physical effects of natural occurring large wood can occur at multiple scales within river corridors (Figure 1.5). Numerous studies have shown that LW increases flow resistance (Shields and Smith, 1992; Manga and Kirchner, 2000; Curran and Wohl, 2003) and creates backwaters (Bocchiola, 2011; Davidson and Eaton, 2013) with an associated decrease in average flow velocity and an increase in flow depth (Gippel, 1995) and spatial variations in sediment entrainment and transport (Keller et al., 1983). Moreover, LW enhances hyporheic exchange (Sawyer et al., 2012) and has the ability to deflect flow toward the bed and banks, thus creating local scour. Large wood can also affect the type and dimensions of bedforms within a river, the location and dimensions of pools, as well as the cross-section channel geometry (Curran and Wohl, 2003; Montgomery et al., 2003).

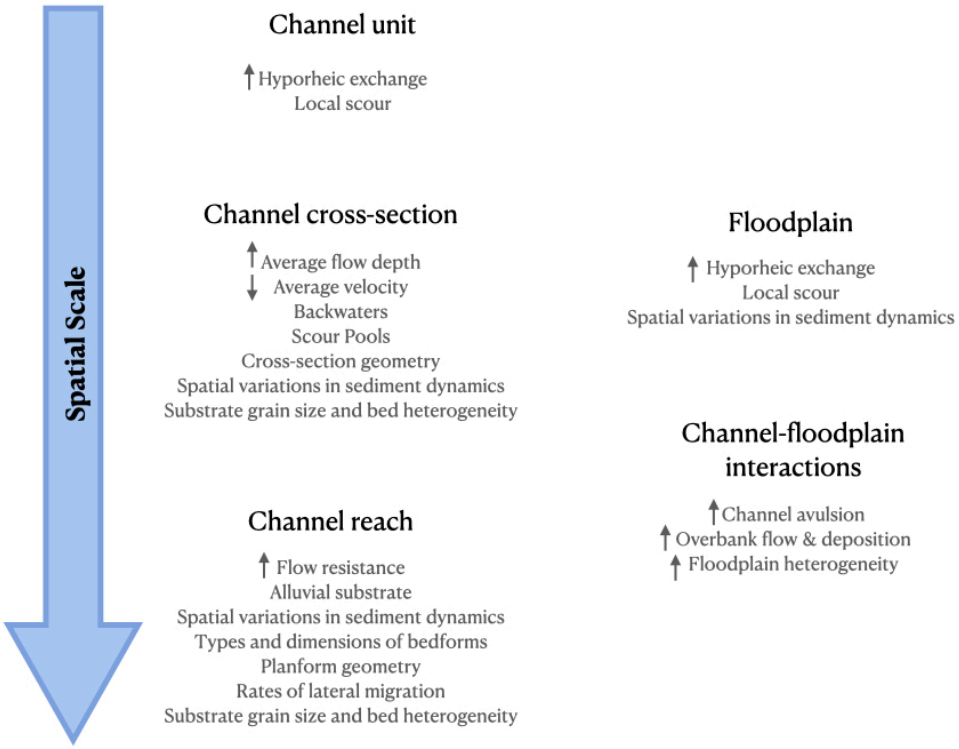


Figure 1.5 Schematic illustration of the multiple physical effects of large wood within the river corridor. *Physical effects are shown in relation to spatial scale and portion of the river corridor. Small black arrows indicate the direction of the change of the physical effects due to presence of large wood. Adapted from Wohl et al. (2017).*

Moving our focus to the whole river corridor, LW can facilitate channel avulsions (Abbe and Montgomery, 2003), increase the occurrence of overbank flooding and thus enhances sediment accumulation in the floodplain (Jeffries et al., 2003; Sear et al., 2010).

The physical effects imposed upon the channel and the floodplain by large wood concur in determining a range of effects on hydraulics, hydrology and ecology which will be addressed in the following Sections (1.6.2-1.6.3).

1.6.2 Effects of large wood on hydraulics, hydrology and flood risk

The effects of large wood and ELJs are site-specific but can generally show a positive flood risk management effect by reducing flood risk locally for small events, increasing hydraulic

roughness, reducing flow velocities, increasing the travel time of the flood wave, creating temporary storage and attenuate flood flows, and increase floodplain connectivity (Gregory et al., 1985; Shields and Gippel, 1995; Abbe and Montgomery, 2003; Curran and Wohl, 2003; Dixon, 2013; Montgomery et al., 2003; Kitts, 2010; Valverde, 2013; Thomas and Nisbet, 2012; Odoni and Lane, 2010; Sear et al., 2010).

Observed and modelled evidence on the effectiveness of LW in reducing and delaying flood peaks is shown in Table 1.2. Observed evidence showed that large wood is generally effective from a flood risk management perspective at the local scale (Wenzel et al., 2014). Modelled evidence showed that large wood placement measures can slow the flood wave locally and delay the timing of the flood peak. Generally, the effectiveness of large wood on flood risk management has been shown to be effective at the reach and small catchment scale (up to 15 km^2) for moderate flood events. However, the positive effects of reducing flood peak height and its timing are usually localised and may depend on catchment characteristics and flood generation mechanisms.

In order to increase their effectiveness and reduce the required maintenance over time, ELJs have been often paired with other NFM and restoration measures such as in-catchment woodland planting or floodwaters retaining structures such as ponds, bunds and wetlands (Dixon, 2013; Thomas and Nisbet, 2012). Large wood was an early adoption of restoration projects, to improve degraded rivers and channels with rapid uptake supported by best practice guidance (Grabowsky et al., 2019). If used correctly, these large wood features can have beneficial physical and environmental effects, which can be harnessed to deliver a suite of benefits in terms of ecological state and flood risk management (Grabowsky et al., 2019). The ecological effects of large wood will be discussed in the following Section.

Table 1.2 Effects of large wood at different scales and for different catchment characteristics.

Study	Type of evidence	Effects	Type of LW	Scale and catchment characteristics
Gregory et al. (1985)	Observed (pre and post-clearing of natural large wood)	100 minutes delay for small events (0.1 m^3/s), 10 minutes delay for discharge of 1 m^3/s	Naturally occurring LW	Channel scale (4 km). Forested catchment (New Forest).
Wenzel et al. (2014)	Observed	The deployment of 9 LW dams was shown to delay the flood wave propagation as a result of increased channel roughness. A small decrease in peak discharge (2.2%), a delay of the flood wave of 166 s and a reduction of 15.5% in flow velocities were observed for a 3.5-year experimental (artificial) flood event.	Artificially installed LW dams.	Reach scale (0.282 km). Small mountain stream in a forested catchment in southeast Germany (Ore Mountains).
Kitts (2010)	Modelled	LW accumulations could alter the timing of small flood peaks by 33%.	Naturally occurring LW of different types (complete, partial etc).	Catchment Scale (12 km^2).

Thomas and Nisbet (2012)	Modelled	5 LW dams were shown to delay travel time of a 1% AEP flood wave by 15 minutes, with a 2.1 m/s reduction in velocity and an increase in water levels due to increased roughness. No changes in flood magnitude.	Restoration of 5 LW dams Simulated via a 70% blockage function.	Reach scale (0.5 km) in a forested part of the Y Fenni catchment (9 km^2).
Dixon (2013)	Modelled	ELJs simulation showed unpredictable results in peak flow discharge at the catchment scale with changes ± 4%. The deployment of ELJs in the downstream part of the catchment resulted in an increase in discharge due to peak synchronisation.	Modelled ELJs were modelled at a spacing of 7-10 channel widths across the catchment.	Catchment scale. Highland Water Catchment, New Forest (26 km^2).
Odoni and Lane (2010)	Modelled	Installation of 100 ELJs could reduce the magnitude of a flood event by 7.5%.	NA	Pickering Beck catchment (68 km^2).

1.6.3 Ecological Effects of large wood

The formation of the so-called debris dams, as a result of woody debris accumulation at a point of obstruction, performs beneficial functions in terms of river ecology and hydro-morphology (Pinto et al., 2019). LW increases channel roughness and creates a more diverse habitat from both a hydraulic and sedimentary point of view (Gurnell and Linstead, 1998; Keller and Swanson, 1979; Montgomery et al., 2003). Moreover, in-channel wood retains organic matter, increasing the local availability of food resources for detritivore communities (Cashman et al., 2016; Flores et al., 2011) and attenuating nutrient enrichment that would otherwise reduce water quality (Krause et al., 2014). As a result, LW has been highlighted as providing essential benefits in terms of biodiversity and habitat improvement (Benke and Wallace, 2010; Benke et al., 1985; Schneider and Winemiller, 2008).

Despite its beneficial functions, large wood has been removed from streams and rivers over long timescales as a result of concerns over conveyance capacity and blockage risk or as a result of changes to land use, flow regulation and installation of embankments (Diehl, 1997; Gippel et al., 1996; Wohl, 2014). However, more recently, the critical role of LW in streams and rivers in terms of both flood risk reduction and on the functioning of the riverine ecosystems has started to be recognised. Many restoration and flood risk management projects have recently made use of in-channel large wood restoration as the main measure to achieve their objectives (Palmer et al., 2010; Wilkinson et al., 2019a; Wohl, 2017). This newly generated interest in LW as a river restoration and NFM technique came with the recognisance of its potential in increasing the storage of water upstream in the channel incision, re-connecting the channel to the floodplain, acting as a sediment trap, and having an impact on channel morphology (Barlow et al., 2014). In addition to flow mitigation and flood risk reduction, it is also suggested that this feature will aid water quality improvement and habitat and biodiversity enhancement (Wasson et al., 2010).

The deployment of Engineered Log Jams (ELJs) into a channel is now often suggested as part of NFM schemes in the UK. However, both the flood risk reduction potential and the environmental co-benefits of NFM structures and, specifically, Engineered Log Jams, are still to be investigated appropriately.

1.6.3.1 Large wood controls on biogeochemical processing, nutrient turnover and in-stream biota

Large wood has an effect on many environmental processes, including nutrient cycling and biogeochemical processing (Table 1.3).

Table 1.3 Environmental processes affected by the presence of large wood. *(Adapted from Magliozzi et al., 2019).*

Environmental Processes	Large wood effects compared to control	References
Hyporheic exchange flow	+	Krause et al. (2014)
Oxygen availability	+	Kaller and Kelso (2007)
Temporal stability	-	Gurnell (2012)
Organic Matter	+	Blaen et al. (2018)
Denitrification	-	Shelley et al. (2017)
Hydraulic Conductivity	+	Mutz et al. (2007)

LW is a natural hydrostatic driver of the increase of hyporheic exchange flow in the river bed (Magliozzi et al., 2019; Krause et al., 2014) and has the potential to enhance Hyporheic Exchange Flow by increasing hydraulic roughness and creating hydraulic gradients (Lautz, et al., 2006; Mutz et al., 2007). Hyporheic exchange flow (HEF) is a subsurface water flow that highly affects river ecology. In particular, HEF controls dissolved oxygen concentrations and fine sediment deposition (Mathers et al., 2014), metabolic processes (Krause et al., 2014), nutrients (Krause et al., 2013) and water temperature (Menichino and Hester, 2014).

The geomorphological changes induced by LW facilitate the deposition of organic material and sediment material and promote nutrient processes such as nitrate accumulation or denitrification (Blaen et al., 2018; Shelley et al., 2017; Craig et al., 2008; Mulholland et al., 2008). These changes in geomorphological and environmental processes have been shown to have an effect on the benthic macroinvertebrate taxonomic structure, in terms of greater abundance, diversity and biomass when compared to sites without wood (Benke and Wallace, 2010; Thompson et al., 2018).

Diffuse nutrient pollution is one of the main water quality problems in agricultural dominated lowland river basins. In particular, the increase in nitrate concentrations in both surface water and groundwater has been shown to have adverse impacts on the environment as well as an economic cost of removal or blending (Craig et al., 2008; Mulholland et al., 2008). Initial evidence of nutrient pollution attenuation induced by the presence of large wood structures were shown in laboratory and experimental field studies (Mutz et al, 2007; Craig et al., 2008). Moreover, the three major drivers of streambed microbial metabolic activity and biogeochemical cycling are directly impacted by the presence of in-stream large wood. These three drivers, hyporheic residence times, temperature and access to bioavailable dissolved organic carbon, lead to the formation of streambed hotspots for nitrogen removal or production (Krause et al., 2014).

1.6.4 Knowledge Gaps

The current scientific knowledge has provided insights and modelling frameworks that can be useful when designing and managing ELJs for different projects. Despite the progress, knowledge gaps remain (Grabowski et al., 2019). These knowledge gaps include the assessment of the suitability of approaches to represent stationary LW in numerical hydraulic and hydrological models (Addy and Wilkinson, 2019; Nisbet et al., 2011). To date, no accepted guidance or bespoke tools are available (Forbes et al., 2015; Metcalfe et

al., 2018; Pinto et al., 2019). In part, this reflects the difficulties in modelling such complex, porous features that can vary in design, setting and form but also the fact that the interest in these structures for restoration and flood management purposes is relatively recent. Not accounting adequately for the presence of in-channel LW can result in erroneous hydraulic predictions at the reach or local scale (Nahorniak et al., 2018; Pasternack et al., 2004).

Moreover, Dadson et al. (2017) and Addy and Wilkinson (2019) highlighted how the current knowledge on the effect of ELJs deployment is also limited to the small scale and for flooding events of small magnitude. Modelling scaled up to a larger catchment scale (>100 km^2) is therefore needed to determine the potential hydrological effects that multiple engineered log jams could have at the large catchment scale. Moreover, various flooding events with varying return periods should be tested when assessing the impact of ELJs at both the reach and catchment scales (Dadson et al., 2017).

In terms of the positive effects on aquatic biodiversity, many studies found that the presence or addition of large wood can benefit freshwater invertebrate communities (Magliozzi et al., 2019; Pinto et al., 2019; Palmer et al., 2010; Miller et al., 2010). However, most studies focused on natural in-stream wood accumulations, applied a plethora of different study designs, and dealt with really different structures of large wood, mainly in semi-natural catchments. To date, only one study has considered the response of freshwater invertebrate communities to ELJs developed during NFM projects, and none have considered other complementary measures to assess the state and functioning of the riverine ecosystem after large wood addition (Deane et al., 2021). Moreover, most studies focused on natural or semi-natural ecosystems and catchments (Magliozzi et al., 2019; Pinto et al., 2019; Pilotto et al., 2016; Pilotto et al., 2014), while little is known about catchments and rivers highly impacted by human activities.

1.7 Aims of the project and research objectives

This thesis aims to understand how Natural Flood Management and, in particular, one technique, the Engineered Log Jam, can affect river hydraulics, freshwater aquatic biodiversity, ecosystem services provision at the reach scale, and flood hydrology at the catchment scale in a rural context.

Within this overarching aim, I addressed the following objectives:

- Chapter 1: To review existing literature about the multiple benefits of Natural Flood Management at different scales and from different perspectives. To narrow the focus on engineered log jams by highlighting the effects of this NFM feature on channel hydraulics, catchment hydrology and freshwater ecology.
- Chapter 2: To understand the changes in channel hydraulics introduced by engineered logjams (ELJs) at the reach scale. This objective will be pursued with hydraulic modelling and data from surveys in the field in order to measure the flow resistance of ELJs. To analyse how ELJs can be represented in a hydraulic model with a particular interest in understanding if Manning's *n* parameter for hydraulic roughness can be considered a good proxy for engineered log jams representation.
- Chapter 3: To determine the sensitivity of a large catchment to changes in channel roughness and to select priority areas within the catchment for the application of the changes in roughness.
- Chapter 4: To understand the impact of ELJs on floodplain inundation, in-stream velocities and peak discharge magnitude and timing at the catchment scale by coupling a hydrological model to simulate the catchment contribution to hillslope runoff and a two-dimensional hydraulic model to simulate in-stream ELJs.
- Chapter 5: To investigate the potential environmental co-benefits of Engineered Log Jams. In particular, to understand the structure of the freshwater benthic

macroinvertebrate communities in response to the new habitats created by ELJs, and to investigate the potential of ELJs in providing an increase in ecosystem services and, specifically, organic matter decomposition, as a result of changes in river morphology following ELJs deployment.

- Chapter 6: To summarise the key findings of this thesis, while highlighting their implications from multiple perspectives. To draw conclusions about potential applications of ELJs in different contexts and at different scales and describe the potential implications for policy and funding.

The research was undertaken in one of the government's NFM trial pilot regions, namely the Evenlode and Littlestock Brook catchments. The Littlestock Brook NFM project was developed by the Evenlode Catchment Partnership and the Environment Agency to reduce flood risks for one of the tributaries of the river Evenlode while, at the same time, improving water quality, decreasing sediment and nutrients transport downstream and achieving the targets set out by the Water Framework Directive. A brief introduction to the study area and the NFM project is provided in the following section.

1.8 Study site: The Evenlode catchment and the Littlestock Brook NFM project

The Evenlode is a large (431 km^2) rural catchment situated in the upper reaches of the Thames basin in southern England (Figure 1.6). In this catchment, the fluvial flooding of 2007 caused extensive damage with 318 properties affected (Old et al., 2019). Lithology within the Evenlode is dominated by the Great Oolite Group, consisting of mudstone and fine-grained limestone. The western part of the catchment is mostly underlain by the Lias Group, consisting of clays, mudstones and limestones.

The Evenlode fulfils an important role for the Thames catchment and the city of Oxford, being one of the four catchments whose output flows into the city. Oxford has a long history of fluvial flooding with the last extensive event occurring in 2013.

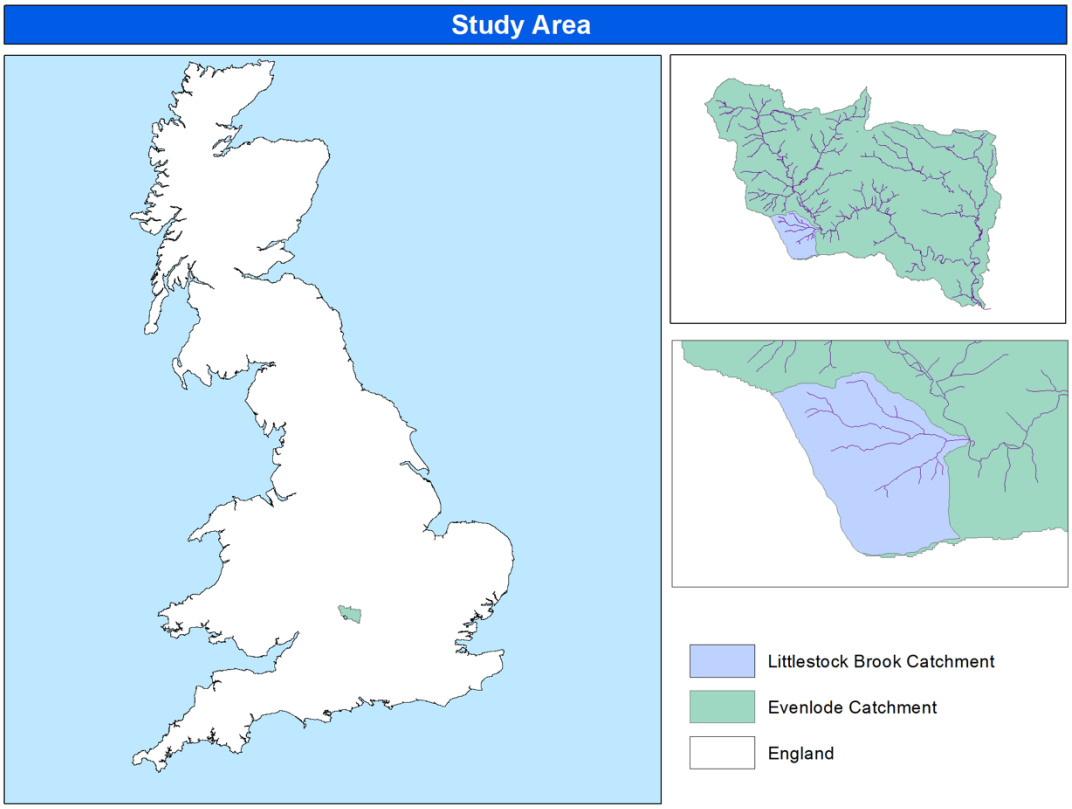


Figure 1.6 The Evenlode and the Littlestock Brook catchments. *The Littlestock Brook is located in the far-left bottom of the Evenlode catchment, flowing through the Wychwoods.*

In recent years, Thames Water and the Evenlode Catchment Partnership have been developing an NFM 5 years project (2016-2021) in a sub-catchment of the Evenlode: the Littlestock Brook catchment (Oxfordshire). The Littlestock Brook catchment has therefore been extensively monitored as part of the NFM scheme developed to reduce flood risk in the village of Milton-under-Wychwood.

The Littlestock Brook is a rural catchment with an estimated area of 16.3 km^2 mainly consisting of farmland and pasture fields with two main urban centres, Milton and Shipton-under-Wychwood, both located in the downstream region of the catchment (Figure 1.6).

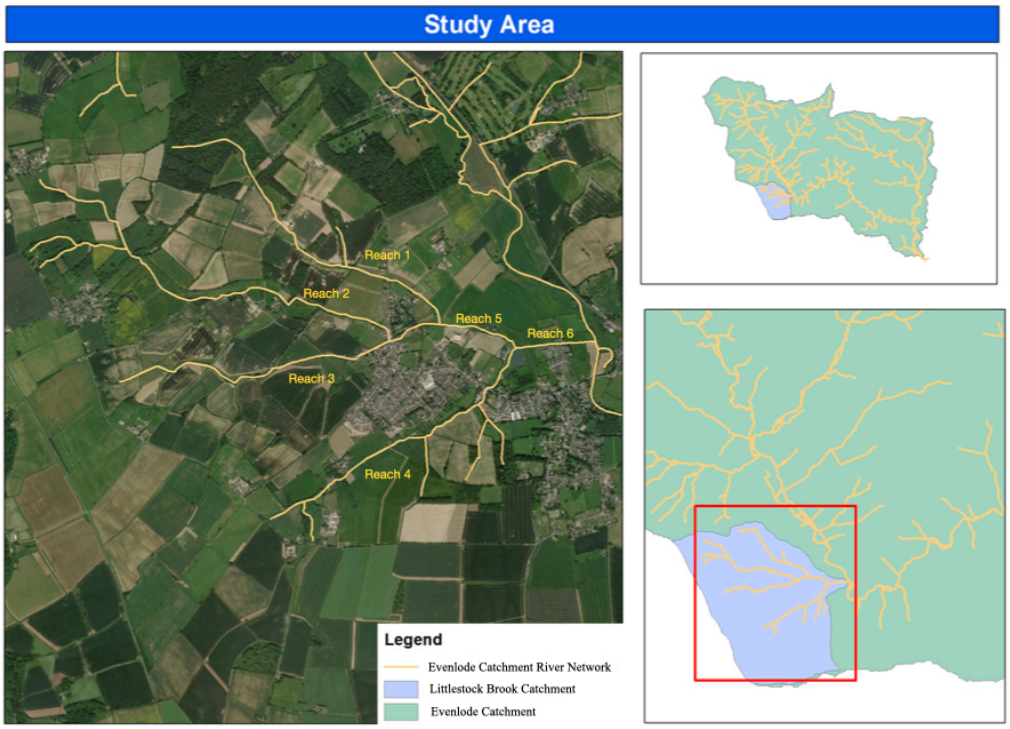


Figure 1.7 The Littlestock Brook catchment. *Milton and Shipton-under-Wychwood are the two main urban areas of the catchment. The Littlestock Brook has its confluence in the Evenlode north of Shipton-under-Wychwood.*

The stream network is formed by small channels and farmland draining ditches and has an approximate total length of 36 km. The average slope of the catchment is relatively steep for England, with elevations ranging from 243 to 91 meters Above Datum (mAD).

This relative steepness, the land-use types, and landcover account for the rapid response to storm events, meaning relatively high discharges with short lead times.

The Littlestock Brook NFM project has involved the design of several nature-based interventions. Over five years, Wild Oxfordshire proceeded with the installation of 42 Engineered Log Jams together with the other NFM measures, including ten corner bunds, three inline ponds and multiple soil scrapes (Figure 1.8).

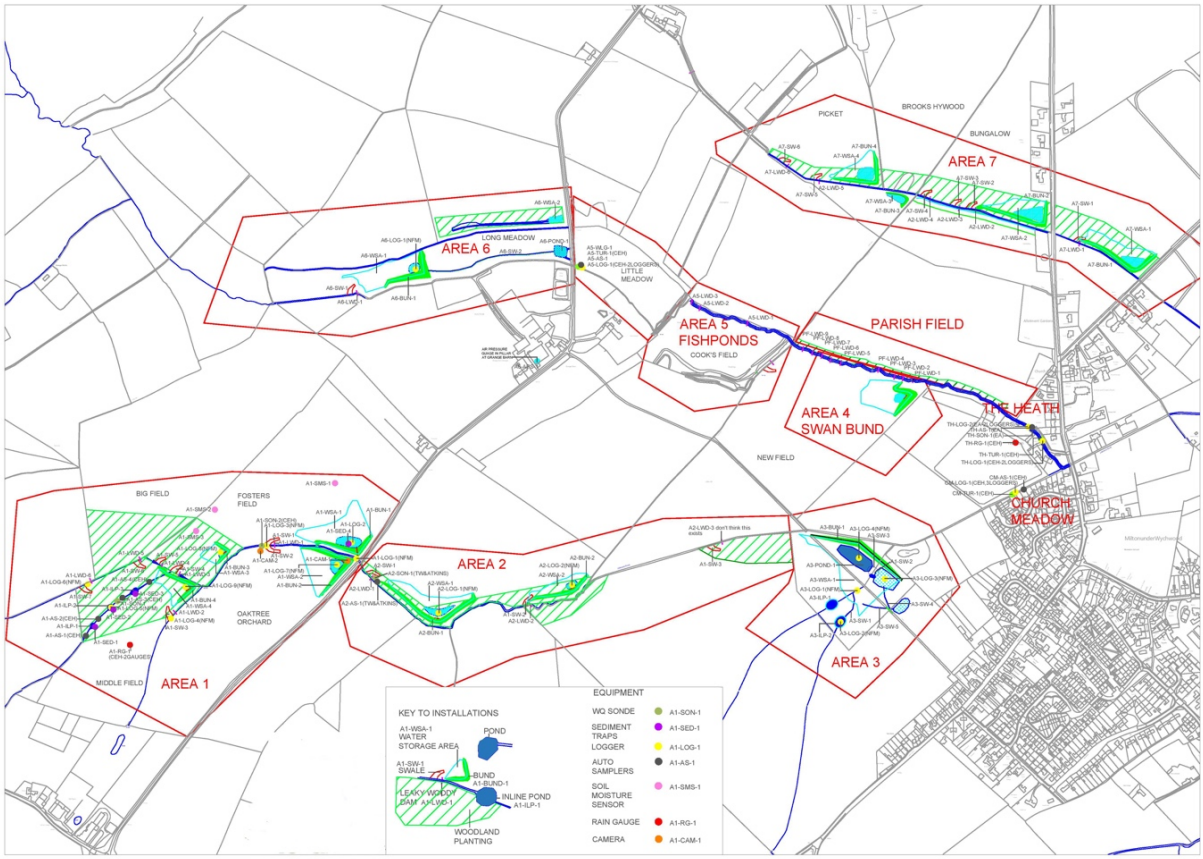


Figure 1.8 The NFM measures installed in the Littlestock Brook catchments*. The Littlestock Brook is a sub-catchment of the Evenlode with four main tributaries. The NFM structures' symbology is included in the legend of the figure. Different areas correspond to different phases of implementation and installation of NFM features (courtesy of Wild Oxfordshire).*

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Chapter 2 : Understanding and modelling the effects of Engineered Log Jams on channel waterflow at the reach scale: a case-study from the Evenlode catchment.

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2.1 Introduction

Large wood (LW) is a natural element of freshwater ecosystems which influences the physical structure of watercourses as it can increase streambed heterogeneity by forming scour pools causing sediment sorting and altering water depths and flow velocity (Rasche at al., 2019; Pilotto et al., 2014). However, LW has been historically removed from river reaches in order to increase channel conveyance and over concerns for blockage risks at structures such as bridges and culverts (Diehl, 1997; Erskine and Webb, 2003; Lassettre and Kondolf, 2012; Wohl, 2014).

In the last two decades LW has been increasingly used as a measure in flood risk management in the form of Engineered Log Jams (Dittrich et al., 2019; Wilkinson et al., 2010a) and re-introduced as a restoration measure in degraded rivers and channels (Addy and Wilkinson, 2019a; Bernhardt et al., 2005; Brummer et al., 2006; Harvey et al., 2018; Kail et al., 2007). ELJs deployed as a NFM measure, are indeed thought to reduce the likelihood of flooding downstream by increasing the temporary storage of water in the channel incision, by reconnecting the channel to the floodplain (Keys et al., 2018; Kitts, 2010; Wenzel et al., 2014).

These effects are the result of increased channel hydraulic roughness (Curran and Wohl, 2003; Kitts, 2011; Sear et al., 2006; Shields and Gippel, 1995) and, consequently, reduced water velocities through the reach. The reduction of water velocities can consequently increase flood wave travel times and provide temporary storage of water in the channel incision (Shields and Gippel, 1995; Gregory et al., 1985; Thomas and Nisbet, 2012).

Engineered Log Jams have also been shown to increase habitat diversity by creating potential new freshwater habitats by altering channel morphology (Abbe and Montgomery, 1996; Doyle et al., 2007), favouring pool formation, (Manners et al., 2007), and impacting the variability of channel width and depth (Smith et al., 1993).

ELJs are therefore increasingly seen as a multi-faceted restoration option with potential benefits for flood mitigation and enhancing aquatic biodiversity and ecosystem functioning (Brookes et al., 2005; Lehane et al., 2002; Shields Jr et al., 2003; Shields Jr et al., 2004). Given these perceived benefits, recently the UK has seen an arising interest in ELJs as a Natural Flood Management technique with funding via the UK Government's Natural Flood Management schemes (https://www.gov.uk/guidance/use-nature-based-solutions-to-reduce-flooding-in-your-area). NFM techniques provide a good alternative method to physical interventions for flood risk reduction, especially for situations in which securing funding and setting aside land for the development of big traditional engineering solutions may be difficult or unfeasible. ELJs are one of the most used NFM measures for the ease of deployment, wide applicability into different contexts and channel types, the low cost required for their installation and their potential to provide a wide array of environmental co-benefits. However, it is widely acknowledged that there are still knowledge gaps relating to the effectiveness of ELJs at large catchment scales (> 100 km^2) and for extreme flood events (Dadson et al., 2017; Wilkinson et al., 2019b) including a need to assess the best methodologies to accurately depict these features in hydraulic and hydrological modelling and understand their impacts on flood events of different magnitudes.

2.1.1 Classification of large wood types

Large wood dams and log jams can be found in the literature with various names and reported under different classifications and naming conventions for the different types of structures and their positioning. For example, the classification proposed by Gregory et al. (1985), following surveys in the Highland Water catchment (UK), is based into four distinct large wood dam types. *Active* dams span the whole channel width and cause a step in the water surface profile; *complete* dams span the whole channel but are not causing a step in

the water profile as they are semi-porous structures; *partial* dams are those composed by logs or large wood attached to the channel bank but not spanning the entire channel width, while *high water* dams included at least one piece of wood bridging across the bank tops (Gregory et al., 1985; Piégay and Gurnell, 1997).

Wallerstein (1997) proposed a similar classification for forested streams in Northern Mississippi (USA), based on the influence of logjams on the hydraulic and sedimentation processes. Wallerstein defined *underflow dams* as the ones effective on the water flow only at relatively high discharge and spanning the whole width of the channel (similar to the *high-water* dams described by Gregory et al.). *Dam jams,* on the other hand, were described as the ones producing a step in the water profile and spanning the whole width of the channel (similar to Gregory's et al. *active* jams). *Deflector Jams*, were the partial dams bank attached which exerted an influence on the water flow by deflecting it to one of the two banks thus affecting bank morphology (similar to Gregory's et al. *partial dams*). *Parallel jams* were those parallel to the water flow, not bank attached and only covering a portion of the surface area. Another more complex classification, including ten types of log jams, was proposed for natural wood formations by Abbe and Montgomery (2003) based on the key members of the wood association structure, their orientation and the presence of loose wood.
For the context of this study, the classification by Gregory et al. (1985) is the most useful one and will be used in this work when referring to specific log jam types.

As explained in Chapter 1, unless stated, I will use the term 'large wood' to refer to the characteristics and the effects at the reach and catchment scale shared by both naturally occurring large wood and artificially re-introduced wood. The term Engineered Log Jam (ELJ*)* will otherwise be used when specifically referring to wood structures artificially re-introduced for natural flood risk management. Moreover, the naming convention created by Gregory et al. (1985) will be used below to refer to the specific types of large wood and ELJs as specified in the section above.

2.1.2 Effects of Large Wood and Engineered Log Jams at the local and reach scales

The presence of in-channel LW can impact both upstream and downstream channel velocity patterns. Upstream patterns are characterised by a constriction and subsequent acceleration of the flow when approaching the obstruction posed by LW, while a flow separation can occur downstream of the obstruction (Abbe and Montgomery, 1996; Shields Jr and Gippel, 1995). Abbe and Montgomery (1996) described how the obstruction created by LW causes flow recirculation that can lead to the formation of a pool just upstream of the obstruction (Figure 2.1). Moreover, the flow acceleration that characterises water downstream of a porous LW obstruction, caused by the constriction of water flow while bypassing the obstruction and the formation of turbulence, can lead to local scour and the appearance of a scour pool downstream of the large wood.

Research has also indicated that the effects of LW on channel hydraulics are strongly dependent on the geometry, complexity and scale of a feature. Generally speaking, the hydraulic resistance generated by large wood is higher for those features that occupy a relatively large portion of the cross-sectional area (Gippel et al., 1996, Manners et al., 2007; Wilcox et al., 2006). However, even a small volume of large wood can offer a relatively high increase in resistance to the water flow (Manga and Kirchner, 2000). Moreover, dense LW structures are associated with the highest drag forces and the lowest upstream velocities. Other characteristics that can influence LW and ELJs include their orientation, with perpendicular logs having a more significant impact on flow resistance (Wilcox et al., 2006; Gippel et al., 1996), and log position in the water column, with logs positioned on the bed or just submerged exerting the highest drag forces (Gippel et al., 1996; Shields Jr and Alonso, 2012). Finally, the flow resistance exerted by LW depends on river discharge: generally, the resistance exerted by large wood decreases as the flow stage increases (Dudley et al., 1998; Wilcox et al., 2006).

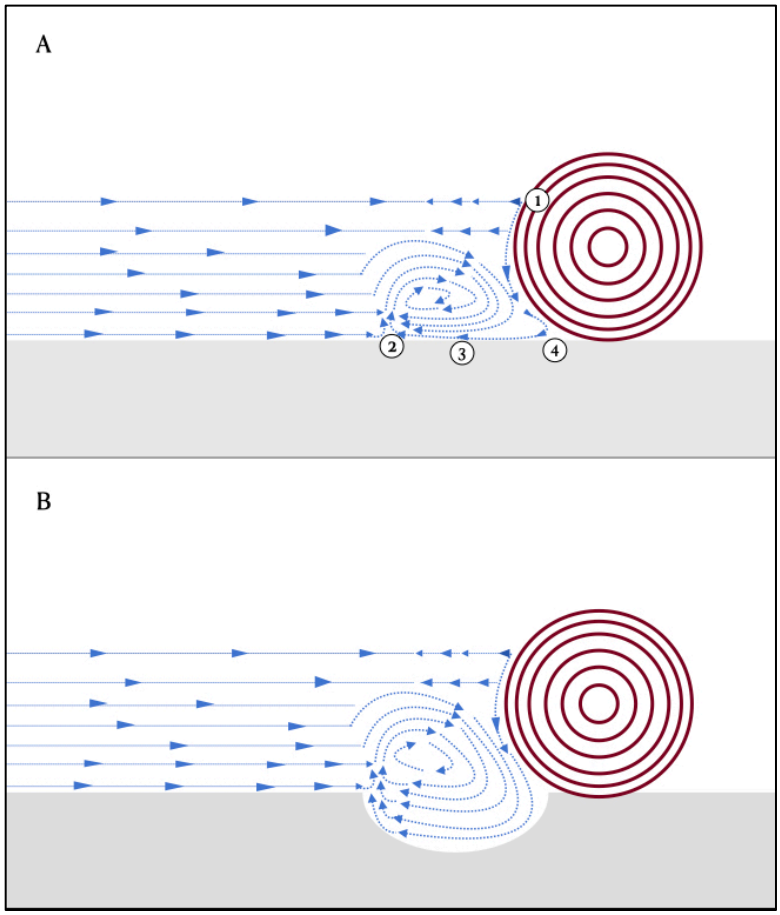


Figure 2.1 Profile for the schematic representation of flow patterns upstream of an obstruction created by large wood*. Figure A: point 1 corresponds to the development of a downward acceleration in the water flow caused by the inversion of the flow direction when hitting the obstruction; point 2 represents the saddle point, and points 3 and 4 delineate the zone of vortex scour. Figure B: an upstream pool formed by the newly established flow patterns against the obstruction.*

Apart from the local disturbance in downstream and upstream velocity patterns, LW and ELJs can also impact water delivery downstream by storing water in the channel incision. When the water approaches an in-channel LW obstruction, there is stalling due to the damming effect with lower water velocities and higher depths upstream of the obstruction. Lower velocities can, in turn, influence flood travel times through the reach (Daniels and Rhoads, 2003; Gregory et al., 1985). Moreover, as shown in Figure 2.1, LW can also increase the occurrence of pools, leading to even slower water velocities as a result of additional ponding of water at these locations. Not all publishes studies, however, indicated benefits. Work by Gippel et al. (1996), for example, suggested that the backwater

and ponding effect upstream of an LW feature may increase the flooding frequency upstream. However, when the floodplain is inundated, it can store water temporarily and flow resistance may increase as the flow interacts with the floodplain roughness (Keys et al., 2018; Jarrett, 1984). The impact of ELJs on floodplain inundation at the reach and catchment level will be addressed in Chapter 4.

2.1.3 Manning's roughness coefficient

Traditionally, roughness coefficients have been used to represent the total resistance to water flow in channels and floodplains. Roughness values are composite terms that embrace all the frictional or retarding effects causing energy losses in the movement of water (Clifford et al., 1992). Total flow resistance consists of several components (Bathurst, 1997), including grain, form, and spill resistance. Grain resistance is the roughness provided by the individual grains and particles that are part of the bed of the channel (Parker and Peterson, 1980). Form resistance is the resistance created by bed forms or other sources of form drag, such as vegetation (Wilcox et al., 2006). The form resistance arises from pressure drag created by the flow separation around an object (Chadwick et al., 2021). Spill resistance is the resistance generated by impromptu velocity changes due to free surface distortion through waves and turbulence (Leopold et al., 2020).

Among the factors contributing to total flow resistance, in-channel LW and ELJs represent one of the most significant contributors (Dixon, 2013).

The most used coefficients of roughness are Manning's n, Chezy C and Darcy Weisbach friction factor (ff). Manning's n is the most commonly used measure of the total resistance offered by channel boundaries and obstructions to the water flow in hydraulic and hydrological models.

Manning's formula (2.1) for streamflow velocity computation has been extensively applied in various studies as a roughness parameter to describe velocity through a reach as a function of the slope of the channel and the hydraulic radius:

$$v = \frac{1}{n} S^{\frac{1}{2}} R^{\frac{2}{3}} \tag{2.1}$$

where v is the reach-averaged velocity, S is the slope of the energy gradient (generally the water surface slope), and R is the hydraulic radius.

Although Manning's equation *(2.1)* was initially developed and tested for uniform flow conditions, it has been widely used in modelling studies for non-uniform flows, given its reliability in different contexts. In modelling studies, Manning's *n* values are typically selected from the tables presented in Chow (1959) and Barnes (1967). However, there is always a certain degree of subjectivity in the choice of Manning's *n*, and the modeller experience is essential in selecting the correct values for Manning's *n*. Manning's *n* value in hydraulic and hydrological modelling is often calibrated in order to allow a best fit between modelled and observed flood events.

Given the variability of large wood structures in size, complexity and shape and the other factors impacting channel roughness, it is challenging to choose Manning's n values to adequately represent these structures since the tabular values were selected for generalised situations. Moreover, Manning's n values vary in both the longitudinal and lateral direction within the channel and are influenced by flow conditions; therefore, it is difficult to represent this heterogeneous situation with a limited number of Manning's n values in modelling studies.

It is thus necessary to quantify the hydraulic resistance associated with large wood in its different forms within channels of various slopes and bedforms to better understand

the hydraulic properties of these structures and parameterise hydraulic and hydrological models.

2.1.3.1 Large wood's roughness

To date, only a few studies have measured the roughness associated with large wood, and there is a need to extend this work to empirically calculate logjam hydraulic resistance at high flows, in different channel types and for different ELJ types.

In addition to being an essential component of flow resistance, LW can contribute to channel roughness by influencing other channel characteristics that contribute to the total flow resistance such as channel geometry, bedform, bank irregularities, stream meander and bed material (Chow, 1959).

Manning's n has been originally developed by Manning (1891) as an empirical method to estimate channel velocity. Manning's n represents the total flow resistance as:

$$n = \frac{1}{v} S^{\frac{1}{2}} R^{\frac{2}{3}} \tag{2.2}$$

Where v is reach-average velocity, S is the slope of the energy gradient (usually the water surface slope), and R is the hydraulic radius.

Large wood in rivers can be a source of significant roughness and consequently influence flow at local and reach scales (Hygelund and Manga, 2003). In-channel large wood can consistently increase the hydraulic roughness of the channel thus being one of the main factors in the reduction of water velocity, in the rise in upstream water levels and attenuation of the peak flood and its timing (Kitts, 2010).

The few studies that have directly measured Manning's n in the field are summarized in Table 2.1. As shown in the table, Manning's *n* coefficients published in the literature were measured for very different stream types, from mountainous step-pool channels (Curran and Wohl, 2003) to straight sand-bed rivers (Shield and Gippel, 1995); for different catchment

sizes, ranging from approximately 10 km^2 (Dixon, 2013; Kitts, 2010) to more than 1000 km^2 (Shield and Gippel, 1995), and for different flow and ELJ types.

In general, measured Manning's n values reported in the literature are in between 0.058-2.22, highlighting a high variability in Manning's n for different large wood types, flow and catchment conditions (Table 2.1). This high variability in the data may be explained by the different LW types, slope of the channel and drainage area of the measured and by the big ranges of flows during which measures were taken (Addy and Wilkinson, 2019).

Most of the published studies were measuring Manning's *n* for natural channels in forested catchments. However, little is known for more heavily modified channels and catchments in agricultural settings and for artificially developed large wood structures developed for flood management purposes.

Table 2.1 Values of Manning's roughness coefficient (n) measured in the field for published studies.

Study	**LW type**	**Reach length**	**Reach slope**	**Channel**	**Flow conditions**	**Mean Manning's n** (range)	**Sample size** (N)	**Drainage area** (km^2)
Gregory, Gurnell and Hill (1985)	Active	NA	NA	NA	NA	**0.224**	1	NA
Shields Jr and Gippel (1995)	Partial	NA	0.0005-0.0008	Straight, sand bed	Range of flows	**0.058** (0.0435-0.0807)	9	> 1000
Shields Jr and Gippel (1995)	Partial	NA	0.001-0.0015	Sinuous, gravel/sand bed	Range of flows	**0.075** (0.0652-0.875)	3	427
Linstead and Gurnell (1998)	Active and complete	80-90 meters	0.0057	NA	NA	**0.677**	8	NA
Linstead and Gurnell (1998)	Partial	80-90 meters	NA	NA	NA	**0.348**	6	NA
Curran and Wohl (2003)	Active (natural)	50-60 meters	0.06-0.18	Step-pool, gravel/cobble bed	Low flow	**0.60** (0.2-1.5)	20	9.6-0.13

Kitts (2010)	Active (overflow)	20 meters	0.0057	Meandering, coarse gravel bed	Low flow	**1.4** (0.58-2.22)	4	11.2
Kitts (2010)	Active (underflow)	NA	0.0057	Meandering, coarse gravel bed	Low flow	**0.27** (0.142-0.398)	9	11.2
Kitts (2010)	Active (deflector)	NA	0.0057	Meandering, coarse gravel bed	Low flow	**0.32** (0.173-0.467)	12	11.2
Dixon (2013)	Active (natural)	20-35 m	0.004	Meandering, coarse gravel bed	High flow	**0.24** (0.137-0.362)	8	9-12
Dixon (2013)	Partial (natural)	20-35 m	0.012-0.013	Meandering, coarse gravel bed	High flow	**0.083** (0.027-0.199)	16	9-12
Dixon (2013)	High Flow (natural)	20-35 m	0.005-0.012	Meandering, coarse gravel bed	High flow	**0.158** (0.059-0.412)	8	9-12

2.1.4 Representation of LW and ELJs in hydraulic modelling

Several different approaches have been used to represent the hydraulic effects of LW and ELJs in hydraulic and hydrological modelling (Addy and Wilkinson, 2019). The most common representation of these structures is via adjustment of channel roughness to simulate LW flow resistance (Kitts, 2010). Other type of LW representation in modelling are the geometrical explicit representation, achieved by either introducing simplified structures such as embankments (Keys et al., 2018) or by modifying the cross-sections to simulate ELJs blockage (Valverde et al., 2013; Rasche et al., 2019) or with non-geometrical representations achieved via modifications of the hydraulic equations to simulate the expected drag forces (Wu et al., 2005). Other less common LW's representations in modelling were hybrids of the ones mentioned above such as the geometry and roughness adjustment applied by Valverde (2013).

The published studies considered different type of ELJs, different channels dimensions and drainage areas and different purposes; of the studies considered by Addy and Wilkinson in their review (2019) only nine were showing calibration and validation results (Thomas and Nisbet, 2012; Valverde, 2013; Pinto et al., 2019; Wu et al. 2005; Kitts 2010; Keys et al., 2018; Rasche et al., 2019; Bair et al., 2019) and only one study tried to compare different ELJ representations (Valverde, 2014).

2.1.5 Knowledge gaps

Despite the scientific progress in understanding the impact of LW and ELJs on river hydraulics and hydrology, significant knowledge gaps remain (Grabowski et al., 2019; Addy and Wilkinson, 2019). These include:

i) the quantification of ELJs associated roughness parameter in different settings and for different ELJs types. Specifically, only a few papers have quantified ELJs

associated roughness and most of them analysed natural settings with natural ELJ association. Little is known for artificial ELJs developed in heavily modified channels.

ii) the assessment of the suitability of approaches to represent ELJs in hydraulic modelling. Most studies represented ELJs through channel roughness modification, geometrical representation and blockage functions. Only a few studies have tried to compare the performances of different ELJ representation in order to establish a guidance for hydraulic modellers.

iii) the effects of the deployment of multiple artificial ELJs into a reach on the flood peak magnitude and travel time for artificial high water ELJs.

2.2 Aim of the chapter and research objectives

The aim of this chapter was therefore to address these knowledge gaps using field measurements and 1D hydraulic modelling using the Hydrologic Engineering Center's River Analysis System (HEC-RAS) to:

- Quantify the hydraulic resistance of high water ELJs deployed during a 5-year NFM project within an agricultural catchment by directly calculating Manning's n parameter using salt dilution.
- Implement a hydraulic 1D model of one reach with multiple ELJs to compare four different ELJs representations in hydraulic modelling
- Evaluate the effects of the deployment of multiple ELJs on the flood peak magnitude and travel time at the reach scale by using the 1D unsteady flow model informed with data collected with field surveys.

2.3. Materials and Methods

2.3.1 Study area

The study area lies within the Littlestock Brook Catchment in Oxfordshire, South-East England. The Littlestock Brook is a small (16.3 km^2) agricultural sub-catchment of the river Evenlode, one of the tributaries of the river Thames (Figure 2.2). This watershed consists mostly of farmland and pasture field with two small urban areas, Milton and Shipton-under-Wychwood. The stream network, formed by small channels and farmland draining ditches, has an approximate total length of 36 km. The Littlestock Brook catchment is characterized by relatively steep slopes for Oxfordshire with elevations ranging from 243 to 91 m.

This research was focused on the small watershed upstream of the village of Milton-under-Wychwood, which drains an area of approximately 6.7 km^2 and which is commonly referred to as "The Heath". This reach has been part of the 5-years Littlestock Brook NFM project led by the Environment Agency with the aid of other associations and stakeholders reunited in the Evenlode Catchment Partnerships. Due to the relatively high gradients of this area and the fact that most of the land cover is highly erodible, the Evenlode Catchment Partnership, decided to deploy 12 ELJs over the reach length with the aim to slow the flow, stabilize the channel bed and therefore retain some of the sediment that was being transported by the river downstream.

2.3.2 Field surveys: data collection and processing

The data of the river channel bathymetry and the ground topography, used to build the terrain model and the cross-sections for the hydraulic model, were collected in the Spring and Summer of 2018-2019. Surveys for the direct measurement of flow resistance were conducted in Autumn and Winter 2020-2021.

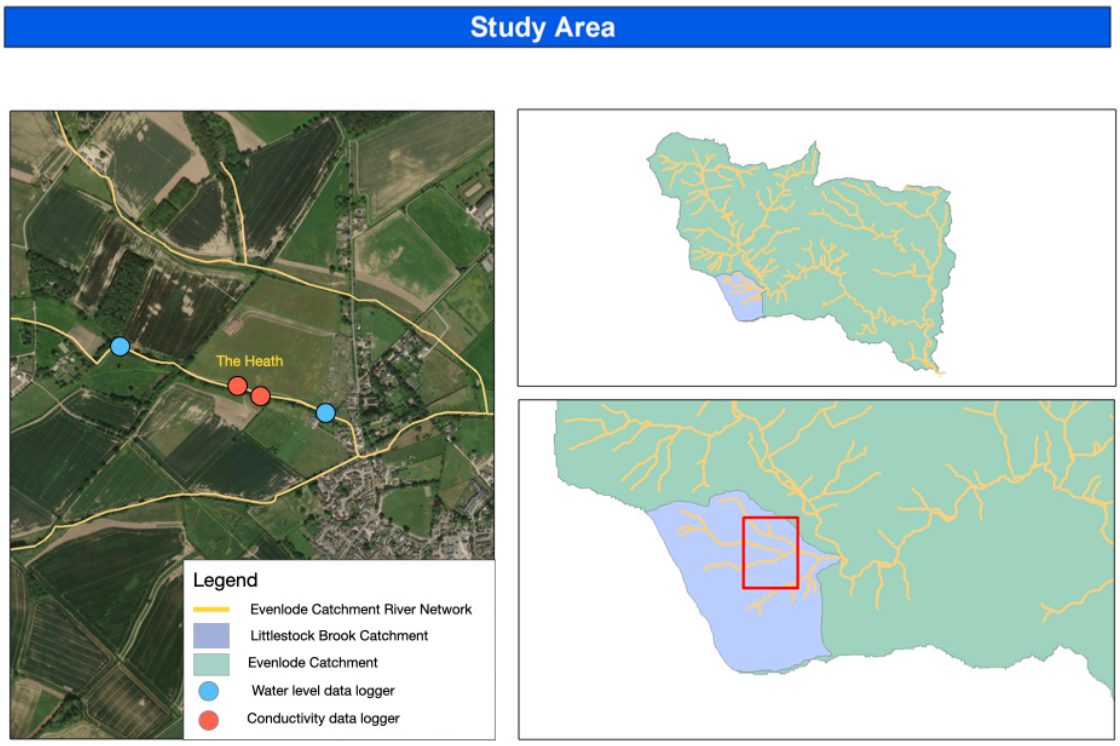


Figure 2.2 The study area selected for this project. *The figure shows the Littlestock Brook catchment and its location within the Evenlode catchment. The drainage area known as The Heath with its river channel is shown on the left. The channel of interest, where 12 high water ELJs were installed is the one in the middle. The four circles represent the approximate locations of the data loggers used to gather water level hydrographs and the conductivity hydrographs used for dilution gauging.*

2.3.2.1 Bathymetry and topography

The channel bathymetry was surveyed in 2018 using a dumpy level and staff at each of the 12 ELJs locations and in additional locations between ELJs across the river reach. For each ELJ's location multiple point were measured: the centre of the channel just upstream and downstream of the ELJ, the extremities of each ELJ and the right and left bank.

The ground topography of the floodplain around the reach was surveyed using a Topcon Real Time Kinematic (RTK) Global Positioning System (GPS) in 2018.

In addition to the topographic and bathymetric surveys, water surface elevations (WSEs) were surveyed at a range of different flows at ELJ locations and at the upstream and downstream end of the reach over a three-year period (2018-2021).

The data collected with the topographic and bathymetric surveys were used to modify the terrain model and cross-sections using the RAS Mapper interface of HEC-RAS.

2.3.2.2 Hydrometric installations

Four hydrometric installations were installed in the 540 m reach chosen for this study were 12 high water ELJs were installed as part of the Littlestock Brook NFM project. Two stilling wells were deployed at the two opposing ends of the reach of interest to contain two Onset HOBO U20L Barometric Water Level Data Loggers (Blue circles in Figure 2.2). Two additional stilling wells were installed at the two ends of a 30 m reach where one of the high water ELJs was installed. These two stilling wells were installed 30 meters apart, with the ELJ in the middle of this study reach, and contained two Onset HOBO U24-001 Fresh Water Conductivity Data Loggers (red circles in Figure 2.2).

The four stilling wells were built by using four Duroscreen pipes with a diameter of 63 mm and 1m length. Before the installation in the field, four holes were drilled on both sides of each pipe to allow the passage of water. The pipes were mounted and fixed on wooden posts that had been previously driven into the bed substrate. After mounting the pipe, the four loggers were deployed into the stilling wells by using a Dyneema cord attached to a lockable pipe plug at the upper end of the pipe. The loggers were attached at the other end of the Dynema cord and constantly immersed in water.

2.3.2.3 Stage Hydrographs

Stage hydrographs for the study channel were collected using an Onset HOBO U20L barometric water level data logger. The calibration of the logged data and its translation to water levels (meters) was realized using water levels measurements collected manually at the start and at the end of the logger deployment and calibrated through the HOBOware software. The stage hydrograph data were collected at 1-minute intervals for roughness measurements while a 15 minutes interval was chosen to record longer time series for HEC-

RAS simulations. The Water Level Loggers were first deployed in the stilling wells on the 27th of November and collected data almost continuously until the end of February. The water level data logged were routineously downloaded and erased from the logger.

2.3.2.4 Roughness measurement in the field

Hydraulic resistance measurements were conducted using a salt dilution method for the reach of interest. The salt dilution method was applied by injecting 5 Liters (L) of water with a Sodium Chloride (NaCl) concentration of 100 g/L at an injection point situated 50 meters upstream of the reach of interest (Figure 2.3). The injection point was selected at a distance sufficiently far upstream to allow full mixing of the salt tracer throughout the water column before reaching the first point of measurement.

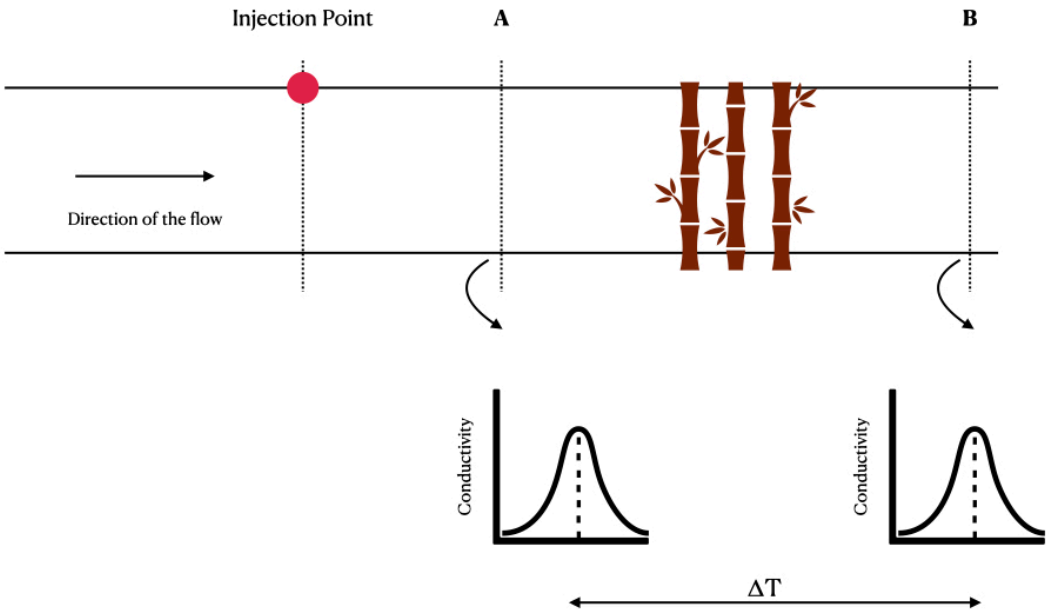


Figure 2.3 Schematic representation of the salt dilution methodology. *The red circle represents the injection point while points A and B represent the locations where conductivity loggers were deployed. Travel time through the reach was measured as the time difference between the centroid of the conductivity curve at point A and B (Adapted from Dixon, 2010).*

Water conductivity was measured with an Onset HOBO U24-001 Fresh Water Conductivity Data Logger at the upstream and downstream end of the reach of interest (A

and B respectively in Figure 2.3). Conductivity measurements were recorded every second thanks to the auto logging feature of the instrument. The data were checked and calibrated, when necessary, against measurement collected using a Hannah HI98129 handheld conductivity meter at the beginning and at the end of each logger deployment. Mean flow velocity was calculated using the travel time of the salt tracer from point A to point B and the distance between the two points (30 m). The mean slope for the channel and the hydraulic radius were measured in the field at point A and B. Manning's *n* parameters were then derived from mean flow velocity, slope and hydraulic radius by using Manning's equation (2.2). Discharge was indirectly calculated by using the average flow velocity through the reach and the cross-sectional area of the downstream end of the 30 m reach of interest. Dilution gauging measurements were all taken in between the 28th November and the 15th December of 2021.

2.3.3 Hydraulic Modelling

2.3.3.1 HEC-RAS model

To model the effects of ELJs on the hydraulics of the channel reach, the Hydrologic Engineering Center's River Analysis System (HEC-RAS) was chosen. HEC-RAS is one of the most used hydraulic models in the world due to its solid computational abilities, fast processing time and free availability through The U.S. Army Corps of Engineers. HEC-RAS was also chosen for the wide range of hydraulic computation available: 1D and 2D solutions of the Saint-Venant equations and 1D-2D hybrid hydraulic models can indeed be built using the HEC-RAS interface.

Specifically, HEC-RAS numerically solves the 1D Saint-Venant equations for the conservation of mass (2.3) and momentum in the x direction (2.4).

$$\frac{\partial H}{\partial t} + \frac{\partial (h v_y)}{\partial y} = 0$$

(2.3)

$$\frac{\partial v_x}{\partial t} + v_x \frac{\partial v_x}{\partial x} + g \frac{\partial H}{\partial x} + g(S_f - S_0) = 0$$

(2.4)

where H is the water surface elevation, h is the hydraulic head, v_x is the velocity in the downstream direction, g is the gravitational acceleration, S_f is the energy slope and S_0 is the channel slope.

2.3.3.2 Representation of ELJs in hydraulic modelling: geometry and roughness

Four different representations of the ELJs were compared in this study: model 1 carries a geometrical representation of the ELJs and the application of a single value of Manning's n through the reach. Model 2 has the identical geometrical representation of ELJs of Model 1 but two different Manning's *n* parameters were applied through the reach, a higher one for the ELJs cross-sections, and a lower one for the cross-sections not directly affected by the ELJs. Models 3 and 4 carry a representation of ELJs with the sole adjustment of Manning's *n* values. In particular, Model 3 has a single value of Manning's *n* applied to the whole reach, Model 4 has two different values of Manning's *n* applied similarly to the ones in Model 2 (Table 2.2). The geometrical representation of ELJs was realised using the Bridge/Culvert Data Editor in HEC-RAS.

The high water ELJs installed at the Heath as part of the Littlestock Brook NFM project are big cylindric wooden pieces of 0.4 m of diameter installed with their ends planted into the channel banks. In order to represent these structures in the hydraulic modelling, quadrangular bridge decks of 0.4 m by 0.4 m were built in the HEC-RAS geometry from the left bank to the right bank using the Bridge/Culvert Data Editor.

The bridge hydraulic performance, as a proxy for ELJ, was modelled using one of the high flow methods of HEC-RAS as suggested in HEC-RAS manual. In particular, as the

bridge was expected to be submerged during the simulations the Energy-based method was chosen for the simulations.

Table 2.2 Summary of the different ELJ representations compared in this study. *NSE and RSR values here reported are the averages of calibration and validation values.*

Model Name	Geometry	Manning's n	Manning's n (ELJ)
Model 1 Geometry + single roughness	12 quadrangular 0.4*0.4 m structures 0.5 m above the river bed	0.042	0.042
Model 2 Geometry + double roughness	12 quadrangular 0.4*0.4 m structures 0.5 m above the river bed	0.042	0.05
Model 3 No geometry + single roughness	N/A	0.042	0.042
Model 4 No geometry + double roughness	N/A	0.035	0.05

2.3.3.3 Model calibration and validation

The calibration and validation of the output hydrograph were realised by using the stage hydrographs and water surface elevations collected with data loggers and during field surveys. The stage hydrograph at the reach outlet was calibrated and validated against the data collected by the water level loggers with a 1-minute time-resolution. The stage hydrograph of a high flow event corresponding to a 1-year return period, measured between the 28th of November and the 3rd of December was used for the calibration. The validation of the stage hydrograph output was performed using a second high flow event observed from the 4th to the 7th of December 2021.

The stage hydrograph data collected for the same two events at the upstream end of the reach were selected as the internal boundary condition for the HEC-RAS model. Normal Depth was selected as the external downstream boundary condition with a slope of 0.0165 as measured in the field. The calibration and validation of the four 1D models carrying different representations of ELJs were performed by adjusting Manning's *n* values across the reach, together with adjustment of the model geometry, initial conditions, and cross-sections to increase model stability.

2.4 Results

The objectives of this chapter were to quantify flow resistance imposed by ELJs by measuring flood wave travel times through a reach and calculating the respective Manning's *n* parameters for different flow conditions, to assess the suitability of different approaches to represent ELJs in hydraulic modelling and the effects of the deployment of multiple artificial ELJs into a reach on the flood peak magnitude and travel time. The first objective was met through the calculation of Manning's *n* based on data collected in field surveys using the salt dilution technique. The second and third objectives were met through 1D Hydraulic Unsteady Flow simulations using HEC-RAS.

2.4.1 Dilution gauging results and comparison with previous studies

Field data from the Littlestock Brook, where Manning's was directly measured for a range of flow magnitudes are shown in Figure 2.4. The data gathered through this study portray an inverse relationship between Manning's roughness parameter and discharge ($\chi^2 = 4.26$, df = 1, $p < 0.001$). The measured Manning's roughness coefficient varied from 0.099 to 0.66, with Manning's *n* values inversely related to the discharge of the channel.

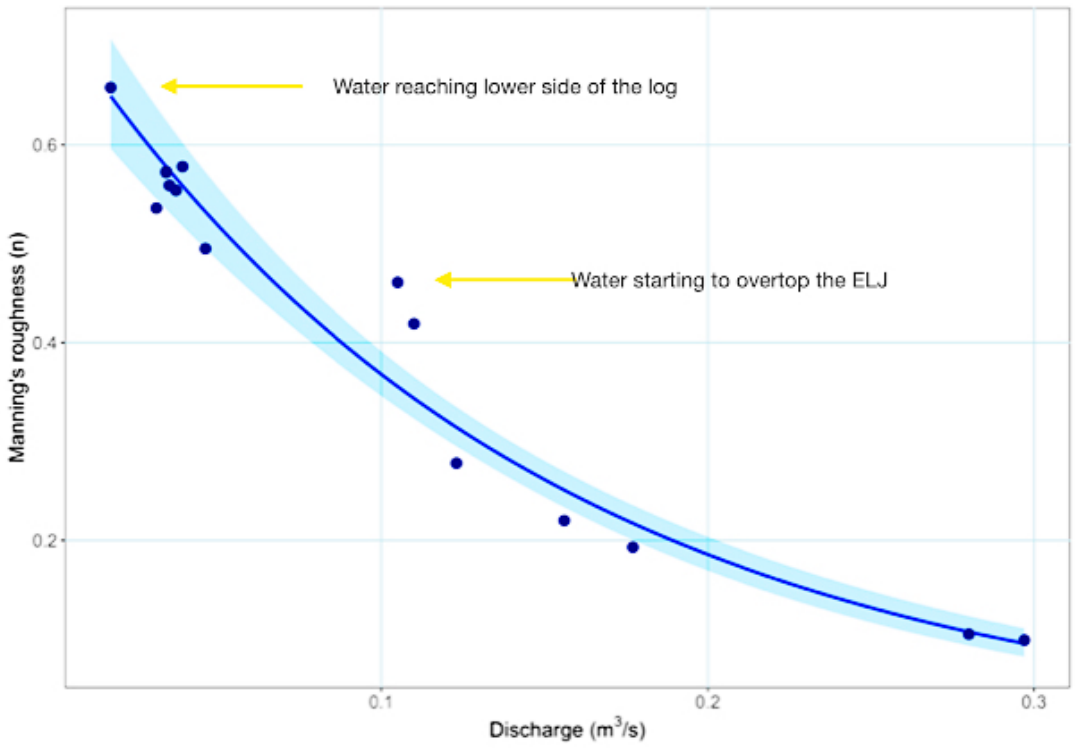


Figure 2.4 Measured values of Manning's roughness coefficient (n) against discharge (m^3/s).

This inverse relationship was also highlighted by Kitts (2010) for a range of different LW structures. The Manning's value observed for the Littlestock Brook are in line with values recorded by Kitts (2010) and Dixon (2013) for High Water/Complete Log Jams in a forested catchment of the New Forest (UK) (reported in Table 2.1 & Figure 2.5).

The data collected in this study with the addition of previously published data were plotted against discharge in order to compare Manning's *n* values across different studies, flows, large wood types and drainage areas (Figure 2.5). As it can be seen in Figure 2.5, this study's measured Manning's *n* values are consistent with the values measured in other studies for similar discharges and large wood types.

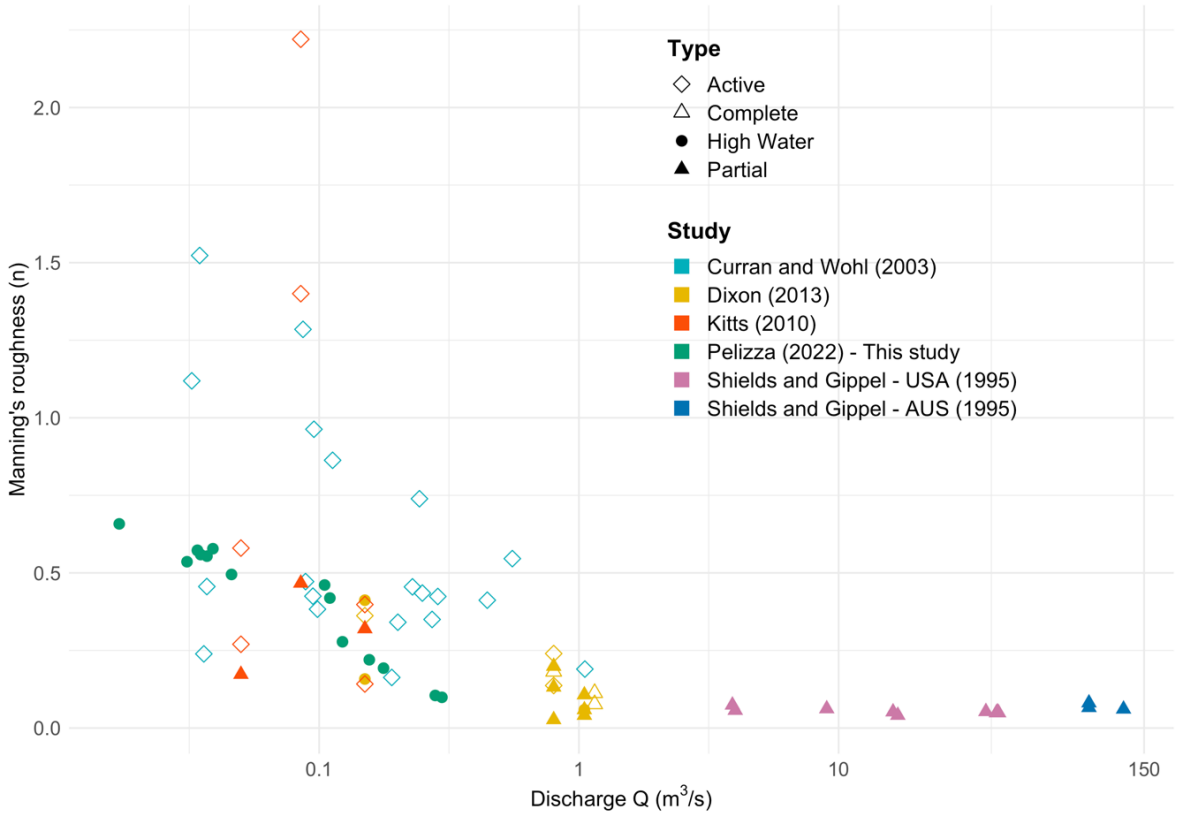


Figure 2.5 Published Manning's n values plotted against Discharge. *The values measured by this study were added to the dataset and are indicated as Pelizza (2022). Discharge data represented in the figure are log-transformed for better visualisation. The different colours represent the different studies while shapes represent the different log jam types*

The relationship between Manning's *n* and discharge (m^3/s) showed in Figure 2.4 holds true also for the collated published values (Figure 2.5). Large wood types are also related to both discharge and Manning's *n* values. Higher discharges correspond to bigger channels were partial log jam types are easier to find. Partial log jams correspond to lower Manning's *n* as a result of opposing to higher discharges and because their influence is smaller as they occupy a relatively smaller part of the channel. This relationship between Manning's *n* values and log jam types is highlighted in Figure 2.6.

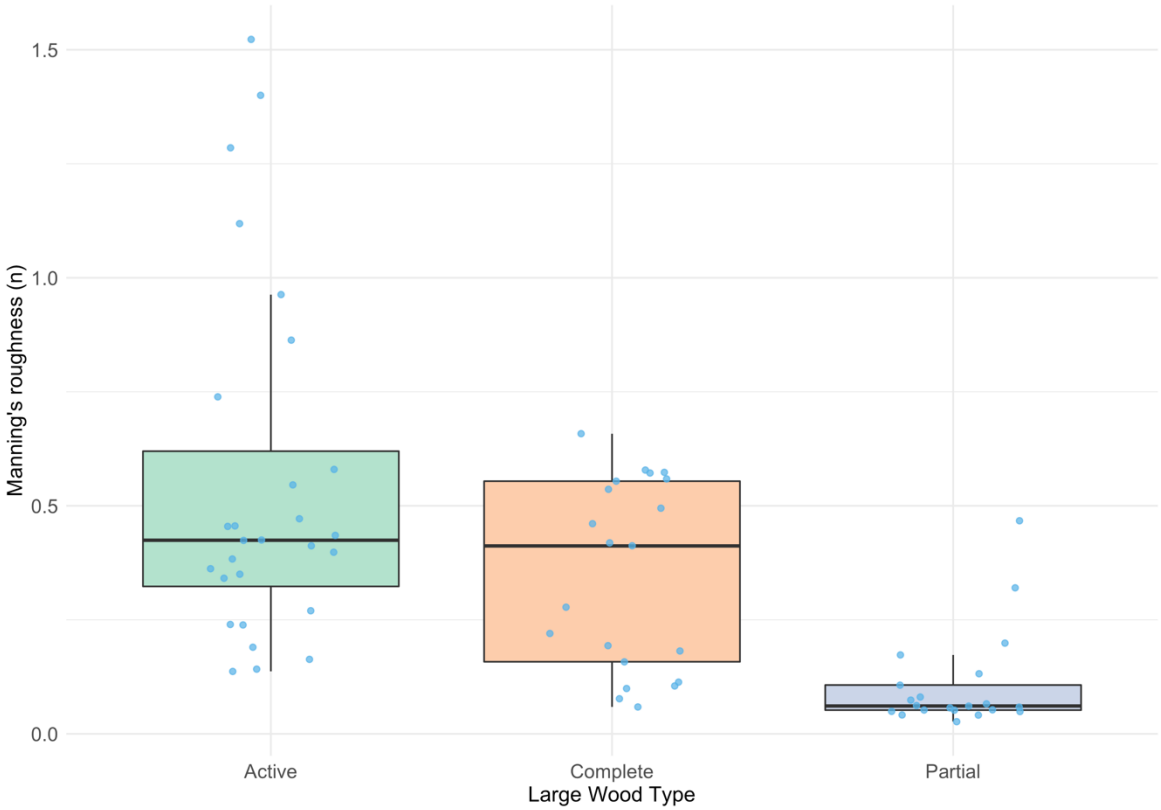


Figure 2.6 Manning's n values for different large wood accumulation types*. The green box is for active dams, orange box for complete dams and the blue box is for partial dams as defined by Gregory et al. (1985). Light blue points represent the individual Manning's n values reported by the studies. Dispersion of the blue points across the x-axis is fictional and used for a better graphical representation. The data from this study, together with the other data measured for high water jams (Dixon, 2013), were collated with complete large wood data for the similarity in their structure and the relatively low numbers of measurement for this type of LW type.*

The differences in recorded Manning's *n* values for high and low flow discharges are shown in Figure 2.7. Manning's *n* values for high flow discharges are lower and less dispersed around the mean value, while Manning's *n* values for low discharges are higher

and with higher dispersion around the mean. shows the same Manning's *n* values collected and reported in Figure 2.5 divided in two categories roughly representing flow conditions at the time of the measurement of Manning's *n* as reported by each study. The data collected in low flow conditions are sensibly higher and with a higher dispersion around the mean. The higher dispersion for low flow data is probably due to the effects of different large wood types and their relationship with channel characteristics and morphology. These differences are probably drowned out at high discharge, hence the lower dispersion for high flow data points.

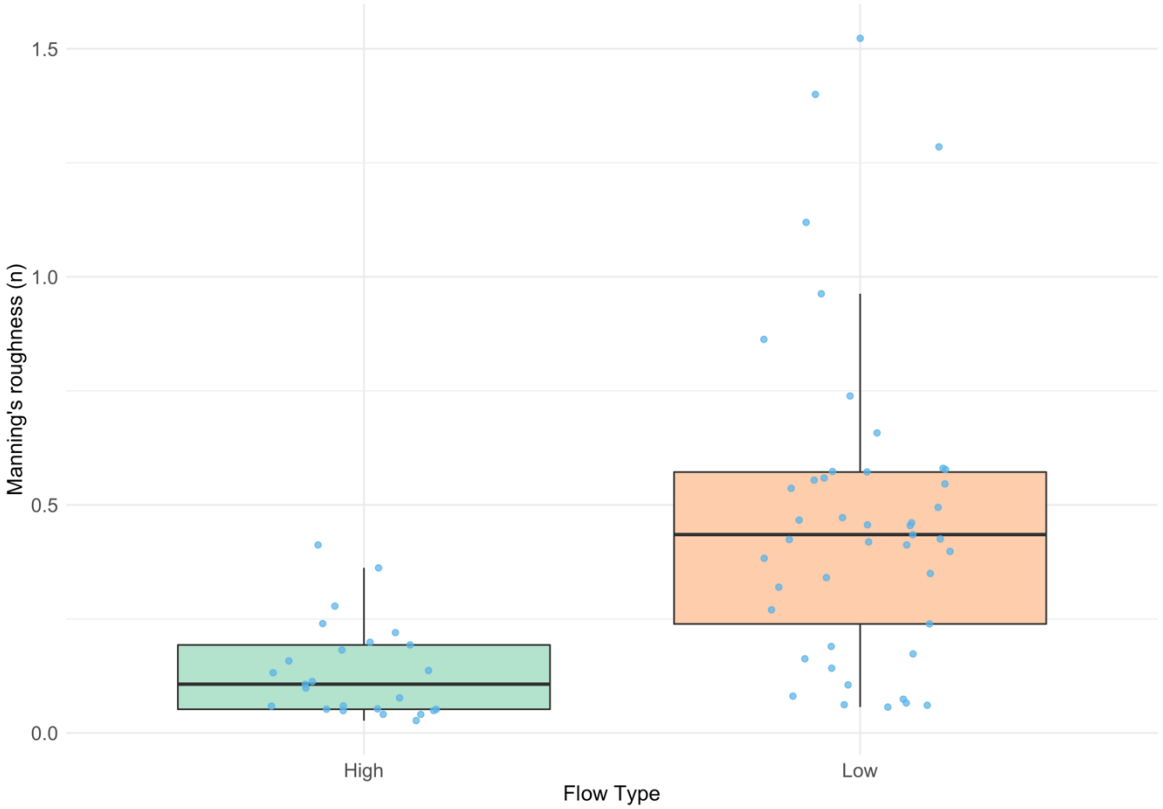


Figure 2.7 Manning's roughness coefficients for low and high flow conditions. *The data represented in this figure are the ones previously shown in Table 2.1 and Figure 2.5.*

2.4.2 Comparison of different ELJs representation

Four different hydraulic models were built to analyse the differences in simulated hydrographs at the reach outlet among the different ELJ representation. The four different hydraulic models and the ELJ representations corresponding to each of these, together with Nash Sutcliffe Efficiency (NSE) and ratio of the root mean square error to the standard deviation (RSR) values for their calibration and validation are shown in Table 2.3.

The calibration and validation of the four different models, based on different ELJs representations, were performed on two different peak flow events: from the 4th of December to the 7th of December (Calibration) and from the 28th November to 3rd December (Validation). All four models showed a better fit for the peak flow used as validation (Table 2.3). However, all the ELJ representations display a very good performance in terms of simulation of the observed hydrographs at the reach outlet with values higher than 0.75 for the NSE parameter and lower than 0.5 for RSR.

Table 2.3 NSE and RSR values for the four different models used in this study, corresponding to four different ELJ representations as a result of a combination of geometry and roughness factors.

Model Name	NSE (Calibration)	RSR (Calibration)	NSE (Validation)	RSR (Validation)
Model 1 Geometry + single roughness	0.76	0.48	0.89	0.32
Model 2 Geometry + double roughness	0.76	0.49	0.78	0.48
Model 3 No geometry + single roughness	0.70	0.55	0.95	0.21
Model 4 No geometry + double roughness	0.80	0.45	0.82	0.41

The data from the validation dataset seem to be better represented by the four different models and this may be due to the fact that all four models struggle to represent the lower flow at the beginning of the simulated period, overestimating it (Figure 2.8). The overestimation of the lower flow is due to the relatively high Manning's *n* used in the simulation that cause water retention in the channel incision, and thus higher water levels at low flows. The modelled hydrograph seems to be flatter than the observed for the peaks in Figure 2.8A, with modelled peaks lower than the observed and low flows higher than the observed. Figure 2.8B shows how all models are able to depict the rising and falling limbs of the peak flows even with low variations in terms of absolute differences in water surface elevations between different days. Considering the different ELJ representation characterising the four different models whose results are shown in Figure 2.8 and Tables 2.2 and 2.3, we can say that there are not big differences in terms of the simulation of the

hydrograph at the outlet of the reach, with very similar values of model performance between the models.

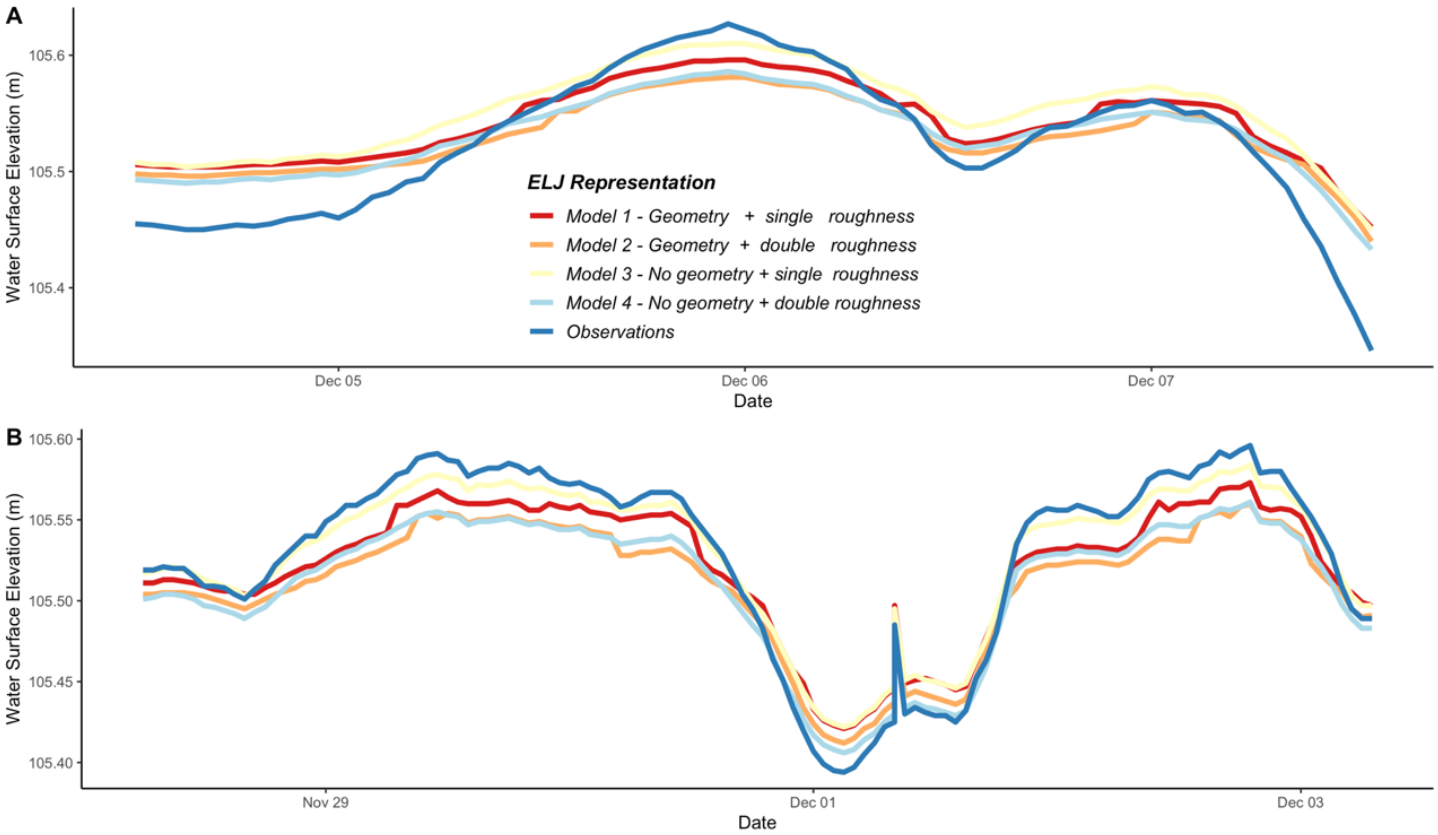


Figure 2.8 Comparison of simulated hydrographs at the reach outlet against observed data. *Figure A. shows the water surface elevations at the reach outlet for 5-7 December, used for calibration. Figure B. shows data from the 28th November to 3rd December, used for validation purposes.*

2.4.2.1 Engineered Log Jams effectiveness in reducing the height of the flood hydrographs and delay its timing

Given the absence of data collected for this specific reach before the implementation of the NFM project and, consequently, the installation of the engineered log jams, it was not possible to perform a before and after comparison of the stage hydrographs for this specific reach. To model the differences between a real life, current scenario with ELJs and a fictional scenario with no ELJs installed in the river, I made use of two different reach representations in the HEC-RAS model, simulating a fictional scenario without the ELJs. This was done in order to compare this fictional scenario, similar to the pre-intervention stage, with Model 1 in order to understand the potential changes introduced by the Engineered Log Jams at the reach outlet. Figure 2.8 shows the hydrographs for the two different scenarios: a scenario with modelled ELJs in the channel (in blue), corresponding to Model 1 and a scenario without ELJs built into the channel geometry (in red).

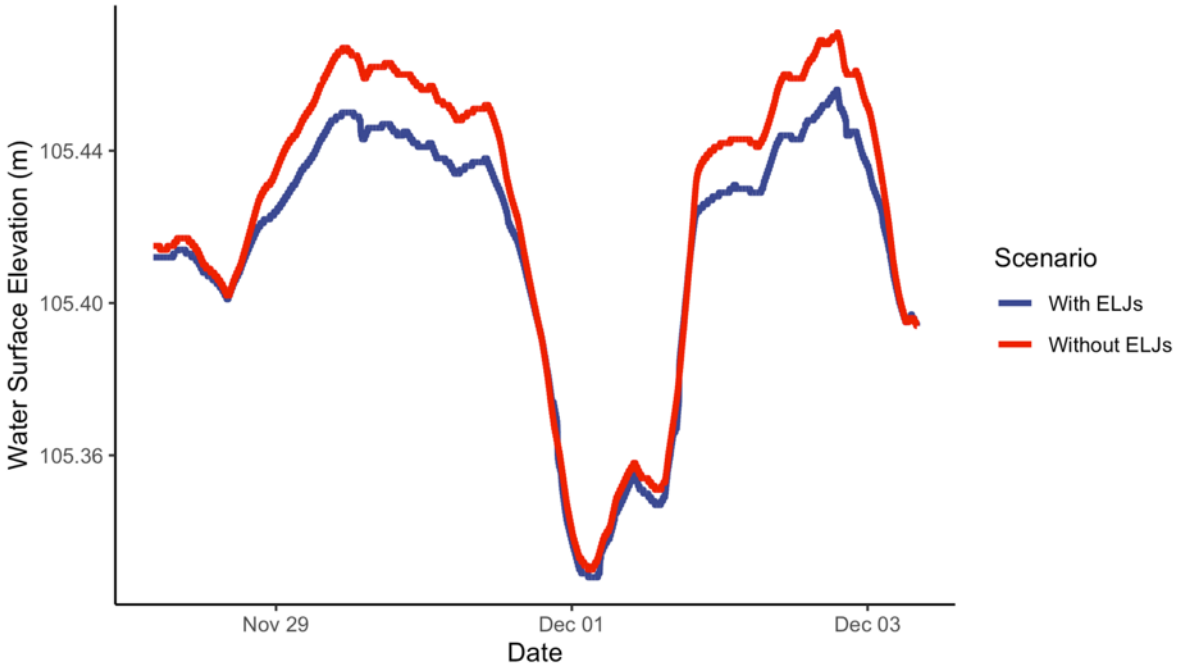


Figure 2.9 Comparison of the simulated hydrograph with a representation of ELJs and a simulated hydrograph representing the fictional scenario of pre-deployment of the ELJs.

The two hydrographs at the reach outlet are consistent one with each other until they reach a certain water surface elevation at which they dissociate. The model without ELJs estimates higher water surface elevations at the reach outlet for the peak discharge, while the model with ELJ produces lower water surface elevations. However, these differences are quite small in percentage and the water surface elevation at the peak flow is 3-5% higher for the scenario of the channel without built in ELJs. This behaviour might be explained by an activation of the high water ELJs once the stage reaches the point of contact with the ELJs thus providing a sudden change in water velocity (further discussed in Chapter 4). The change in water velocity as a result of the obstruction of the ELJs will cause an increase in water retention upstream of ELJs with higher water levels. This could potentially provide a flattening of the peak of the stage hydrograph, as water is released more slowly through the reach, and a slight delay in the timing of the peak.

The relationship between Manning's *n* and the discharge shown in Figure 2.4 and 2.5 can be in part explained by the functioning of high water ELJs (Figure 2.10, Figure 2.11) and large wood accumulations in general.

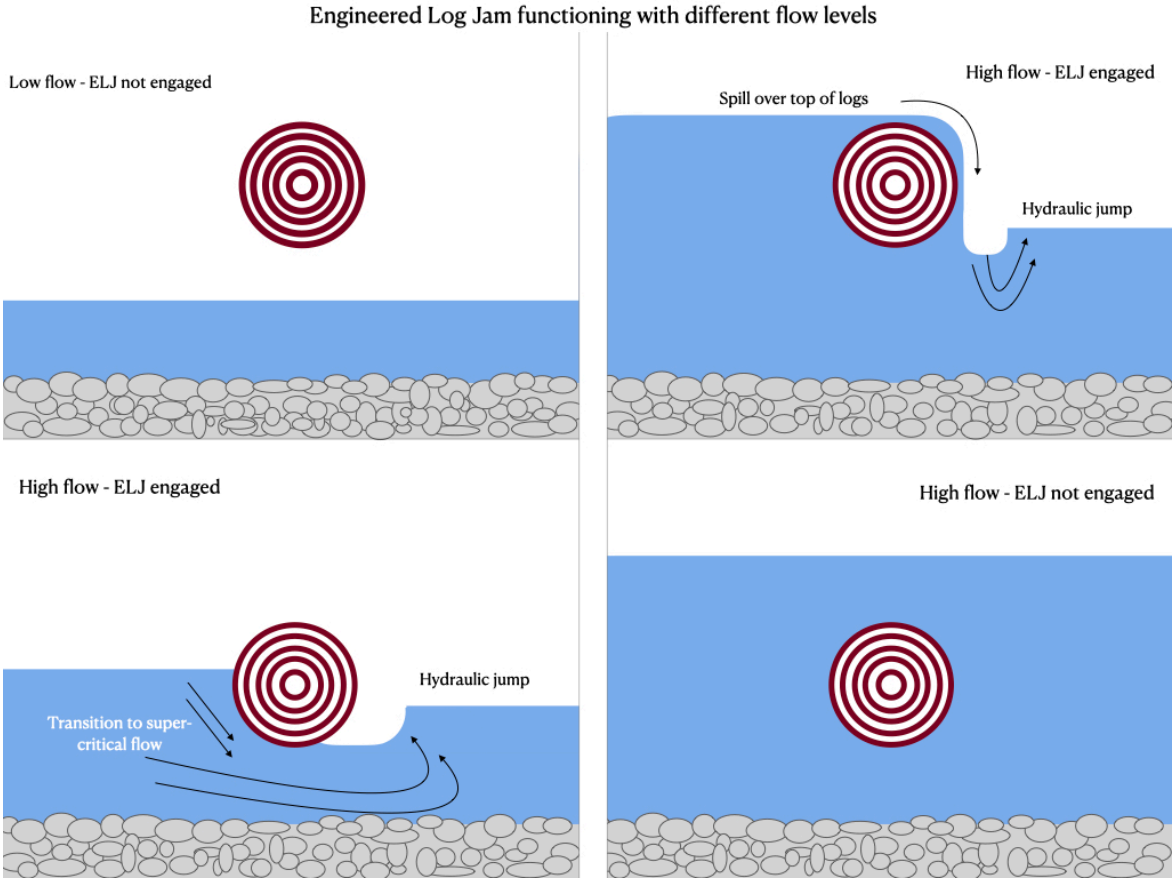


Figure 2.10 Exemplification of the functioning of ELJ at different flow levels. *At low flows and high flows ELJs won't be engaged or only exert little effect on the water flow. At higher stage the water hits the ELJ, thus causing a transition to supercritical flow that produces turbulence and possible hydraulic jumps at the downstream end of the ELJ, thus causing an increase in the resistance to water flow. When the stage reaches the point of the maximum height of the ELJ, water will start to spill over the top of the logs causing turbulence and hydraulic jumps, thus increasing resistance to the water flow.*



Figure 2.11 High water ELJs effects at different stages as observed in the field*. The two photos on top were taken from the top of the left bank side. The photo at the bottom was taken standing in the river. The photo at the bottom was taken during the night, the lighting is due to the headtorch used to take the picture. In the photo at the bottom it can be seen the same ELJ of the two photos at the top completely submerged by the flow of water (the location of the ELJ is in line with the turbulence that can be seen in the middle of the photo).*

2.5 Discussion

The aim of this chapter was to quantify the hydraulic resistance of high water ELJs deployed during a 5-year NFM project within an agricultural catchment by directly calculating Manning's *n* parameter using a salt dilution gauging method, to implement a hydraulic 1D model of one reach with 12 ELJs to compare different ELJs representations in hydraulic modelling and to evaluate the effects of these structures on the peak flow magnitude and timing.

Manning's roughness coefficient measured through this study showed consistency with the values observed by other studies for similar type of ELJs. The data of this study, collected for a variety of different flows, were highly dependent on the discharge (Figure 2.4).

The stage hydrograph of two flood events were used to validate and calibrate four different models with different ELJ representation. All four models showed a good fit and a good overall performance even if all models were struggling in depicting the higher and lower flows. These results are in concordance with Valverde (2013) who showed how both roughness adjustments and explicit geometrical representation were able to correctly depict the water surface elevations close to a partial Engineered Log Jam. In our case, it seems that all four different representations can accurately depict the hydrograph at the reach outlet after the calibration. However, Model 1, characterised by an explicit geometrical representation of the ELJ, required little calibration compared to its roughness only model counterpart (Model 3). The geometrical representation of ELJs as bridge decks using the Bridge/Culvert Editor in HEC-RAS showed a really good performance with little calibration needed. This methodology, used for the first time, can be particularly helpful to model ELJs with a simple structure such as high water ELJs.

The comparison of two modelled scenario, one representing the reach with the ELJs and the other on the reach without the ELJs, provided insights on the possible differences in

the stage hydrograph at the reach outlet. The behaviour of the model and the two hydrographs that dissociate one from the other at the peak flow seem to represent well the functioning of a high water ELJs or of a sequence of them as per this case study. However, the differences in the water surface elevations at the peak flow were negligible (3-5%) and the timing of the peak flow didn't seem to be delayed by the presence of the ELJ, according to the simulation.

2.6 Conclusions

In this study I demonstrated that the Manning's *n* values observed for the study reach are in concordance with those collected for similar reaches in the UK and for similar type for ELJs. Artificial ELJs thus seem to exert similar effects on channel roughness to their native and natural counterparts. As expected, Manning's *n* roughness for reaches characterised by ELJs addition are dependent on the flow discharge and can be explained by their specific structure and by the volume of cross-section occupied by the ELJs with changing flows.

My study also demonstrated that 1D hydraulic modelling with HEC-RAS represents a good solution to represent ELJs effects on output stage hydrographs. Different ELJs representation (geometry, roughness and geometry-roughness hybrid) can perform accurately and represent faithfully the effects of ELJs on the output hydrograph if calibrated and validated with observed data from field monitoring.

As highlighted by Valverde (2013), different ELJs representation perform better for low and high flows. Generally speaking, the hybrid solution of geometrical adjustment and roughness change seems to be a reliable solution in most cases and could also be preferred in the planning phase of NFM projects.

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Chapter 3 : The potential effects of catchment-wide changes in channel roughness on peak flow magnitude and timing in a large rural catchment.

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3.1 Introduction

Floodplains and catchments have been extensively modified by human populations across centuries to exploit water and food sources and to build infrastructure and properties. The progressive land use changes within catchments involved encroachment of urban centres and infrastructures into floodplains, a progressive change in the land cover and extensive modification to river networks in order to use water for private properties, agriculture and farming. Land use changes within the catchment area, and especially those adjacent to river channels, have the potential to modify and affect hydrological processes.

In particular, the removal of the natural vegetation cover for cropland planting and other farming purposes can lead to soil compaction and changes in soil properties that, in turn, can decrease soil infiltration capacity. Urbanisation can also impact the hydrology of a catchment by enhancing runoff because of reduced vegetation cover, reduced soil permeability and reduced water storage in the soil. These transformations can lead to an increase in the volume of surface runoff as well as in the velocity and concentration time of storm runoff.

The transformations of the catchment land have, consequently, over centuries, led to changes in the catchments' response to rainfall patterns. This, together with other climatic and environmental factors, has led to higher peak flows compared to the pre-development stage (Liu et al., 2004), thus increasing flood risk of vulnerable areas.

Traditionally channelization, river straightening, and river clearing have been the most common approaches taken in order to reduce local flood risk by increasing the flood conveyance of the channels. However, increased flood conveyance through these approaches has been shown to increase the risk of flooding downstream, thus potentially moving the issue of flooding downstream more than trying to mitigate the risks from flooding for different parts of the catchment (Gippel, 1995). Channelization, river

straightening and clearing have also caused ecological losses resulting from increased water velocities and reduced habitat diversity.

Due to the issues associated with traditional river management practices and with 'hard' engineering solutions, such as levees, dams and dykes, Natural Flood Management (NFM) techniques are increasingly being considered as an alternative for flood risk mitigation. NFM is often seen as a holistic approach to catchment-based interventions that aim at reducing flood risk while restoring or enhancing the catchment's hydrological characteristics. NFM involves different types of interventions at different scales: from the local scale, with in-channel interventions such as engineered log jams, re-meandering and off-channel storage areas, to the catchment scale with land use change, afforestation measures and new management practices for agricultural land (Forbes et al., 2015; Lane, 2017). However, it is also widely acknowledged that NFM measures are highly varied in spatial availability and economic expenditure, and they have different time spans of action and activation. As an example, afforestation measures within the catchment will require a good spatial availability and a long time span to become effective; on the other hand, small-scale structures, such as ELJs, could be readily effective after installation and require less space and economic expenditure. Various NFM measures should thus be considered to address different issues and starting conditions based on the different expected outcomes.

Natural large wood and artificially re-introduced ELJs are thought to reduce the likelihood of flooding downstream by increasing hydraulic roughness and drag forces, thus reducing flow velocities and storing water temporarily in the channel incision (Keys et al., 2018; Kitts, 2010; Wenzel et al., 2014). The reduction in velocities through reaches and the increase in water storage can delay the movement of peak flows through the catchment's river network following heavy rainfall and consequently reduce the magnitude of flood peak levels and delay its timing. Moreover, when largely present across a catchment, large wood is thought to reduce the likelihood that the peak flows from multiple tributaries will arrive simultaneously downstream, thus decreasing the risk that the capacity of the channels will

be exceeded where higher-value land, assets and infrastructure may be located (Gippel, 1995; Gregory et al., 1985; Lane, 2017a; Reaney et al., 2011; Thomas and Nisbet, 2012).

Engineered Log Jams with varying designs have been recently installed in many small streams across the UK, as part of NFM projects, to reduce downstream flood risks (Addy and Wilkinson, 2019; Barlow, 2014; Kail et al., 2007; Thomas and Nisbet, 2012; Wilkinson et al., 2010). However, empirical evidence of the catchment scale effects of engineered log jams is scarce and confined to small catchments (<15 km^2). For example, a study by Wenzel et al. (2014) used nine artificially deployed ELJs in a controlled reach of approximately 280 meters inserted in a steep headwater catchment in Germany (2 km^2). They observed a peak reduction of 2.2% and a delay in the timing of 166 seconds for a 3.5-year recurrence interval flood. Another study by Kitts (2010) showed a decrease of 21% in flood peak magnitude and a 33% increase in flood peak travel time for flows at around a 2-year recurrence period for low gradient streams in a small catchment in the New Forest. However, the changes were part of a more general stream rehabilitation with wood accumulation and meander restoration and later studies, in the same study area, showed how these effects tended to be drowned out for flows over a 2-year recurrence (Dixon et al., 2016). Therefore, while it is known that the hydraulic effects of large wood and ELJs on flow resistance and floodplain re-connection can result in changes in stream discharge, the UK Evidence Directory (Barlow, 2014) and recent review papers (Dadson et al., 2017; Lane, 2017; Wilkinson et al., 2019) stated that there is still a significant knowledge gap in understanding the effects of LW and ELJs on big flood events and in large catchments (>100 km^2).

The Evenlode is a large (431 km^2) rural catchment that has been extensively modified during the last centuries with an increase in the land used for farming and agriculture. It has experienced severe floods in the previous decades. Indeed, it is one of the four catchments that flow into the city of Oxford that has been affected by flooding in 2007 and 2013. This catchment has been part of a recent Natural Flood Management project which

involved deploying a set of NFM measures, including ELJs. The extent of the catchment, its rural nature, and the deployment of ELJs in a heavily modified environment, make the Evenlode a perfect study area. The choice of a large catchment poses some constraints on the type of model that needs to be used for the simulation and, consequently, on the type of representation of ELJs in such a model. For these reasons, I decided here to use a semi-distributed hydrological model and to perform a roughness sensitivity analysis for the Evenlode catchment as a proxy for widespread ELJ implementation within the catchment. The differences between the effects of a widespread change of roughness across the river network and the widespread installation of ELJs will be discussed in the conclusions.

The aim of my research, discussed in this chapter, was therefore to understand, by applying a semi-distributed hydrological model of the Evenlode catchment, if catchment-wide channel roughness adjustments have the potential to alter the peak flow magnitude and timing at a large catchment outlet.

3.2 Aims and Research Objectives

The overarching aim of this part of my DPhil research was to examine the sensitivity of a large rural catchment, the Evenlode, to catchment-wide changes in channel roughness. Flood peak magnitude and timing were used to monitor the response of the catchment to the changes in this variable. To address this aim and build on the results and field observations of Chapter 2, I undertook the four interlinked objectives:

1) Development of a calibrated and validated hydrological model for the Evenlode catchment that could accurately depict the discharge at the catchment outlet using a semi-distributed hydrological model: the Soil and Water Assessment Tool (SWAT).
2) Simulation of an extensive change in channel roughness within the catchment to understand the potential effects on the peak flow magnitude and its timing for flooding events with different recurrence intervals.

3) Determination of priority areas where roughness should be preferentially manipulated on the basis of the contribution to the peak flow and water yield of different parts of lands within the catchment.
4) Comparison of the different scenarios based on the definition of priority areas.

3.3 Materials and Methods

3.3.1 Study area

The Evenlode catchment lies to the west of Oxford in South-East England and is situated northwest of the Thames River Basin. The Evenlode covers an area of 431 km^2 and flows in a south-easterly direction through the counties of Gloucestershire and Oxfordshire to finally converge with the Thames at Cassington, near Oxford. The area experiences a temperate maritime climate, with an average annual minimum temperature of 5.9 °C and a maximum of 13.4 °C and characterised by a yearly precipitation of 809 mm (Met Office, 2020). The catchment lithology is dominated by the Great Oolite Group, mainly consisting of limestone and mudstone in the eastern part of the catchment. In contrast, the western part is underlain primarily by the Lias Group, characterised by a composition of clays, mudstones and limestones. The former group covers an area of 45.7% of the catchment's surface and is characterised by moderate permeability bedrock, while the latter is characterised by very low permeability bedrock and covers 46.3% of the catchment area. The topography is characterised by an upland area along its western border, characterised by relatively steep slopes, and a lowland, flatter area in the southern part of the catchment.

The Evenlode is a rural catchment whose land use is dominated by arable land and grassland covering respectively 48.5% and 31.4% of the catchment surface (Figure 3.1). The urban centres are limited to small villages and cover 4.9% of the catchment area.

The low permeability bedrock in the western part of the catchment, its relatively steep slopes, and a land use dominated by arable land concur in determining a responsive regime with the potential for flash floods.

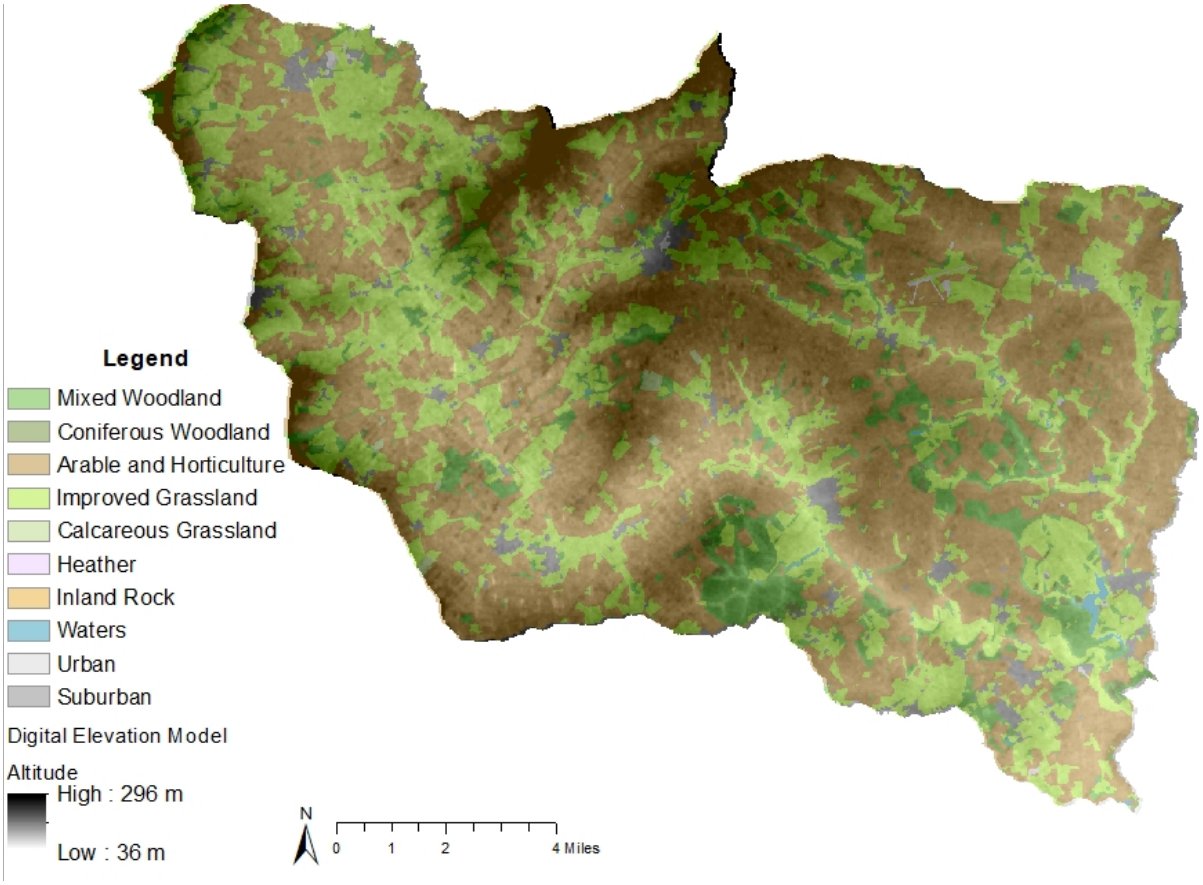


***Figure 3.1 The Evenlode catchment land utilization and altitude map**. The map was obtained from the Land Cover Map 2015 from CEH (Rowland et al., 2017) and the Digital Elevation Model ASTER global v.2 (Meyer et al., 2011). Further details in section 3.3.2.1.*

3.3.2 The Soil and Water Assessment Tool (SWAT)

SWAT is a semi-distributed and physically based model which operates at a continuous time-step (Arnold et al., 2012). The model is designed as a flexible infrastructure to simulate the effects of changes in management practices and land use on the water, sediment, nutrient and pollutant fluxes within a catchment area.

To develop a hydrological model for my study catchment, the Evenlode, the ArcGIS-ArcView extension and interface for SWAT was used. In particular, ArcSWAT version 2012.10.0.14 was applied in this study (Texas A&M University, 2015). The methodology outlined in Winchell et al. (2013) and Arnold et al. (2012) was applied to build the model of the study catchment.

SWAT uses a spatial discretisation of the catchment in multiple sub-basins, which are further split into Hydrological Response Units (HRUs). HRUs are lumped units that consists of homogeneous land use, slope and soil characteristics (Arnold et al., 2012).

The hydrological cycle is simulated by SWAT based on the equation of water balance:

$$SW_t = SW_0 + \sum_{i=1}^{t}(R_{day} + Q_{surf} - E_a - W_{seep} - Q_{gw})$$

(3.1)

2here SW_t is the soil water content after t days (mm H_2O), SW_0 is the soil water content at the beginning of the simulation, R_{day} is the amount of precipitation of the ith day, Q_{surf} is the amount of surface runoff of the ith day, E_a is the amount of evapotranspiration (mm H2O), W_{seep} is the amount of water lost as seepage, Q_{gw} is the amount of water lost as groundwater recharge.

SWAT physical processes are divided into two distinct phases. First, the land or terrestrial phase of SWAT considers the fluxes of water, sediment, nutrients and pesticides of each sub-basin. Secondly, the in-stream or routing phase of SWAT simulates the movement of water, sediment, nutrients and pesticides through the channel network to the channel and catchment outlet (Arnold et al., 2012). In addition to keeping track of mass flow in the channel, the water phase models the transformation of chemicals in the stream and streambed (Neitsch et al., 2011).

In order to simulate the water, sediment, nutrients and pesticides fluxes through the study catchment, SWAT requires inputs from geographical and meteorological data. The geographical datasets include a Digital Elevation Model (DEM), a soil map, and a land-cover map. Required meteorological inputs include daily observations of precipitation, solar radiation, mean wind speed, maximum and minimum temperature, and mean relative humidity. The details of the datasets collated to input into the SWAT model are outlined below.

3.3.2.1 Geographical Datasets

The Digital Elevation Model (DEM) applied within this study has a resolution of 30 m and was obtained from ASTER global digital elevation model version 2 (Meyer et al., 2011). To generate the final DEM map used in this study, the tiles for the study area were merged into a mosaic and then cut to the catchment shape in ArcGIS.

Land cover within the study area was identified following the UK Centre for Hydrology and Ecology (CEH) Land Cover Map 2015 (LCM2015) raster dataset with a resolution of 25m (Rowland et al., 2017). The Land Cover Map divides the land cover into 21 distinct classes based on the Broad Habitats defined within the UK Biodiversity Action Plan.

Soil types for the Evenlode catchment were obtained from the National Soil Map (NATMAP) vector dataset (Cranfield University). A simplified map of the composition of soil associations for the Evenlode, obtained from NATMAP, is shown in Figure 3.2. The NATMAP vector is characterised by 300 distinct Soil Associations, each composed of multiple Soil Series and with distinctive hydraulic properties. In order to simulate the fluxes of water and nutrients through the catchment, SWAT requires for each layer of each soil type a measure of the depth of the soil layer, moist bulk density, available water capacity, saturated hydraulic conductivity, sand, silt, clay and organic carbon content, the maximum rooting depth within the soil profile, the fraction of porosity from which anions are excluded, the moist albedo of the soil surface and erodibility (Arnold et al., 2012). These variables for each soil type and layer were calculated from the NATMAP Horizon Fundamentals and Horizon Hydraulics datasets using R Software (Team R, 2013). The resulting dataset was then used as input for SWAT's *usersoil* table.

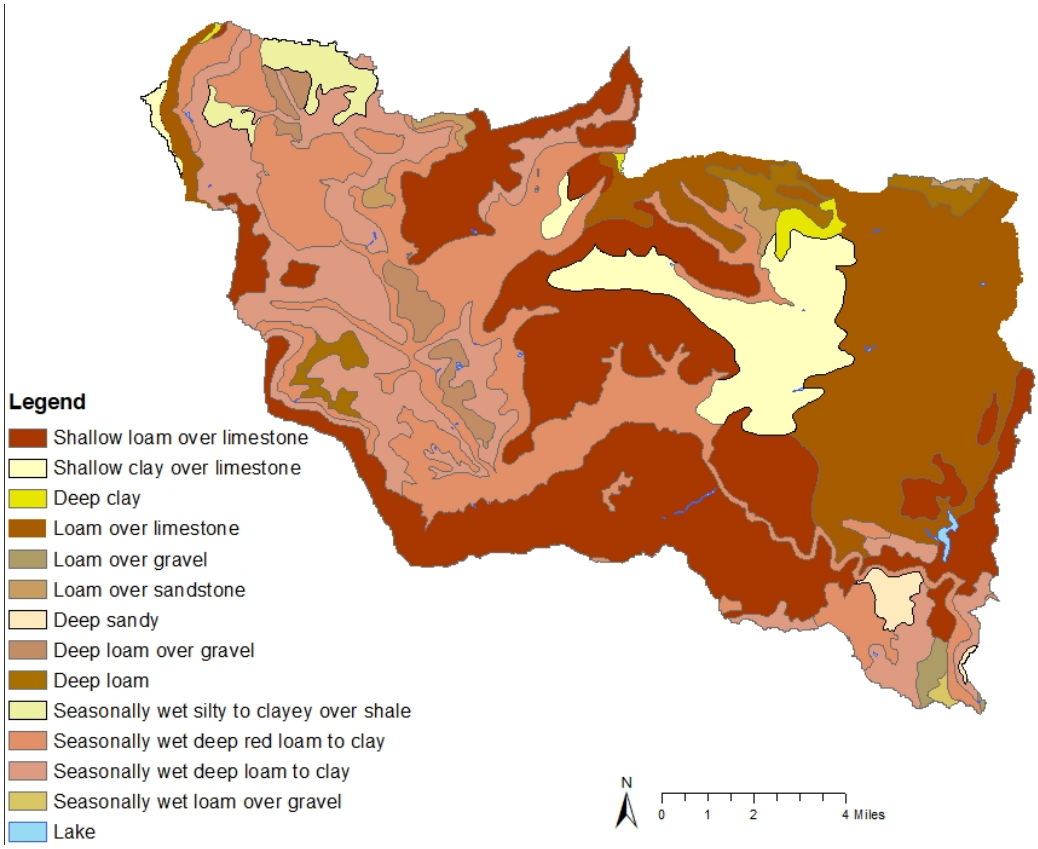


Figure 3.2 Map of soil types for the Evenlode catchment as defined by NATMAP dataset.

3.3.2.2 Meteorological data

The daily meteorological inputs required by SWAT are precipitation, mean wind speed, maximum and minimum temperature, solar radiation and mean relative humidity (Arnold et al., 2012). These data were obtained from the National Aeronautics and Space Administration (NASA) Agro-Climatology datasets for dates between 1st January 1990 and 31st December 2020. NASA Agro-Climatic data are available in the open access archive of the Prediction of Worldwide Energy Resources (POWER) project (http://power.larc.nasa.gov). This archive contains long-term estimates of meteorological variables derived from NASA's Global Model and Assimilation Office (GMAO) and the Goddard Earth Observing System assimilation model version 4 (GEOS-4) and version 5 (GEOS-5), in addition to the surface solar energy fluxes obtained using satellite observations. The data, available since 1983, are characterised by 3-hourly, daily, and monthly time steps. The resolution for the radiation datasets is 1.0° latitude by 1.0° longitude while ½° latitude by ⅝° longitude for the meteorological data sets.

For this study, the daily estimates for six data points across the study area were used as meteorological inputs for SWAT. NASA Agro-Climatology was chosen for the wide data availability and after the comparison with other freely available datasets during the model calibration and validation processes.

3.3.2.3 Channel roughness manipulation

The modelling approach used in this work involves quantifying the effects of changes in channel roughness on the discharge at the catchment outlet for flooding events of different magnitude. As explained above, SWAT in-stream phase simulates the movement of water, sediment, and nutrients through the channel network. The roughness of the channels within each SWAT sub-basin can be controlled via two parameters: CH_N (2) and CH_N (1). The former is the Manning's *n* parameter for the main channel of each sub-basin, whose value is required by the model and specified in one of the input files of SWAT

(*.rte*). The second parameter, CH_N (1), represents Manning's *n* value for tributary channels within the sub-basins and must be specified in a second input file (*.sub)* that contains all the sub-basin specific parameters.

Manning's roughness coefficient was used to test the catchment response to changes of in-channel roughness, after an initial calibration of the SWAT model for the Evenlode catchment. The Manning's *n* values suggested in the SWAT input manual (Arnold et al., 2011), following Chow (1959), are displayed in Table 3.1.

Table 3.1 Values of Manning's roughness coefficient for channel flow (Chow, 1959). *Adapted from (Arnold et al., 2011)*

Channel type	Range of values	Median value
Excavated or dredged		
Earth, straight, uniform	0.016-0.033	0.025
Earth, winding and sluggish	0.023-0.050	0.035
Not maintained, weeds and brush	0.040-0.140	0.075
Natural streams		
Few trees, stone or brush	0.025-0.065	0.050
Heavy timber and brush	0.050-0.150	0.100

To study the sensitivity of the catchment to in channel roughness variation, the values of the two SWAT parameters describing Manning's *n* values for the main channel of each sub-basin and its tributaries, CH_N (2) and CH_N (1), were modified with a set of values obtained from Table 2.1 and the direct measurements of Manning's *n* in a representative reach of the catchment whose results were shown and discussed in Chapter 2. The values used in the sensitivity analysis were comprised between 0.014 and 0.6 with progressive increases of Manning's *n* values to quantify the sensitivity of the modelling approach to the modifications in this parameter.

In most figures of the results section, three values of Manning's *n* were chosen for the graphical representation. These values were selected as they were deemed representative

of the range of Manning's *n* values found in the literature and observed in the field for high water ELJs. The overview and discussion of Manning's *n* values directly measured in the field during this study and the comparison with previous publications were aforementioned in Chapter 2.

3.3.3 Model calibration and validation

The simulated daily discharge (m^3/s) at the catchment outlet was calibrated and validated against the observations obtained from the UK National River Flow Archive (https://nrfa.ceh.ac.uk/data/station/info/39034) for station 39034 – Evenlode at Cassington Mill in order to produce a reliable hydrological model of the Evenlode catchment. The calibration was performed on a three-year time window (2009-2011), while the model validation was performed over the following three years (2011-2013). The performance of the model was evaluated at every step of the calibration process with an objective evaluation based on three quantitative statistics: Nash-Sutcliffe Efficiency (NSE), Percent Bias (PBIAS), Ratio of the Root Mean Square Error to the standard deviation of the measured data (RSR).

3.3.3.1 Determining the goodness of fit using the Nash-Sutcliffe Efficiency coefficient (NSE)

The Nash-Sutcliffe Efficiency coefficient (NSE) is a normalised statistic which describes the "goodness of fit" of simulated data by determining the relative magnitude of the residual variance compared to the measured data variance (Nash and Sutcliffe, 1970). NSE is computed through equation 3.2:

$$NSE = 1 - \frac{\sum_{i=1}^{n}(Y_i^{obs} - Y_i^{sim})^2}{\sum_{i=1}^{n}(Y_i^{obs} - \overline{Y^{obs}})^2} \quad (3.2)$$

where n is the total number of observations, Y_i^{obs} is the value for the observed variable at the ith time-step, Y_i^{sim} is the value simulated by the model for the variable of interest at the ith time-step, and $\overline{Y^{obs}}$ is the mean value of the observed data for the time interval considered.

NSE can vary between -∞ and 1, where 1 describes a perfect fit between the model predictions and observations. An NSE value between 0 and 1 represents an acceptable model performance, whilst values lower than 0 indicate that the mean value of the observations is a better predictor of the variable than the actual simulated data point, thus indicating unsatisfactory model performances. NSE is the most used parameter for the goodness of fit, thus providing extensive information on reported values for comparison across studies and, as such, has been recommended by Servat and Dezetter (1991).

3.3.3.2 Determining the tendency of simulated data with Percent Bias

Percent Bias (PBIAS) measures the average tendency of simulated data to under- or overestimate their observed counterparts. The optimal value of PBIAS is zero, meaning a perfect agreement between simulated data and observations. Negative values represent a general tendency of the model to overestimate their observed counterparts, while positive values indicate a model underestimation bias.

PBIAS is calculated with Equation 3.3:

$$PBIAS = \frac{\sum_{i=1}^{n}(Y_i^{obs} - Y_i^{sim}) * 100}{\sum_{i=1}^{n}(Y_i^{obs})} \quad (3.3)$$

where n is the total number of observations, Y_i^{obs} is the observed value of the variable of interest at the ith time-step, and Y_i^{sim} is the simulated value for the variable of interest at the ith time-step. Acceptable model performance ratings for PBIAS are shown in Table 3.3.

Table 3.2 Reported performance ratings for PBIAS for SWAT model. *Adapted from Van Liew et al. (2007) and Singh et al. (2004), and Moriasi et al. (2007).*

Performance Rating	PBIAS value	Reference
Satisfactory	PBIAS < 15%	Santhi et al. (2001)
Satisfactory	PBIAS < 20%	Bracmort et al. (2006)
Very good	PBIAS < 10%	Van Liew et al. (2007)
Good	10% < PBIAS < 15%	Van Liew et al. (2007)
Satisfactory	15% < PBIAS < 25%	Van Liew et al. (2007)
Unsatisfactory	PBIAS > 25%	Van Liew et al. (2007)

3.3.3.3 Determining the mean square error with the RSR coefficient

The ratio of the root mean square error to the standard deviation of the measured data (RSR) was derived from the root means square error statistics (RMSE). RMSE is one of the most commonly used error indices in statistics (Chu and Shirmohammadi, 2004; Singh et al., 2005). The RSR was developed to include a measure of the standard deviation of observed data into the formulation of the RMSE, thus incorporating a scaling/normalisation factor for comparison across studies.

RSR varies from an optimal value of zero to large positive values. A value of zero corresponds to zero RMSE, or residual variation, and a perfect model fit.

RSR is computed as the ratio of the RMSE to the standard deviation (STDEV) of observed data as exemplified by Equation 3.4 (Moriasi et al., 2007):

$$RSR = \frac{RMSE}{STDEV_{obs}} = \frac{\sqrt{\sum_{i=1}^{n}(Y_i^{obs} - Y_i^{sim})^2}}{\sqrt{\sum_{i=1}^{n}(Y_i^{obs} - \overline{Y^{obs}})^2}} \quad (3.4)$$

where n is the total number of observations, Y_i^{obs} is the observed value for the variable of interest at the ith time-step, Y_i^{sim} is the simulated value for the variable of interest at the ith time-step and $\overline{Y^{obs}}$ is the mean of the observed data of the variable of interest.

3.3.3.4 Performance and goodness of fit indicators

Moriasi (2007) reviewed the values reported in other published studies for the most used quantitative statistics to rate model performance of SWAT and other hydrological models. Moriasi et al. (2007) also provided a qualitative marking of model performance based on NSE, RSR and PBIAS (Table 3.4).

Table 3.3 Summary of general performance ratings for recommended statistics as suggested by Moriasi et al (2007) on the basis of data reported in Table 3. 2 and Table 3.3 for NSE and PBIAS respectively. *Adapted from Moriasi et al. (2007).*

Performance Rating	**NSE**	**RSR**	**PBIAS (%)**
Very good	$0.75 \leq NSE \geq 1.00$	$0.00 \leq RSR \geq 0.50$	$PBIAS \leq \pm 10$
Good	$0.65 \leq NSE \geq 0.75$	$0.50 \leq RSR \geq 0.60$	$\pm 10 \leq PBIAS \geq \pm 15$
Satisfactory	$0.50 \leq NSE \geq 0.65$	$0.60 \leq RSR \geq 0.70$	$\pm 15 \leq PBIAS \geq \pm 25$
Unsatisfactory	$NSE \leq 0.50$	$RSR > 0.70$	$PBIAS \geq \pm 25$

3.3.4 Priority areas for changes in channel roughness

In this chapter, I evaluate the sensitivity of a large rural catchment to marked changes in channel roughness that can be manipulated through various management interventions. Many NFM measures such as Engineered Log Jams, river restoration and re-meandering have the potential to reduce water velocities in the channel by increasing the in-channel roughness, the ponding of the flow and the storage in the channel incision.

. The aim of this chapter is to understand how changes in this parameter affect catchment response. To fully understand catchment sensitivity to roughness is thus important to apply a wide range of roughness values and to define priority areas that might represent an optimal benefit/cost ratio in terms of catchment response to roughness changes.

To define the priority areas for changes in roughness, we should consequently seek to prioritise those areas with fast runoff travel times and higher water yield to partially counteract these factors with slower water transport through the channel. However, it should be noted that the reduction in the water transport through a catchment could cause the opposite effect with the synchronisation of its peak flow with the peak flow coming from one or more other tributaries.

Runoff travel times through the catchment were derived from a digital elevation model, from which flow direction and runoff accumulation were calculated for each grid cell. The DEM characteristics are described in Section 3.3.2.1. SAGA GIS 7.9 was used to fill the DEM sinks and calculate the slope and catchment area. The tool Isochrone Variable Speed of the Hydrology library of SAGA GIS was used to produce a raster of travel times (hours, h) for the catchment. The isochrones variable speed module of SAGA calculates travel times from each grid cell to the catchment outlet based on Manning's n values as an input raster, using a kinematic wave function. The Manning's n raster values were derived by using the landcover map for the Evenlode catchment (CEH, 2015) and the values from Chow (1959) (Table 3.5).

Table 3.4 Manning's n values used in SAGA GIS for the isochrones module to describe the travel times through the catchment relatively to the land cover for the Evenlode catchment as defined by CEH (2015).

Landcover type	Manning's n (surface roughness)
Arable Land	0.035
Bareland	0.025
Forest	0.12
Grassland	0.035
Shrubland	0.1
Water Courses	0.02
Water Bodies	0.02
Urban and semi-urban	0.013

The output raster obtained from the travel times analysis was then used to calculate the contribution of each grid cell to the peak flow with a methodology described in O'Connell (2007) as a Source-Pathway-Receptor model. A histogram of travel times from each cell to the basin outlet was generated as a synthetic hydrograph of flow through the catchment shown in Figure 3.3. From the histogram a relative index of contribution to the peak flow was computed across the catchment following Wilebore and Benz (2017):

$$T_{rel} = 1 - \left| \frac{T_{out}[i] - T_{peak}}{T_{max}} \right| \tag{3.5}$$

where T_{rel} is the relative contribution of a grid cell to the peak flow at the catchment outlet, T_{out} is the travel time from location i to the outlet, T_{peak} is the time at which the peak flow through the catchment occurs and T_{max} is the value of the maximum travel time from anywhere in the catchment to the outlet (Figure 3.3).

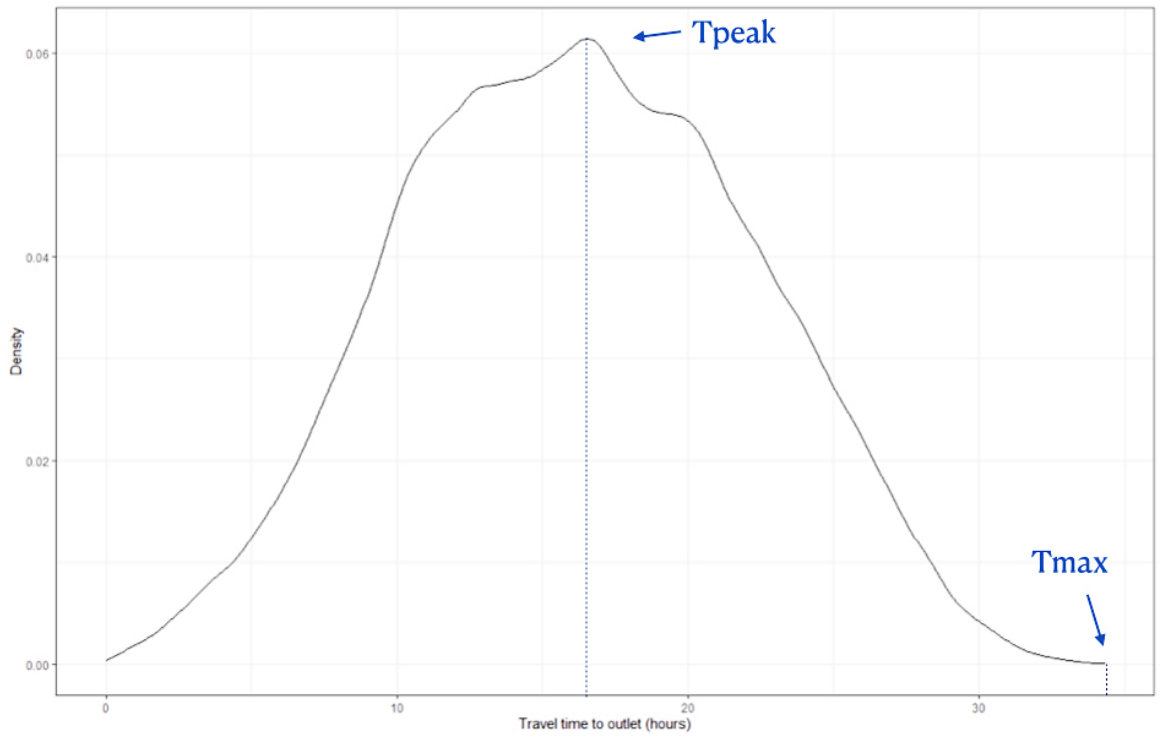


Figure 3.3 Density of the catchment grid cells for specific travel times (hours). *Tmax and Tpeak represent, respectively, the maximum travel time through the catchment and the travel time at the peak flow.*

The water yield of different sub-basins was gathered by SWAT *output.sub* table that lists the set of output variables at the sub-basin level of SWAT. In particular, the WYLD variable represents the water yield (mm H_2O) from each sub-basin (Arnold et al., 2011). WYLD is the net amount of water leaving the sub-basin and contributing to the streamflow during the time-step (e.g. daily, in this study). The water yield values for the 36 sub-basins were ranked and used as an additional measure to evaluate the sub-basins contribution to the peak flow.

The priority areas for roughness changes within the catchment were defined by ranking the catchment sub-basins for their average relative contribution to the peak flow and their water yields. After ranking the sub-basins, two different scenarios were applied: a scenario corresponding to modifying the in-channel roughness for the river networks of 5 sub-basins in the western part of the catchment (Uplands Scenario); the second scenario

corresponded to changing roughness parameters in 4 sub-basins in the eastern part of the catchment characterised by lower elevations and flatter slopes (Lowlands Scenario).

3.4 Results

3.4.1 Model Calibration and Validation

The results of the calibration and validation processes are shown in Figure 3.4, while the calculated values for the coefficients NSE, PBIAS and RSR, that evaluate model performance and goodness of fit, are shown in Table 3.6.

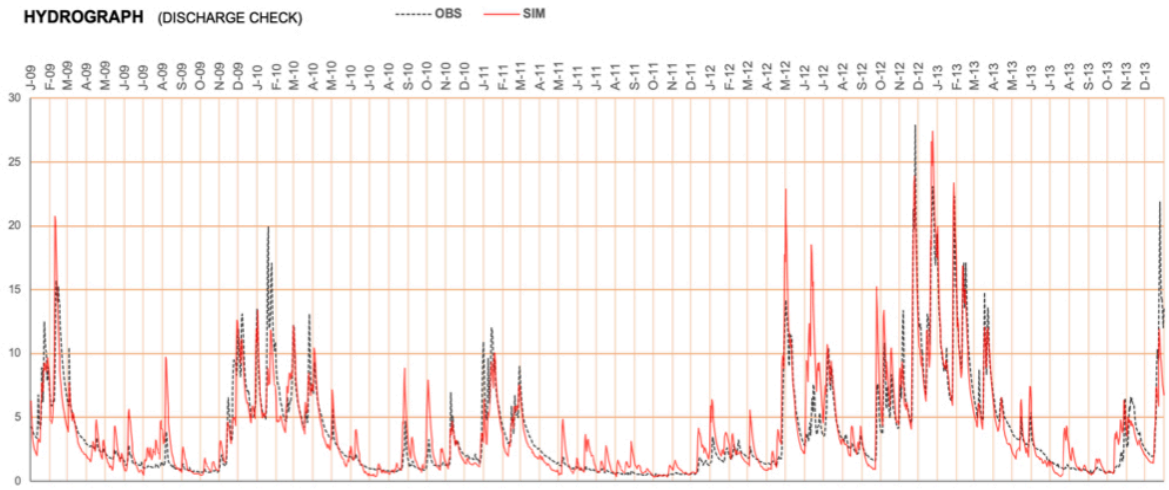


Figure 3.4 The discharge (m^3/s) at the Evenlode catchment outlet for four years (2009-2013). *Observations (black dotted line) are here compared to SWAT simulated data (red line). Values for the goodness of fit of the simulated data shown in this figure can be found in Table 3.6.*

The model performance is very good, as defined by Moriasi et al. (2007) for both the calibration and validation periods (Table 3.6). In particular, the most extreme events, corresponding to higher peaks, are well captured by the model.

Table 3.5 Statistics to evaluate model performance and goodness of fit.

Parameter	Calibration	Validation	Whole period
NSE	0.77	0.77	0.77
R^2	0.77	0.81	0.80
RSR	0.48	0.48	0.48
PBIAS	0.7%	7.4%	4.2%

3.4.2 Roughness sensitivity analysis

The sensitivity analysis performed here aimed to evaluate the effects of a conceptual river rehabilitation through NFM measures, particularly ELJs implementation, for flooding reduction purposes in the Evenlode catchment.

The hydrographs for a 25-year flood event and a 1-year flood event are shown respectively in Figures 3.5 and 3.6. The different colours in the two figures correspond to identical SWAT simulations but with a different parameterisation for Manning's roughness coefficient, whose values are shown in the legend. As shown by the figures, increasing Manning's *n* values correspond to a "flatter" peak flow and to a shift forward of the flood wave. These changes in the peak flow height and its timing are conserved for flood events of different magnitude and are remarkable also for a modest increase in the channel roughness (e.g. Manning's n = 0.050). The timing of the peak flow for flood events corresponding to return periods between 1 and 25 years, simulated in the period 2009-2013, was showing a shift forward of approximately 20 hours for values of Manning's *n* between 0.050-0.10. Moreover, a shift forward of the peak flow of approximately 35 hours was observed for values of Manning's *n* higher than 0.15.

The absolute change in peak flow discharge at the outlet of the Evenlode catchment is shown in Figures 3.7 and 3.8. The absolute change in discharge was obtained by synchronising the different hydrographs at the time of the peak flow (point 0 on the x-axis).

Figure 3.7 shows the absolute change in discharge (m^3/s) for the 25-year recurrence flood event whose hydrograph is shown in Figure 3.5. As it can be observed in both Figures 3.7 and 3.8 and as evident from their hydrographs shown in Figures 3.5 and 3.6, respectively, the peak flow height is markedly reduced for increasing values of Manning's *n*.

The relative reduction in the height of the peaks expressed as a percentage for a 25-year flood event and for a 1-year flood event is shown in Figures 3.9 and 3.10. As shown in Figure 3.9 for a 25-year flood, we can see a reduction of around 2.5% in the peak flow discharge for values of Manning's *n* of 0.050, representing a small variation in channel roughness from the basal value of the calibrated model (0.014). Higher values of Manning's *n* (0.1 and 0.15) reach more remarkable peak reductions with a 9-10% decrease in discharge at the outlet. More extreme values of Manning's *n* exert a higher influence on the peak flow with a reduction between 12 and 17.5% for Manning's *n* between 0.30 and 0.60.

Figure 3.10 shows similar results for the peak discharge reduction for a 1-year flood event. The lowest variation in Manning's *n* corresponds to a small percentage reduction in discharge at the Evenlode outlet (2%). Higher values for channel roughness show more marked differences in discharge, with a reduction of 5 and 7.5% for Manning's *n* values of 0.10 and 0.15 respectively. The increase in roughness compared to the basal model produces an expected pattern: the days preceding the peak flow are showing an increase in the discharge for higher values of Manning's *n* and the same holds true for the days following the peak flow.

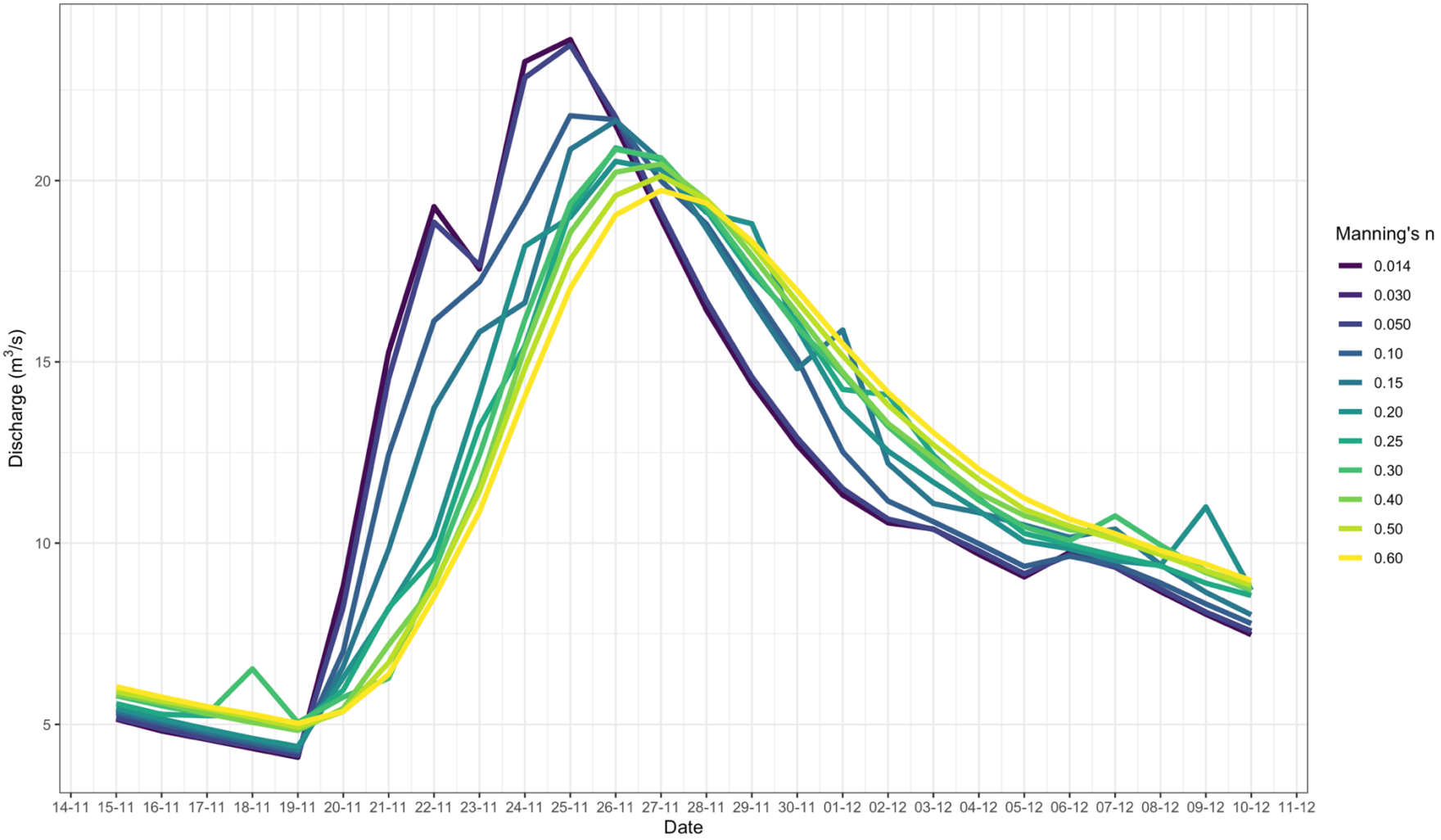


Figure 3.5 Modelled hydrographs for a 25-years flood event. *Different colours correspond to the several different Manning's n values applied in the simulation*

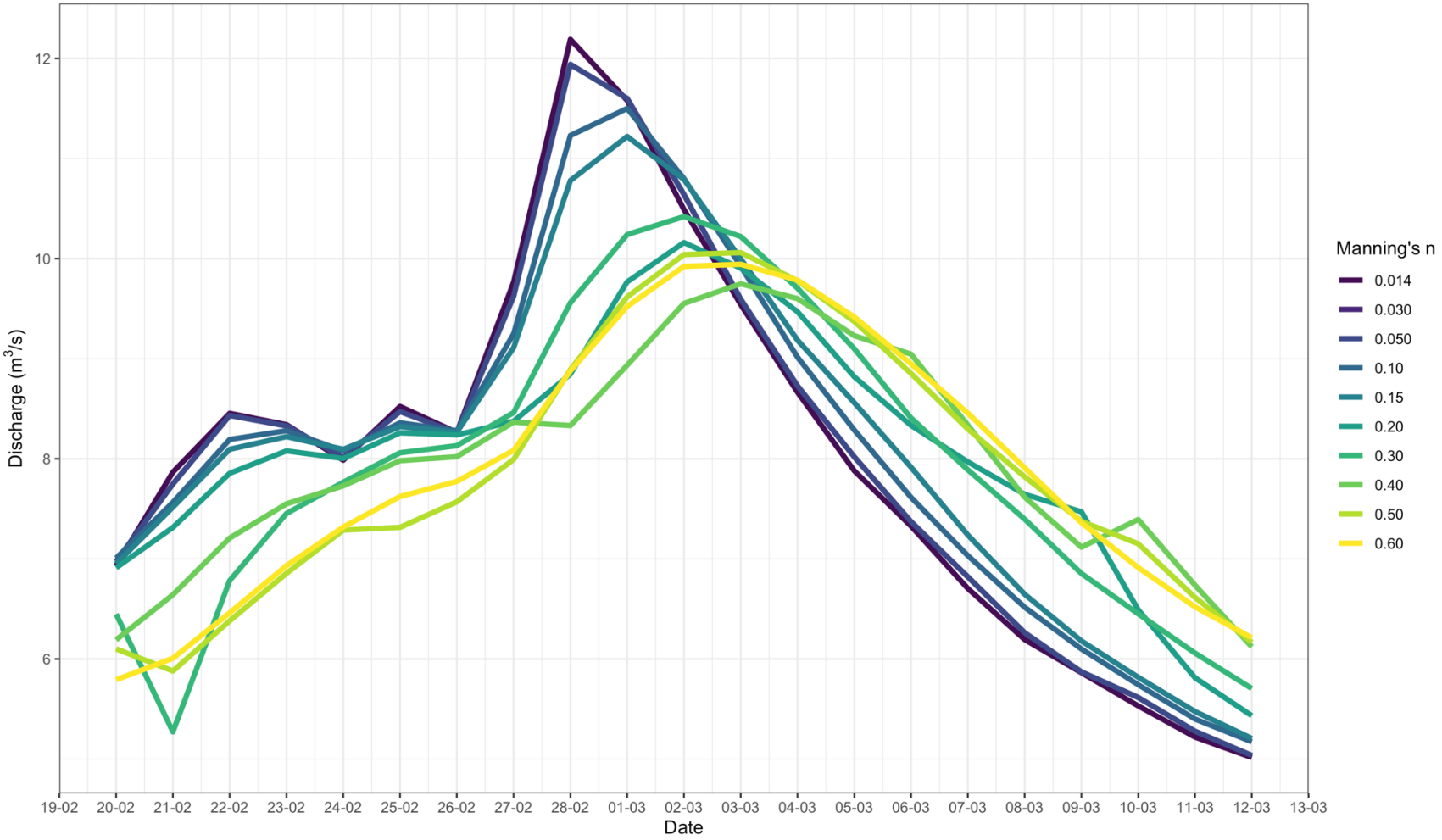


Figure 3.6 Modelled hydrographs for a 1-year flood event. *Different colours portray the different hydrographs simulated with a range of Manning's n values.*

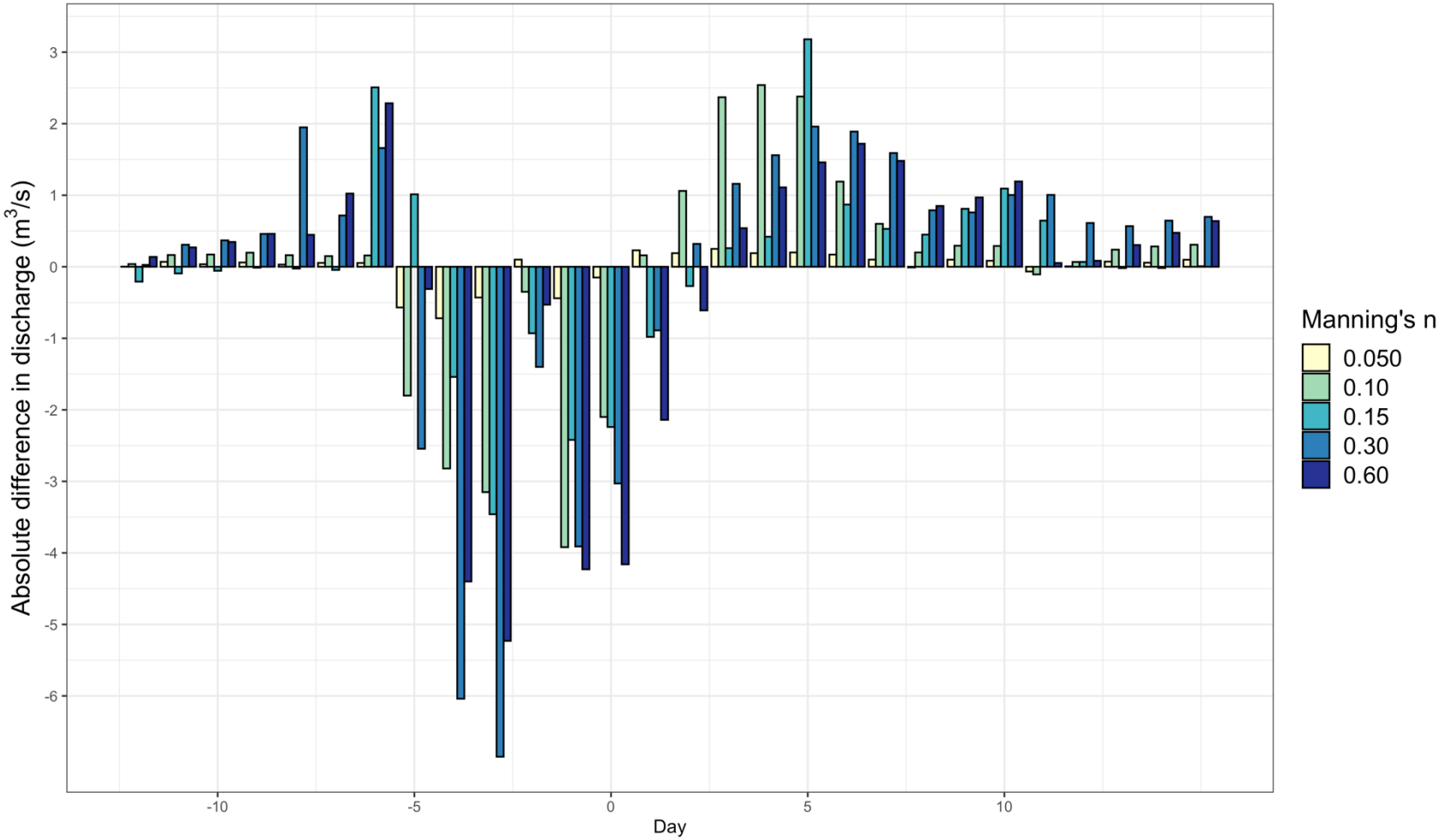


Figure 3.7 Absolute difference in discharge at the catchment outlet (m^3/s) for different simulated hydrographs on the basis of the different Manning's n values applied. *The values are referred to the 25-year flood event of Figure 3.5. Peak timings were synchronised in order to compare peak flow discharges.*

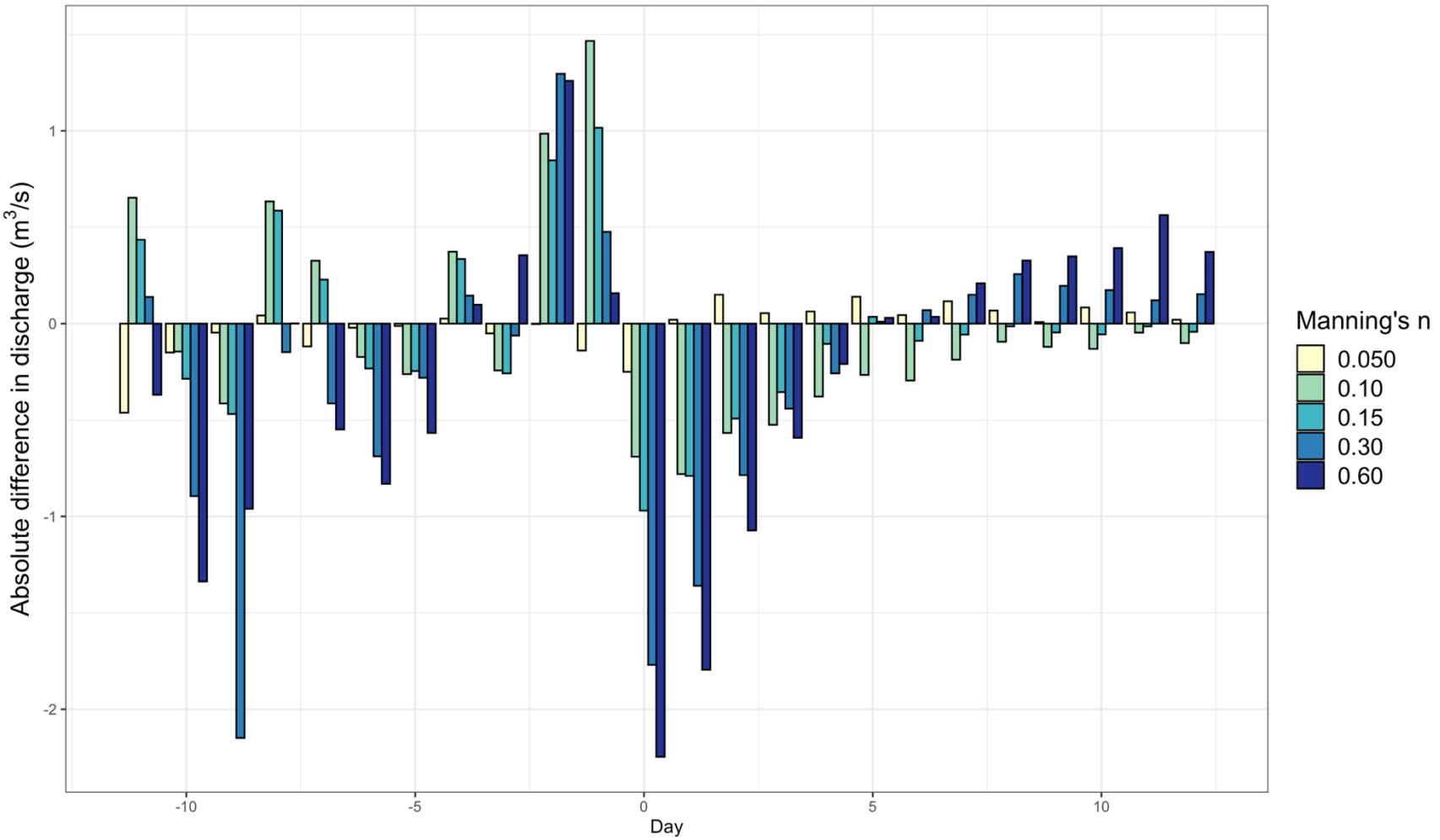


Figure 3.8 Absolute difference in discharge at the catchment outlet (m^3/s) for different simulated hydrographs on the basis of the different Manning's n values applied. *The values are referred to the 1-year flood event of Figure 3.6. Peak timings were synchronised in order to compare peak flow discharges.*

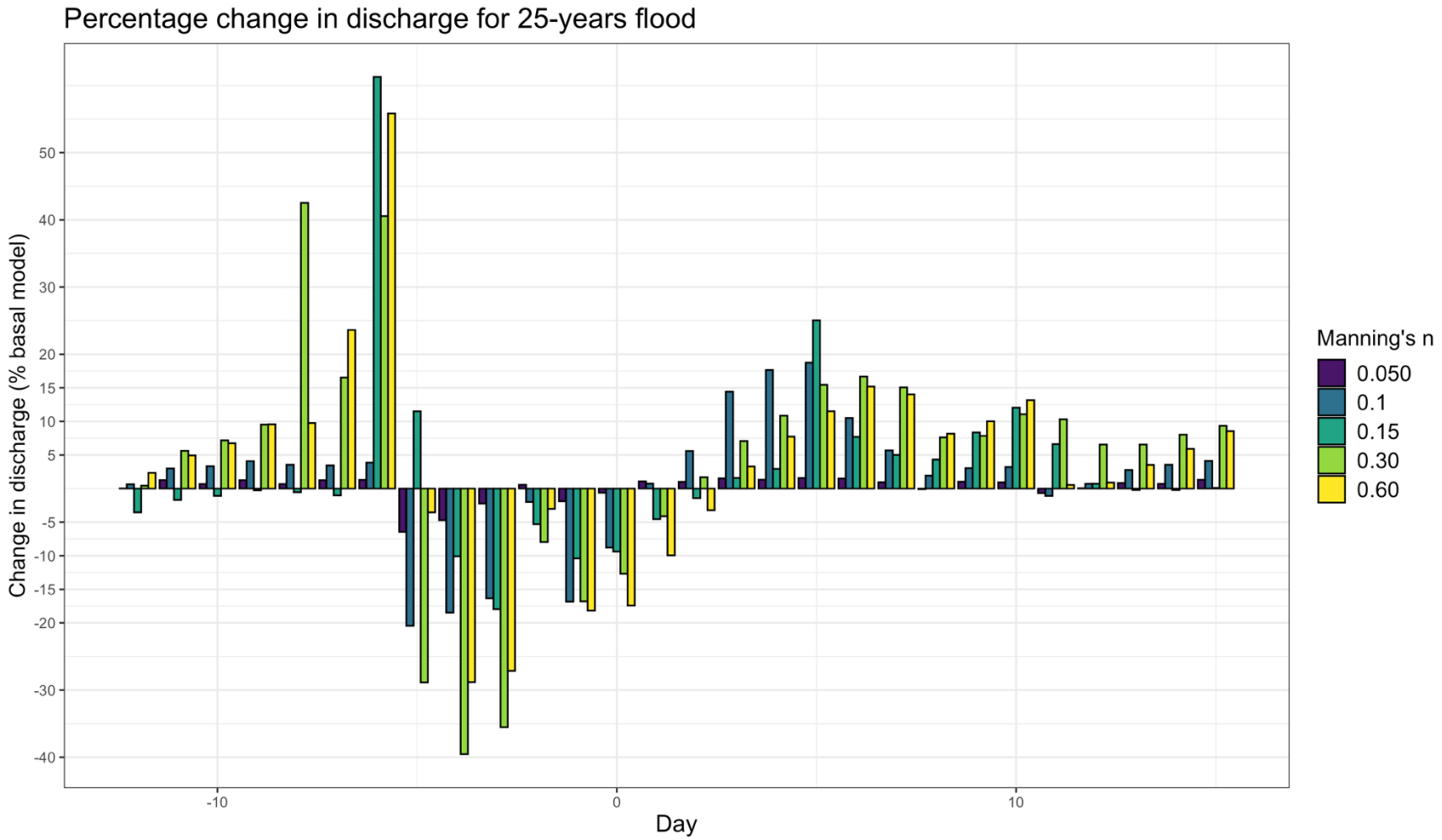


Figure 3.9 Change in discharge at the catchment outlet for a 25-years flood event, expressed as percentages. *Different bar colours correspond to the different Manning's n values used in the simulations.*

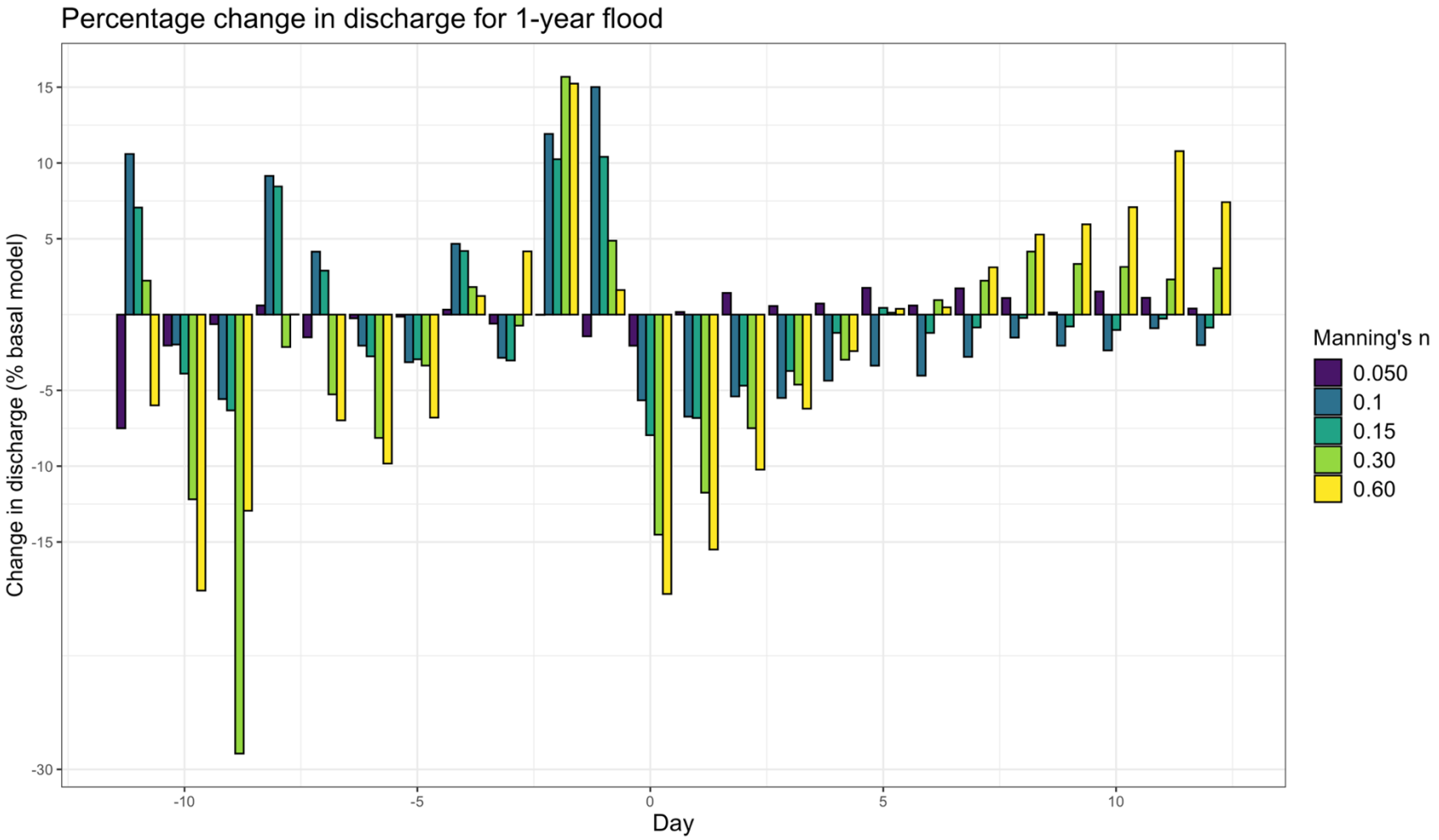


Figure 3.10 Change in discharge at the catchment outlet for a 1-year flood event, expressed as percentages*. Different bar colours correspond to the different Manning's n values used in the simulations.*

3.4.3 Travel times, water yield and scenario selection

The travel times through the catchment (hours, h) computed with the isochrone speed tool are shown in Figure 3.11. Results indicate that travel times through the catchment area are up to a maximum value of 34 hours, with the peak flow travel time being at approximately 16 hours (Figure 3.3). As we can appreciate from Figure 3.11, lower travel times correspond to areas with an extensive presence of channels and tributaries of the river network and closer to the catchment outlet. In order to de-synchronise peaks arriving at the catchment outlet from several different sub-basins at the same time, it is of primary importance defining the index of contribution to the peak flow of the different sub-basins and areas of the catchment.

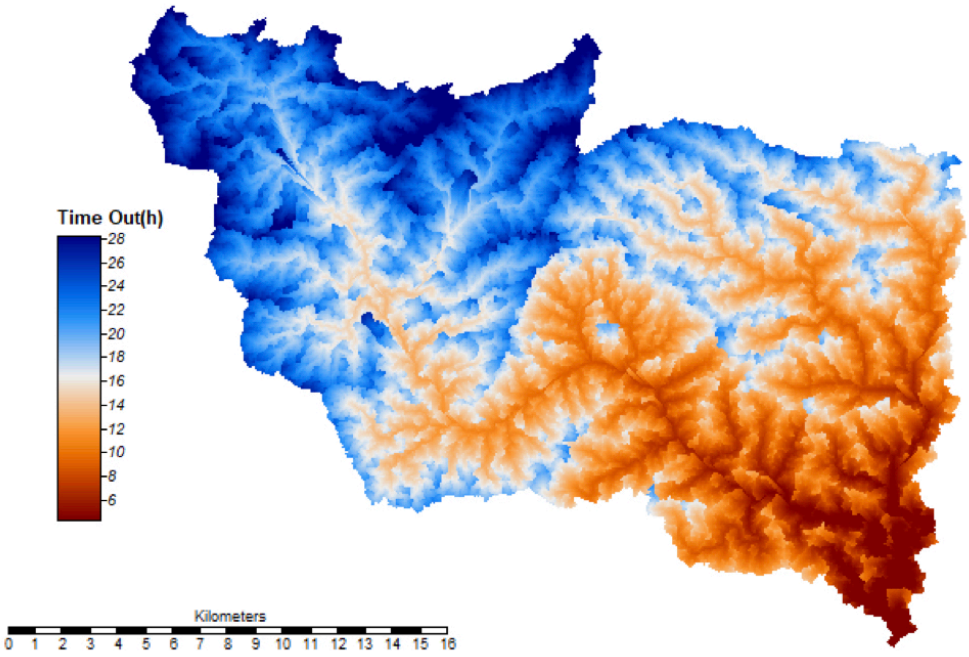


Figure 3.11 Travel times (h) from each grid cell to the catchment outlet based on Manning's n values for different land cover types.

The raster resulting from the definition of the relative index of contribution to the peak flow is shown in Figure 3.12. Results indicate that there are two main areas of contribution to the peak flow based on the index. The first is the area situated in the south-western part of the catchment while the second is the one situated in the upper-eastern part of the catchment. These two areas of the catchment, with the relative sub-basins drawn in SWAT were the ones chosen as priority areas for changes in channel roughness.

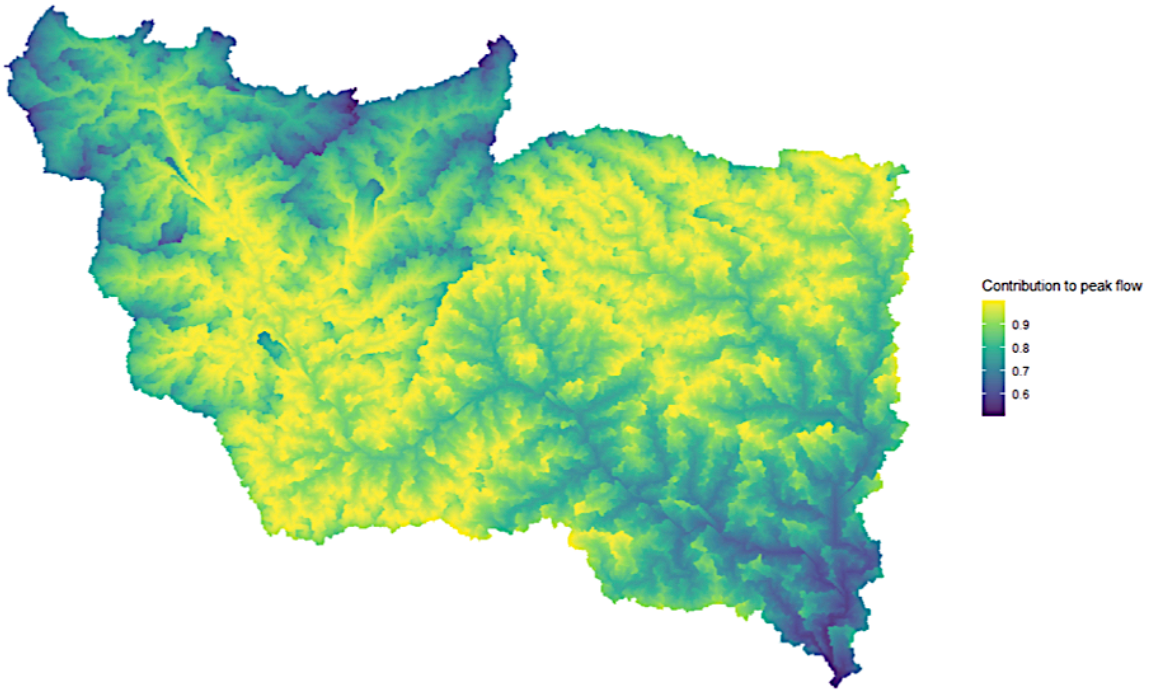


Figure 3.12 Index of the contribution to the peak flow of the catchment land. *The T_{rel} index for land contribution to the peak flow can assume values in between 0 and 1. Higher values represent a major contribution to the peak flow. Yellow areas will thus exert the highest contributions to peak flow at the catchment outlet.*

The choice of priority areas was made also on the basis of the water yield values obtained from SWAT outputs for the sub-basins (Figure 3.13).

As a result of the choices made on the contribution to the peak flow and the water yield from different sub-basins, two different scenarios were chosen, each representing a similar surface area of approximately 80 km^2. The two scenarios are a so called "Uplands" scenario, representing sub-basins in the western part of the catchment (Sub-basins 16, 18,

19, 23, 30, 35, 36 - Figure 3.13) and a "Lowlands" scenario representing sub-basins in the eastern part of the catchment (Sub-basins 7, 9, 14, 26, 27, 28, 29 – Figure 3.13).

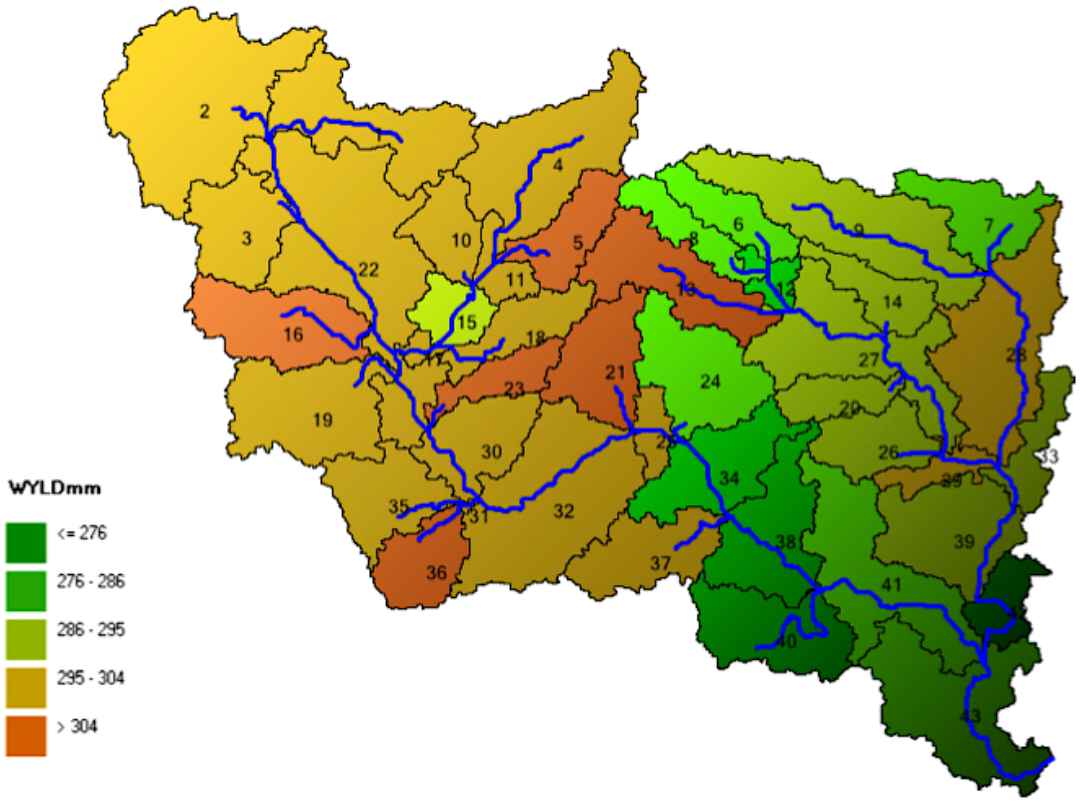


Figure 3.13 Modelled water yield values (mm of water, WYLDmm) for the different sub-basins of the Evenlode catchment as calculated with SWAT model.

The simulated hydrographs for the two different scenarios and for varying Manning's *n* values for a 25-years and a 1-year flood events are shown in Figure 3.14 and 3.15. The two scenarios reached similar results in terms of peak flow height reduction and delay in timing. In particular, as shown in the figures, the changes in channel roughness across the prioritised portion of the catchment, chosen for both scenarios, don't seem to exert appreciable effects on the catchment outlet even for the highest values in Manning's *n*. However, they are still providing a limited reduction of the peak height of between 2 and 7% for Manning's *n* values comprised between 0.15 and 0.60, as it can be seen in Figures 3.16 and 3.17, where percentage changes are shown for the 25-years and 1-year flood events, respectively.

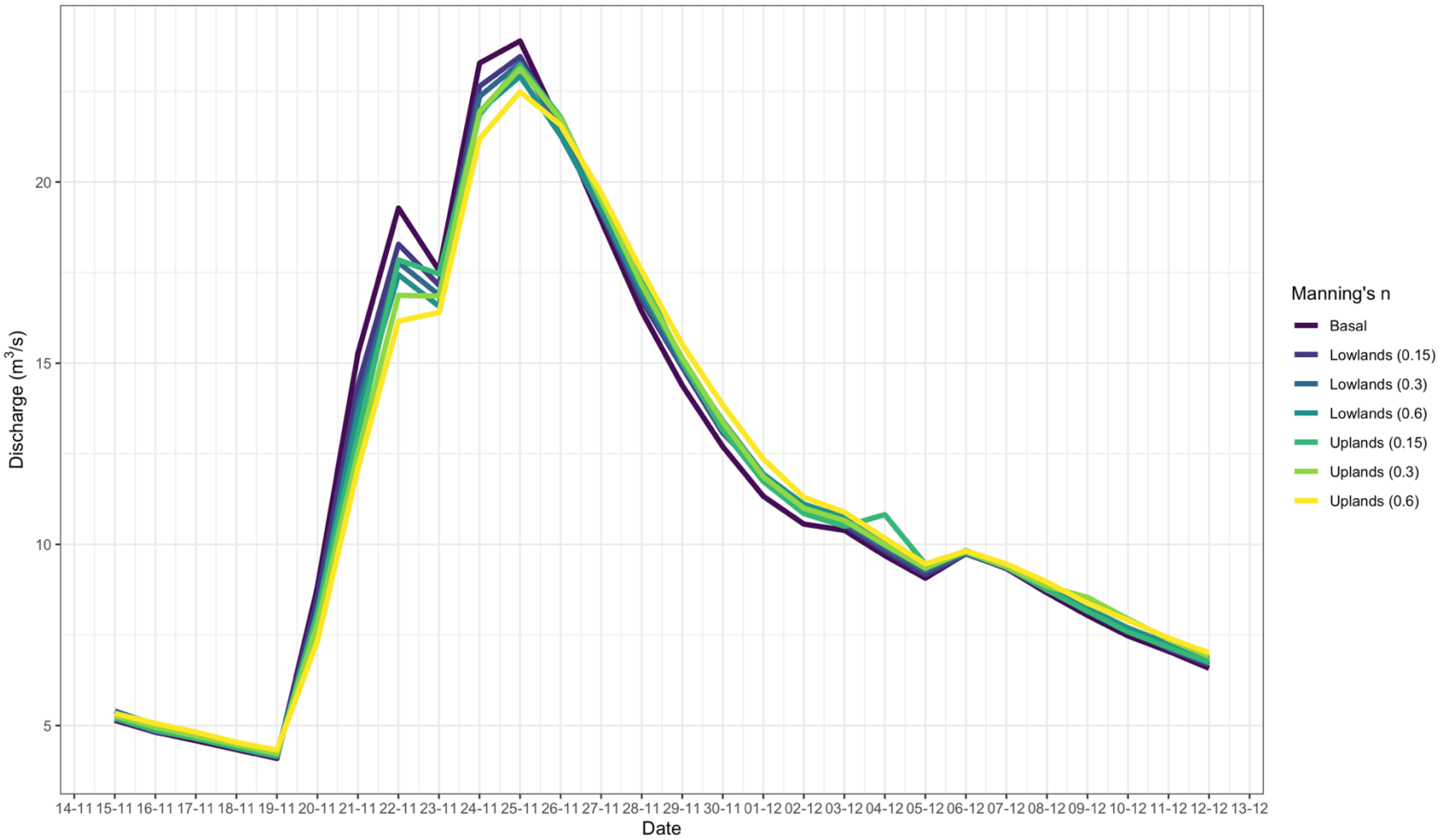


Figure 3.14 Modelled hydrographs for a 25-year flood event. *Different colours represent different simulations made for two different priority areas (Lowlands and Uplands) and for three different Manning's n values. The darker colour represents the basal model calibrated on the observed data.*

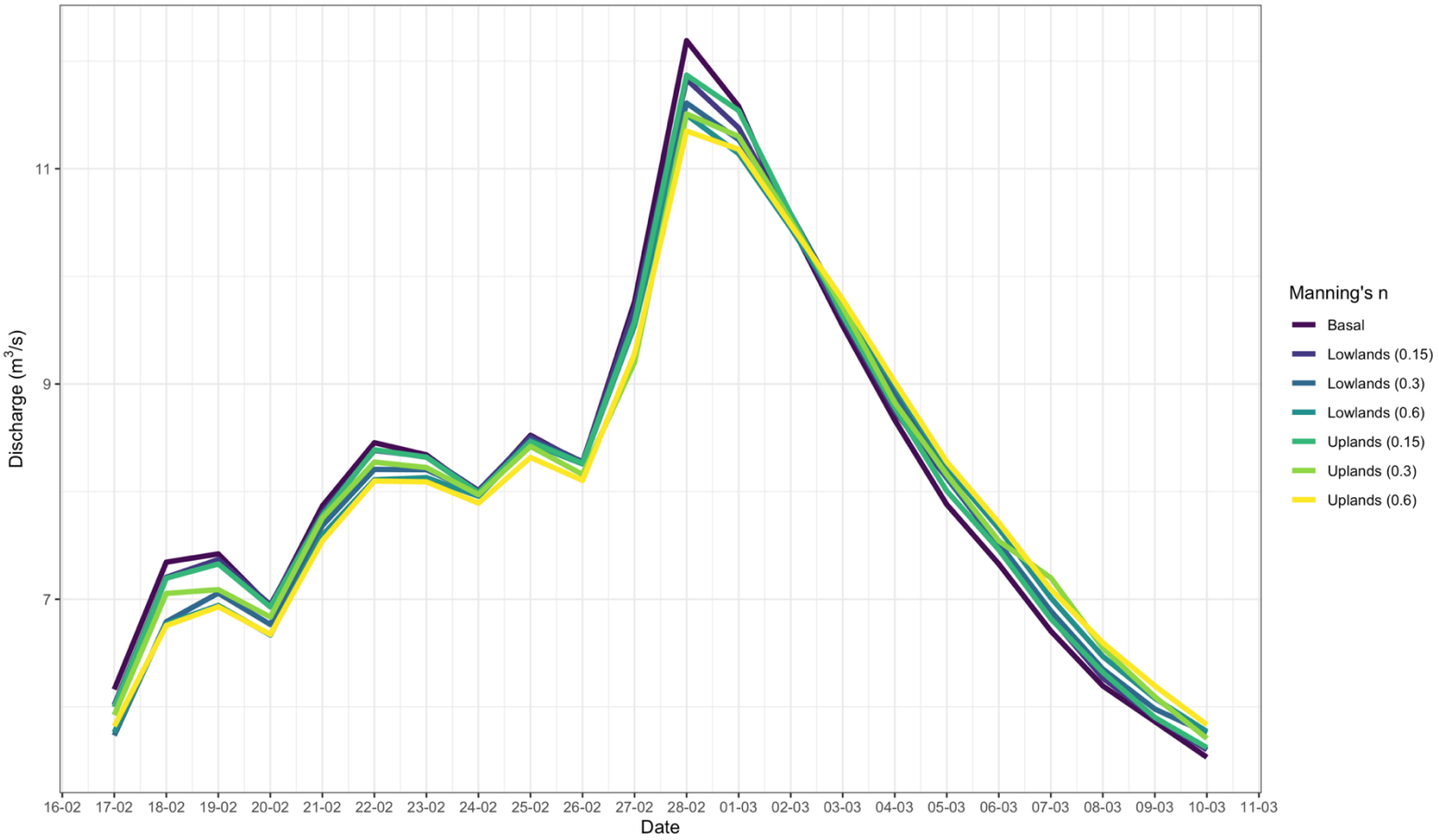


Figure 3.15 Modelled hydrographs for a 1-year flood event. *Different colours represent different simulations made for two different priority areas (Lowlands and Uplands) and for three different Manning's n values. The darker colour represents the basal model calibrated on the observed data.*

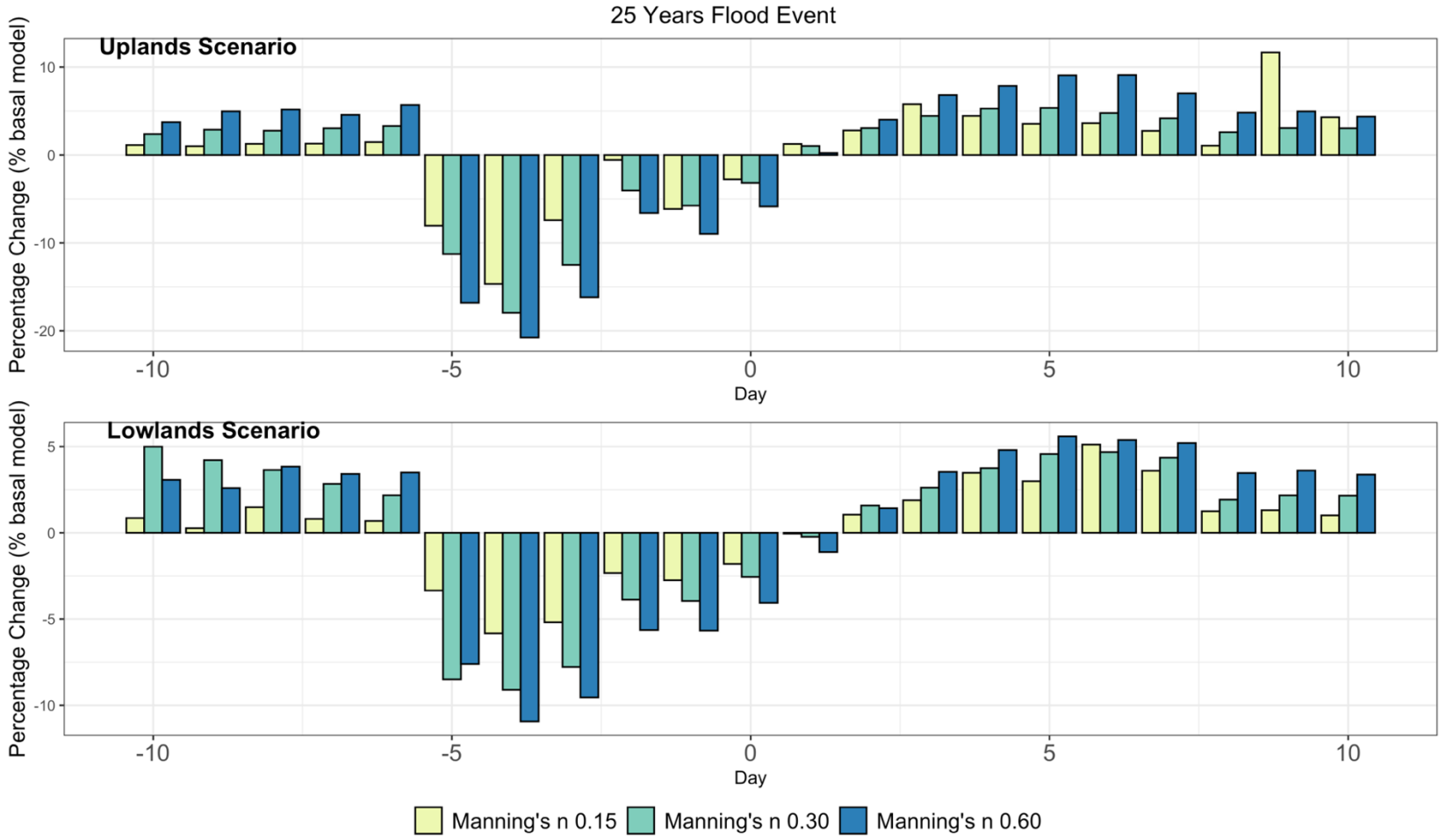


Figure 3.16 Percentage change in peak flow discharge for a 25-year flood event for two different scenarios of priority areas based on contribution to peak flow and water yield. *Day 0 corresponds to the day of the peak flow. Negative values of the x-axis correspond to the days preceding the peak flow passage through the outlet. Different colours are for the different Manning's n values applied to the simulation.*

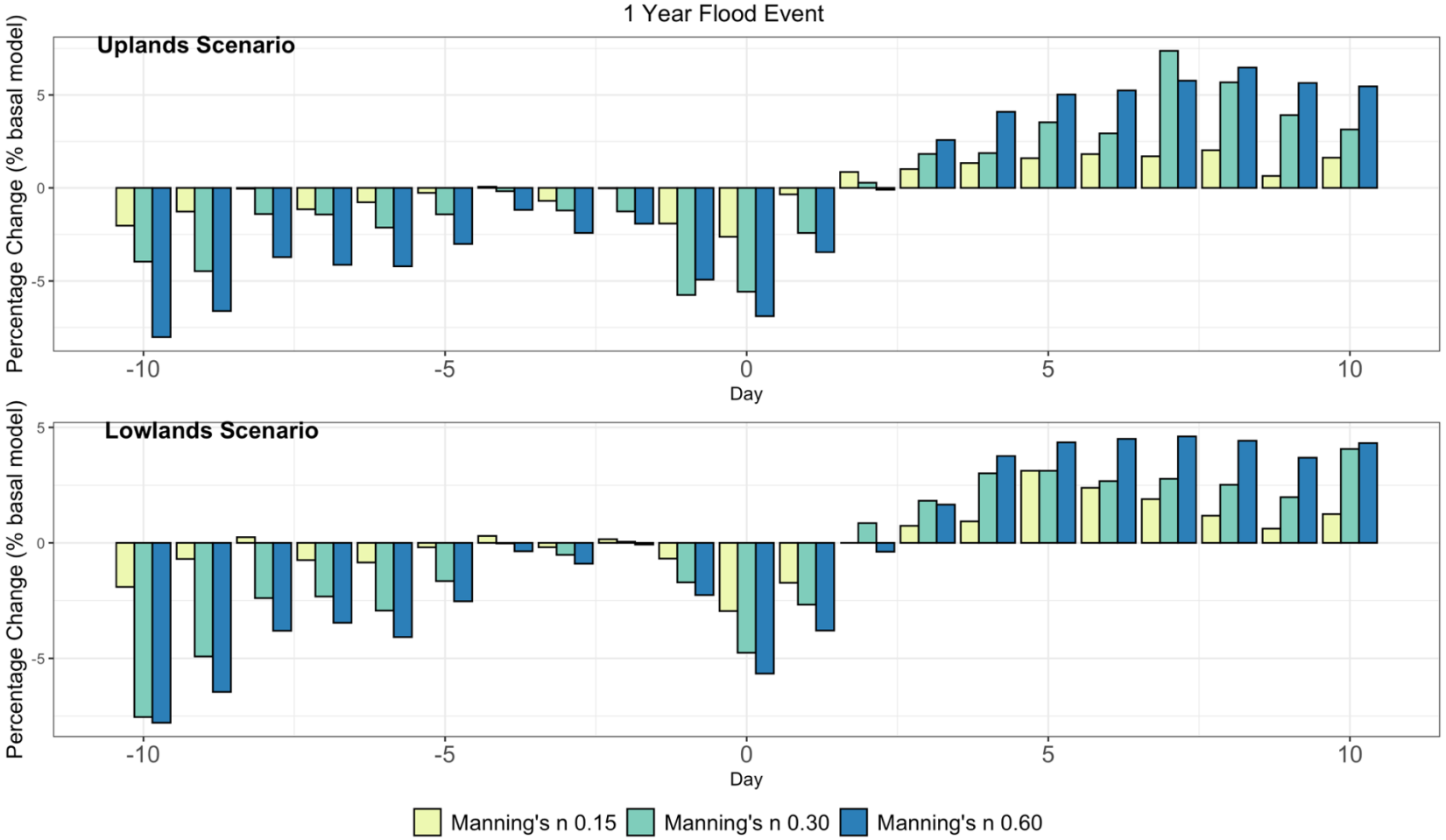


Figure 3.17 Percentage change in peak flow discharge for a 1-year flood event for two different scenarios of priority areas. *Day 0 corresponds to the day of the peak flow. Negative values of the x-axis correspond to the days preceding the peak flow passage through the outlet. Different colours are for the different Manning's n values applied to the simulation.*

3.5 Discussion

The objectives of this study were to build a calibrated and validated hydrological model for the Evenlode catchment to test the catchment sensitivity to a catchment-wide modification of in-channel roughness and then determine priority areas for a change in channel roughness that would given the maximum reduction in peak flow magnitude.

Through the application of a semi-distributed hydrological model, SWAT, I was able to simulate the river Evenlode daily discharge at the catchment outlet. The objective quantification of the model goodness of fit, obtained through three different statistical coefficients, was deemed as 'very good' based on the comparison with values for these three coefficients published by other studies that made use of the SWAT model (Moriasi et al., 2007).

The objective of this chapter was to understand catchment sensitivity to in-channel roughness by applying the data measured in the field for the reach scale of Chapter 2 to the wider scale of the Evenlode. The wide application of changes in roughness might thus be considered as a proxy for the widespread installation of ELJs. However, some differences must be noted.

Manning's *n* is highly dependent on the discharge as seen in Chapter 2 (Figure 2.4-2.5). In this study, due to the limitations of SWAT, a single value of Manning's *n* was applied for each model run. This is equivalent to having a never changing Manning's *n* regardless of the flow conditions. Moreover, applying a single *n* value to the whole river or catchment corresponds to implying that an ELJ is installed every 30 m in the river network and that their performance is neither affected by the presence of the backwater effects of other ELJs nor by the discharge.

In terms of understanding the sensitivity of the catchment to channel roughness, the results of this modelling study show that a change in roughness, across a wide area of land within the catchment, could decrease the height of the peak flow and delay its timing

considerably for a range of events comprised between 1 and 25 years return events. The simulation results indicate a decrease in peak discharge of a minimum of 2% to a maximum of 20% and a delay in the timing of the peak flow in between 10 and 50 hours.

These scales of reduction in peak discharge are in line with the simulations of Liu et al (2004) who applied a similar modelling approach in the Rhine River Basin and with the observations from Thomas and Nisbet (2012) for a smaller catchment in the UK. Interestingly, the change in Manning's *n* values for channels in the river network are as effective in reducing a 25-year flood event as a 1-year flood event. However, in reality, roughness varies with the discharge of the channel, thus making more difficult to maintain high hydraulic resistance, and hence roughness, as the stage and discharge increase. Moreover, since channel roughness is influenced by discharge as shown in Chapter 2 and as described in Odoni and Lane (2010), the restoration of a river reach to high values of Manning's *n* values could be more difficult for deeper flow depths since it would involve bigger structures and, potentially, more local floodplain inundation issues. Moreover, Manning's roughness coefficients are site-specific, they may be influenced by channel characteristics such as meandering and bed substrate, and they depend on flow depth and seasonal accumulations of debris. All these factors are unique to each channel and vary in time and space. For this reason, applying a singular value of Manning's through a wide area, for different flow conditions and for flood events with different recurrence intervals may misrepresent the actual resistance to flow and introduce uncertainties. In particular, Manning's *n* values vary with variation in discharge as explained through Chapter 2 and as noted and showed by many other studies (Addy and Wilkinson, 2019b; Curran and Wohl, 2003; Dixon et al., 2016). Changes in Manning's *n* that cause a shift to a value of n equal to 0.150 for the 1-year event may not be sufficient to maintain the same Manning's *n* value of 0.150 for the 25-year flood. For the high water ELJs used as an example throughout Chapter 2, the effectiveness for different flood events may be particularly dependent on the maximum

height of the ELJ (Figure 2.10-2.11). For these reasons, to maximise the effectiveness of ELJs in increasing Manning's *n* values for flood events with different return periods, the structure of ELJs should be designed on the basis of the most extreme events the measure is targeted to.

In addition, careful consideration of the impacts of the ELJs on floodplain inundation should be taken in the design phase. Raising the maximum height of a high water ELJ to impact the most extreme floods could increase the occurrence and extent of floodplain inundation as shown in Keys et al. (2018) for small woody debris. The more frequent and extensive floodplain inundation could, on the one hand, have a beneficial effect on the reduction of the flood peak by temporarily storing floodwaters in the floodplain; on the other hand, it could lead to negative effects if the flooded area is high-value land or occupied by properties and infrastructures.

Given the importance of the effects of ELJs on floodplain inundation and their potential feedback mechanisms for flood risk, floodplain inundation caused by ELJs presence will be addressed into Chapter 4.

3.6 Conclusions

Engineered Log Jams are not the only NFM or stream restoration measure that could increase the channel roughness and consequently have an impact on catchment hydrology. In particular, all those measures that tend to re-create river natural hydraulic and morphological characteristics, such as re-meandering, in-channel vegetation and non-clearance of debris, could result in an increase of the in-channel roughness.

ELJs are an important feature for NFM projects, given the economic feasibility of their deployment and the relative singular low impact in terms of space occupied. Moreover, ELJs can provide additional benefits in terms of retention of sediment, enhanced water quality, increased habitat variability, thus favouring biodiversity and the creation of spaces for

recreation. These strategies are therefore essential for a modern approach to catchment-based flood management and represent a good cost/benefit ratio for the decrease of local flood risk for small catchments.

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Chapter 4 : A hydrological-hydraulic modelling cascade to investigate the effects of engineered log jams on floodplain inundation and peak flood magnitude and timing.

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4.1 Introduction

In the UK, since the 20th century, the runoff retention capacity of catchments has been decreasing (Metcalfe et al., 2018). This decrease has been suggested as one of the main contributors to the frequency and magnitude of flooding downstream (O'Connell et al., 2007). The factors leading to the reduction in runoff retention capacity are mainly due to urbanisation, changes in land cover, more intensive agricultural practices and improved field drainage (Metcalfe et al., 2018; Wheater and Evans, 2009). Traditionally, flooding risks have been dealt with hard engineering structures, such as flood walls and levees, and activities such as river clearing and dredging. These interventions aimed to increase water retention and local channel conveyance, thus reducing water levels locally and avoiding spillage into the floodplain. However, evidence has indicated that whilst reducing the problem upstream, this can prove problematic for areas downstream that are subject to higher discharges as a result of the improved channel conveyance and the input discharges from other tributaries. Moreover, hard engineering structures may not always be applicable for local interventions due to the high costs of their construction and maintenance (Hankin et al., 2019; Metcalfe et al., 2018)

From 1998 to 2013, extensive flooding events across Europe caused damages estimated at £54 billion (European Environment Agency, 2016). In response to these, the EU promulgated the European Floods Directive (2007/60/EC), which required member states to evaluate flooding risks and act to mitigate them. Among other suggestions, the Flood Directive recommended the use of "natural water retention" for flood mitigation purposes. Moreover, the Floods Directive had to be implemented in coordination with the Water Framework Directive (2000/60/EC), which required member states to implement river basin management plans that also resulted in an improved ecological and chemical status of the rivers. In the UK, following the extensive flooding of 2007, the Pitt Review (2008) also made the same policy recommendation towards "working with natural processes". In this policy context, flood

mitigation approaches are referred to in several different terms, including Natural Flood (Risk) Management (NFM/NFRM) and Nature-Based Solutions (NbS). NbS have gained popularity across Europe and the UK to reduce flood risk (Hankin et al., 2019; Lane, 2017b). These measures aim to restore or enhance the natural hydro-morphological characteristics and processes within the catchment floodplain and river network while, at the same time, enhance or restore environmental co-benefits such as improved soil and water quality, biodiversity and habitat provision. Thanks to the increasing interest in such NbS and especially in those built as a result of NFM schemes, there has been a growing number of investigations focusing on the effects of these measures in decreasing flood risk (Metcalfe et al., 2018; Nicholson et al., 2012; Nicholson et al., 2020).

NFM measures include small-scale features like engineered log jams (Addy and Wilkinson, 2019b), leaky dams (Metcalfe et al., 2018), runoff attenuation features (Nicholson et al., 2012; Nicholson et al., 2020), and river restoration for re-meandering and floodplain reconnection (Keys et al., 2018; Liu et al., 2004), together with more widespread solutions such as afforestation (Carrick et al., 2019) and peatland restoration to increase roughness (Shuttleworth et al., 2019). Among NFM measures, the in-channel deployment of large wood and engineered log jams (ELJs) has been one of the most widely applied NFM measures across the UK. ELJs have the potential to add frictional resistance to the water flow, thus retaining water in the channel incision, and reduce water velocities (Addy and Wilkinson, 2019b; Dixon et al., 2016; Odoni and Lane, 2010). Moreover, large wood and ELJs can affect multiple processes at both the local reach scale and at the wider catchment scale (Figure 4.1).

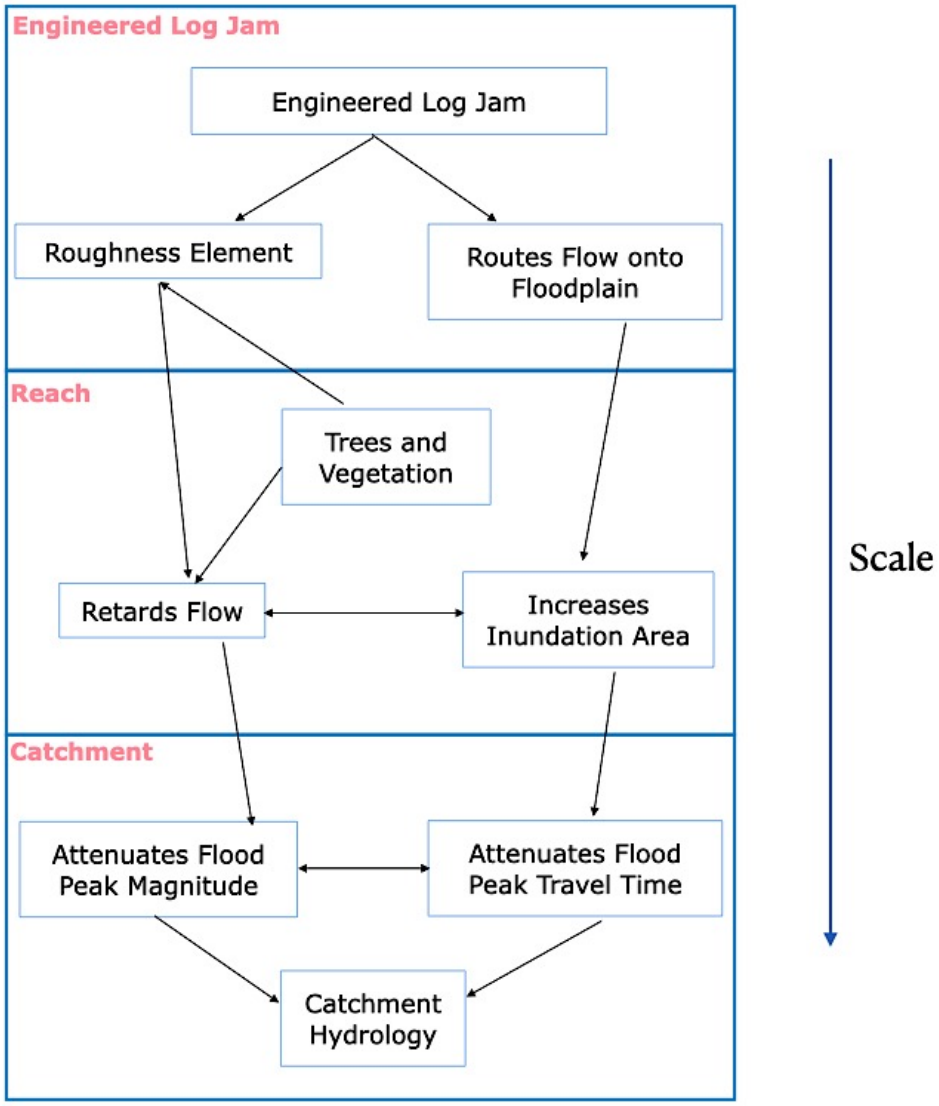


Figure 4.1 The interconnected effects of large wood on water flow at different scales.
Adapted from Kiesel (2013).

At the local scale ELJs decrease flow velocities in the channel by adding resistance thus slowing the velocity of the waterflow and providing a temporary storage of water in the channel incision by increasing water levels. This in turn increases the potential for floodplain inundation as a result of higher water levels upstream of the ELJs (Keys et al., 2018). The local effects can in turn slow the flow at the reach scale and increase the inundation area, thus providing a temporary storage of floodwaters into the floodplain. These effects may in turn delay the movement of peak flows through the river network following heavy rainfall.

A delay in the timing of the peak flow and a reduction of its magnitude can be the final result of the deployment of ELJs through the catchment. When used across a catchment, ELJs are indeed thought to reduce the likelihood that the peak flows from multiple tributaries will arrive simultaneously downstream, thus decreasing the risk that the capacity of the river network will be exceeded where higher value land, assets and infrastructure may be located (Dixon et al., 2016; Thomas and Nisbet, 2012).

Apart from the advantages for flood risk management, the success of ELJs as an NFM measure has been prompted by the additional environmental co-benefits they can provide, such as sediment and nutrient retention, habitat restoration and increased biodiversity (Burgess-Gamble et al., 2017). Moreover, these features are widely applicable thanks to their economic feasibility and ease of deployment in a variety of different settings (Hankin et al., 2019).

Together with the raising interest in ELJs for flood retention purposes, a few studies have focused on the effectiveness of these at a variety of scales and for different types of ELJs and large wood. In particular, Thomas and Nisbet (2007) estimated a reduction in flow velocities and a 15 minutes retardation in the downstream flood peak following the restoration of 5 large wood dams in a 0.5 km reach. Keys et al. (2018) and Quinn et al. (2013) showed how large wood was increasing floodplain inundation by inducing a backwater effect and higher water levels upstream.

Even with the increasing number of studies focusing on NFM measures and their effectiveness, only a few studies to date have considered their effects at the catchment level, and the different modelling strategies that can be employed to determine their effectiveness.

The simulation of small-scale features such as ELJs to depict their effects at the catchment scale is indeed a challenge that hydrologists and river engineers had been trying to solve since these measures started to be largely deployed. In order to test the effects at the catchment scale, strategies for the distributed modelling of ELJs and other small-scale NFM

measures to test their effects at the catchment scale should rely on a representation of hydrological responses accurately, and then represent the changes incurred to those processes as a result of small-scale distributed NFM intervention realistically (Hankin et al., 2019).

While a single hydrological or hydraulic model can prove helpful in those situations where we are interested in either the wide catchment scale or the hydraulic model scale, there are situations in which a combination of models can improve the overall accuracy of the representation of processes. Semi-distributed watershed models like SWAT, used in Chapter 3, can indeed be inadequate in simulating the in-channel process and the spatial extent of floodplain inundation at the reach scale and its effects on the delivery of the waterflow downstream. On the other hand, hydraulic models are not computationally efficient to depict larger scales and they do not consider all of the hydrological processes that brings to the formation of the waterflow into the channel.

The Soil and Water Assessment Tool (SWAT) can simulate hillslope hydrology for a variety of systems providing inputs to each subbasin's main river channel in terms of water and nutrients. In contrast, one-dimensional (1-D) and two-dimensional (2-D) hydraulic models, like the widely used Hydrological Engineering Center River Analysis System (HEC-RAS), can simulate channel and floodplain hydraulics at finer spatial resolutions. HEC-RAS is a malleable hydraulic model that can be applied to various scenarios and simulate the multiple components of the hydraulics of channel flow and floodplain hydraulics, from water surface elevations to flow velocity. For these reasons, modelling strategies that involve model coupling and modelling cascades can prove useful in depicting the effects of hillslope hydrology and also in explicitly representing the in-channel hydraulic processes.

Hydrologic and hydraulic modelling cascades have been previously implemented in various contexts with different hydraulic and hydrological models (Guse et al., 2015; Hankin et al., 2019). However, despite the increasing usage of modelling cascades in the last few years,

given the improved flexibility of the individual models, little has been done to evaluate the effects of NFM measures, such as ELJs, using modelling cascades on a variety of characteristics of channel flows and floodplain hydraulics. Moreover, no study has specifically focused on high flow ELJs deployment within rural or highly modified catchments and river networks. Filling this knowledge gap by deploying a modelling cascade to fully understand the effects of high flow ELJs was, therefore, an important research aim of my thesis, as described in the remainder of this chapter.

4.2 Aims and Research Objectives

The overarching aim of this part of my DPhil research was to model the effectiveness of high flow engineered log jams in reducing flood peak magnitude and delay the peak flow timing for a small catchment (16.3 km^2). Specifically, I aimed to use a modelling cascade to understand the local scale effects of ELJs on floodplain inundation, water surface elevations and waterflow velocities. The hydrological model will allow the depiction of the runoff generation at the catchment scale, while the implementation of a 2D hydraulic model, built on the inputs from the hydrological model, will allow the explicit representation of high flow ELJs at the local scale and their local effects on water flow and floodplain inundation. The local scale effects will then be upscaled to the catchment scale by the explicit representation of the catchment's river network in the hydraulic model. The simulation of the in-channel hydraulics for the whole catchment will help in understanding the effects of high flow ELJs on the magnitude and timing of peak flow at the catchment scale.

To address this overarching aim and building on the results of the simulation outlined in Chapter 2 and 3 and the field surveys of Chapter 2, I undertook the following interlinked objectives:

1) Estimate the effects on water flow velocities and water surface elevations of the deployment of ELJs, and how these may impact and increase floodplain inundation.
2) Simulate the effects of the local storage provided by the floodplain inundation on the peak flow magnitude and its timing at the catchment level (16.3 km^2) for events of different magnitude.
3) Compare the simulation of the effects of different scenarios characterised by varying number of ELJs deployed within the catchment, in order to understand the sensitivity to the number of ELJs.

4.3 Materials and Methods

4.3.1 Study Area

The Littlestock Brook is a small, 16.3 km^2, rural catchment situated in the south-eastern part of the Evenlode catchment (Figure 4.2). This catchment's land use is mainly composed of pastures and arable land with two small villages, Milton and Shipton under-Wychwood. Milton-under Wychwood has been affected by the flooding events of 2007 and 2013. Moreover, the river network of the Littlestock Brook has been impacted by other environmental issues caused by sediment travel downstream and high phosphorus concentrations in its waters. In order to decrease the in-channel water velocities, increase water retention in the channel incision and decrease sediment and nutrient transport downstream, a 5-years Natural Flood Management project has been developed in this catchment. This has allowed the deployment of more than 40 ELJs within the catchment plus other NFM solutions, including corner bunds to retain flood waters temporarily, soil scrapes to connect the channel to the bunds and afforestation measures within the catchment.

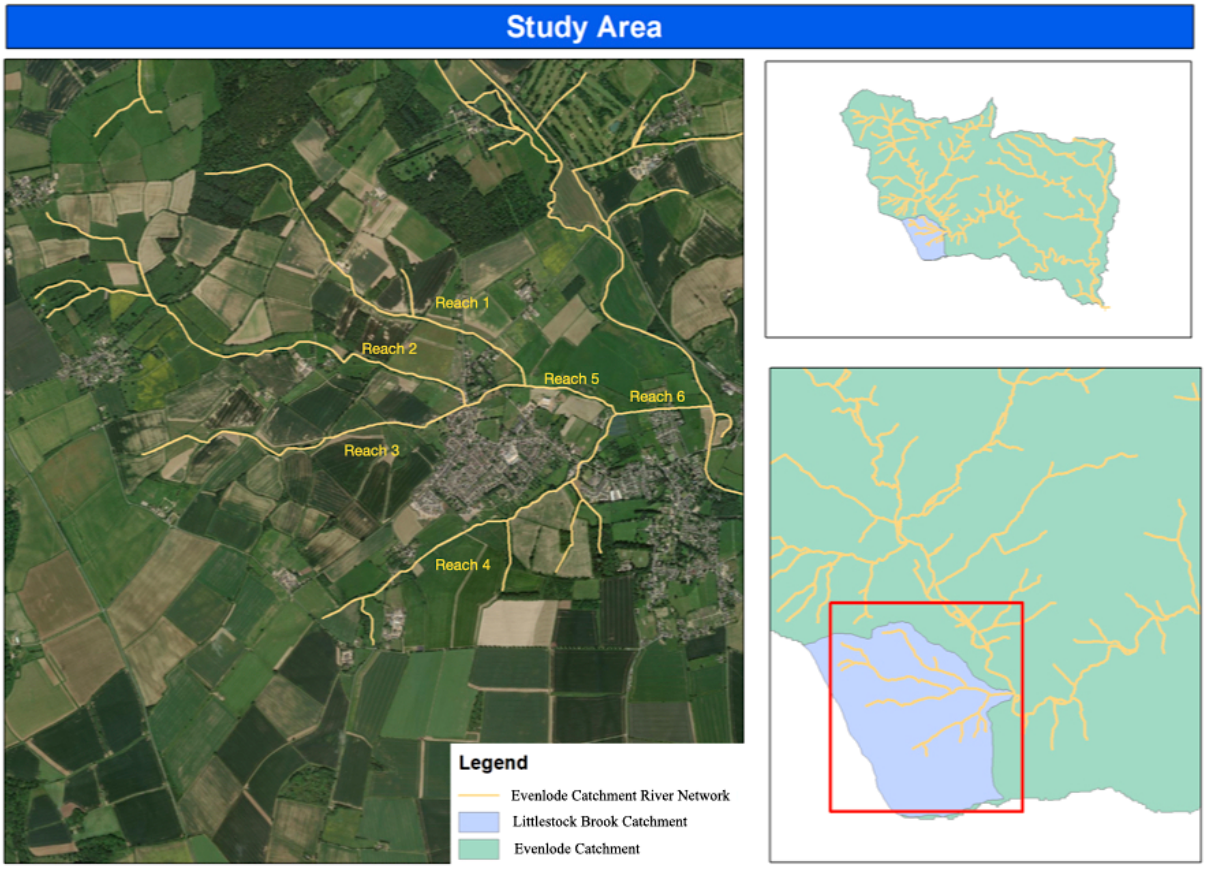


Figure 4.2 Map of the study area. *The figure shows the location of the Littlestock Brook catchment (blue) within the Evenlode catchment (green). The river network of the Evenlode catchment is drawn in yellow. The figure on the left shows an enlarged view of the four main channels (Reach1-4) of the Littlestock Brook's river network. The Littlestock Brook joins the Evenlode north of the village of Shipton-under-Wychwood (Reach 6).*

4.3.2 Modelling Approach

To achieve the main objectives of this chapter, the application of a cascade modelling approach that couples the Soil and Water Assessment Tool (SWAT) with the Hydrologic Engineering Center's River Analysis System (HEC-RAS) was deployed. This modelling approach takes advantage of the computational efficacy of the catchment hydrology represented by a semi-distributed hydrological model, SWAT, coupled with the 2D hydrodynamic modelling capabilities of HEC-RAS. This approach was initially designed to consider large-scale effects of small-scale features (Keisel et al., 2013). More recently,

modelling cascades have been developed to depict the effects of small-scale features and impacts on the catchment's hydraulic and hydrologic behaviour (Hankin et al., 2019).

The structure and methodology here applied are similar to the one by Metcalfe et al. (2018) but using 2D-hydrodynamic modelling to better account for the effects of floodplain inundation and the direction of the waterflow that leaves the channel. Maintaining a continuous spatial connection between the models allows simulation of water fluxes from the watershed to the channel scale for evaluating ELJs effects. Here I used the consecutive application of two different models: a hydrologic model to represent catchment-scale hillslope processes (ArcSWAT) and a two-dimensional hydraulic model to explicitly represent ELJs at the reach scale and their effects on waterflow regulation at both the local and catchment scales. By coupling models this way, it is possible to capture features that have a significant impact locally while also representing the catchment in a computationally tractable way.

First, the Soil and Water Assessment Tool (SWAT) was used to simulate the impact of catchment characteristics, climate and land use on the catchment water and sediment fluxes. SWAT is a freely accessible tool developed by Texas A&M and the USDA that has been used worldwide at various scales, from small catchments to river basins. Within my modelling framework, SWAT was used to simulate runoff contribution from the wider catchment to the reach scale. The SWAT model was run for the entire Evenlode catchment, of which the Littlestock Brook is a sub-catchment. Since the Evenlode is a gauged river with daily flow data recorded since 1969 at Cassington Mill, modelling the entire Evenlode will allow for the calibration and validation of the SWAT model using the gauged flow data.

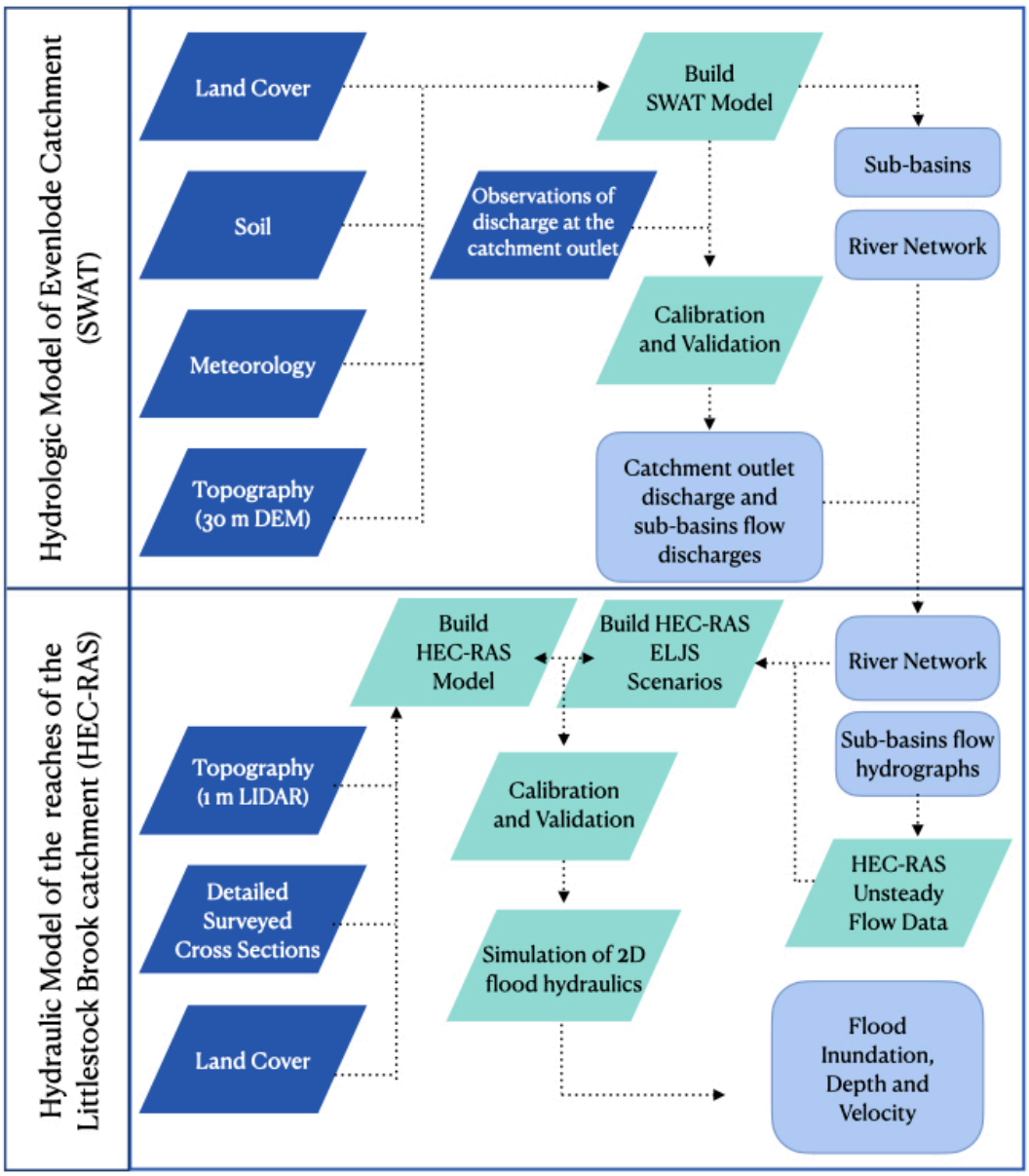


Figure 4.3 Schematic of the cascade framework proposed in this study*. Dark blue polygons refer to input datasets and surveyed data. Green polygons refer to modelling processes, while light blue polygons represent model outputs.*

SWAT has a simplified representation of the catchment using sub-basins and Hydrologic Response Units (HRUs). While this simplification allows the model to be computed quickly across very large river basins, it also makes the model too coarse for depicting changes in water flow at finer resolution. In my model cascade, smaller-scale channel processes such as the hydraulic processes that can be affected by the installation of engineered log jams, were simulated at the reach-scale using the HEC-RAS model. The outputs of the SWAT simulation

(hourly stream flow) formed the boundary conditions to the HEC-RAS model as shown in Figure 4.3. To connect the two models, the SWAT catchment map and river network were used as inputs for the geometrical representation of the catchment into the HEC-RAS model domain, thus providing the continuity needed in the modelling cascade (Figure 4.3).

4.3.2.1 SWAT

The Soil and Water Assessment Tool (SWAT) is a semi-distributed hydrological model that represents catchment hydrology by dividing the catchment into functional sub-basins that are then further discretised into homogenous hydrological response units (HRUs). HRUs are functional units that corresponds to unique combinations of land use, soil and slope. In the SWAT model, physical processes are separated in two phases: the land phase and the channel or routing phase. HRUs don't need to be spatially contiguous in SWAT, but are lumped together as single units. SWAT calculates the loadings for each HRU and these loadings are then summed to determine the total loading of the sub-basins (land-phase). Each sub-basin is assigned to a main channel which then transfers loadings from the sub-basin to the other reaches within the catchment (water-phase). SWAT uses Manning's *n* equation to define the rate and velocity of streamflow.

4.3.2.2 HEC-RAS

The Hydrologic Engineering Center's River Analysis System (HEC-RAS) is one of the most used and well-known software for hydraulic modelling given its strong computational abilities, rapid processing time and free licensing through the U.S. Army Corps of Engineers (https://www.hec.usace.army.mil/software/hec-ras/), which allows the two-dimensional depiction of the flow within the channel and floodplain. The two-dimensional modelling of HEC-RAS is based on the numerical solution of the 2D Saint-Venant equations for conservation of mass (4.1), conservation of momentum in the x direction (4.2) and conservation of momentum in the y direction (4.3).

$$\frac{\partial H}{\partial t} + \frac{\partial (h v_x)}{\partial x} + \frac{\partial (h v_y)}{\partial y} = 0 \tag{4.1}$$

$$\frac{\partial v_x}{\partial t} + v_x \frac{\partial v_x}{\partial x} + g \frac{\partial H}{\partial x} + g \left(S_f - S_0\right) = 0 \tag{4.2}$$

$$\frac{\partial v_y}{\partial t} + v_y \frac{\partial v_y}{\partial y} + g \frac{\partial H}{\partial y} + g \left(S_f - S_0\right) = 0 \tag{4.3}$$

where H is the water surface elevation, h is the hydraulic head, v_x is velocity in the downstream direction, v_y is the velocity in the transverse direction, g is the acceleration due to gravity, S_f is the energy slope and S_0 is the channel slope.

In HEC-RAS the Saint-Venant Equations (Equations 4.1– 4.3) are numerically solved by using a finite volume approximation discretised in time and space. HEC-RAS can solve the full Saint-Venant equations or use the diffusive wave approximation. When using the diffusive wave approximation, the first and second terms of Equations 4.2 and 4.3 are considered null. For my research, the hydraulic model was run using the diffusive wave approach due to increased computational speed and for the higher model stability. The diffusive wave

approximation is common to 2D hydrodynamic models and is based on the consideration that the pressure and slope components are of an order of magnitude greater than local and convective acceleration terms. These terms are thus neglected in the numerical solution of the equations (Henderson, 1966).

In the initial modelling phases, both the diffusive wave approach and the complete solution of the equations were tested. Since they were producing similar results and the computation with the full solution of the equations required longer run times, the diffusive wave approach was chosen for the subsequent model runs. Keys et al. (2018) and Quirogaa et al (2016) showed that the diffusive wave approach was giving comparable results to the complete solution of the three Saint-Venant equations, despite the simplification of the Saint-Venant equations it uses, and its application is acceptable for 2D modelling of flood events.

4.3.2.2.1 Model Geometry

HEC-RAS model geometry was built on the basis of the river network imported from ArcSWAT as a shapefile, as shown in Figure 4.4. After pre-processing the river network, a 2D mesh perimeter overlay was drawn along the central reaches with a cell size of 20 by 20 meters. This cell size mesh was chosen after multiple runs to evaluate the effects of different mesh resolutions on the results produced by the model. Breaklines were drawn for the channel banks and the stream centreline to align computational cells to the waterflow and to the channel banks. The breaklines were drawn with a near spacing of 0.5 m and 4 near repeats (breaklines are represented in pink in Figure 4.4). The spacing of the breaklines was chosen in order to have an area of higher resolution for the channel reach and along the banks, and based on the recommendation in the HEC-RAS manual to have 5-10 cells across the channel width.

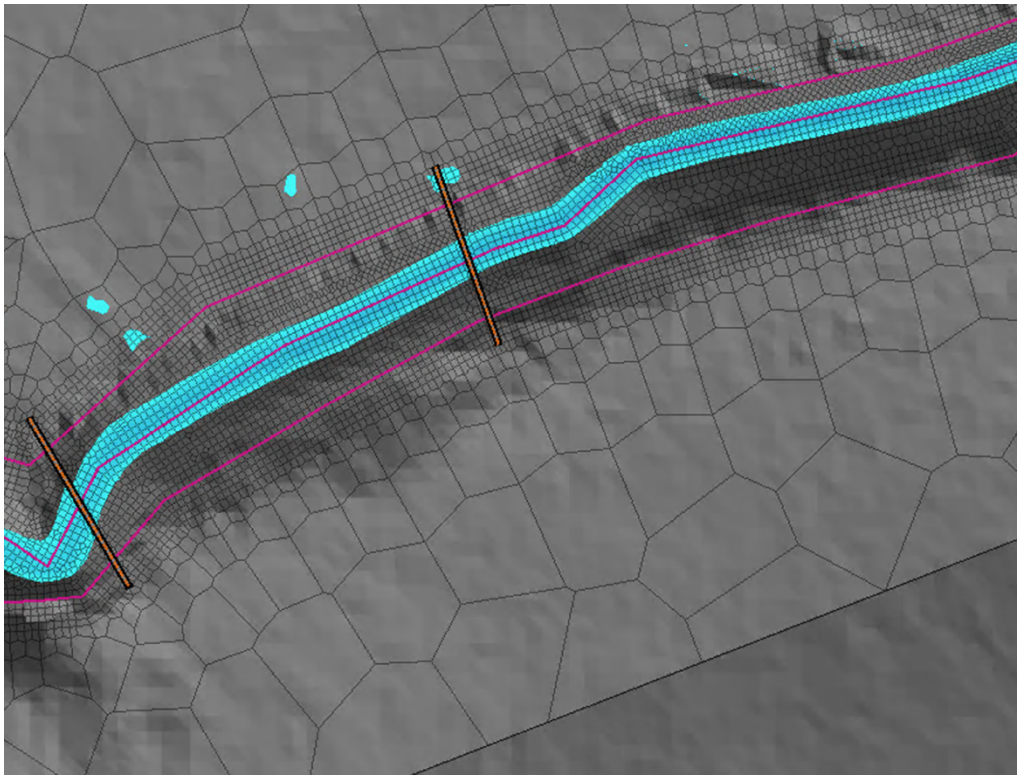


Figure 4.4 HEC-RAS 2D geometry with underlying terrain model. *Pink lines represent the three breaklines used in the geometry. Black contours show each 2D computational cell's edges (one computation point lies in the middle of each cell). The brown lines perpendicular to the channel represent the ELJs, and the waterflow passing through the channel is represented in light blue.*

4.3.2.2.2 ELJs explicit representation

The HEC-RAS 2D geometry described in the previous Section (Section 4.3.2.2.1) was used as the basal model for calibration and validation. After model calibration and validation, ELJs were progressively drawn into the 2D geometry to simulate the three modelling scenarios described in Section 4.3.3.

The ELJs were drawn into HEC-RAS as informed by the modelling of Chapter 2. In particular, the Surface Area/2D Connection layer in HEC-RAS was used to draw hydraulic structures into a 2D mesh as follows: first, the ELJs centreline was drawn into the 2D mesh with a 0.5 m spacing; second, after drawing the centreline, the high flow ELJs were drawn with the bridge deck option in HEC-RAS as quadrangular pieces of 0.4 by 0.4 m with the same explicit geometrical representation used in Chapter 2. The space between the high flow ELJs and the channel bed, represented within the catchment, was between 1 and 0.2 m, depending on the local flow conditions observed in the field for different reaches within the catchment. The Hydraulic Property Table parameters were adjusted to provide the maximum stability to the model and corrected over multiple runs. The schematic representation of high flow ELJs in HEC-RAS is shown in Figure 4.5.

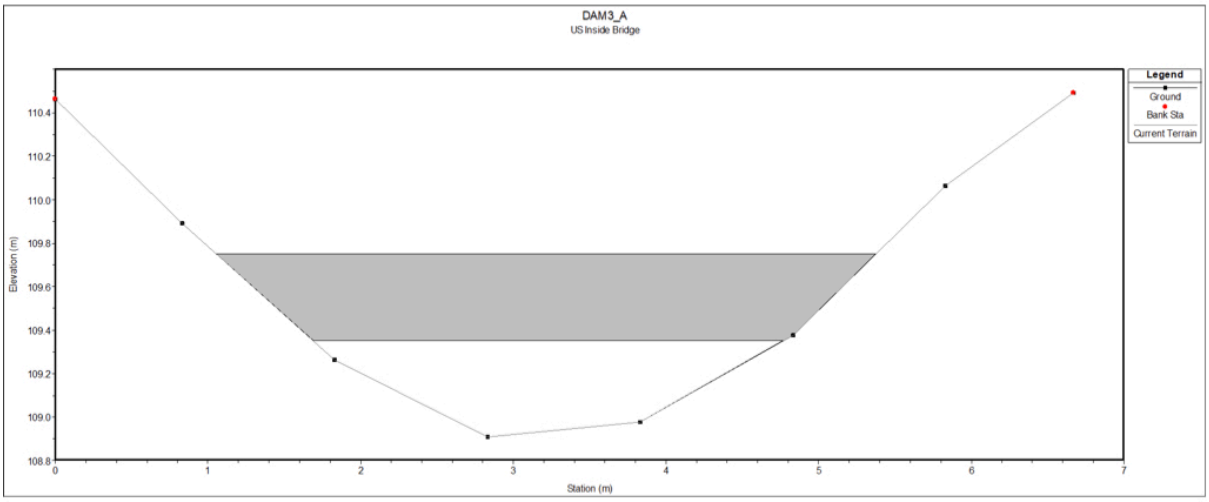


Figure 4.5 Schematic representation of an ELJs cross-section in HEC-RAS. *The y-axis represents the elevation (m) while the x-axis represents the distance from the left bank station (m). The grey box in the middle of the figure represents the ELJs drawn in the HEC-RAS geometry as a bridge deck. The grey line represents the underlying terrain while the red dots represent the left and right bank stations.*

4.3.3 Modelling alternative scenarios

To assess the influence of different numbers of ELJs at the catchment scale, different modelling scenarios were built and run to allow comparisons. Three different scenarios were simulated: a small deployment of ELJs with 20 ELJs built across the catchment, a medium deployment of ELJs with 40 ELJs deployed across the catchment and a large deployment scheme with 70 ELJs deployed across the catchment.

The first 20 ELJs (first scenario) were added to the HEC-RAS geometry in Reach 1 and 2 (Figure 4.2) with distances between the lower part of the ELJs and the channel bed of 0.2 and 0.4 m respectively. These ELJs were built into the geometry at the locations of the ELJs implemented for the Littlestock Brook NFM project. The additional 20 ELJs of the second scenario (40 ELJs) were added to Reach 3 and 4 with distances between the lower part of the ELJs and the channel bed of 0.2 m. Additionally, 30 ELJs were drawn into the HEC-RAS geometry for the third scenario. These ELJs were drawn as follows: 5 ELJs each for Reach 1-6 with distances of 0.2, 0.4, 0.2, 0.2, 0.8, 1 m respectively.

The location of the dams within the catchment was chosen on the basis of the land use of the catchment and on the deployment of ELJs during the Littlestock Brook NFM scheme. ELJs were firstly drawn in the HEC-RAS geometry at the locations where they were deployed during the NFM project. Finally, additional dams were deployed within the catchment where land use was either pasture, woodland or arable. Moreover, ELJs were deployed in locations far off infrastructure and private properties and placed at a minimum of 30 m distance apart.

Alternative flood events were modelled to determine the impacts of ELJs across events of different magnitude. In particular, the input flows for the channels of each sub-basins were selected from SWAT outputs on the basis of selected date with high flow events. In order to compare events with different magnitudes, flood events with a return period of 1 and 25 years were selected for the simulations.

4.3.4 Input dataset and model parametrisation

HEC-RAS 2D modelling necessitates input datasets for terrain, Manning's *n* and boundary conditions to initiate and run the model computations. The terrain model was generated from Light Detection and Ranging (LiDAR) data collected by the Environment Agency and OsMaps Terrain 5, both downloaded through the Digimap portal (https://digimap.edina.ac.uk/). The Environment Agency LiDAR data for the study area has a resolution of 1 meter, while OS Terrain 5 has a resolution of 5 meters. Since LiDAR data were not available for the entire catchment, a mosaic of the two datasets was obtained in ArcGIS v. 10.8 and then cut to the catchment contour. The lower part of the catchment has then a resolution of 1 meter while the upper part of the catchment has a resolution of 5 meters (Figure 4.6).

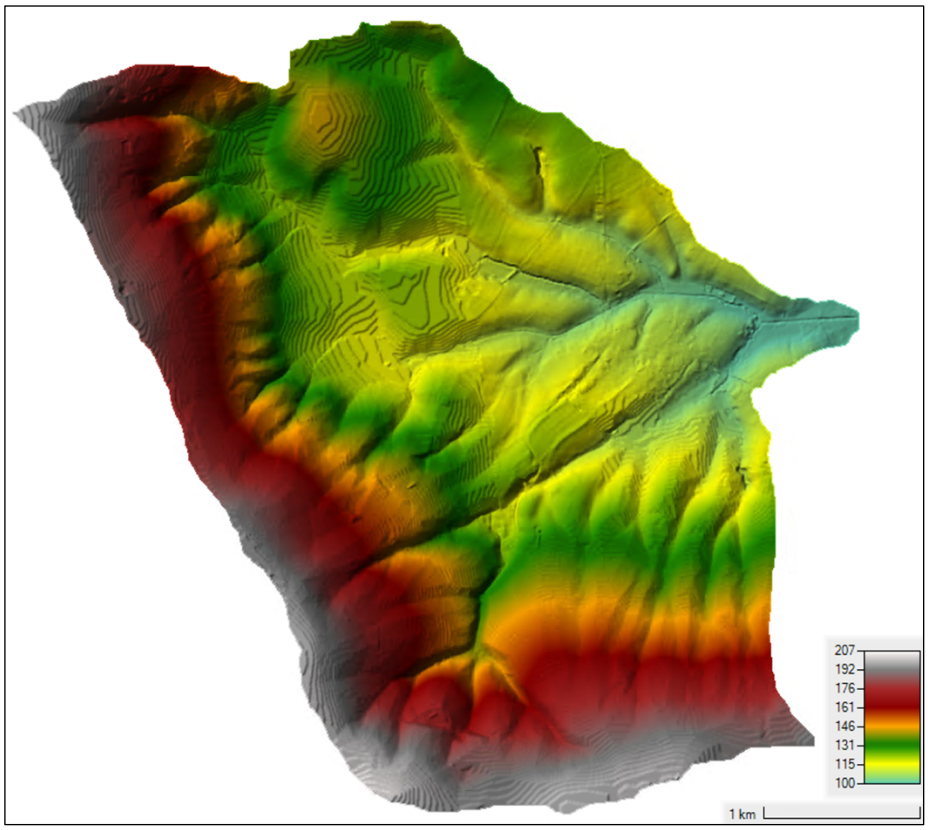


Figure 4.6 Terrain model for the Littlestock Brook catchment. *The legend portrays the colour scale for elevation within the catchment in meters.*

The terrain model was then imported into HEC-RAS via the RAS Mapper interface and the river network was modified manually to adjust channel sections and correct inconsistencies in the terrain (Figure 4.7).

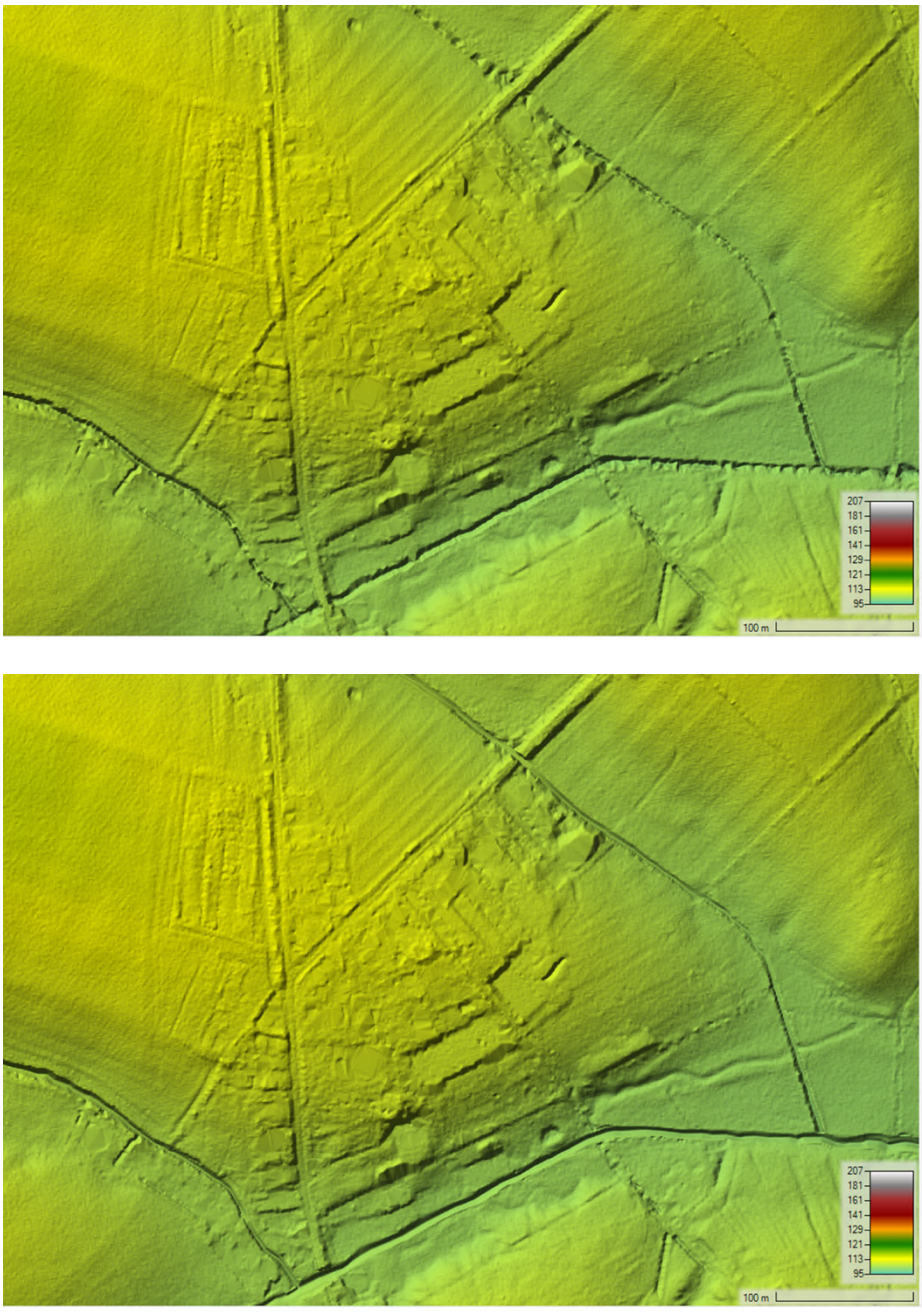


Figure 4.7 Channel corrections for the lower catchment *(Top: raw terrain model, bottom: corrected terrain model). The differences can be seen in the bottom left corner for one tributary channel and in bottom right corner for the main channel and for another tributary that flows towards north-west. The legend portrays the scale of colours for different elevations (meters, m).*

Manning's *n* values are required in HEC-RAS to define surface roughness of the channel and floodplain. In order to define the different land use classes and assign the given Manning's *n* for each of them, the CEH Land Cover (LC) map described in Chapter 3 was used. Since the aforementioned LC map has a resolution of 30 m, it was checked and corrected with web imagery data in RAS Mapper. Manning's roughness coefficients were firstly assigned using Chow's Open Channel Hydraulics (1959) and then calibrated as necessary to improve model performance. The Manning's n values are listed in Table 4.1.

Table 4.1 Manning's n roughness values assigned to different land cover types.

Land Cover	Manning's Roughness Coefficient *n*
Arable	0.05
Grassland	0.045
Woodland	0.09
Semi-urban	0.01
Channel	0.06

The boundary conditions for the 2D model were obtained from SWAT model runs. The sub-basin inlets for each tributary were obtained for the catchment by running SWAT hourly. The flow hydrographs were then used as the inlet boundary conditions for the 2D model, while Normal Depth was used as the outlet boundary condition.

The computational step of 2 seconds (s) was chosen in order to meet the Courant condition for model stability:

$$\Delta t > \Delta x/v \tag{4.4}$$

where Δt is the computational time step (s), Δx is the space step (m) and v is the speed of the flood wave (m/s) (Pappenberger, 2005). The speed of the flood wave was calculated from the field surveys described in Chapter 2 and extended to the other catchment reaches.

4.3.5 Model Calibration and Validation

The basal 2D geometry without ELJs was used to calibrate and validate the modelled outflow from the Littlestock Brook catchment with the flow simulations by SWAT.

This method for calibration was chosen because the calibration of the simulated peak flows with observation was not possible in this case. The peak flows of the Littlestock Brook are indeed influenced by the performance of the several different NFM measures that were installed within the catchment area before the start of this project. It was thus not possible to isolate the effects of these NFM measures on the peak flow in order to create a basal model without any NFM measure built in.

To calibrate the model, Manning's n values for the channel and floodplain were adjusted to minimise error and improve the overall model fit. Manning's n values for the channel in the 2D geometry, ranging from 0.03 to 0.075, were tested against the flow simulations used for calibration. A final value of 0.06 was applied into the 2D geometry for

the channel. Ten different flow hydrographs were chosen to calibrate and validate the model. In particular, ten flow hydrographs corresponding to the 1, 2, 5, 10, and 25-year event were extracted from the SWAT time series. Five flow hydrographs were used to calibrate the model and five hydrographs were used to validate the model.

The flow hydrographs used for calibration, validation and scenario analysis were extracted from the time series of the SWAT daily simulation of Chapter 3. In particular, the observations for the peak flows of the Evenlode catchment were ranked to define the return periods of different peak flow events. After defining the average magnitude of peak flows with a determinate return period, I selected the flow hydrographs that were closer to the simulated flow hydrographs by plotting observed values against simulated values. After selecting the peak flow events with closer fit, I extracted the flow hydrographs from SWAT hourly simulation.

For the three additional scenarios with several built in ELJs, Manning's n values were kept at the same calibrated values and modelled ELJs geometry were used to simulate the effects of ELJs. Previous studies have utilised similar techniques for calibrating hydraulic models with ELJs (Keys et al., 2018).

4.3.6 Simulation of experimental floods events

The simulation of the 1-year and 25-year flood events was used to better depict the effects of ELJs on reach velocities, water surface elevations, floodplain inundation extent and the reach's peak flow magnitude and travel time. Moreover, this was applied to the catchment scale with scenarios portraying a varying number of ELJs built in the catchment.

The 1-year and 25-year flood events were extracted from SWAT hourly simulation as explained in Section 4.3.5.

4.4 Results

4.4.1 Model Calibration and Validation

The performance and goodness of fit for the basal model without any ELJs are shown in Table 4.2 using the coefficients NSE and RSR. The model fit and performance are very good as defined by Moriasi et al. (2007) for both the calibration and validation periods and for different peak flow events with varying return periods.

Table 4.2 Statistics to evaluate model performance and goodness of fit.

Parameter	Calibration					Validation				
Return Period (years)	**1**	**2**	**5**	**10**	**25**	**1**	**2**	**5**	**10**	**25**
NSE	0.76	0.84	0.78	0.81	0.86	0.82	0.83	0.79	0.86	0.85
RSR	0.48	0.45	0.49	0.46	0.47	0.45	0.44	0.48	0.43	0.45

4.4.2 Simulation of experimental flood events

4.4.2.1 Floodplain Inundation

Across the catchment, inundation surface area increased with ELJs addition (Table 4.2). The maximum modelled inundation surface areas of the scenarios with the addition of ELJs were found to be higher than the inundation surface areas of the basal modelled scenario without any built-in ELJs. The inundated area average depth was also found to be slightly increasing for increasing numbers of ELJs deployed across the catchment for the 1-year recurrence event. Figure 4.8 exemplifies the case of the local floodplain inundation of one section of the reach in the scenario with ELJs against the basal scenario without ELJs.

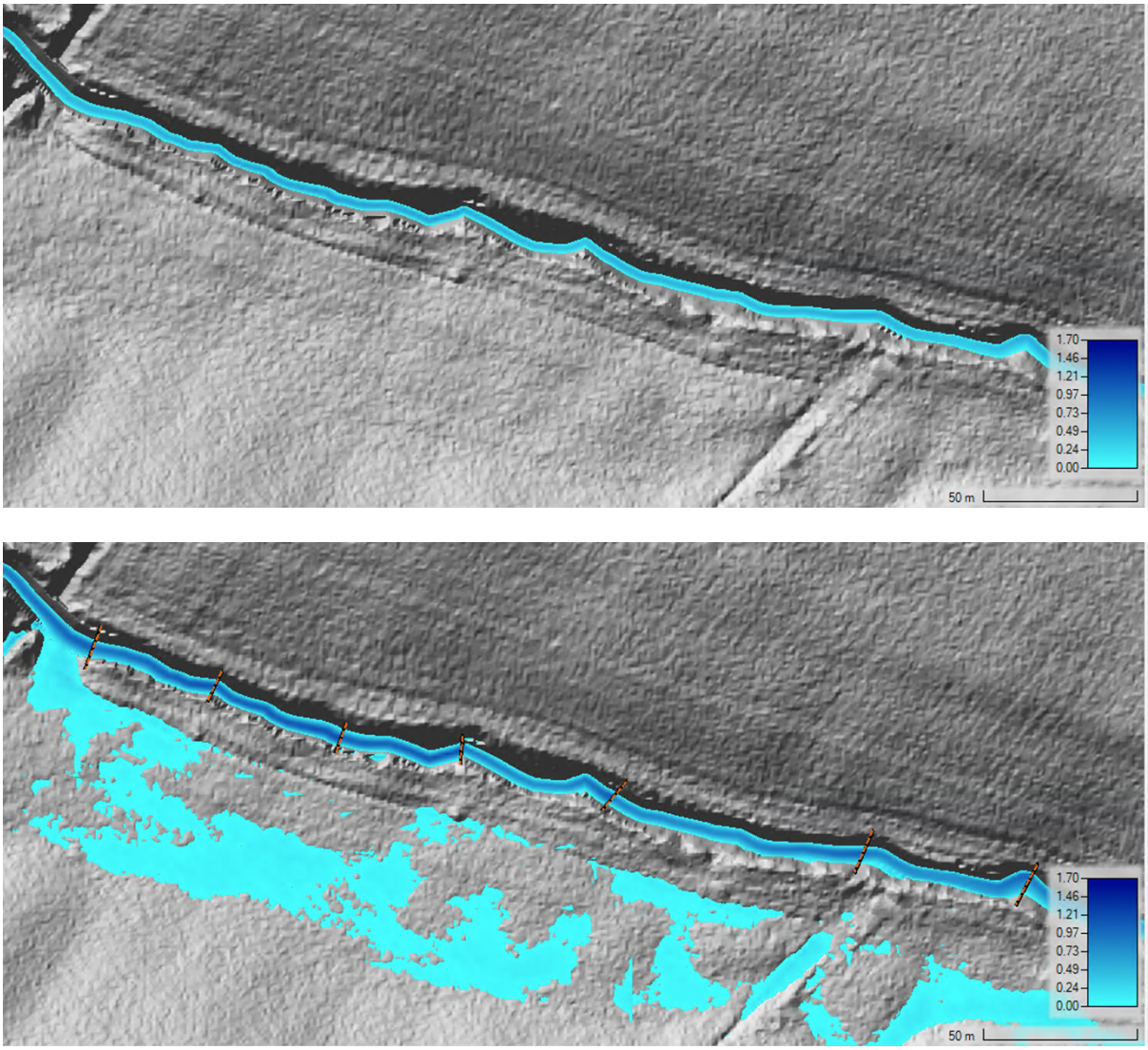


Figure 4.8 Comparison of two different scenarios for a 25-year flood event. *Top: reach without any ELJs. Bottom: reach with six ELJs built in sequence. The blue scale represents varying values of depth (m).*

Table 4.3 Average inundation depth and maximum inundation area for the 1-year flood event and the 25-year flood event.

Flood return period	Variable	NO ELJs	20 ELJs	40 ELJs	70 ELJs
1-year	Max Inundation Surface Area (m^2)	128	572	750	1143
25-years	Max Inundation Surface Area (m^2)	651	1151	1312	1789
1-year	Average Inundation Depth (cm)	11.0	10.8	13.5	16.7
25-years	Average Inundation Depth (cm)	18.6	19.1	18.9	20.1

4.4.2.2 Flow Velocities

Velocity dynamics for reaches with ELJs also varied between simulation with ELJs and without ELJs. For a 1-year flood event, a decrease in flow velocities of 10-15% was observed at every active ELJ. The decrease in flow velocities was also propagated upstream and downstream of the ELJ. Approximately a 30 meters reach was affected by the implementation of each ELJ as shown in Figures 4.9 and 4.10. In particular, a deceleration of the waterflow starting circa 10-15 meters upstream of each active ELJ can be observed, with a maximum deceleration observed at each ELJ's location. For the simulated scenarios a decrease in waterflow velocities can also be observed after each ELJs.

Most of the high flow ELJs shown in Figure 4.8 and 4.9 were activated at the maximum peak flow, with the local water surface elevation reaching the minimum "deck" elevation of each ELJ, as reports in Table 4.3. ELJ 1 and ELJ 2, were only barely activated by the water surface elevation at the time of the peak flow (Table 4.3).

Table 4.4 ELJs percentage activation for the different flood events represented in figure 4.8 and 4.9. *Percentage activation was calculated in terms of the stage against the ELJ with 100% being the maximum stage before the ELJ is overtopped and 0% being the stage for which waterflow pass under the ELJ.*

ELJ Number	1-year flood percentage activation	25-years flood percentage activation
ELJ 1	1-5%	70-75%
ELJ 2	1- 5%	70-75%
ELJ 3	30-35%	Overtopped
ELJ 4	1-5%	55-60%
ELJ 5	30-35%	Overtopped
ELJ 6	25-30%	75-80%

ELJs 1-4 were distanced 40 meters apart and this distance seem to be sufficient to provide a continuum in the velocity decrease. ELJs 4-5 and 5-6, on the other hand, are further apart (50 and 85 meters) and these distances, together with a steeper terrain, interrupted the continuum in the decrease of water velocities between ELJs. This suggest that for higher flow velocities the effects of ELJs may be limited to shorter reaches.

Flow velocities for the 25-year flood event were almost equally affected by the presence of ELJs in the simulation. However, the decrease in flow velocity is more constant throughout the reach and the influence of each ELJ propagates upstream with a marked 20-30% decrease in flow velocities for each section upstream of the ELJ (Figure 4.11).

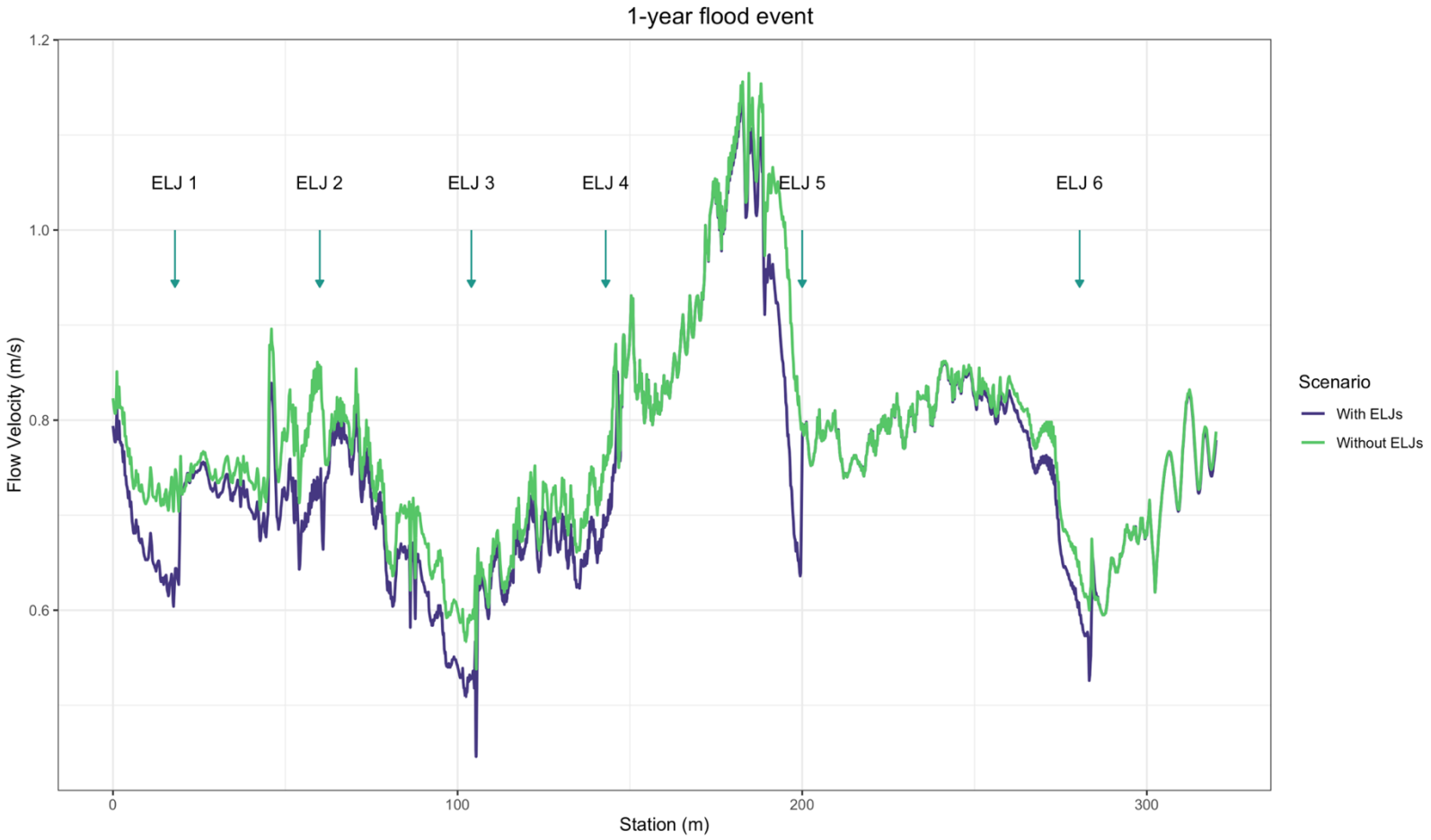


Figure 4.9 Comparison of the simulated peak flow velocities through a 300 m reach for a 1-year flood event. *The different colours refer to two different scenarios: one representing the basal model without any ELJ, the second one with six ELJs deployed in sequence into the reach.*

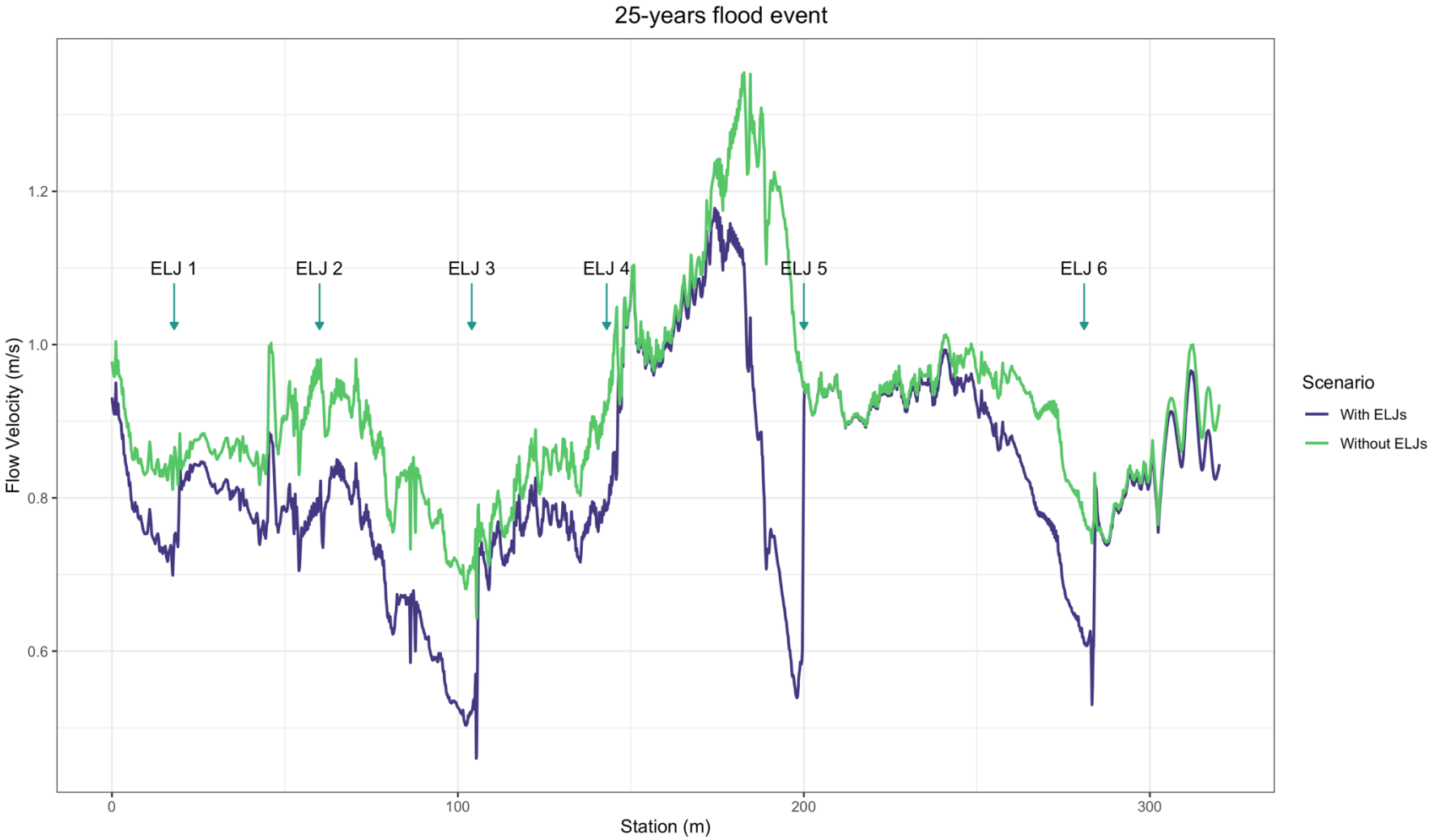


Figure 4.10 Comparison of the simulated peak flow velocities through a 300 m reach for a 25-year flood event. *The different colours refer to two different scenarios: one representing the basal model without any ELJ, the second one with six ELJs deployed in sequence into the reach.*

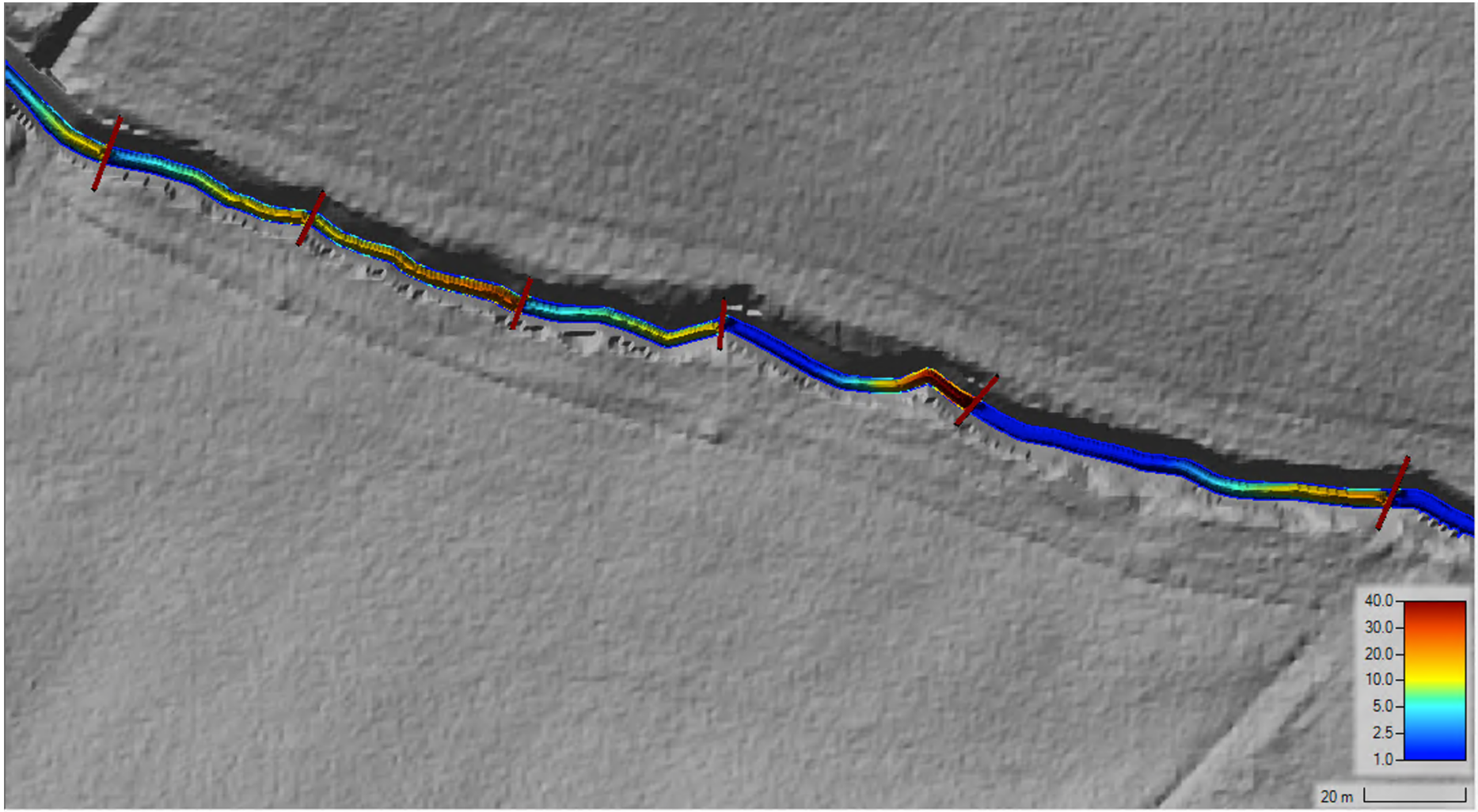


Figure 4.11 Percentage reduction in flow velocity between the two simulated scenarios for a 25-year return period flood event*. ELJs are represented by maroon lines perpendicular to the channel direction (ELJs dimensions were exaggerated for better visualisation).*

4.4.2.3 Effects on the catchment peak flow

My results indicate that the effects of the addition of ELJs at the catchment scale depend on the simulated flood event considered. The effects of the ELJs deployment (Figure 4.12) for the 25-year flood are relatively marginal: the deployment of 40 to 70 ELJs across the catchment delayed the peak flow by 4-5 hours and caused a decrease in the maximum flow discharge of approximately 5 %. The addition of 20 ELJs across the 16.3 km^2 catchment, delayed the simulated peak flow by 3 hours while the maximum peak discharge remained approximately equal to one of the scenarios without ELJs. For the 1-year flood, the deployment of ELJs was more influential with a decrease in the peak flow maximum discharge of 10-20% depending on the number of ELJs deployed and a delay in the timing of the peak of 4 hours on average (Figure 4.13).

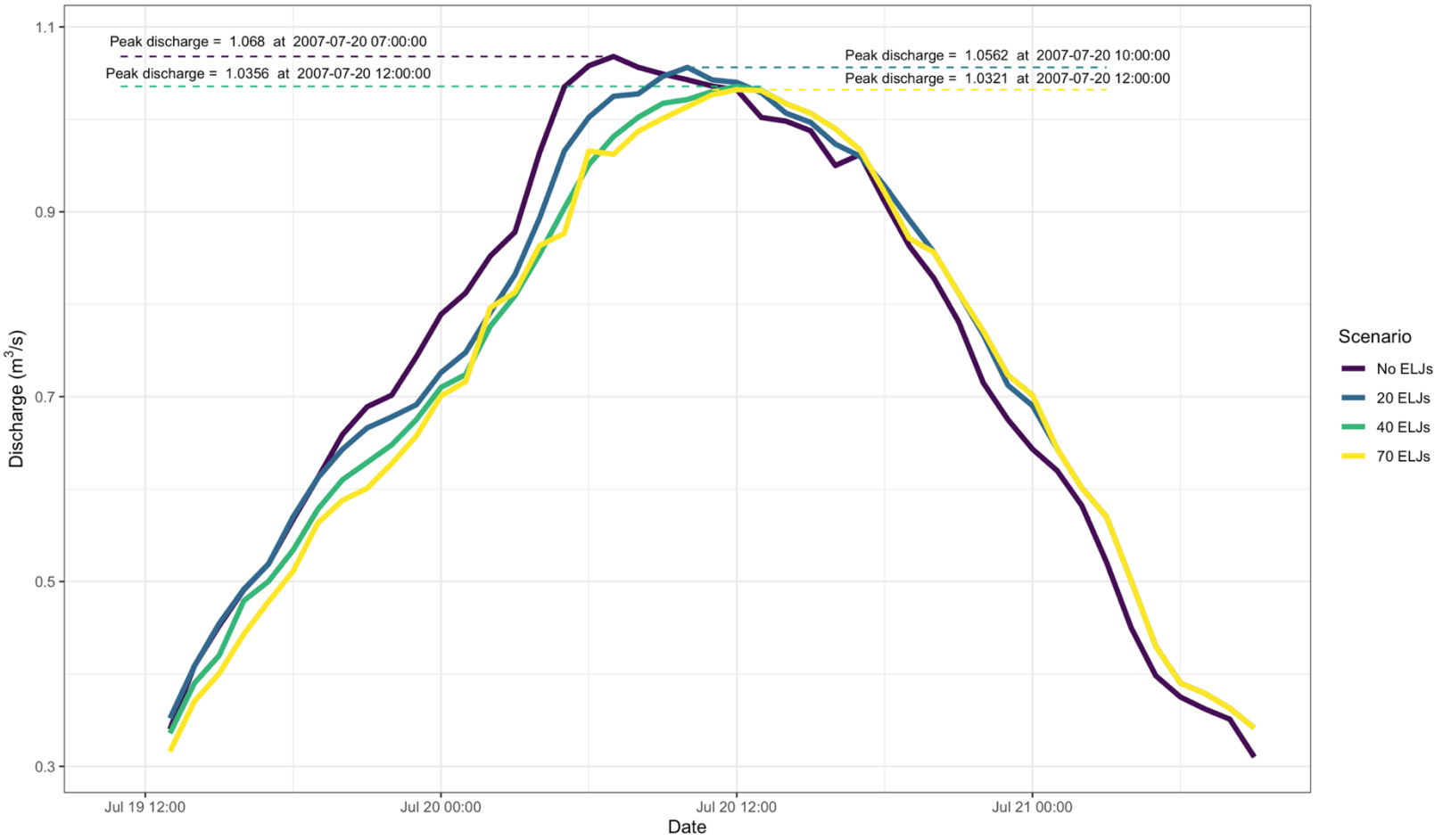


Figure 4.12 Peak flow hydrographs for a 25-year return period flood event for the four different scenarios. *The first scenario represents the behaviour of the catchment without any ELJ deployed (represented in dark purple) while the other three scenarios (in blue, green and yellow) represent different numbers of ELJs deployed throughout the catchment.*

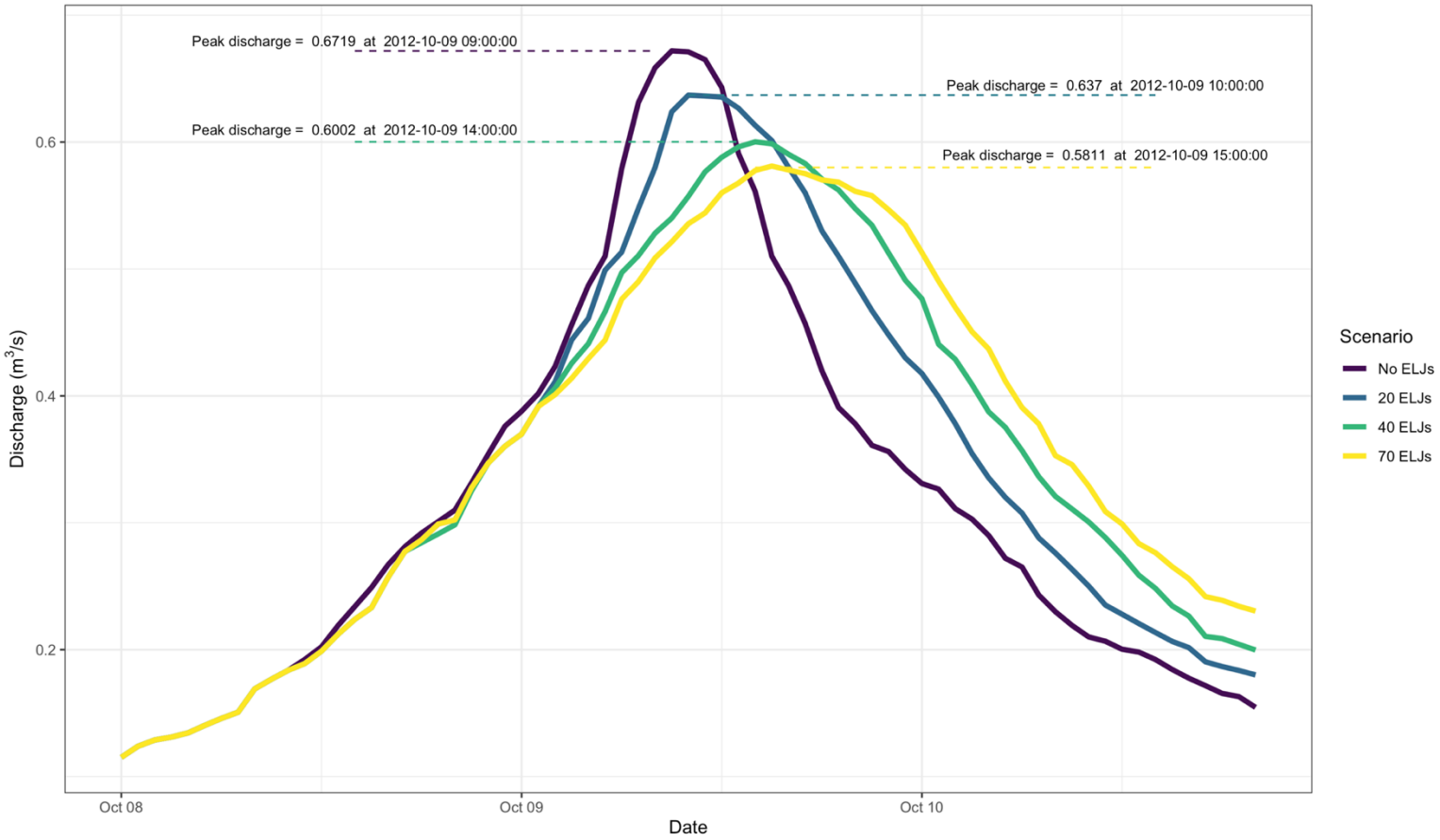


Figure 4.13 Peak flow hydrographs for the 1-year return period flood event for four different scenarios. *The first scenario represents the behaviour of the catchment without any ELJ deployed (represented in dark purple) while the other three scenarios (in blue, green and yellow) represent different numbers of ELJs deployed throughout the catchment.*

4.5 Discussion

The overarching aim of this research was to build a hydrological-hydraulic modelling cascade to better depict the effects of high water ELJs on flood dynamics and floodplain inundation at the reach and at the catchment scale (16.3 km^2). Specifically, the aims were to estimate the extent of ELJs-induced floodplain inundation and their effects on channel processes, to simulate the effects of floodplain water storage on the peak flow magnitude and its timing at the catchment scale, and to simulate and compare scenarios characterised by varying number of ELJs.

As anticipated, high water ELJs proved to be highly effective in reducing local in-reach flow velocities for flooding events with varying return periods. Moreover, a higher reduction in longitudinal velocity was prompted by ELJs for higher flows. This was due to the engagement of a larger portion of ELJ due to the raising stage (Table 4.3). These findings are in line with what has been observed in previous research (Gippel et al., 1996; Turcotte et al., 2016); high water ELJs proved to be effective also in increasing floodplain inundation extent, duration and depth for the study reach, supporting the observations collected by Keys et al. (2018) for Atlantic headwater streams and the remarks in Wohl (2014).

The results of the modelling outlined above for a rural, highly modified catchment in South East England is particularly important for future NFM project design in similar catchments. This is because the integration of different NFM measures could further enhance some processes initiated by ELJs. In particular, floodplain inundation, together with the temporal water storage provided, may be enhanced by building temporary retention structures such as corner bunds. Channel scrapes are also an NFM technique of simple development that could increase the extent and frequency of floodplain inundation by aiding channel-floodplain reconnection. The findings of my research are also consistent with general notions outlined in previous studies. In particular, as reported in Gippel et al. (1995) and Wohl et al. (2017), the

traditionally used river snagging and clearing, can increase channel conveyances locally, by favouring faster and shallower flows, while at the same time interrupting channel-floodplain connection and decreasing local flooding. This may in turn increase water delivery downstream where major value land or infrastructure are present. Keys et al (2018) suggested that the parts of the channel that have higher velocities and shallow depths are the ones with the highest floodplain inundation potential. In addition, Curran and Wohl (Curran and Wohl, 2003) suggested that steeper channels had the highest flooding potential. My study catchment and its river network, however, suggested that the floodplain inundation potential is mainly due to channel and floodplain geometry more than the longitudinal velocity (i.e. velocity head) and the water depth in the reach (Darby, 1999; Bates and De Roo, 2000; Curran and Hession, 2013).

My results also indicated that the number of ELJs deployed into the catchment was highly influential for many of the variables analysed and for flood characteristics. In particular, water flow velocities were impacted by each ELJ for a stretch of the channel of approximately 30 meters in length. Placing consecutive ELJs, at a 30-40 meters distance, could thus provide almost a continuum in the effects of ELJs on flow velocities through the reach. Moreover, the floodplain inundation extent and depth were related to the number of ELJ simulated in the river network. This is probably due to the propagation of their effects on longer stretches of the channel, thus increasing flooding potential for an increased proportion of the channel network. However, the number of ELJs does not seem to have a big effect on the peak discharge at the catchment outlet in case of extreme flood events (return interval of 25 years or higher). These findings are in line with what has been observed by other studies (Keys et al., 2018; Hankin et al., 2019).

For the study catchment, the de-synchronisation of peak flows coming from tributary catchments is not impacting the peak flow while the widespread deployment of more ELJs into the catchment seems to be more important for moderate flood events.

At the catchment scale, the 1-year return period flood was highly impacted by ELJs presence with a reduction in peak flow discharge. For more extreme events with return periods of 25 years or higher, the peak flow discharge magnitude had only a minor shift even for the scenario with the highest numbers of ELJs. These findings reflect what has been observed in literature for different studies and different types of large wood (Dadson et al., 2017; Dixon et al., 2016; Keys et al., 2018; Metcalfe et al., 2018). However, I must here note that the findings are relative to a specific type of ELJ with a certain design and ELJ that occupy a bigger portion of the cross-sectional area of the channel may be more effective in reducing extreme flood events at the reach scale.

4.6 Conclusions

In this chapter I used a SWAT/HEC-RAS modelling cascade to evaluate the effects of the installation of high water ELJs on channel hydraulics and floodplain inundation at the reach scale, and on peak flow magnitude and timing at the catchment outlet. The results indicate that ELJs deployment in a rural catchment with a high responsive regime is an effective NFM technique to target flood dynamics and characteristics. In particular, ELJs proved to be effective in reducing streamflow velocities and increasing floodplain inundation extent.

Further studies will be fundamental to understand how the positioning of ELJs through a catchment is important in determining their effectiveness. Moreover, future research should focus on the effects of the positioning of high water ELJs in the channel cross-section to maximise their effectiveness for a range of flood events. Lastly, it is crucial to research how different NFM measures might be integrated together to enhance their effectiveness on flood risk reductions at the catchment level.

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Chapter 5: The effects of NFM on ecosystem status: are high flow engineered log jams beneficial to the structure and functioning of aquatic riverine ecosystem?

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5.1 Introduction

Naturally occurring Large Wood (LW) and artificially deployed Engineered Log Jams (ELJs) represent a significant source of hydraulic roughness that can alter channel flow dynamics and patterns (Montgomery et al., 2003; Mutz, 2000; Pilotto et al., 2016; Wohl, 2017). As highlighted in Chapter 4, large wood presence and re-introduction as ELJ, leads to lower in-reach flow velocities and increases the heterogeneity in water depths and velocities across reaches (Pilotto et al., 2014). Moreover, large wood has the ability to influence the physical structure of watercourses as it increases streambed heterogeneity by creating scour pools and riffles (Abbe and Montgomery, 1996).

The increased spatial heterogeneity of physical conditions is often followed by a higher diversity of biotic communities. In particular, the studies by Beisel et al. (2000) and Pilotto et al. (2014) investigating, respectively, the impacts of the spatial heterogeneity of a river bottom and of large wood on invertebrate communities, showed an increase in alpha-diversity. Moreover, the meta-analysis by Miller et al. (2010), looking at the macroinvertebrate responses to in-stream restoration of physical habitat heterogeneity, showed a positive effect on macroinvertebrate richness. The LW-induced spatial heterogeneity of physical conditions may also result in changes in ecosystem functioning and services, such as primary production, sediment retention and organic matter distribution (Pilotto et al., 2016).

Most studies on the effects of the presence of large wood on macroinvertebrate communities have been carried out in mountainous streams close to natural conditions, mainly in North America. In comparison, only a few studies have been conducted in European catchments and for lowland reaches (Gerhard and Reich, 2000; Hoffmann and Hering, 2000; Pilotto et al., 2014; Vaz et al., 2014). Moreover, most studies conducted to date have looked at large wood in streams with a good ecological state, while little has been done on highly modified rural catchments and/or for streams that were exposed to high modifications.

The purpose of different projects, and the range of management goals they aim to achieve, define the design chosen for large wood re-introduction and its difference to natural large wood occurrences. For example, large wood, installed as part of habitat restoration projects, try to resemble more closely the characteristics of naturally occurring large wood (Figure 5.1 e, f, g, h), namely its heterogeneity, seasonal variability and porosity. In comparison, Engineered Log Jams and leaky dams installed as part of NFM projects are built to maximize the effects on channel hydraulics, on waterflow velocities and to retain water in the channel incision, while little consideration is given to habitat restoration and other environmental benefits. NFM projects also tend to prioritize larger diameter timbers (Turcotte et al., 2016), dense and simple structures (Manners et al., 2007; Wilcox et al., 2006), usually oriented perpendicularly to the flow. These ELJs are designed to span the entire channel causing a step in the waterflow cross-section at all flow conditions (Figure 5.1 c), or only at high water conditions while the basal waterflow is not affected (Figure 5.1 a, b, d).

Despite the large body of literature showing the effect of large wood on freshwater macroinvertebrate communities (Appendix A), most studies to date have regarded either sampling of naturally occurring LW or re-introduced LW for restoration projects. These projects have often involved different quantification methodologies and sampling approaches, making a comparison between studies challenging. In addition, the assessment of the ecological status of rivers and streams has traditionally relied almost exclusively on measurements of ecosystem structure (e.g. invertebrate community composition, water quality) and neglected the assessment of ecosystem processes (e.g. primary production, organic-matter decomposition) (Chauvet et al., 2016). This focus on the ecosystem structure can sometimes create situations for which the outcome of a restoration or NFM project, or the assessment of human impacts on streams and rivers, are only partially understood.

Moreover, many ecosystem processes are not necessarily linked to ecosystem structure and there is a general lack of substitutes for direct measurements of ecosystem services (Bunn et al., 1999). Organic matter decomposition is a function of riverine ecosystems that has been shown to be affected by large wood. Large wood is indeed responsible for an increase in sediment storage and organic matter retention as a result of the increase in hydraulic roughness and change in channel morphology (e.g. creation of pools). Organic matter and nutrient storage at LW sites are the result of the increased flow resistance due to the presence of the LW obstruction. This accumulation of organic matter and nutrients at large wood sites may, in turn, decrease the spiralling lengths thus increasing the availability of energy and nutrients to micro- and macro-consumers, hence potentially enhancing secondary production (Smock et al., 1989). The importance of large wood and the so-called debris dams as sites for organic matter storage in streams was first quantified by Bilby and Likens (1980) who quantified that 75% of total organic matter in a first-order stream was stored in dams or behind them. These quantitative data were further confirmed by Smock et al. (1989) who found that two low-gradient headwater streams were storing 21% and 85% of the organic matter at debris dams' sites along the channel. Moreover, the storage of organic matter and nutrients at woody debris sites has been shown to create 'hotspots' that were largely responsible for the reach-scale ecosystem respiration, a measure of stream metabolism and ecosystem functioning (Blaen et al., 2018).

Given the peculiar design of ELJs in many NFM projects (Figure 5.1), it is of primary importance to understand if the macroinvertebrate communities and ecosystem functioning are affected by the particular type of ELJs deployed in the context of NFM projects. To date, only a few studies have considered the re-introduction of large wood and Engineered Log Jams and their influence on invertebrate communities and on ecosystem functioning. Moreover, most of the previous studies on the effects of LW on macroinvertebrates were mainly focused on the freshwater ecosystems in mountainous settings, characterized by steep slopes and coarse

substrates, while only a few studies have been conducted for European and UK lowland reaches and even fewer have considered the deployment of ELJs for Natural Flood Management purposes.

Given the peculiarity of ELJs structures and their recent deployment in highly modified rural catchments with designs that favour hydraulic resistance more than the suitability for ecological co-benefits, it becomes of primary importance to understand the ecological effects of high water ELJs, installed as part of NFM schemes, on the structure and functioning of riverine ecosystems.



Figure 5.1 Differences in LW and ELJ structures. *Different structures for large wood installed as part of NFM schemes (top) and natural occurring large woody debris accumulating in different streams and rivers (bottom). Pictures a, b, d are from the Littlestock Brook NFM scheme; c shows an engineered log jam installed as part of the Pickering NFM project; e, f, h represents large woody debris in North America and Northern Europe (Gurnell et al., 2019; Wohl, 2014); g shows natural large woody debris accumulation in the Evenlode catchment (courtesy of David Gasca-Tucker).*

5.2 Aims and Objectives

Based on the findings of my literature review and the existing knowledge gaps, I addressed the following objectives in order to assess the ecological benefits of high water ELJs into the context of NFM projects.

1. Understand if high water ELJs, built under an NFM scheme, can increase the diversity of the benthic macroinvertebrate communities.
2. Assess if the morphological changes induced by high water ELJs can alter the pattern of accumulation of organic matter and affect its rate of decomposition.
3. Understand how high water ELJs can provide additional environmental co-benefits and how these should be specifically targeted in the project design phase.

5.3 Materials and Methods

5.3.1 Study Area

The Evenlode is a headwater catchment (431 km^2) of the River Thames flowing from Moreton-in-Marsh to Eynsham in South England between the Gloucestershire and Oxfordshire regions. The Evenlode is a flashy clay catchment characterized by a hydrologically responsive flow regime. The catchment has a legacy, like many other agricultural catchments in the UK, of land drainage operations and milling which resulted in over-deepening and straightening of streams and channels and in a general disconnection from the floodplain. The flashy clay soils have been under-drained to enhance agricultural productivity. This may have increased the catchment responsiveness to precipitation events (Old et al., 2019).

The legacy of human modification and land-use, together with the soil characteristics, make the Evenlode a catchment prone to flooding. The pilot NFM scheme implemented by the

Evenlode Catchment Partnership, together with multiple local stakeholders and Thames Water, regards a sub-catchment of the Evenlode: the Littlestock Brook. The Littlestock Brook is a small rural catchment (16.3 km^2) flowing through the villages of Shipton- and Milton-Under-Wychwood. The catchment has been affected in the past by various flooding events, the most important of which happened in 2007.

The Littlestock Brook is characterized by four headwater arms joining together in Shipton-Under-Wychwood before the brook merges with the Evenlode flowing from the North-West. The pilot NFM scheme has been implemented in the Littlestock Brook catchment and 42 ELJs were installed in three of the four headwater arms of the Littlestock Brook.

The sampling was carried out on the three streams where ELJs have been implemented in sequence and in two different streams used as control sites. One of the control streams is the fourth arm of the Littlestock Brook flowing through Shipton-Under-Wychwood (Figure 5.2, bottom right sampling site represented by circular symbols), which has not been impacted by NFM measures. The other control stream that was sampled is located in another sub-catchment of the Evenlode, the Westcote catchment (Figure 5.2, top left sampling site represented by circular symbols). The Westcote catchment, which adjoins the Littlestock Brook catchment on its north boundary, was chosen because of its proximity, similar land-use and catchment characteristics. The details and justification of the sampling and data analysis methodologies are further discussed in sections 5.3.2, 5.3.3 and 5.3.4.

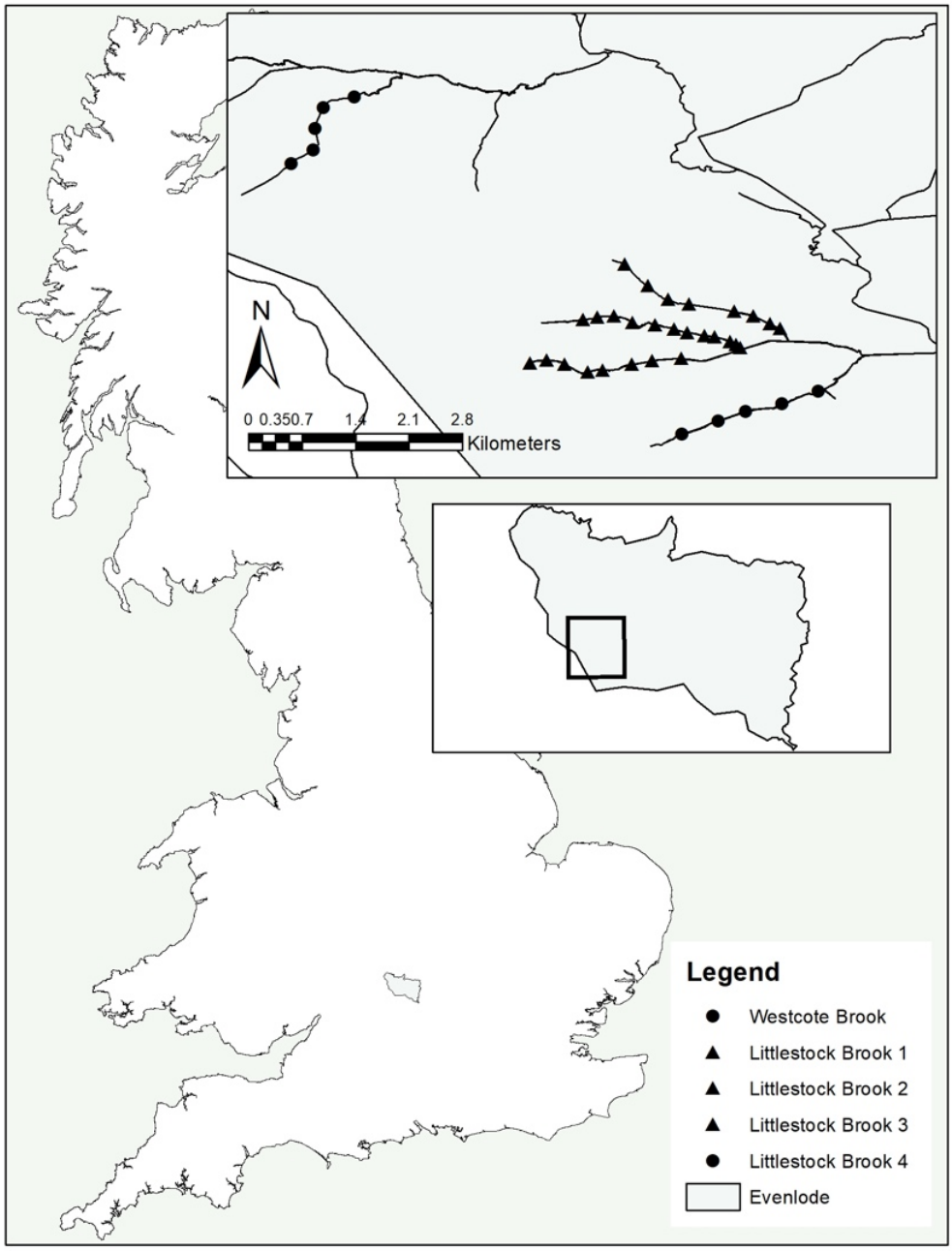


Figure 5.2 Study area and sampling sites. *The top right quadrant shows the five streams that were sampled and the approximate locations of each sampling site. Triangular sampling sites are the ones in reaches where ELJs were deployed. The control sites, represented by circular shapes, were sampled on control reaches not modified during the Littlestock Brook NFM project. The naming of the surveyed reaches in the legend is from north to south. The location of the sites within the Evenlode catchment is represented with the quadrant in the figure below. The position of the Evenlode in South England is showed in the underlying figure.*

5.3.2 Experimental Design

Benthic macroinvertebrate communities were sampled to assess the impact of the new engineered log jams on the in-stream diversity. Organic matter decomposition rates were measured using the cotton-strip assay developed by Tiegs et al. (2007; 2013) to assess the effect of the installation of ELJs on organic matter decomposition rates.

A Control-Impact design (space for time substitution) was chosen for the data collection as a BACI (before after control impact) design was not possible due to the inconsistency of the data collected before the implementation of the NFM trial project.

Macroinvertebrate communities' samples and cotton-strips were collected and deployed at forty sites in five different reaches (Figure 5.2). A total of 40 samples of macroinvertebrate were collected and a total of 136 cotton-strips were retrieved from the river after the incubation period. For the scope of this analysis the four different branches, from which the Littlestock Brook originates, were named with subsequent numbers from north to south (Figure 5.2). Westcote Brook is the stream on the top-left corner of Figure 5.2. The shapes of points in Figure 5.2 represents the type of the site. For clarity and to simplify the figure, the upstream control sites (see description below) on Littlestock Brook 1-3 were portrayed with the same triangular shapes as the impacted and downstream sites since the aim of the figure was to demonstrate the difference among the five reaches and their location.

The choice of samples sites was carried out as follows:

1. On impacted streams the reach was sampled before and after each ELJ of the sequence, thus providing 19 impacted sampling locations. One upstream control was sampled on each one of the three impacted streams. Five control sites were sampled on each control stream.

2. Three downstream sites were chosen on each impacted channel. The three sites were positioned at circa 25, 50 and 75 meters downstream of the last ELJ of the sequence on each impacted stream. The site 75 meters downstream of the last ELJ of the sequence of the reach named Littlestock Brook 2 (Figure 5.2) was not accessible, thus only 2 downstream sites were sampled for this channel. In total 8 downstream samples were analysed in the lab.

5.3.3 Benthic macroinvertebrates

5.3.3.1 Field Sampling procedures

Benthic macroinvertebrates were sampled at all sites shown in Figure 5.2 in two subsequent weeks in late spring (June 2019). Benthic communities of macroinvertebrates were sampled using a 500 μm mesh net Surber Sampler using standard field procedures (Dobson, 2013; Murray-Bligh et al., 1997). The usage of a Surber Sampler instead of a common kick-net allowed for a semi-quantitative estimate of the macroinvertebrate richness at the family level and their abundance by combining the area defined by the frame of the Surber Sampler (425 cm^2) and a fixed sampling effort (3 minutes).

The sampling procedure used for this study involved a first visual assessment of the different mesohabitats along the reach. The fixed 3-minute sampling effort was then divided between the identified mesohabitats by their percentage occurrence following Barbour's protocol (Barbour et al., 2001). After planning the sampling according to the different mesohabitats present at each site location, the sampling involved the dislodging of the benthic macroinvertebrates by scraping the channel bed with the heel of the foot inside the frame of the Surber Sampler. The "kick" sampling with the Surber Sampler was always realised facing upstream, thus allowing the dislodged macroinvertebrates to be collected in the Surber Sampler net. After finishing the time allotted for each site sampling (3 minutes in total), the

macroinvertebrates caught in the net were sorted, placed in plastic pots and preserved in 70% ethanol to be taken to the lab. In the lab, each sample was processed by removing the substrate, and then macroinvertebrates were sorted and identified to family level by using a stereomicroscope. During the sampling, total dissolved solutes, electrical conductivity, pH, temperature and dissolved oxygen were measured. The level of the water was recorded at a representative location for each site.

5.3.3.2 Diversity of macroinvertebrate communities

The diversity of the macroinvertebrate communities and the effects of ELJs on local diversity were evaluated by using the Shannon Index for diversity (Shannon, 1948). Shannon's H was chosen as it evaluates both the abundance and diversity of the species composing the sampled community. Moreover, Shannon H is often used for macroinvertebrate diversity in riverine sites as it has been shown to better discriminate between sites when compared to other diversity and richness indexes (Wilsey et al., 2005).

The Shannon Index characterizes diversity with an equation based on the number of species present (species richness) and the number of organisms per species (species evenness):

$$H = -\sum_i p_i \, ln(p_i)$$

(5.1)

where p_i is the proportion of individuals of the ith species in a whole community, defined as:

$$p_i = n/N$$

(5.2)

where *n* is the number of individuals of the ith species, or other taxonomic unit, and N is the total number of individuals in the community.

5.3.3.3 Data Analysis

Sampling sites were categorised in three groups for analysis: impacted sites (ELJs), control sites (Control) and downstream sites (DS). The control sites were the sampling sites above the intervention zone in reaches were ELJs were implemented and the sampling sites of the two control, non-impacted, reaches (Westcote Brook and Littlestock Brook 4).

The Shannon Index for each sampling sites was calculated using the package *vegan* in R (R Core Team, 2017). Differences among the different sampling sites groups were calculated using one-way ANOVA with a post-hoc Tukey test in R (R Core Team, 2017).

In order to identify similarities and dissimilarities between different sampling sites, a hierarchical clustering analysis was carried out in R. The hierarchical clustering was produced using agglomerative clustering algorithms. Agglomerative clustering algorithms, as opposed to divisive clustering algorithms, evaluate distances between the "leaves" of the dendrogram starting from single-element clusters (bottom-up approach) and thus tend to favour the formation of small clusters. The hierarchical clustering was carried out using R's *stats* package and the *factoextra* package for the enhanced graphical visualisation of the dendrogram.

5.3.4 Organic matter decomposition

Organic-matter decomposition is a function of riverine ecosystems that has been shown to be affected by large wood (Smock et al., 1989). In particular, in-stream large wood plays a role into retaining and trapping drifting organic matter, thus making it available for benthic detritivorous invertebrates (Scealy et al., 2007). The retention of organic matter induced by large wood might thus have an effect on macroinvertebrate community composition.

In order to test if organic-matter decomposition was influenced by the installation of high water ELJs, a cotton strip assay was used to measure organic matter decomposition for ELJs and control sites (Latter and Howson 1977, Harrison et al., 1988). The cotton-strip assay

(Tiegs et al., 2013) makes use of a standardized cotton material to measure decomposition as a proxy for plant litter since constituted for the 95% by cellulose. The usage of a standardised material in substitution for plant litter allows to bypass some of the constraints of the typical litter bags in terms of litter availability and diversity.

Cotton strips (20 x 85 mm) were prepared from bolts of unprimed 12-ounce, heavy-weight cotton fabric. Nine cotton strips were prepared for each sampling site. Three strips were placed in the stream by tying them to a wooden post placed in the bed substrate, while the remaining six cotton strips were enclosed in litter bags with two different mesh sizes and tied in place at the same location. Three cotton strips were enclosed into a coarse mesh size (1 cm) nylon bag while the last three cotton strips were enclosed into a fine mesh nylon bag (500 μm). The choice to use bags of different mesh sizes was intended to isolate the microbial decomposition of organic matter by excluding most of the invertebrates with the fine mesh bag.

The samples were incubated in the stream for a 15 days period in Spring 2019 and then retrieved, washed in 70% ethanol and air dried before measuring their tensile strength.

The incubation period was intended to yield approximately 50% of tensile-strength loss, a level of decay for which the test is believed to be the most sensitive as suggested by Tiegs et al. (2013).

The tensile strength was measured by placing the ends of the cotton strips in the grips of a tensiometer. The max tensile strength was determined by pulling the strips ends apart at a rate of 2 cm/min until the strip tore as suggested by Tiegs et al (2013).

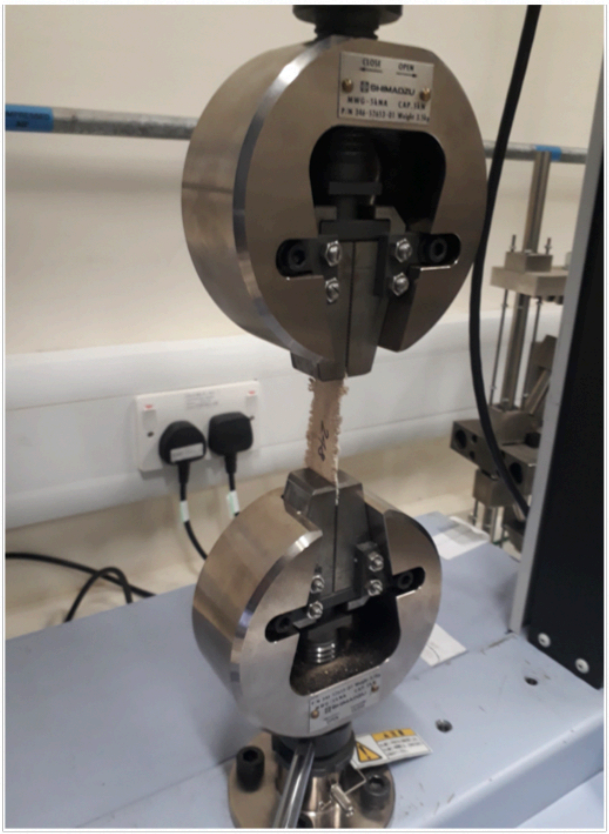


Figure 5.3 Tensile strength measurement performed on a cotton strip. *Cotton strips are placed between the two opposing grips of the tensiometer and pulled apart with a constant work per unit of time.*

The recorded tensile strength was then used to calculate tensile strength loss as the reduction in tensile strength from the day of the start of incubation and expressed as a percentage with Equation 5.3.

$$Tensile\ Loss = \left[1 - \left(\frac{Tensile\ Strength_{treatment\ strips}}{Tensile\ Strength_{reference\ strips}}\right)\right] * 100/Incubation\ Time \quad (5.3)$$

where $Tensile\ Strength_{treatment\ strips}$ is the maximum tensile strength recorded for each of the strips incubated in the field, and $Tensile\ Strength_{reference\ strips}$ is the mean of the maximum tensile strength of 20 strips that were not incubated in the field, but were cleaned in ethanol and dried before testing them as controls.

5.4 Results

5.4.1 Benthic Macroinvertebrate communities

Results from measurement of the structure of the benthic macroinvertebrate community, at different sites and for the different surveyed reaches, indicated a clear difference in the Shannon index among different waterbodies (Figure 5.4), implying that the environmental and abiotic characteristics of each stream have an influence in determining the biodiversity of each stream. Specifically, sites impacted by the Engineered Log Jams deployment following the Littlestock Brook NFM trial scheme are the first 32 sites from the left in Figure 5.4. Littlestock Brook 1, Littlestock Brook 2 and Littlestock Brook 3 have a higher Shannon index, thus higher levels of diversity when compared to the sampling sites placed within control reaches.

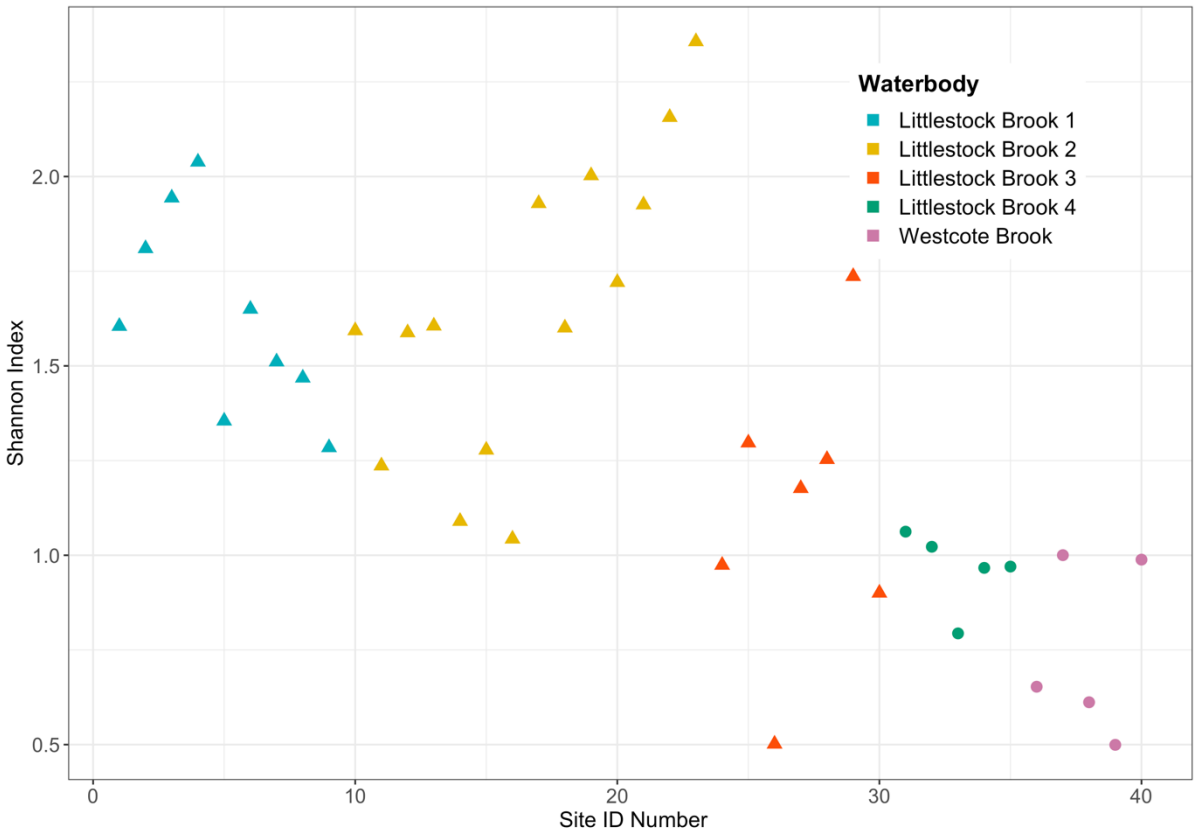


Figure 5.4 The Shannon index of each sampling site. *Site ID numbers from 1 to 40 are portrayed on the x-axis. The values of the Shannon index are portrayed on the y-axis. The legend shows the different colours used to differentiate the different channels. The different shapes represent control (circles) and impacted reaches (triangles) as shown in Figure 5.2 and described above.*

The differences between sites impacted by the installation of ELJs, control sites, and sites downstream of an ELJs sequence are shown in Figure 5.5. The highest diversity was recorded for impacted sites (ELJs) which are the ones in closest proximity or in between engineered log jams. Control sites collated (Control) are those with the recorded lowest diversity, while downstream sites (DS) in reaches with ELJs were found to have intermediate Shannon indices. Upstream sites in reaches with ELJs were sampled at 15 and 30 meters upstream of the first woody dam of the sequence. For this reason, they were considered normal control sites and collated to the sites on control reaches for this analysis. The pattern depicted by Figure 5.5 was confirmed with the statistical analysis: sites impacted by engineered log jams have significant higher biodiversity than control sites and reaches (tested with ANOVA, $P <$

0.01). These results allowed me to reject the null hypothesis of no difference between impacted and control sites and to accept the alternative hypothesis of a statistically significant difference in benthic macroinvertebrate diversity between control and impacted sites (with ELJs).

It is also interesting to note the lack of a statistically significant difference in the Shannon Index when comparing control sites and downstream control sites on impacted reaches (Figure 5.5). An ANOVA test between the two groups represented by upstream and downstream sites in the impacted reaches indicates that even if they seem to have different overall biodiversity, upstream and downstream sites did not show a statistically significant difference ($P > 0.15$). This result, therefore, suggests that the effects of Engineered Log Jams on macroinvertebrates are very localized and not felt further downstream.

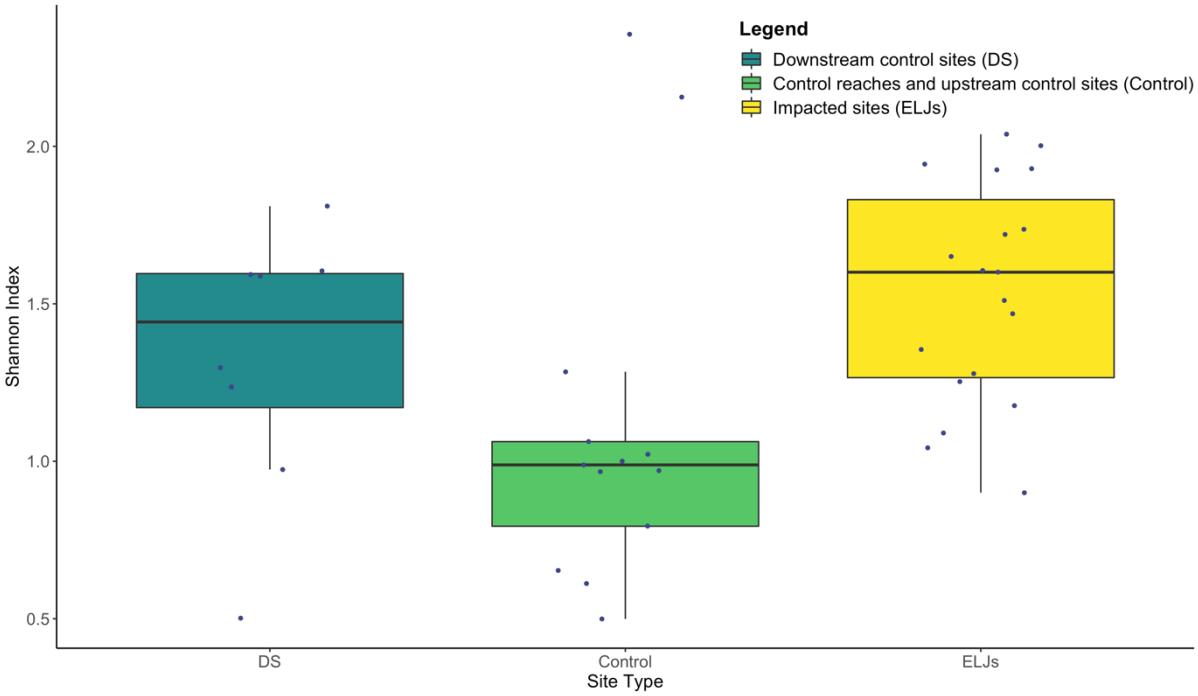


Figure 5.5 Differences in Shannon Index values between different site types. *Shannon Index values are here categorised for site type. In the figure, from left to right, downstream sites (DS, dark green) are sites downstream of ELJs, Control sites are sites upstream of ELJs collated with control reach sites (light green) and ELJs sites are sites in proximity of engineered log jams (yellow). The small circles represent individual data points. The circles' dispersion in the horizontal direction within each category (Site Type) is an artefact used for better visualisation of individual data points.*

The comparison of average abundance for each family and for different site types is shown in Figure 5.6. For a better representation, the rarest taxa were not included in the figure. Further details on average abundance and richness for the different site types is shown in Table 5.1.

The upstream and control sites collated (Control in Figure 5.6 and Table 5.1) exhibit a higher average abundance per site compared to ELJs sites and sites downstream of ELJs. However, control sites also show a lower richness in taxa when compared with downstream and ELJs sites. This confirms the results obtained for the Shannon Index, as control sites seem to be dominated by a high abundance of fewer taxa but with an overall lower taxa diversity. In contrast, ELJs sites and downstream sites show higher species richness per site and also a higher total richness when pooling together the number of taxa per site type (value in parenthesis in Table 5.1).

Table 5.1 Data of richness and abundance for different site types. *The values represent average number of specimens per site (abundance) and total number of taxa per site type (richness).*

Site type	Average richness per site (Total richness per site type)	Average abundance per sampling site
Downstream Sites (DS)	12.4 (33)	253
Control Sites (CONTROL)	8.9 (26)	357
ELJs Sites (ELJs)	11.8 (42)	297

The findings of Figure 5.5 were further confirmed by the hierarchical clustering of sampling sites shown in Figure 5.7. The clusters shown in Figure 5.7 describe the similarity of macroinvertebrate communities of different sampling sites. As it can be seen in the figure, the sampling sites in the right cluster are mainly the ones on the reaches Littlestock Brook 4 and Westcote Brook, used in my sampling design as the two control reaches for the space for time substitution. According to the hierarchical clustering the macroinvertebrate community

composition of the Littlestock Brook 4 and Westbrook seem rather different from the community composition of the reaches impacted by the presence of ELJs.

Figure 5.6 shows that Gammaridae is the dominant family in all site types. Gammaridae represent the dominant macroinvertebrate species, in terms of biomass, in many freshwater ecosystems (MacNeil et al., 1997). Gammarids are important keystone species as they play an important role in the litter breakdown processes of the detritus cycle and constitute an important element in food webs as prey for secondary consumers. Interestingly, a relative high number of Baetidae specimens were found across the different site types and especially abundant in control sites. Baetidae are active swimmers in their larval stage and represent a widespread and abundant family. Baetidae are mostly shredders that have been shown to feed on particulate organic matter in mountain streams (Molina et al., 2004). However, Baetidae preferentially act as collectors-gatherers in more fragmented habitats with higher microbial activity, where coarse particulate organic matter (CPOM) is less abundant (Tomanova et al., 2006).

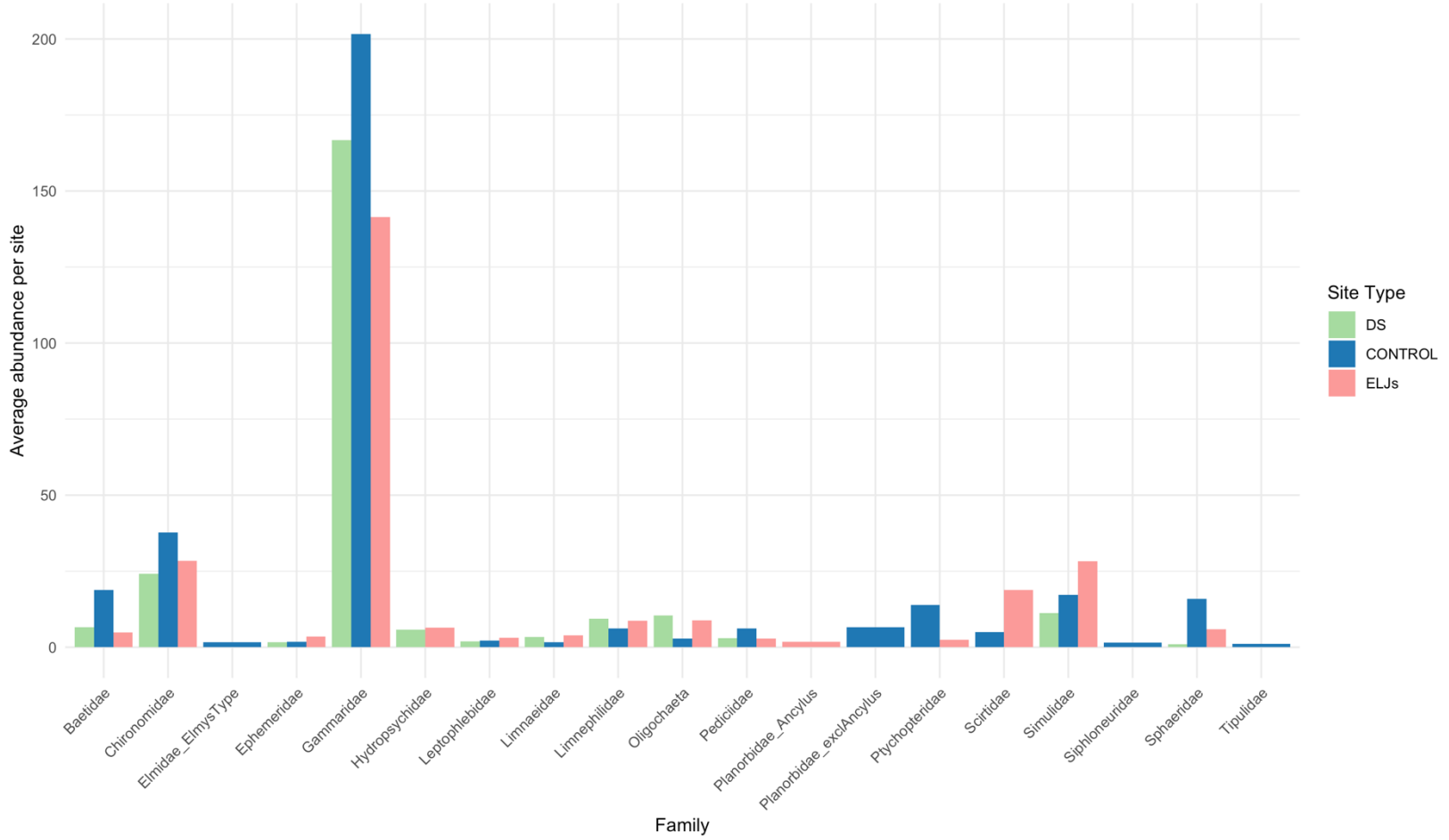


Figure 5.6 Comparison of family abundance per site type. *The x-axis portrays the most occurring families in the sampled region. The y-axis portrays average taxa abundance per site. The different bar colours differentiate the different Site Types: Downstream sites (DS), Control sites (CONTROL), and ELJs sites (ELJs).*

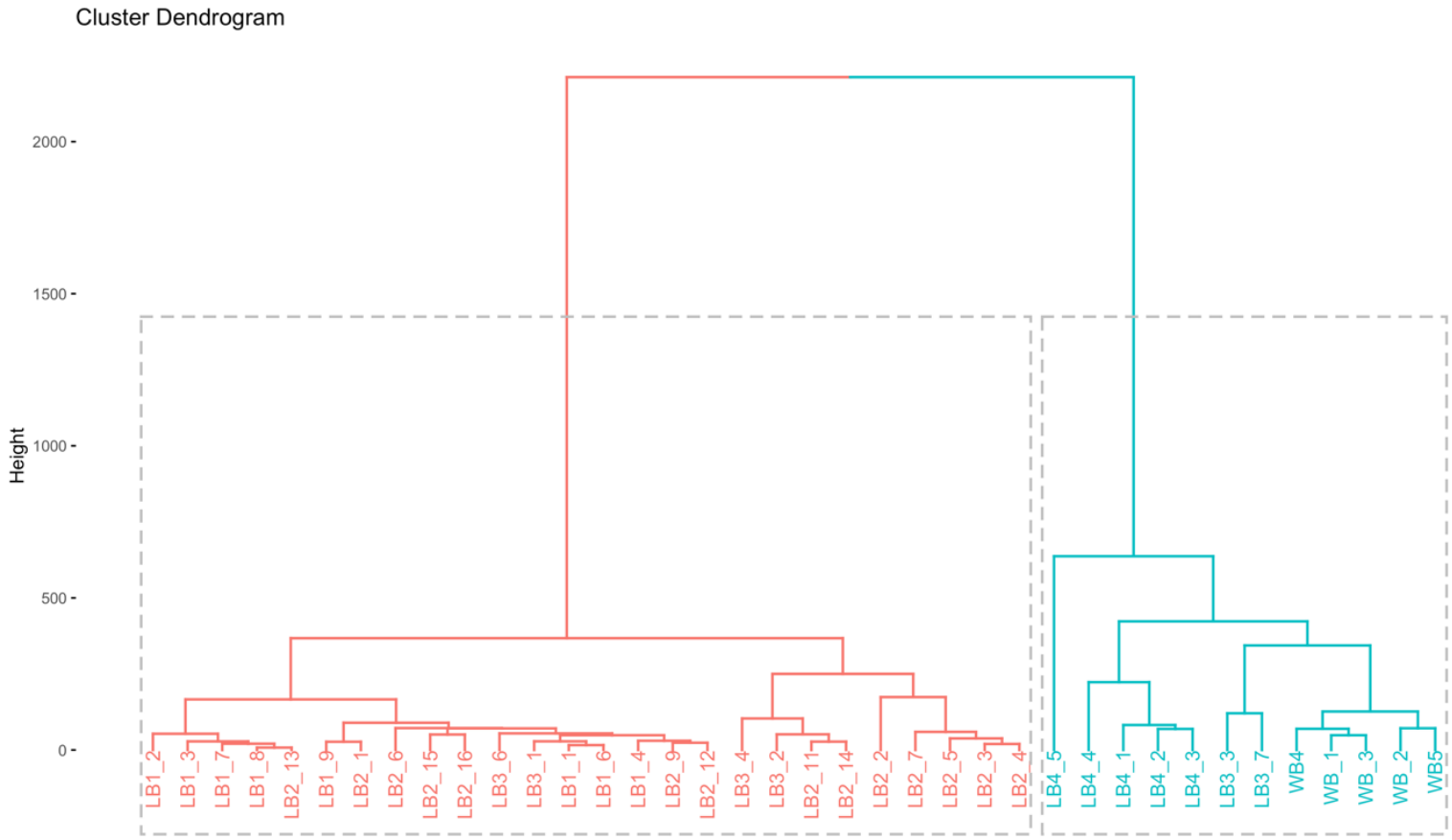


Figure 5.7 Hierarchical clustering of the sampling sites. *Labels refer to the reaches code and the site number within the reach. LB, LB2, LB3, LB4 and WB stand for Littlestock Brook 1, Littlestock Brook 2, Littlestock Brook 3, Littlestock Brook 4, and Westcote Brook respectively. The two different colours represent the two main clusters of the dendrogram.*

5.4.2 Organic matter decomposition

The experimental results of the quantification of organic matter decomposition rates did not show a significant difference in patterns of organic matter decomposition among the different site types (Figure 5.8)

The daily tensile strength loss was higher for the cotton strips not enclosed in a litter bag while the lowest tensile strength loss, corresponding to the lowest daily decay of the cellulose of the cotton strips, was associated with cotton strips enclosed in a coarse mesh bag. Cotton strips enclosed in a fine mesh size bag were showing intermediate values of daily tensile strength loss. The difference in the means for the different deployment modality (coarse mesh bag, fine mesh bag and no bag) was significant as tested with ANOVA ($p < 0.02$). These patterns were consistent even across the four different types of sampling sites (Figure 5.8).

Even if no difference was observed in pattern of organic matter decomposition among the different site types for cotton strips not enclosed in bags, control reaches and ELJs-impacted sites showed a difference in daily tensile strength loss for cotton strips enclosed in litter bags (Figure 5.8).

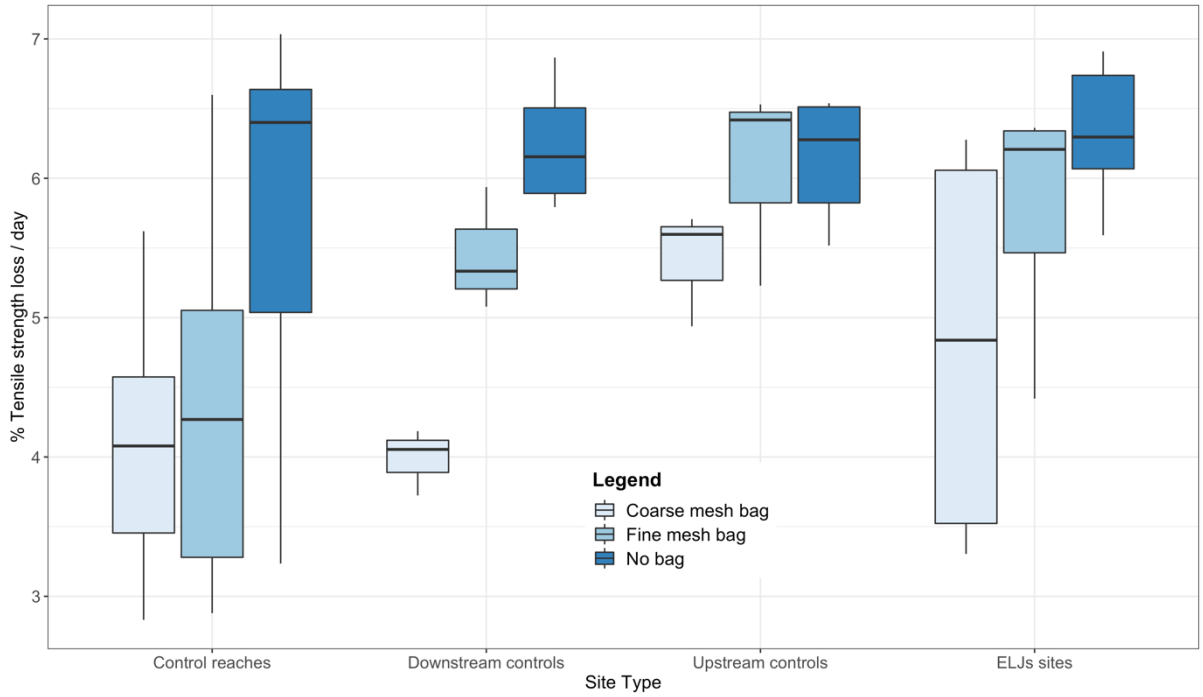


Figure 5.8 Tensile strength loss for different sampling sites. *The colour of the boxplot reflects, within each type of sampling site, the placement modality into litter bags with different meshes or without litter bag. In the figure, from left to right, Control reaches represent the sampling sites in the channels used as controls, not impacted by the presence of artificial ELJs; Downstream controls are the sampling sites downstream of ELJs; Upstream controls are sampling sites located upstream of the reaches impacted by ELJs installation, ELJs sites are the sampling sites in close proximity and directly impacted by the installation of ELJs.*

5.5 Discussion

This study provides initial clear evidence that ELJs can have a positive impact on localised diversity of the communities of freshwater invertebrates. A statistically significant increase in diversity was observed at all sites that were close to ELJs suggesting that high water ELJs, installed into channels as part of NFM projects, give additional environmental co-benefits thus addressing one of the main knowledge gaps highlighted by Dadson et al. (2017). However, my study also indicates that the effects leading to the establishment of higher diversity are probably localised and influenced by the micro- and meso-habitat characteristics established by each single ELJ more than influenced by the sequence of log jams implemented in the reach.

These findings are also consistent with the findings of previous published studies (Appendix A) and with the hydro-morphological changes imposed by ELJs on the riverine habitats that were shown in Chapter 2 and 4. Moreover, as highlighted by Deane et al., (2021), the deployment of ELJs into streams seem to have an almost immediate effect on the macroinvertebrate communities. After only a few months from the implementation of Large Woody Debris, Deane et al. (2021) observed an increase in macroinvertebrate abundance and taxa richness. This is probably due to the fact that the hydraulic effects of ELJs on the channel hydraulics are immediate and due to the drag and damming effects they pose upon the waterflow.

In contrast to measures of biodiversity, I found no appreciable difference in ecosystem functioning, as measured by decomposition rates of organic matter, among sites impacted by the presence of ELJs and unimpacted sites. The placement modality of cotton strips had an effect on decomposition rates. In particular, cotton strips enclosed in litter bags were showing lower decomposition rates when compared to cotton strips not enclosed in bags. This might reflect both a reduced accessibility to the cotton strips for macroinvertebrates and a reduction in palatability due to the presence of a nylon mesh close to the cotton strips or a decrease in microbial activity due to a bag effect.

This is partially in contrast to earlier studies that showed that the efficiency with which streams store and process carbon partly depends on the abundance of retention zones (Peterson et al., 2001; Hall et al., 2002). The presence of channel-spanning log jams has been highlighted as one of the key differences that promotes the creation of retention zones in headwater streams (Beckman and Wohl., 2014). However, the creation of biogeochemical hotspots with high rates of carbon decomposition depends on the persistence of the log jams for at least a few years (Beckman and Wohl., 2014). The results showed in this study for organic matter decomposition may thus be dependent on the relatively new deployment of the ELJs into the channel, or as a

result of the particular type of ELJs examined here, the high water ELJ, which does not directly affect the water flow at low flow conditions.

5.6 Conclusions

This study showed clear evidence of the beneficial effects of high water ELJs, developed in the context of NFM projects, on riverine ecosystems and provided new evidence of the environmental co-benefits of NFM. The results indicated that benthic macroinvertebrate diversity was higher at sites impacted by the installation of Engineered Log Jams, probably reflecting the heterogeneity in micro- and meso-habitats established by the presence of ELJs. This impact was however very localised and did not influence the benthic macroinvertebrate diversity of sites downstream of the ELJs. There was also no evidence to support the hypothesis that the higher retention of organic matter, due to the presence of ELJs, resulted in higher organic matter decomposition rates and thus an increased ecosystem functioning.

5.7 Appendix A: Literature Review

A literature search was conducted in 2019 including publications from 1975-2018, using two databases: ISI Web of Science and Scopus. Search keywords were [Invertebrates] AND [Woody dam OR Leaky dam OR Large Woody Debris OR Large wood OR LWD OR Debris dam OR Log Jam] AND [Stream OR River]. After the first scan of the title and abstract, the full body text of 32 papers was analysed completely (Table 5.2). Just under half of them (15) showed relevance to the subject of assessing the impact of large wood in streams on invertebrate communities, and these were further analysed and summarized (Appendix A).

These 15 studies were almost evenly distributed between those that used a Control-Impact (CI) design (space for time substitution) (8) and studies which used a Before-After-Control-Impact design (BACI) (6). The former represents a sampling design which involves the comparison of control sites with impacted sites. In freshwater studies CI normally includes control sites on rivers different from the one impacted but chosen because of the proximity to the impacted sites and on the basis of similar characteristics. It is assumed that the control rivers are not impacted by the intervention/impact. The BACI design involves instead a comparison of the after-impact conditions at the site with the conditions before-impact, as well as with the conditions of control sites before and after the date of the impact. This approach, therefore, provides both temporal and spatial controls.

Most studies (8) in the review (Appendix A) showed an increase of either macroinvertebrate richness or abundance or a change in the structure of benthic invertebrate communities (feeding groups or specific taxa) associated with LW or restored woody debris while one third (5) didn't show any appreciable change in the macroinvertebrate communities. The two remaining studies showed statistically significant changes in the biodiversity of the benthic invertebrate communities. However, these studies were using different approaches that

included the evaluation of wood quantity (Pilotto et al., 2016) and wood complexity (Schneider and Winemiller, 2008).

It is clear that there is considerable variability in the outcome of many river restoration projects on benthic macroinvertebrate communities, where large woody debris were the main restoration measure. Miller et al. (2010), for example, in an extensive meta-analysis of restoration projects concluded that LW did have an influence on the macroinvertebrate community biodiversity. However, while the highlighted trends were generally tending to an increase in biodiversity, not all projects showed statistically significant biodiversity improvements.

The literature review has highlighted some key findings:

- The studies showed that both naturally occurring and reintroduced wood have a tangible effect on macroinvertebrate communities as reported in previous published reviews (Miller et al., 2010; Palmer et al., 2010).
- The sampling design choice for the study did not impact the outcome, as the study conclusions did not show a trend in the type of sampling design applied.
- Invertebrate communities have been quantified across different geographic areas and with different land-uses and catchment characteristics, but the effect of large wood on invertebrate communities was largely consistent.
- The studies have shown a large variation in the approaches and the units of measure used to understand the structure of the communities.
- Only a few studies were considering other types of measure different from structural metrics.
- No studies have been carried out to date on high water ELJs associated with NFM projects that differs from naturally occurring large wood in key structural characteristics such as orientation, density and porosity.

***Table 5.2 Review table of studies linking freshwater invertebrates and large wood**. Response variables: Abundance (A), Biomass (B), Behavioural Traits (BT), Density (D), Feeding Group Composition (FG), Taxonomic Richness (R), Secondary production (SP); taxonomic identification levels: Family (F), Genus (G), Species (S).*

Study	**Location** **Stream name**	**Land use** **Stream characteristics**	**Response Variable and taxonomic level of the identification/taxa included**	**Type of restoration action or impact being monitored** **Monitoring duration** **Type of design**	**Significance of effects**
Benke et al (1985)	Georgia (USA), Satilla River	Satilla River: Low gradient coastal plain blackwater river	Variables: B, D, SP Identification: G, S (when possible)	Naturally occurring woody debris (SNAGS). 12 months (samples taken every 2-4 weeks) CI design	SNAG habitat had highest diversity, highest biomass and secondary production per unit of habitat surface.

Smock et al. (1989)	Virginia (USA), Buzzard Branch and Colliers Creek	Two low-gradient, blackwater streams. Buzzards Branch had a sandy/silty substrate and a permanent flow throughout the year with a full canopy and meandering through a small forested floodplain. Colliers Creek was a bottomland stream with a substrate of clay and silt with deposits of organic matter, with intermittent flow (dry in the summer or just with standing water in pools). Colliers Creek braided through a forested narrow floodplain which was completely inundated from December to May.	Variables: FG, B, D	Naturally occurring woody debris dams (any wood debris with a dimeter bigger than 5cm and spanning at least one-fourth of the channel). Artificial dams were placed in Buzzards Branch: the manipulation involved the implementation of High, Low dam frequency and no dam in three different sites. 12 months (6 sampling in total) BACI design	Macroinvertebrate density and biomass in Buzzards Branch dams were 5-10 times greater than on sandy sediments. This reflected high aggregation of organic matter in dams and the greater stability of dams compared to shifting sand. Organism abundance in streams is influenced by dam abundance. Debris dams also affected functional feeding-group composition of the macroinvertebrate community. Shredders were dominant or co-dominant organisms in terms of biomass in dams in both streams. Increased dam abundance thus increased shredder abundance and the relative proportion of shredders in the stream.
Wallace et al (1995)	North Carolina (USA)	Cunningham Creek. Forest with mixed hardwoods and a dense understory of rhododendron resulting in year-round shading. Cobble-sized substratum with localized areas of boulder and bedrock as well as depositional areas. Elevation around 900 m a.s.l. The Creek is situated in the Appalachians mountain range.	Variables: B, D, FG, R Identification: S (Chironomids only)	Three stations where a log (20-32 cm in diameter) was placed in the centre of the station and anchored with stakes rock or existing tree trunks. 8 months (4 samples in total, 3 before and 1 after) BACI design	Invertebrate abundances were significantly greater at debris-dam transects after log addition than those of other sites and periods. Abundances were 24X those prior to log addition and following log addition exceeded those of reference riffles by >8X. Biomass significantly greater after log addition. Abundance of scrapers and filterers decreased significantly at debris-dam transects following log addition, collectors and predators increased.

Gerhard & Reich (2000)	Germany	Lüder (meadows and pastures) Jossklein (beech and non-native spruce forest)	Variables: D, R Identification: F, S (Plecoptera, Ephemeroptera, Coleoptera, Trichoptera, Amphipoda and Mollusca)	LWD accumulated naturally or built-in (Jossklein), built-in only (Lüder) 1 single sampling CI design	Of the 41 species found within the Plecoptera, Coleoptera, Trichoptera, Amphipoda and Mollusca half were in microhabitats occurring only in restored reaches. Highest densities were found in the restored reach for all microhabitats.
Lemly & Hildebrand (2000)	Virginia (USA)	North Fork Stony Creek: a third order low gradient stream at 920 meters of elevation circa in the Appalachian Mountains. Surrounded by a mature mixed soft- and hardwood forest.	Variables: D Identification: F, S (when possible)	600 meters reach divided in two 250 m sections with 50m buffer zone in between. First section (upstream) used an unmanipulated control site while LWD was added to the second section. Location and positioning of logs were subjectively determined to enhance stream habitats. 12 months (2 samples in spring, one before and 1 after) BACI Design	Insect community structure as measured by abundance of functional feeding groups remained similar spatially and temporally. However, community structure differed substantially between pools and riffles, and LWD additions greatly shifted the pool/riffle relationship.
Larson et al. (2001)	Washington (USA)	Four urban streams but with different levels of human disturbance.	B-IBI score (ten matrices of taxa richness and evenness, disturbance tolerance and life history attributes) Identification: G or F	Anchored and unanchored LWD added to the streams. Upstream and downstream monitoring sites. 60 months CI design	Little influence on the benthic invertebrate community. The physical characteristics in the reach that did change displayed no clear relationship to biological condition.
Johnson et al (2003)	Michigan and Minnesota (USA)	Agriculture with small proportion of forested lands.	Variables: FG, BT, R	Natural large wood Only 1 sampling for each of the two regions (September 1994 and September 1998) CI design	A substantial portion of local diversity can be attributed to the presence of wood habitats. A mean of 55% and 26% of taxa in Michigan and Minnesota, respectively, were added to a reach when a wood habitat was present representing a substantial contribution to the local biodiversity. Large wood plays an important role in structuring the composition and function of the macroinvertebrate community.

Rosi-Marshall et al (2006)	Michigan (USA)	Cook's Run is a tributary of the Paint River part of the Lake Michigan watershed.	D, R, FG Identification: F or G	k-dam structures which are designed to create pool habitats (it was a restoration project for fish) 48 months CI design	Macroinvertebrate density, diversity and functional feeding group composition were not significantly affected by the treatments.
Lester et al. (2007)	Victoria (Australia)	8 perennial streams on intensively-grazed, commercial agricultural properties (beef, dairy and sheep production)	D, R, FG Identification: F	Volume of wood determined from natural woody debris in streams, added and anchored in place. 15 months BACI design	Across the common habitats sampled treated sites supported a greater number of families than untreated sites. Family richness for most functional feeding groups was also higher at treated sites. No difference found for sensitive families (EPT taxa).
Schneider & Windemiller (2008)	Texas (USA), Brazos River	Lowland river ecosystems	Variables: A, R Identification: F	40 small woody debris patches with high or low level of complexity One single survey (14 days after implementation) CI design	Small patches of wood were associated with greater abundance and diversity of macroinvertebrates in shallow habitats. The richness tended to be greater within more complex habitat patches in both habitat types.
Wellnitz et al (2014)	Minnesota (USA), Blind Temperance Creek	Forest, 3rd order with gravel bed	Variables: R, D Identification: F (Chironomids), Genus (other groups)	Installed log-jams with upstream controls on the same river. 12 months (2 sampling in Spring of subsequent years) BACI design	Overall taxonomic richness and abundance remained the same at both control and impact sites. However, both these parameters increased for Ephemeroptera and Plecoptera.
Pilotto et al. (2014)	Poland	Pliszka River: low-gradient sand-dominated lowland river with mostly forested riparian zones.	Variables: R Identification: S or G (mostly)	Naturally occurring LWD. 1 single sample (Oct- Nov) CI design	Benthic samples taken near large wood and on the large wood surface were characterised by significantly higher values of the diversity metrics (Shannon Wiener and eveness) than the control samples. Large wood areas showed values 81,110, 108 and 168% higher than in the controls for abundance, taxa richness, evenness and Shannon-Wiener diversity index, respectively.

Pilotto et al. (2016)	Poland	3 near-natural rivers: Płociczna, Korytnica and Pliszka rivers. Low gradient, sand dominated lowland rivers with largely forested floodplains.	Variables: R Identification: S or G (mostly)	Naturally occurring LWD. 1 single sample (April-May) Sampling based on 2 levels: wood rich (WR) and wood poor (WP) sites	Macroinvertebrate community structures were consistently distinct between samples at WP sites and those immediately adjacent to wood. The wood surface was associated with the highest abundance of macroinvertebrates while the highest taxonomic richness and diversity was found adjacent to the wood. Beta diversity was higher in WR sites and increasing with the wood present. The highest alpha diversity was recorded around the LW and taxonomic richness and Shannon-Wiener diversity were on average 30% and 45% higher there than in wood poor sites.
Thompson et al. (2017)	England, 5 different streams	Lowland chalk rivers	Variables: R Identification: S (mostly)	Restoration activity 12 months MBACI Design	Invertebrate alpha diversity increased by five species in restored and target reaches relative to controls. Differences in invertebrate assemblages between habitat types revealed that the addition of LW habitat enhanced overall beta diversity within restored and target reaches relative to controls.
Pinto et al (2019)	England River Blackwater and River Whitewater	Blackwater catchment is predominantly urbanised catchment underlain by superficial gravels and London Clay Whitewater is a predominantly rural catchment underlain by clay	Variables: A, R	60 months Design with 4 reaches where wood was introduced (R1a, R1b, R1c and R2) 2 reaches with naturally occurring woody dams (N1 and N2) and 3 controls (2 at restored sites and 1 at the natural wood site) CI Design	No significant differences between large wood reaches and the respective control sites for both abundance and diversity

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Chapter 6 : Summary of the results, Discussion and Conclusions

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6.1 Introduction and summary

Large wood is an integral part of freshwater ecosystems that provides a plethora of ecosystem services and affects river hydraulics, channel morphology and catchment hydrology. Traditionally, however, large wood has been removed from rivers and channels. Moreover, rivers have been straightened, channelized and dredged. The main reason for these modifications was to improve navigability, provide a constant output of water for agriculture and farming, increase channel conductance, and decrease the local flood risks by lowering water levels and increasing water flow velocities locally. However, the importance of large wood for riverine ecosystems has been recently re-discovered and the operations of de-snagging, dredging, and channelization have been more and more discussed and frowned upon (Wohl, 2017).

The newly developed interest in large wood presence and the understanding of the benefits it provides on channel hydraulics, sediment retention and biodiversity made the re-introduction of large wood a sought-after restoration technique (Gurnell, 2012; Miller et al., 2010). In the last few decades, the Water Framework Directive implementation in the UK, together with the direction taken by the UK government on the environment, with its 25 years Environmental Plan, and the funding allocated for "Working with Natural Processes", have promoted the use of large wood as a restoration and a flood risk management technique. In particular, NFM has been increasingly implemented as a flood risk management strategy on those occasions for which a traditional hard engineering solution was not a viable choice because of the extent of the problem or because there were concerns over the potential implications for nature and freshwater ecosystems.

The introduction of ELJs in rivers and channels, in particular, has been an extensively used restoration technique thanks to its economic feasibility, ease of deployment and additional environmental co-benefits. Despite the wide application of ELJs for NFM projects, outstanding

knowledge gaps remain to fully understand their effects on channel hydraulics, catchment hydrology and on the wide array of environmental benefits and ecosystem services they may enhance or provide. In particular, Addy and Wilkinson (2019) and Dadson et al. (2017) highlighted that ELJs have been shown to have effects at the small scale and for small flood events but there remain uncertainties on their effects for large catchments (>100 km^2) and for large flood events (25 years recurrence period or more). Addy and Wilkinson (2019) also highlighted how there is a need for more guidance on their representation in hydraulic and hydrological models at a variety of spatial scales. Dadson et al (2017), Lane (2017) and Ellis et al. (2021) highlighted how little is known about the potential environmental co-benefits of NFM techniques and, more specifically, of ELJs.

Given these general knowledge gaps, the overarching aim of my thesis was to understand the effects of high water ELJs on local reach hydraulics, catchment hydrology and freshwater ecosystem structure and functioning. To address this aim I focused on the following specific objectives:

- Chapter 1: To review the scientific literature on NFM and ELJs and identify the current knowledge gaps, to introduce the scope of this research and to highlight the importance of this research and how the knowledge accrued could help in overcoming barriers and setting up opportunities for successful implementation of ELJs into a catchment's river network.
- Chapter 2: To measure the hydraulic resistance provided by high water ELJs by directly quantifying Manning's roughness with field surveys; to compare different methodologies used to simulate ELJs effects at the reach scale and their effects on the peak flow magnitude and timing at the reach outlet.
- Chapter 3: To simulate the sensitivity of the catchment and its river network to changes in channel hydraulic roughness and to determine priority areas where an increase in

channel roughness could have the greatest effect on the peak flow at the catchment outlet.

- Chapter 4: To understand the impact of ELJs on floodplain inundation, in-stream velocities, and peak discharge magnitude and timing at the catchment scale by coupling a hydrological model to simulate the catchment contribution to hillslope runoff, and a two-dimensional hydraulic model to simulate in-channel ELJs and water flow dynamics.
- Chapter 5: To investigate the potential environmental co-benefits of ELJs in terms of biodiversity and ecosystem services by analysing the structure of the in-channel macroinvertebrate communities and the organic matter decomposition rates in channels where ELJs were installed.
- Chapter 6: To summarise the findings of this research, their limitations and future directions for research. To draw conclusions and discuss how the findings of this research can inform stakeholders, set opportunities for the successful implementation of ELJs/NFM, help to overcome barriers to ELJs/NFM implementation and provide policy and research future perspectives.

6.2 Key findings and Discussion

This thesis addressed the effects of ELJs from both a flood management perspective and an ecological perspective and considered the different scales at which ELJs can exert effects on channel and floodplain hydraulics and on field and catchment hydrology for small and large catchments. I will hereby summarise the key findings by considering the effects of ELJs at the different scales that were analysed in this study.

6.2.1 Channel and reach scale

The effects of the installation of a sequence of high water ELJs on channel conveyance at the reach scale were measured in the field by quantifying hydraulic resistance with Manning's *n* values (Chapter 2 - Figure 2.4). The data collected in the field showed Manning's *n* values ranging from 0.09 to 0.6 for a short reach (30 m) containing one high water ELJs. These values were in concordance with other studies in other catchments in the UK (Dixon et al., 2013; Kitts et al., 2010; Gregory et al.,1995; Linstead and Gurnell, 1998). Interestingly, all these studies measured Manning's *n* in reaches of forested catchments where the presence and accumulation of large wood in the channels was a naturally occurring phenomenon. My study is the first one that has measured the additional resistance provided by ELJs in highly modified channels and for artificial high water ELJs installed as part of an NFM project. These findings justify the installation of high water ELJs as part of NFM projects. High water ELJs analysed in this study were indeed shown to exert hydraulic effects equivalent to naturally occurring large wood.

The increase in channel roughness that follows ELJs installation causes a decrease of water flow velocity of up to 40% upstream of the ELJs for a 25-year flood event (Figure 4.11). The decrease in flow velocities, and the resulting higher diversity in flow velocities across both

the longitudinal and lateral directions of the channel, will establish a higher habitat heterogeneity that could be beneficial for the invertebrate and fish communities. Moreover, habitat heterogeneity will also increase as a result of the formation of pools close to ELJ structures (Abbe and Montgomery, 1996). Pool formation is due to the flow patterns arising as a result of the blockage imposed by the ELJ and the formation of circular flow patterns against the ELJ structure (Figure 2.1).

The increase in channel roughness and the resulting lower in-reach velocities and pool formation due to the presence of installed ELJs are also important in increasing hyporheic residence times (Section 1.6.3.1). Higher residence times are one of the main drivers of streambed microbial metabolic activity and biogeochemical cycling that can lead to the formation of hotspots for nitrogen removal or production (Krause et al., 2014).

Measured Manning's *n* values showed a dependence on flow magnitude as highlighted in the literature (Addy and Wilkinson, 2019). During the rising limb of the flow hydrograph, Manning's *n* for the reach of interest decreased as discharge increased (Figure 2.4). This dependence of Manning's *n* on flow magnitude has important implications in the choice of the structure of high water ELJs before installation. Indeed, both the distance between the lower part of a high water ELJ and its width and height should be carefully considered. First, the distance between the bottom of the ELJ and the channel bed should be chosen on the basis of the normal low flow and stage level that should be allowed to pass beneath the structure. Second, the height of the high water ELJs is important in determining the maximum flow and stage level that will be impacted by the ELJ. Turcotte et al. (2016) found that the hydraulic resistance exerted by large wood pieces is proportional to the cross-sectional area of the channel they occupy. However, larger and higher structures could increase the frequency and extent of floodplain inundation as a result of higher water levels upstream of the structure.

The change in channel hydraulics induced by ELJ can lead to an increase in habitat heterogeneity with a higher variability in flow velocity and depth conditions, water depth and velocity, increased hyporheic exchange fluxes and increased bioavailability of dissolved organic carbon. These large wood related changes have the potential to benefit the in-stream biota. The response of macroinvertebrate communities to the presence of ELJs was analysed in this thesis with important key findings. Macroinvertebrate communities at ELJs sites showed a higher taxonomical diversity compared to control sites (Figure 5.4-5.5 and Table 5.1). This higher diversity was mainly due to a higher species richness while control sites were shown to have an overall higher abundance but mainly due to the presence of ubiquitous dominant families. The higher macroinvertebrate diversity of ELJ sites might be correlated to an increase in habitat heterogeneity due to the presence of ELJs. This could have important implications in the near future with the creation of a market for biodiversity credits and ecosystem services.

6.2.2 Small catchment scale

The effects of the wide implementation of ELJs in a small rural catchment (the Littlestock Brook, 16.3 km^2) were quantified by using a hydrological-hydraulic modelling cascade. The application of the modelling cascade allowed for a depiction of the hydrology of the catchment by simulating the hillslope runoff. Furthermore, it was also possible to take advantage of hydraulic modelling to depict the 2D hydraulics of the channel and floodplain with an explicit representation of ELJs in the 2D geometry.

The cascade modelling framework applied to the small catchment scale showed an increase in floodplain inundation extent and frequency proportional to the number of ELJs installed in the catchment. The hydrograph at the catchment outlet showed a decrease in the peak flow discharge for small magnitude flood events and a delay in peak flow timing at the catchment outlet for both small and moderate flood events. In particular, the effects of wide

implementation of ELJs was remarkable for the 1-year flood event with a reduction in the peak flow height of 10-20%, depending on the number of ELJs installed, and a delay of the peak flow timing of 5-6 hours for the scenarios with 40 and 70 ELJs. The effects of the catchment-wide installation of ELJs were instead negligible for the 25-year flood event, with an implementation of 40-70 ELJs that was producing only a 5% decrease in peak flow discharge. However, the peak flow timing was delayed by 5 hours for both the 40 and the 70 ELJs scenarios.

These findings have important implications for the uptake of NFM projects and, more specifically, for the installation of ELJs. As shown in this thesis (Chapter 4) the decreased flood risk at the catchment outlet may come at the expense of increased floodplain inundation in the upstream part of the catchment. The catchment-wide installation of ELJs, to achieve a reduction in flood risk downstream, will thus imply to 'Make Space for the River' in another part of the catchment. As explained in Section 1.5.3 this can create upstream-downstream conflicts between different stakeholders and landowners and act as a barrier to the implementation of NFM projects. In order to increase their effectiveness, ELJs have been often paired with other NFM and restoration measures. In particular, since ELJs promote floodplain inundation, it has proved particularly effective the association with floodwater retaining structures such as ponds, bunds and wetlands (Thomas and Nisbet, 2012).

6.2.3 Large catchment scale

Natural Flood Management has been shown to have effects on flood hydrology for small catchments and for small or moderate flood events, while evidence of their effectiveness for larger catchments and for extreme flood events is simply not available. Extending the results obtained at the small scale (5-15 km^2) to greater spatial scales is extremely difficult methodologically. In fact, it is difficult to apply NFM solutions over a very large area and

obtain data in an experimental way because the spatial scale of interest increases together with the probability of confounding processes.

The values of roughness measured in Chapter 2 (Figure 2.4-2.5) were applied as channel roughness parameters in a hydrological model of the Evenlode catchment, in order to test the catchment's sensitivity to changes in channel roughness. This methodological approach has been already used before to test headwater rehabilitation measures (Liu et al., 2004). The idea behind this approach is that the commonly used techniques for river restoration, such as re-meandering, large wood or boulder addition, increase the channel roughness.

The sensitivity analysis for the Evenlode catchment showed a low sensitivity of the modelled catchment to changes in roughness. In order to impact peak flow height and timing, an increase in Manning's *n* of two to three times should be achieved across the whole river network of the Evenlode. Moreover, as seen in Chapters 2 and 4, ELJs exerted their effects on a limited length of the channel (approximately ten channel widths in this case). If ELJs were the restoration measure chosen to achieve the change in roughness, it would require an incommensurable number of them to achieve a 2.5% reduction in peak flow discharge (e.g. Manning's n =0.05). Moreover, ELJs performance is highly affected by the discharge. Exerting a high resistance to flow under high flow conditions could thus be even more difficult.

The results of the sensitivity analysis were really interesting for their implications. In particular, as other studies have shown before, NFM measures, and especially ELJs, are probably unfit to exert effects on peak flow magnitude at the large catchment scale. However, as this thesis has highlighted, they can be effective for small catchments and moderate flood events. Moreover, when deployed in sequence in small river reaches they are beneficial for biodiversity and many environmental and ecological processes.

Future research should set aside the negligible effects of NFM at the large catchment scale and try to understand better the effects of ELJs on ecosystem processes. In particular, the

effectiveness of ELJs in reducing diffuse nutrient pollution is of particular importance, given nutrients impact on the environment as well as the economic cost of their removal or blending. Clear scientific evidence on the effects of ELJs on nutrient cycling could help overcoming some of the barriers to the implementation of ELJs and favour an higher uptake of NFM projects.

6.3 Discussion and Conclusions

Traditional Flood Management (TFM) typically diverges the flow of flood waters away from the areas at risk through the implementation of hard engineering structures such as flood walls, dams, raised embankments and levees. This approach has historically been employed by local or regional governing bodies responding to public demands after major flood events (Butler and Pidgeon, 2011). The installation of TFM structures is often a fragmented response to flood risk management that only considers the immediate risk area without a wider focus on the potential catchment implications such as the increase of flood risk for a downstream area or the environmental effects of a spatially intensive measure (Demeritt and Nobert, 2014).

In contrast, NFM aims to reduce flood risk by protecting, restoring and emulating the innate hydrological and morphological processes and characteristics of catchments, using Nature-based Solutions as opposed to traditional "hard engineering" structures. NFM intervention can be classified under three different types of manipulation to reduce runoff generation. In particular, runoff reduction is realised with land use and land management changes, by temporarily storing runoff into the floodplain, by building storage retention measures such as ponds and wetlands, or by reducing conveyance thus 'slowing the flow' (Lane, 2017). Engineered Log Jams are one of the NFM interventions that intervene on the conveyance of flood waters to the downstream part of the catchment through the river network. ELJs are one of the most used NFM interventions for their economic feasibility, ease of deployment and the wider environmental co-benefits they are assumed to deliver (Burgess-Gamble, 2017; Ellis et al., 2021).

This research investigated the effects of high water ELJs on hydraulics and hydrology at multiple scales and on the environmental co-benefits they are assumed to provide.

The aim of the sections below is to draw conclusions from the research carried out and presented in this thesis about the key opportunities for implementation of ELJs, how to

overcome barriers to ELJs implementation, and future perspectives in research, policy and water management (Figure 6.1).

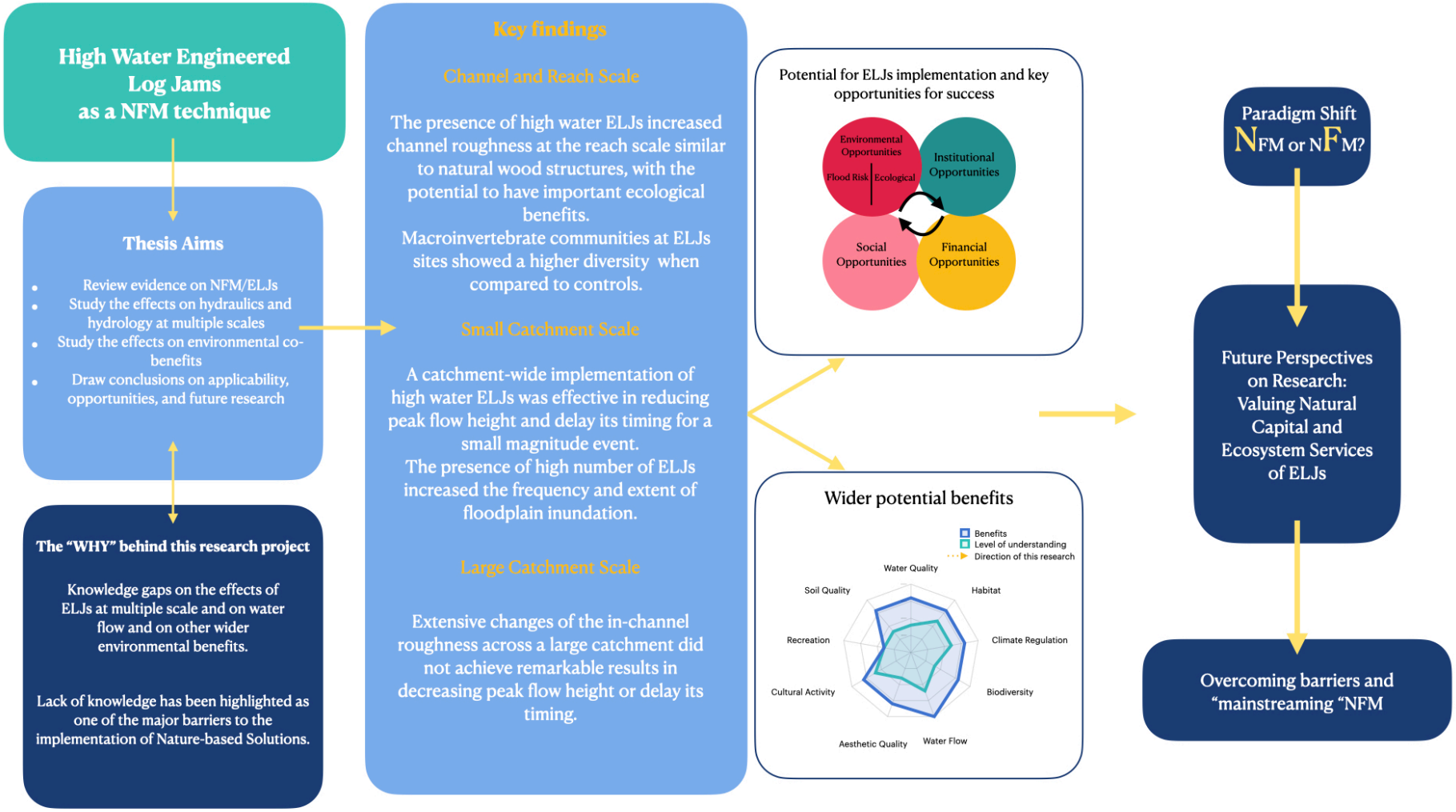


Figure 6.1 Conceptual abstract.

6.3.1 Key opportunities of success for implementing High Water Engineered Log Jams in watercourses

NFM has recently become a prominent strategy for flood risk mitigation in countries all over the globe (Sudmeier-Rieux et al., 2021). In particular, NFM is a spatial extensive flood risk mitigation technique that separates from more traditional engineering solutions that are spatially intensive (Thaler et al., 2023).

There are many key opportunities for the successful implementation of NFM projects. As highlighted by Thaler et al. (2023), the successful and sustainable implementation of NFM depends on the interconnection of different factors that can be divided into four broader categories: Environment Opportunity, Social Opportunity, Financial Opportunity, Institutional Opportunity.

In the following sections, the opportunities of success for the implementation of Engineered Log Jams will be drawn from the results of this thesis and from the wider literature for each separated category.

6.3.1.1 Environment Opportunity for ELJs

The Environment Opportunity for the implementation of ELJs is mainly set out by the physical and the environmental/ecological characteristics of the catchment. As highlighted above, ELJs are an NFM measure that manipulates the conveyance to reduce rapid runoff generation and delivery downstream through the river network (Lane et al., 2017). Moreover, ELJs may provide additional ecological benefits in terms of biodiversity, water quality and sediment retention. The Environment Opportunity could be thus furtherly divided into a Flood Risk Mitigation Opportunity and an Ecological Opportunity. The Flood Risk Mitigation Opportunity depends on catchment characteristics and its state. The catchment's land use, geology and river network are important in determining the type of flooding and its drivers

and, as a result, in delineating the best NFM measures that should be implemented. It is hereby suggested that the scheme developed by Thaler et al. (2023) on the key opportunities for successful implementation of NFM could be modified as presented in Figure 6.2 to better consider the specific case of ELJs. Thaler et al. (2023) suggested that this scheme could provide guidance and facilitate both documentary analysis and the interviews with stakeholders.

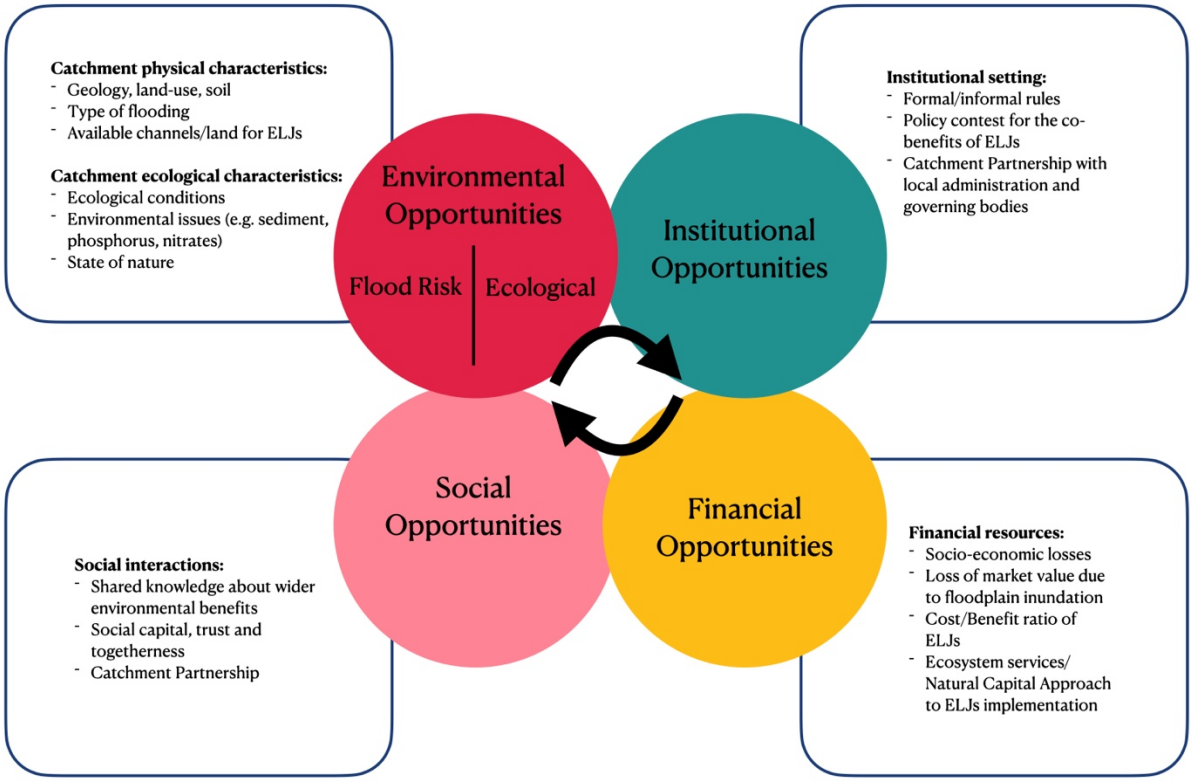


Figure 6.2 Key opportunities of success for ELJs implementation. *Adapted from Thaler et al.(2023).*

6.3.1.1.1 Flood Risk Opportunity

Drawing upon the results of this research project, presented in Chapters 2-4, and the wider published literature, the Flood Risk Mitigation Opportunity for ELJs can arise in catchments whose river networks have been highly impacted by artificial modifications. In particular, river channels, that have been straightened, dredged and excavated in the past to increase water conveyance, can represent a good target for the implementation of ELJs in their

multiple forms. High water artificial ELJs have indeed been shown to increase channel roughness by this study (Chapter 2 - Figure 2.3) confirming the results from previous publications for naturally occurring high water jams (Kitts et al., 2010). Moreover, a wide implementation of high water ELJs has been shown by this study to be beneficial in terms of the reduction of moderate flood events at the catchment scale for a small rural catchment (the Littlestock Brook) with a responsive regime. The Flood Risk Opportunity could thus arise in highly modified rural catchments where there are specific flooding problems even for flood events of moderate magnitude. However, a precautionary approach should be used in the planning phase of ELJs implementation. High water ELJs may indeed increase floodplain inundation frequency and spatial extent as shown by this study (Chapter 4), and by previous studies for different wood structures (Keys et al., 2018).

6.3.1.1.2 Ecological Opportunity

Drawing upon the results presented in Chapter 5 and the wider literature on the ecological effects of the presence of large wood in river systems, the Ecological Opportunity of success for the implementation of ELJs can arise for those river channels and networks that have been extensively modified by human activities. In particular, large wood has been shown to provide an increase in the abundance and taxa richness for macroinvertebrates (Deane et al., 2021; Lester et al., 2007), an increase in key ecosystem processes such as sediment, organic matter and nutrient storage (Smock et al., 1989; Bilby and Likens, 1980) an increase in hyporheic exchange flow and oxygen availability (Krause et al., 2014; Kaller and Kelso, 2007), and to induce channel morphological alterations and provide refugia and food for invertebrates and fishes (Chapter 1-Section 1.6.3). This thesis has shown how ELJs reintroduced in largely modified agricultural channels can still provide an increase in macroinvertebrate diversity, even for simpler wooden structures such as the high water ELJ. In the case of ELJs, the

Ecological Opportunity may even be predominant over the Flood Risk Opportunity. This might be the case in catchments highly impacted by human activities, with a river network that doesn't reach a good ecological status or with other environmental issues that may be, at least partially, improved by the presence of ELJs within the river network. As an example, diffuse agricultural pollution and sediment transport are two common environmental issues of catchments in the UK that could be targeted by the implementation of ELJs. NFM features like ELJs and Wetlands should thus be taken in high consideration also for their high potential in providing environmental co-benefits. As highlighted in Dadson et al. (2017), these environmental benefits may be so preponderant when considering their implementation, that it would be sufficient for them not to have any negative impacts on flood risk. For this reason, it is timely to consider a paradigm shift towards a major focus on the benefits of NFM from an ecosystem service perspective more than from a purely flood risk perspective. The paradigm shift will be further discussed in section 6.6.2.

6.3.1.2 Social Opportunity

The land needed for NFM implementation can often be mobilized in the part of the catchment with lower land values, to protect and mitigate the flood risk of downstream higher-value areas (Morris et al., 2016). However, as shown in Chapter 4, high water ELJs, deployed throughout the catchment to mitigate flood risk at the catchment outlet, were found to have the potential to increase the frequency and extent of floodplain inundation upstream.

This transfer of flood risks between the upstream and downstream parts of the catchment can lead to intensive conflicts and a lack of acceptability from different stakeholders. Social capital, trust and social norms might increase the acceptance of the transfer of risks between upstream and downstream areas (Thaler et al., 2023).

6.3.1.3 Financial Opportunity

The third category of factors that influence the implementation of NFM is the financial context of the catchment and how it will be modified by NFM. For example, NFM interventions might reduce the potential land available for livestock or agriculture, thus reducing the potential to generate agricultural incomes (Collentine & Futter, 2018). The proposed NFM should thus be financially viable from the perspective of both the implementer and those acted upon. From the perspective of the latter, complying with the needs of the proposed NFM must be economically viable from a business perspective (Thaler et al., 2023). For example, if their land must be repurposed for NFM, they should receive a beneficial impact in terms of compensation. Zooming on the NFM measure that is the subject of this thesis, the Engineered Log Jam, it must be noted that, compared to other NFM measures more spatially intensive, they require little space for implementation and the impact on the agricultural land would be minimal when compared to other NFM measures. ELJs are implemented within the channel and thus would not require the land to be completely repurposed. However, ELJs implementation requires to 'make space for water' and, as a result, some floodplain inundation should be expected.

6.3.1.4 Institutional Opportunity

NFM implementation usually involved a set of different actors and stakeholders. The institutional setting for NFM regulates the social interactions between different actors and stakeholders, the types of compensation, and the role of public administration and the private sector in flood risk management (Hartmann et al., 2019; McCarthy et al., 2018). ELJs implementation should consider the institutional settings and, in particular, the boundaries between different rural properties in case of floodplain inundation. In particular, strong social relationships between landowners could favour the installation of ELJs as these could impact

neighbouring properties and, in case of funding, it should be clear who should benefit from these and in which proportions. In this context, catchment partnerships could be particularly important in favouring the social interaction between all the private and public actors and stakeholders.

6.3.2 Natural Flood Management: time for a Paradigm Shift

The comparison of Natural Flood Management (NFM) to more Traditional Flood Management (TFM) techniques often focuses on the fact that NFM aim is to restore or enhance the hydrological morphological and hydraulic characteristics of the catchment and its river network by using "green infrastructure" or "natural" infrastructure. When discussing NFM we are thus usually mentioning its "natural" character as opposed to the "hard" or "grey" engineering structures of TFM. However, this focus on the naturality of these solutions tends to shift when evaluating the effects of the implementation of these measures. As highlighted by Lane (2017), the 'Flood' in NFM is given much more attention than those wider environmental co-benefits that are being suggested by the term 'Natural'. This focus can be seen through most of the available literature, where even outstanding reviews about NFM (Dadson et al., 2017; Lane, 2017) tend to address the wider environmental co-benefits of NFM in a short section at the end of the paper. The major focus on the 'Flood' benefits is also evident throughout this thesis. However, the sole focus on the 'F' in NFM is unlikely to be sufficient to justify the implementation of these measures, especially at the larger scales. Moreover, the only quantification of the effects on flood risk is likely to underestimate the wider benefits that come from slowing the flow in terms of ecology, social benefits and broad environmental aspects (Figure 1.3).

The focus on the 'F' in NFM is probably linked to both a perception of flooding as a frightening natural hazard and the grounding of flood risk management in the quantitative determination

of economic losses. The focus on the flood risk effects of NFM, while neglecting or marginally addressing the wider environmental benefits, has also been highlighted by O'Donnell et al (2017) as one of the barriers to the implementation of Green and Blue Infrastructure.

NFM can represent a paradigm shift in the approach to flood risk management when compared to traditional flood management techniques. A paradigm shift intended as a transition by which a dominant paradigm is superseded by a new paradigm that is based on a different conceptual framing (Kuhn, 1996) and entails those involved to both think and act differently (Bark et al., 2019). A paradigm shift from traditional flood management to natural flood management should thus embrace and focus on the "natural" aspect of NFM which is what truly differentiates NFM from TFM and has important implications from a policy and environmental perspective.

6.3.3 Policy implications

Since the 1st January of 2021, the UK definitely left the European Union after the transition period of Brexit. The exit from European Union included the withdrawal from the Common Agricultural Policy (CAP) and the associated funding mechanisms that determined how the agricultural sector was financially supported by the EU. The UK government established a new UK subsidy regime for agriculture (Environmental Land Management System) that was recently updated by DEFRA (2023). The Environmental Land Management Scheme has set out three schemes to pay for environmental and climate goods and services: the Sustainable Farming Incentive (SFI), the Countryside Stewardship (CS) and the Landscape Recovery.

Under CS the government is promising to provide water storage within floodplains, control excess silts and nutrients from entering surface and groundwater, and support the management of rivers, stream and floodplains. Furthermore, CS guaranteed to enhance water

quality, manage and reduce flooding, re-connect channels to the floodplain, increase in-stream habitat diversity and populations of native species in channels and floodplains, finally enhancing rivers' natural functionality.

This poses ELJs development in an excellent position for a wider application within catchments. Indeed, ELJs have been shown, by this work and other published studies, to provide an increase in biodiversity prompted by habitat heterogeneity (Chapter 5), to enhance water quality by increasing residence times and hyporheic exchange flows (Krause et al., 2014; Chapter 2), to re-connect highly modified channels to the floodplain (Chapter 4) and finally to provide flood water storage within the floodplain (Chapter 4) and, eventually, in the channel incision (Chapter 3). Obtaining additional funding for specific NFM measures, such as ELJs, is essential to further embrace a paradigm shift for a more naturally-based flood risk management (Bark et al., 2019). In particular, payments under the ELM schemes could overcome one of the barriers to the implementation of NFM as landowners and land managers has been shown to expect a full compensation for these types of measures (Bark et al., 2019).

6.3.4 Future perspective on research

The research on ELJs effects from both a flood risk and a natural or environmental perspective is still in its early stages. The wider literature has highlighted many potential benefits of naturally occurring large wood in rivers. However, ELJs and other wooden structures, such as leaky barriers and large woody debris, implemented in the context of NFM, may be quite different in structure from their natural counterparts. For this reason, more scientific evidence is needed to understand the potential flood risk and wider environmental benefits that these artificial structures could provide and to assess if there are any differences among different structures in delivering these benefits.

In order to assess the effectiveness of ELJs or other NFM structures more experiments with controls or full BACI experiments are needed. Baseline monitoring before the implementation of NFM structures should become the norm in NFM projects.

NFM can deliver multiple ecosystem services. However, as highlighted in Section 6.6.2 research on NFM has almost always focused preferentially on the 'Flood' part in NFM. As a result, the ecosystem services of NFM are only assumed and quantitative data to support assumptions is often not available (Ellis et al., 2021). Most of the literature on NFM alludes to its multiple co-benefits, but many of these benefits lack full understanding and quantitative support from published literature (Figure 6.3).

The research on NFM needs to move towards a more integrated assessment of the wider benefits of NFM. An interdisciplinary ecosystem services approach, quantified by natural capital (Daily et al., 2009; Laurans et al., 2013), could be suitable to provide empirical evidence of the wider benefits of the multiple NFM techniques.

The research on the flood risk benefits of NFM showed that NFM has the potential to decrease flood risk at the small catchment scale and for events of moderate magnitude. However, NFM has wider benefits that transcend the flood risk benefits. If we can help individuals and institutions to recognize the value of the wider benefits of NFM, investments in these measures could greatly increase. Unfortunately, research on NFM has neither developed a sound understanding of NFM's wider benefits nor the scientific basis for their monetary evaluation, thus hindering a policy revision and allocation of funding.

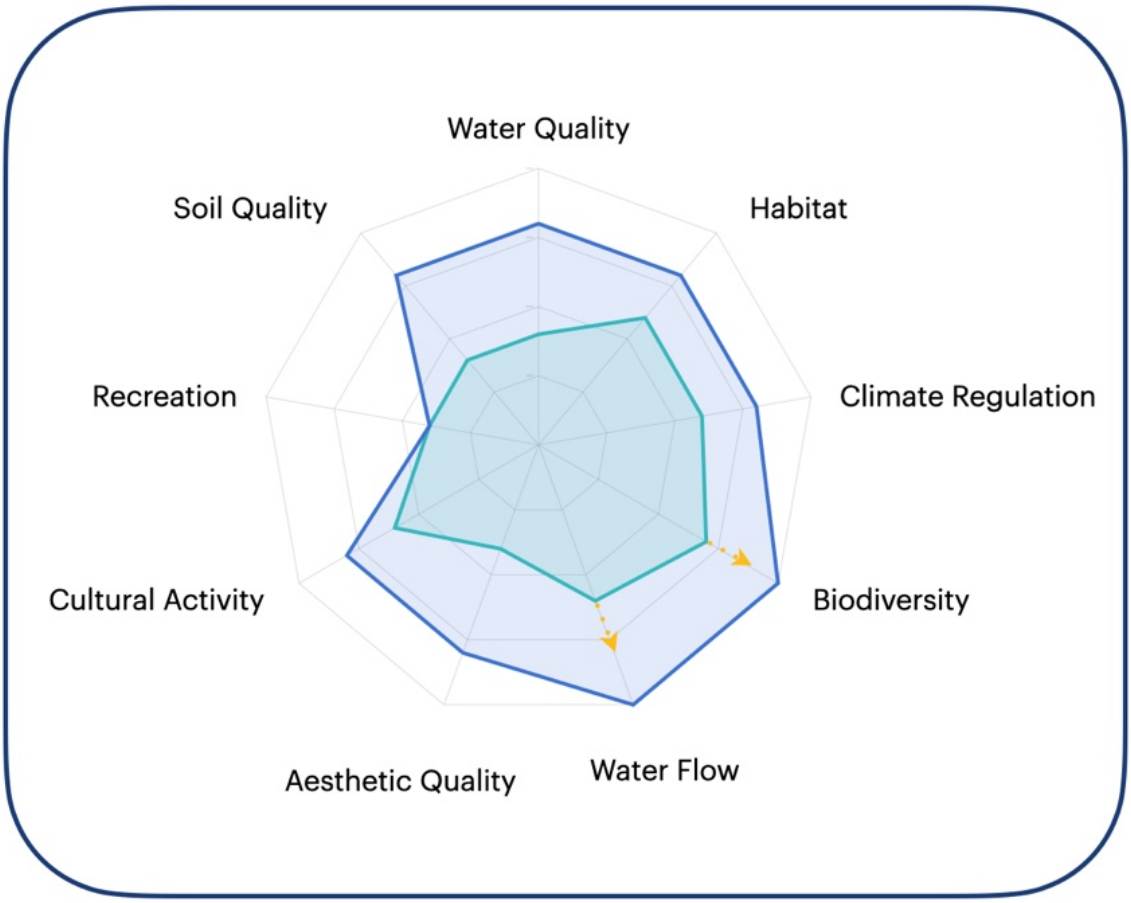


Figure 6.3 The potential benefits of Engineered Log Jams on ecosystem services*. The potential benefits of ELJs (dark blue) are plotted against the present level of understanding of the ecosystem services provided by ELJs (green) as concluded from the available literature. The scale used in this figure goes from 1 to 5 with 1=poor benefit/limited understanding, 5=great benefits/great understanding. As shown, all categories have a gap in potential benefits versus our current understanding. The small orange arrows show the direction taken by this thesis to increase our knowledge and understanding. Adapted from Burgess-Gamble (2017) and Ellis et al. (2021).*

6.3.5 Overcoming barriers and mainstreaming ELJs and NFM

The Engineered Log Jam is a prominent NFM solution that offers multiple environmental benefits and demand stronger consideration among flood risk management strategies. However, there are still complex and diverse barriers to NFM uptake and implementation. These barriers include the biophysical characteristics of catchments, the lack of scientific evidence of NFM effectiveness, a lack of governance over the maintenance of NFM projects, land managers' past experiences and beliefs and problematic interactions between different actors and stakeholders (Wells et al., 2020).

To overcome the current barriers to the implementation of ELJs, an analysis of the opportunities for success is fundamental. In particular, this analysis must identify the biophysical and ecological characteristics of the catchment, the key actors and stakeholders, the impacts of the proposed implementation, the funds necessary for its realisation, and the decision-makers who have the power to establish the business case for effectively plan and deliver the project in the catchment (Thaler et al., 2023).

The dynamic complexity of the NFM delivery and the numerous barriers to its implementation require a multi-stakeholder collaborative partnership that could evaluate the environmental, social, financial and institutional settings for NFM implementation.

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"All that was good in me thrilled in my heart at that moment, all that I hoped for in the profound, obscure meaning of my existence. Here was the endlessly mute placidity of nature, indifferent to the great city; here was the river beside these streets, around these streets, waiting for the city to die, to cover it with timeless water once more. There came over me a terrifying sense of understanding about the meaning and the pathetic destiny of men. The river was always there, a patient white animal, waiting for men to die, for civilizations to flicker and pass into the darkness. Then men seemed brave to me, and I was proud to be numbered among them. All the evil of the world seemed not evil at all, but inevitable and good and part of that endless struggle to keep the water down."

Adapted from John Fante – Ask the Dust.

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