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Abstract

There is an increasing emphasis on using natural processes, including riparian forest restoration, to enhance the ecological, hydrological, and geomorphological functioning of watercourses. However, we have insufficient knowledge on how the supply and retention of in-channel wood from riparian forest stands changes with age, with inferences typically based on data from terrestrial forests. This presents a challenge in estimating the efficacy and functional lifespan of restoration projects. In this paper we use a riparian forest growth model to show there is a lag of up to 40-50 years between the start of forest growth and trees delivering wood to the channel that is large enough to resist fluvial transport, anchor logjams and so increase channel complexity and hydraulic resistance. Resource managers need to account for realistic timescales over which changes promoted by riparian woodland restoration will occur and may need to consider using interim engineered logjams as the forest develops.

Introduction

Recent advances in forest ecohydrology research have shown the multiple benefits of riparian forests and wood in rivers. Riparian forests influence forms and processes within the channel (Gregory et al., 1991; Gurnell et al., 2002; Montgomery et al., 2003); acting as a source of dead large wood, both to the floodplain surface leading to greater geomorphic complexity (Jeffries et al., 2003; Sear et al., 2010; Polvi and Wohl, 2013), and to the river channel where it enhances geomorphological and hydraulic heterogeneity (Piégay and Gurnell, 1997; Gurnell et al., 2002; Montgomery et al., 2003; Dixon, 2016). The greater complexity induced by wood in turn supports greater habitat diversity (Collins et al., 2002; Johnson et al., 2005) and thus can potentially support greater ecological abundance and diversity. Riparian forests can also reduce delivery of diffuse pollution through trapping of fine sediment runoff from agricultural land (Cooper et al., 1987; Daniels and Gilliam, 1996; Lowrance et al., 1997), storing sediment behind logjams (Davidson and Eaton, 2013; Wohl and Scott, 2017), remove nitrogen and phosphorous from runoff and sub-surface flow (Peterjohn and Correll, 1984; Lowrance et al., 1997; Wang et al., 2012; Sutton-Grier et al., 2013), and enhance stream metabolism (Blaen et al., In Press). Forested floodplains are a source of particulate organic matter to the channel (Gurnell et al., 2002), provide shade (Montgomery et al., 2003), help regulate water temperature (Garner et al., 2015; Garner et al., 2017; Ouellet et al., 2017; Dugdale et al., 2018) increase bank stability through root reinforcement (Shields Jr and Gray, 1992; Beechie et al., 2006) and

decrease the erosive power of the channel (Gregory et al., 1985; Manga and Kirchner, 2000; Fisher et al., 2010).

In a mature floodplain forest system trees act to drive a large wood cycle (Collins et al., 2012), whereby large wood from fallen trees alters in-channel process, either protecting areas of forest from erosion to allow riparian trees to reach greater size, or diverting flow to increase bank erosion and recruit more wood to the channel. Mature floodplain forests are therefore highly dynamic systems with the forest acting as an ecosystem engineer to alter the river environment (Collins et al., 2012; Gurnell, 2014). The increased geomorphological complexity of the channel and the floodplain surface has been shown to increase flood wave travel times (Ghavasieh et al., 2006; Thomas and Nisbet, 2007; Thomas and Nisbet, 2012), with a mature floodplain forest and abundance of logjams in the channel shown to be effective at reducing flood peak height at the catchment scale (Dixon et al., 2016).

As knowledge of the benefits of mature floodplain forests has increased, policy and practice both in the US and Europe has turned towards encouraging riparian forest restoration and protecting riparian forests; initially for ecological reasons (Naiman and Décamps, 1997; Broadmeadow and Nisbet, 2004; Nislow, 2005), and at relatively small scales (Nislow, 2010), but more recently as a component in natural flood risk management, or "working with natural processes" (Defra, 2007; Lane, 2017; Mondal and Patel, 2018; Nilsson et al., 2018), and advocated at entire catchment scales.

Regardless of the spatial scale, or management objective, in order to use floodplain forests to deliver societal or ecosystem benefits resource managers need to be able to understand and predict their influence on processes over time. In natural flood management (NFM) projects, flood risk managers are raising questions about how to specify or estimate the changing performance of introduced wood structures and newly planted riparian woodland. We, therefore need to understand the timescales over which a new stand of floodplain forest trees develops and matures, and how the maturity of the stand is linked to production of large wood and thus in-channel processes. The complexity of forest ecosystems makes it challenging to develop conceptual models of forest growth (Botkin et al., 1972), which is particularly the case in riparian forests (Robertson and Augspurger, 1999; Warren et al., 2016). It is not possible to uncritically apply knowledge of upland terrestrial forest plots to riparian forests, as riparian plots are subject to allogenic disturbances from the fluvial system. Site specific erosion and deposition as well as lateral channel migration lead to destruction of land as well as creation of new emergent land surfaces (Naiman and Décamps, 1997) and areas for seedling colonisation (Van Pelt et al., 2006). In the presence of active erosion there can be chronic stress reducing riparian vegetation community structure at the eroding edge leading to the possibility of retrogression where succession is not unidirectional towards an increasingly mature vegetation community, but can move back towards earlier, less complex communities (Décamps et al., 1988; Kupfer and Malanson, 1993). Furthermore, the establishment of stable logjams has been linked to the presence of large key pieces (Montgomery et al., 2003), which can only be delivered by trees of a sufficient size and maturity, hence in-stream wood function is likely to increase later in stand development (Keeton et al., 2007). However, logjam structures are complex accumulations of a range of wood sizes, anchored by large pieces but packed with smaller branches, twigs and leaves that create seasonal and inter-annual variability in hydraulic performance (Kitts 2011; Millington & Sear 2017). Given the complex interplay between the riparian forest and the channel processes, it is likely that the scale and type of processes occurring in forested river channels will change with different stages of maturity in the floodplain forest.

Naiman et al (1998) proposed a conceptual model of riparian forest succession on bare earth (Figure 1) with four stages of development. Following initial establishment there is a second phase of stem exclusion where all growing space is occupied and species or specimens with a competitive advantage can expand into space occupied by other specimens, out competing and eliminating them. New plant colonisation is mostly excluded and vertical sorting and stratification occurs. In the third phase an understory develops through the establishment of shade tolerant species and gap phase regeneration following mortality of large trees leading to multiple canopy levels. In the final stage there is an old growth assemblage where mortality opens up gaps in the canopy as an autogenic regeneration process (Naiman et al., 1998). Growth rates and time between stages will vary with

species composition, disturbance regime and sites. This approach has been further developed to consider how nutrient retention (Valett et al., 2002) and light availability (Warren et al., 2016) changes for different successional stages. As riparian forests age, wood is likely to play a greater role in the aquatic environment (Kasprak et al., 2012) and studies have found correlations between in-stream large wood loadings and the age of the dominant canopy trees (Hedman et al., 1996; Meleason et al., 2003; Cordova et al., 2007; Warren et al., 2009; Brooks et al., 2012). The mechanistic model of Meleason et al (2003) for managed riparian forests in the US Pacific North-West found there was a lag of around 50 years before wood beings to accumulate in the channel, with a slowdown in accumulation rates from around 250 years onwards and maximum wood volumes achieved at 525 years. Empirical models (e.g. Warren et al., 2009) show that wood inputs and wood loading increase with stand age, however there is a need to established whether these trends are linear, or whether loadings move through different phases in association with different successional stages. Therefore, there is a need for a new conceptual model of floodplain forest development which specifically links changes in the maturity of the forest stand with associated changes in the scale and type of influences by in-channel large wood and logjams.
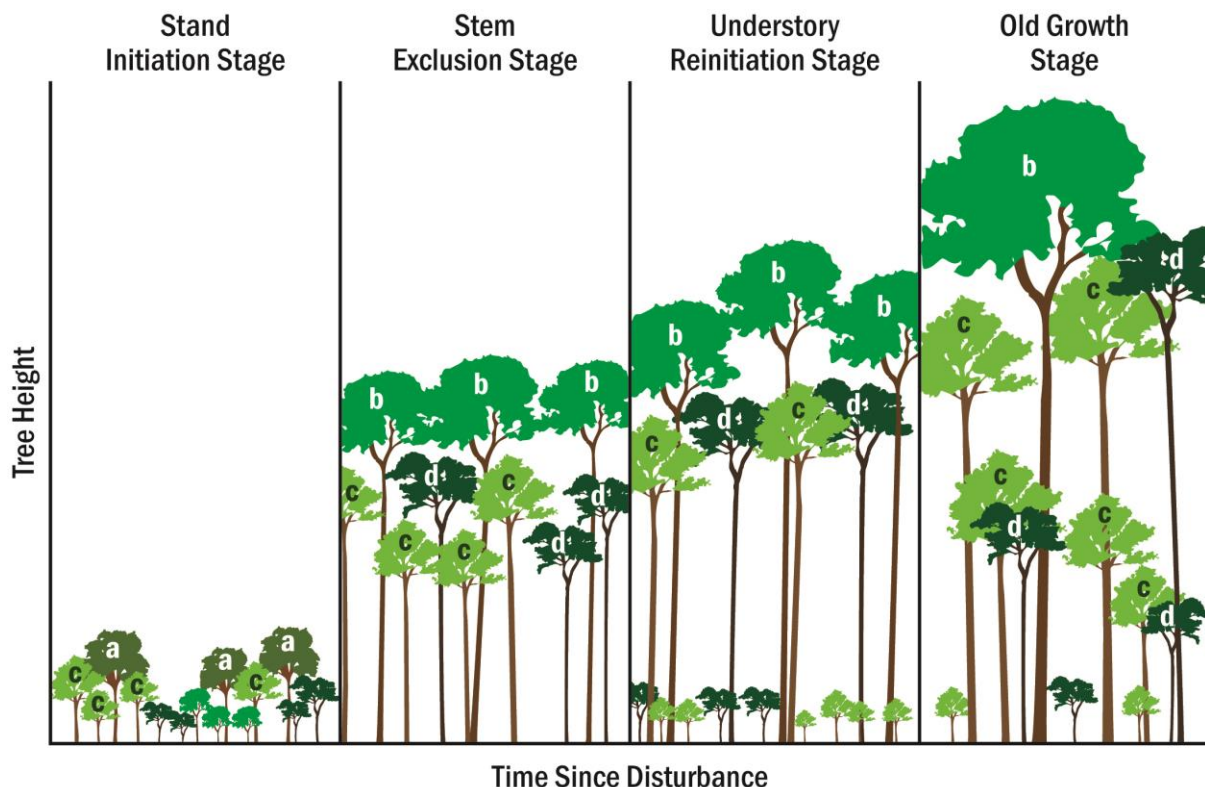

Fig. 1. Four stages of forest development on a 'bare earth' site, e.g. following a disturbance. In stand initiation stage all tree species (a-d) establish soon after disturbance. However, during the development of the riparian forest the dominant tree species will change over time as stem numbers decrease and vertical stratification of species occurs (After Naiman et al., 1998

The objectives of this study are to use a numerical model with forest growth and deadwood components to explore how riparian forest growth is linked to loadings of in-channel wood and to expand upon the conceptual model of a riparian forest proposed by Naiman et al (1998) to include logjam development and compare to empirical findings on wood loading and stand age (Warren et al., 2009). The specific aims of this study are to; i) determine whether in-channel wood loads increase linearly with forest age, ii) determine if riparian forest species composition affects the successional stages of forest development, iii) determine at what stage in forest development management objective such as provision of natural flood management benefits could expect to be realised.

Methods

In order to derive predictions of in-stream large wood loads and the complexity of floodplain surfaces over time following a programme of riparian forest regeneration, a numerical modelling approach was adopted to simulate forest growth and succession. Numerical models of riparian forest growth are comparatively rare worldwide and none exist for a UK context (Broadmeadow, 2012), therefore a numerical model for the North-Eastern United States (NE-CWD, Nislow, 2010) was used, which incorporates growth, dead wood and riparian dynamics of both broadleaf and conifer species (Lester et al., 2003). The use of numerical models is well established to predict upland forest plot growth and harvest yields (e.g. Botkin et al., 1972; Phipps, 1979; Randle, 2000; Busing and Solomon, 2004; Huber et al., 2013; Mikac et al., 2013), as well as forest

succession (Shugart and West, 1977; Pearlstine et al., 1985). However, comprehensive studies of riparian forest growth are few, although conceptual models of riparian forest succession have been proposed (Fonda, 1974; Hawk and Zobel, 1974; Pabst and Spies, 1999; Nierenberg and Hibbs, 2000) and numerical models developed (Phipps, 1979; Décamps et al., 1988; Hanson et al., 1990; Nuttle and Haefner, 2007). Despite challenges in modelling complex riparian areas it is established that where appropriate old growth reference conditions do not exist, vegetation simulation models can be useful in understanding riparian forest dynamics (Kasprak et al., 2012).

Model Description

The upland and riparian Northeastern Coarse Woody Debris (NE-CWD) model was developed between the United States Department of Agriculture (USDA) Forest Service Northern Research station and the University of Massachusetts, Amherst, based upon a stem growth model NE-TWIGS, created by Hilt & Teck (1989) .

NE-CWD (Lester et al., 2003) incorporates live tree dynamics such as seedling regeneration, ingrowth (the growth of trees into the smallest measured size fraction in the model) and tree growth, and death at the individual tree/subject level. Dead wood dynamics (snag fall rates, log breakage and decomposition) are incorporated to predict residence times of dead wood. In addition to live and dead wood dynamics the model also incorporates riparian dynamics with the input of riparian logs through bank erosion

and the transport of in-stream large wood by river flow.

Functions for ingrowth, diameter growth and mortality are derived from NE-TWIGS (Hilt and Teck, 1989). Snag fall rates are based on forest inventory data from Massachusetts, Maine and New England and snag fall angles and log breakage rates are based on data from Bragg et al (2000). Bank erosion functions are based on data from Idaho (Meleason, 2001 in Lester et al, 2003), decomposition and decay rates were derived from values cited in the literature for the North-eastern USA and are specified at the species level (Lester et al., 2003). A full model description is included in Lester et al, (2003). In previous applications of NE-CWD, in-stream wood loads were shown to be much higher than those found in natural managed forest streams, with highest accumulation rates found for 100-150 years after stand initiation (Nislow, 2010). However, the disparity between modelled and managed loads was explained in terms of the legacy of previous deforestation and forest management which have exerted a strong long-term influence on the structure and function of ecosystems (Jones et al., 1999; Bragg, 2000; Nislow, 2010).

Modelling approach

The modelling approach used herein is explicitly an exploratory or heuristic modelling exercise. The objective is not to model a specific forest, or to deliver quantitative predictions; instead we used the model with generalised parameters in order to understand the broad relationship between forest age and in-stream wood loadings and how these vary for different forest compositions. The results of the

model are therefore not treated as predictive or quantitative, and are analysed based on relative differences within and between modelling scenarios in order to understand how the trajectory of in-stream wood loadings change with floodplain forest development.

Ingrowth functions within NE-CWD are derived from an earlier growth model called NE-TWIGS, this is based on data from the USDA Forest Inventory and Analysis program for the 20 states in the North Eastern USA. This region is characterised by a cool-moist-temperate climate, with acidic, nutrient poor forest soils (Shifley et al., 2012). Within this region, forests cover 42% of the total land area, with around 70% of the forests between 40-100 years old (Shifley et al., 2012), with the most common forest types being Oak-Hickory and Maple-Beech-Birch forests which occupy 35% and 29% of forest cover, respectively (Smith et al., 2009). The dynamics for snag fall, bank erosion and log breakage in NE-CWD are based on a wide variety of studies and are assumed to be fundamental processes which are not dependent on climatic variables (Liu and Malanson, 1992). Variations in soil type, temperature, climate and elevation will lead to variations in tree growth and mortality between regions (Liu and Malanson, 1992). However, it is important to note there is a great deal of variation in live tree growth rates, deadwood accumulation (Hély et al., 2000; Christensen et al., 2005; Lombardi et al., 2011) and decay rates (Boddy and Swift, 1983), at both an inter-continental but also intra-regional, and even intra-basin level (Hough-Snee et al., 2016). Therefore, results of any forest growth modelling exercise which are not explicitly calibrated

to a given stand should not be treated as explicit quantitative predictions; rather they should be used to inform management on the trajectory and magnitude of likely forest growth.

Model set up

A Monte Carlo approach is used with 100 replicates of the initial model set up run and averaged. Live trees, deadwood and snags are reported on a per unit area basis for the floodplain components, and on a per reach length basis for in-channel components. Model timestep is yearly and output is written for every 5 years of model simulation for a total model simulation time of 200 years. Windstorm and harvesting options are disabled, and decay of both logs and snags results in their removal from the simulation when their density is below 1.6 kg/m^3

Tree composition

Four modelling scenarios for different forest composition were run; i) mixed beech, birch, oak and pine ("mixed"), ii) mixed beech, birch and oak ("deciduous", iii) pine monoculture ("conifer"), and iv) beech monoculture ("beech"). The first two forest types characterise valley-bottom forest composition in temperate lowland rivers in both the north-eastern US and Europe (Peterken and Hughes, 1995), the second two represent commercial plantations and mixed forest cover, respectively.

Plot parameters

The plot area is 4000 m^2 with distance from the stream edge to the edge of the plot, perpendicular to the channel set at a minimum of 30 m, at which distance the in-

channel biomass is independent of plot dimensions. Plot slope angle is set to 0.260 degrees and channel width is set at 4.3m. These values correspond to typical lowland 2nd and 3rd order head water streams in the UK for which existing field data on stream characteristics, logjam and dead wood dynamics and abundance are available (e.g. Gregory et al., 1985; Gurnell et al., 1995; Gurnell and Sweet, 1998; Jeffries et al., 2003; Dixon and Sear, 2014; Dixon, 2016). The initial model set up includes a single tree in the smallest diameter class for each species, and otherwise the plot is bare earth.

Results and Discussion

All model runs show a number of similarities in forest composition relative to stand age and a similar pattern of forest development over time (Figures 2-4). The initial phase of development shows a rapid increase in total forest biomass over the first 80 years of forest growth, with biomass at, or asymptotically approaching, a maximum value at around 100 years. All model runs approach an equilibrium state at around 200 years where successful ingrowth of new trees is dependent on gap phase regeneration following the death of larger specimens (Figure 2). However, overall live biomass shows a slight decline over the final 100 years of the three scenarios including deciduous trees. Initially, (prior to ~25 years) there is negligible deadwood biomass either on the floodplain or in-stream (Figures 3 and 4). In the mature phase of the model (>100 years) all scenarios are at, or are asymptotically approaching, a maximum dead wood biomass value, although this maximum value varies between forest types. The total dead wood biomass values (Figure 3)

partly constrain the loadings of dead wood to the channel, with the difference between the total (Figure 3) and in-channel (Figure 4) dead wood values representing wood removed through fluvial transport plus an additional bank erosion input delivered to the channel. The model removes all in-channel wood less than 30 cm diameter; the remaining in-channel deadwood is therefore the stable fraction of wood delivered the channel and is a proxy for pieces of wood which can act as key pieces to anchor logjam formation. It is probable that some, if not all, transported in-stream wood will be trapped and deposited within the catchment (e.g. Braudrick et al., 1997; Gurnell et al., 2002; Bocchiola et al., 2008; Dixon and Sear, 2014), as stable large wood and logjams have been shown to be effective trapping locations for mobile wood in the channel (Davidson et al., 2015), furthermore, it is probable that some longer pieces less than 30cm in diameter would also remain stable . Although these values will be an underestimate of total in-stream dead wood biomass it is a useful measure of the likely relative abundance of logjam features in small and medium-sized forest streams (Gurnell et al., 2002), furthermore total and average in-stream wood volumes have been shown to be heavily dependent on the largest pieces of wood (e.g. Meleason et al., 2007). Values for in-channel deadwood in Figure 4 show some temporal fluctuation due to the influence of large pieces of wood within a relatively small plot size, as well as biomass being represented by density, which varies with decay stage of dead wood. The values for total dead wood

biomass illustrate the differences between tree species, with forest scenarios including conifers producing a greater volume of dead wood compared to mixed deciduous stands, with beech stands producing comparatively little dead wood (Figure 3). However, conifers produce dead wood of relatively small size fractions which is largely removed from the channel through fluvial transport, resulting in low levels of in-channel dead wood for conifer stands compared to mixed and mixed deciduous stands (Figure 4). Validation of riparian forest model output is problematic because there are relatively few studies reporting in-stream deadwood biomass in the context of forest stand age, or in comparison with live tree biomass for unmanaged forests unconfounded by historic management, or older 'carry over' wood from prior stands (see Table S1 in supplemental information). Even in the absence of field validation, forest modelling of processes, directionality and composition are still recognised as heuristically useful (Hanson et al., 1990). In order to constrain the magnitude of natural in-stream wood loadings values from the literature are plotted on Figure 4 as bars showing reported data ranges, these show values are of the same order of magnitude, with NE-CWD values likely underestimating total natural loadings as discussed above. Warren et al (2009) compared in-stream wood volumes and stand age in 28 streams in the North Eastern US. Their results are plotted on Figure 4 and demonstrate that measured volumes are broadly comparable to those predicted by the model.

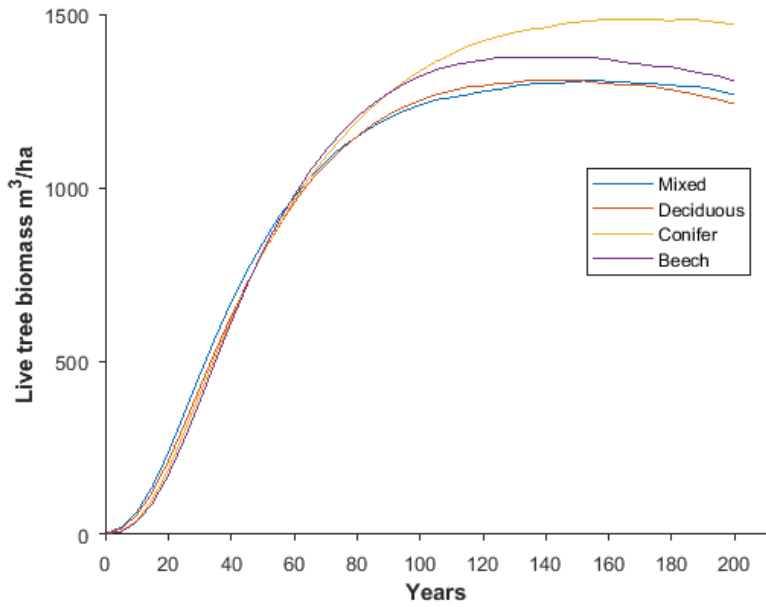Figure 2 – Accumulation of live tree biomass over time for the four different forest composition scenarios. Values shown are means of 100 Monte Carlo simulations.
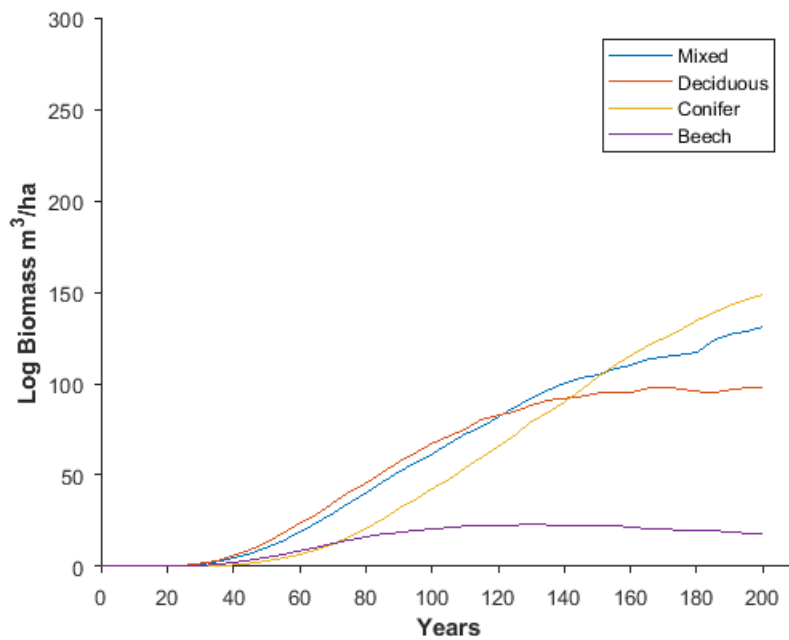

Figure 3 – Accumulation of dead wood biomass on the floodplain as logs for the four different forest composition scenarios. Note this figure excludes standing/rooted dead wood. Values shown are means of 100 Monte Carlo simulations.
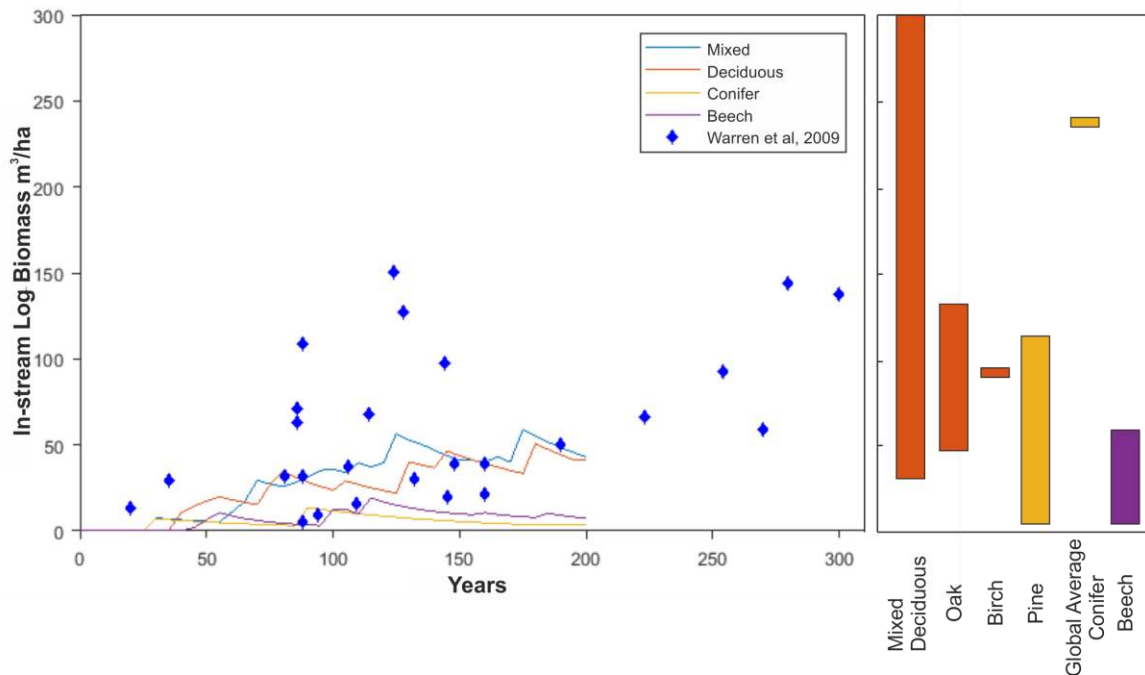
Figure 4 – Accumulation of in-stream dead wood biomass as logs for the four different forest composition scenarios. Values shown are means of 100 Monte Carlo simulations. Fluctuations relate to the relatively small plot size and thus influence of individual very large logs on totals. Field data for in-stream wood loads against estimated forest age from Warren et al (2009) are plotted on the graph for comparison. Literature values for in-stream deadwood are shown as ranges on the right (Harmon et al, 1986; Gurnell et al, 2002; Dixon, 2013; see Table 1, Supplemental information for full details of literature values)

The behaviour of the mixed and mixed deciduous stands is broadly similar in terms of live tree growth rates and dead wood accumulation both on the floodplain and in the channel. Beech stands produce much lower volumes of dead wood (Figure 3) and are thus characterised by low accumulation of logs on the floodplain and in the channel. However, although the values differ, the broad trajectory of riparian forest development is similar to the mixed stands. Conifer (*Pinus Sp.*) stands differ as they produce abundant dead wood on the floodplain as logs, but little in-stream dead wood; this is due to the logs being relatively small diameter which are readily removed through fluvial transport. Within the NE-CWD model for the combination of slope and channel size only deadwood in excess of ~30cm diameter will remain immobile, and the majority of conifer dead wood produced by the model is smaller than this.

The complexity of forest ecosystems makes it difficult to develop conceptual models of forest growth (Botkin et al., 1972), and this is especially true for riparian forests with additional allogenic disturbances (Hanson et al., 1990; Warren et al., 2016). The conceptual model proposed in Figure 5 shares similarities with observed riparian forest growth in other environments (e.g. Nanson and Beach, 1977; Naiman et al., 1998; Van Pelt et al., 2006), as well as other modelled results (Meleason et al., 2003);

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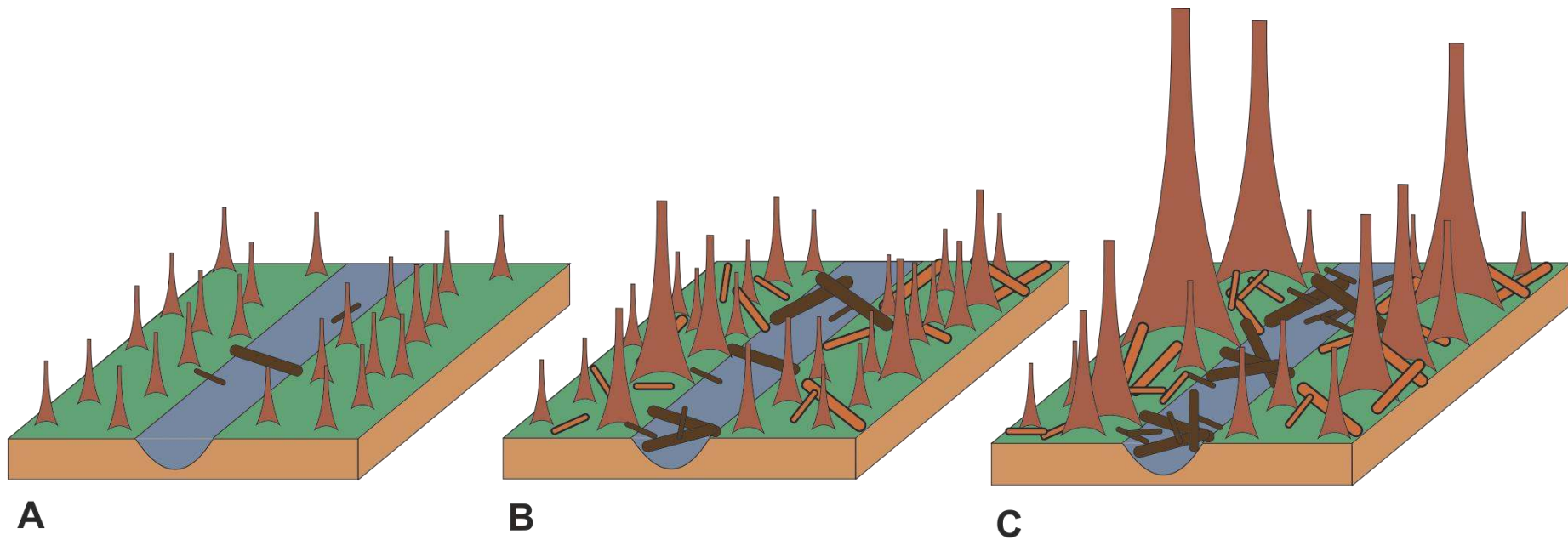
Figure 5 – conceptual model of broadleaf riparian forest succession following forest restoration to a bare earth site.

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

and the model expands on the theories put forward by Naiman et al (1998) (Figure 1) to explicitly include deadwood and the fluvial environment into a model of riparian forest succession. Notably, Van Pelt et al (2006) described a 300-year vegetation chronosequence for mixed riparian forests of the Pacific North West where deadwood was observed to be absent in young forests, as trees are not of a sufficient size to generate significant logs, with large logs not appearing until well into the second century post establishment (Van Pelt et al., 2006), with similar observations made by Cordova et al (2007) for young forests in the US Midwest.

Management Implications

The model results in Figures 2-4 and resulting conceptual model (Figure 5) suggest that the ecohydrological and geomorphological functioning of riparian forests are non-linear. This agrees with previous studies on the relationship between riparian forest age and function (Gregory et al., 2003; Keeton et al., 2007; Warren et al., 2009; Lorimer and Halpin, 2014; Reilly and Spies, 2015; Warren et al., 2016). The role of floodplain forests in natural flood management would likely be minimal until the second phase of the conceptual model (Figure 5) when the floodplain forest structure begins to be become more complex and dead wood delivered to the channel begins to reach sizes that are stable, which the numerical model suggests would be 25-40 years after stand establishment. Maximum effects on attenuating flood travel wave time would not be achieved until the final phase of the conceptual model which the numerical

model suggests could take more than 100 years post-stand establishment (Dixon et al., 2016).

The results of the modelling show that differences in forest composition partly control the timing and quantity of deadwood in the river network. Mixed deciduous woodland delivers the highest levels of floodplain deadwood biomass, as well as the highest levels of large, stable in-channel deadwood pieces, compared to conifer or beech woodlands (Figure 2-4). Therefore, in order to maximise the ecosystem service benefit of in-channel wood resource managers should focus on establishment of mixed deciduous riparian woodland.

It is important to note that the modelling does not include grazing, harvesting or other human intervention in forest growth. Previous modelling studies have shown that harvesting reduces the rates of delivery of deadwood to forested streams, attributed to reduced competition and thus mortality rates (Laser, 2007). Conversely, other studies have shown in-stream wood to be elevated by older 'carry over' wood from stand replacing disturbances in the study area (e.g. Hedman et al., 1996). Additional management or pressures on forest stands are likely to reduce the delivery of deadwood to the channel and thus resource managers need to consider the implications for on-going management. Managers may need to plan to monitor and intervene in restoration or natural flood management schemes to ensure objectives for biodiversity, wood accumulation, and natural capital are met.

Even given uncertainties in precise growth rates and timings of forests reaching

specific phases of development in the model there is a clear lag between the establishment of a new forest stand and the accumulation of stable pieces of large wood in the channel (Figure 4). This lag time varies with different forest compositions, but is at least 20-40 years, with peak in-channel large wood values not reached until around 100 years; this broadly agrees with other work showing in-stream wood loads are low for young riparian forest stands (Cordova et al., 2007) and reach a maximum in the later stages of succession (Richmond and Fauseh, 1995; Hedman et al., 1996; Gregory et al., 2003; Meleason et al., 2003; Keeton et al., 2007; Warren et al., 2009). It will therefore be important for resource managers to communicate with stakeholders to manage expectations for the delivery of benefits from restoration or natural flood management schemes. One possibility to bridge the gap between forest establishment and delivery of stable in-channel wood is to use engineered structures, such as engineered logjams, to provide some of the benefits of natural logjam structures in the interim period before the forest is able to provide a steady supply of deadwood to the channel (e.g. Bouwes et al., 2016). However, it will also be important to recognise that the increased connectivity between the channel and floodplain promoted by such structures may in turn influence the development of the forest stand.

The quantitative results of the modelling are uncertain, particularly once we move beyond 50-75 years of forest growth, this is because there are fewer studies in the literature reporting values for dead wood related to precise forest age for older, forests without the influence of

mangement, and relatively few studies reporting in-channel deadwood values related to forest age for riparian forest stands. However, the results of the exploratory modelling are still useful for guiding resource managers even the absence of comprehensive validation. Results indicate important differences between forest types, and establish estimates of timescales and magnitude of deadwood accumulation. Riparian forest growth models need improvement, both in terms of the representation of different processes and impacts, such as grazing and pathogens, as well as the mechanisms of fluvial transport of large wood. There is a need for much more data on the growth of riparian forest stands of different compositions and ages and their relationship to in-channel large wood loadings and logjam formation, particularly for stands without a legacy of management, or 'carry over' wood, in order to establish data sets for model calibration and validation.

Conclusion

1. Results from exploratory numerical modelling of riparian forest growth show that there is a lag of 20-40 years between the establishment of a new forest stand and the delivery of stable in-channel deadwood. This means resource managers need to be aware that Natural Capital benefits, including natural flood management are unlikely to be realised during this initial phase of forest growth without additional management intervention, for example using engineered logjams.

2. There are differences in deadwood biomass, both on the floodplain and in the

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channel for different forest compositions. Mixed deciduous, and mixed deciduous/conifer stands deliver higher volumes of deadwood biomass compared to beech or conifer stands. Therefore, to maximise deadwood delivery to the channel, and large, potential stable pieces of deadwood, resource managers should focus on locally appropriate mixed deciduous woodland species.

3. Model results suggest it will take in excess of 100 years after establishment of a new riparian forest stand before the forest reaches sufficient maturity to provide maximum benefits for natural flood management through development of a complex floodplain surface and abundant in-channel deadwood. Resource managers will, therefore, need to work with stakeholders to manage expectations as well as consider intervention initiatives to ensure that biodiversity and flood management aims are met.

4. Finally, this paper highlights the need for a) improved evidence to link stand composition and age with hydromorphologically relevant measures of wood loading and size, and b) development of modelling tools suited to emerging requirements to forecast the hydromorphological benefits of working with natural processes such as riparian and floodplain forests.

Acknowledgements

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Supplemental Info/Appendices

Location	Dominant tree species/Forest type	In-stream wood loading (m³/ha)	Reference
Worldwide Average	Conifer plantation	240	(Gurnell et al., 2002)
New Hampshire, USA	Conifer, unmanaged	30-80	(Harmon et al., 1986)
North-Western USA	Mixed, unmanaged	812	(Wohl and Jaeger, 2009)
New York and New Hampshire, USA	Mixed, unmanaged	6-237	(Warren et al., 2009)
Colorado, USA	Old Growth Spruce and Fir	92-254	(Richmond and Fauseh, 1995)
California, USA	Giant Redwood, unmanaged	240-4500	(Harmon et al., 1986; Keller et al., 1995 in Harmon et al., 1986)
USA	Fir, unmanaged	50-216	(Lambert et al., 1980; Harmon et al., 1986)
Tennessee, USA	Norway Spruce, unmanaged	140-220	(Harmon et al., 1986)
Idaho, USA	White Spruce, unmanaged	50-88	(Harmon et al., 1986)
Alaska, USA	Sitka Spruce, unmanaged	55-300	(Swanson et al., 1984 in Harmon et al., 1986)
British Columbia, Canada	Sitka Spruce, unmanaged	320-1700	(Toews and Moore, 1982 in Harmon et al., 1986; Hogan, 1987 in Harmon et al., 1986)

USA	Pine, unmanaged	30-82	(Sacket, 1979 in Harmon et al., 1986; Harmon et al., 1986)**
Idaho, USA	Pine, unmanaged	2.5-120	(Harmon et al., 1986)
Washington, USA	Douglas Fir, unmanaged	308-1421	(Franklin et al., 1981 in Harmon et al., 1986; Huff, 1984 in Harmon et al., 1986; Harmon et al., 1986)**
Oregon, USA	Douglas Fir, unmanaged	45-1200	(Harmon et al., 1986)
California, USA	Douglas Fir, unmanaged	10-1200	(Harmon et al., 1986)
California, USA	Giant Sequoia, unmanaged	555-1000	(Harmon et al., 1986)
USA	Birch, unmanaged	82	(Harmon et al., 1986)
USA	Yellow Poplar, unmanaged	51	(Harmon et al., 1986)
USA	Mixed Oak, unmanaged	46-94	(Harmon et al., 1986)
USA	Chestnut Oak, unmanaged	132	(Harmon et al., 1986)
Tennessee, USA	Mixed hardwood, unmanaged	40-300	(Harmon et al., 1986)
Nisqually River, Washington, USA	Mixed, unmanaged (>80% conifer)	633	(Collins et al., 2002)
Snohomish River, Washington, USA	Immature riparian forest, leveed river	52	(Collins et al., 2002)

Stillaguamish River, Washington, USA	Immature riparian forest, leveed river	24	(Collins et al., 2002)
South-Eastern USA	Mixed Deciduous	92-339	(Hedman et al., 1996)

Table S1 – showing in-stream large wood loading for forests of varying types and with varying dominant species across a range of locations. These data show in-stream large wood loadings are highly variable even within the same type of forest and same geographical region.