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ESEX Commentary

Wolf-triggered trophic cascades and stream channel dynamics in Olympic National Park: a comment on East *et al.* (2017)

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Overview

East *et al.* (2017) concluded that long-term channel widths and braiding of Olympic National Park rivers were significantly influenced by peak flow magnitude and were inversely associated with a slow decline in the abundance of Roosevelt Elk (*Cervus elaphus*) over a period of 65 years. We consider some of their results and conclusions to be in error since the effects of highly altered riparian plant communities, effects that began after the extirpation of gray wolves (*Canis lupus*) in the early 1900s and which became increasingly severe during the remainder of the twentieth century, were largely ignored or misrepresented. Overall, we conclude that the evidence provided by East *et al.* (2017) does not diminish the importance of trophic cascades whereby the loss of an apex predator allowed a native large herbivore to increasingly alter riparian plant communities over time, thus contributing to greater channel widths and braiding of the park's rivers.

Outline

In their recent study, East *et al.* (2017) utilized a chronosequence of 1939 to 2013 aerial photographs for the Hoh, Queets, and Quinault Rivers in the western portion of Olympic National Park to quantify long-term changes in channel planform and braiding. As part of their study, two working hypotheses were identified. Their primary hypothesis (i.e. flow hypothesis) indicated that 'river discharge drives channel morphology' and thus greater channel width and braiding should correspond to greater flow magnitude. From their analysis, they concluded that channel planform of the park's western rivers was driven primarily by hydrologic forcing (peak flows) with significantly greater active channel widths in recent decades. They also investigated an alternative hypothesis (i.e. biological hypothesis) whereby Roosevelt Elk abundance may drive channel morphology. However, they found increased active channel widths and braiding occurring during a period when elk abundance was slowly decreasing. In general, their study attempted to provide an alternative viewpoint on several topics that we evaluated approximately a decade earlier (i.e. Beschta and Ripple, 2008). Herein, we offer comments and concerns about their two main hypotheses using results from Beschta and Ripple (2008), as well as other studies, for comparative purposes. Although the East *et al.* (2017) study also included the Elwa River, a river that drains the northern portion of Olympic National Park, our earlier study only considered the three major western rivers and thus we confine our comments to these rivers.

Biological Hypothesis

Highly productive coastal forests occur along the western Olympic Peninsula due to large amounts of annual precipitation, temperature regimes moderated by the Pacific Ocean, and substantial amounts of sunlight in the summer months (Franklin and Dyrness, 1973; Grier, 1979; Balian and Naiman, 2005), resulting in high levels of above- and below-ground biomass (Jackson *et al.*, 1996; Hudiburg *et al.*, 2009). Within riparian areas, various coniferous and deciduous woody species collectively provide an array of ecological functions, including the mediation of microclimates, shading of river waters in summer, sources of organic matter to soils and rivers, and stabilizing river banks, floodplains, and terraces (National Research Council, 2002; Hupp and Bornette, 2003; Bennett and Simon, 2004). During high flows, river banks occupied by densely occurring woody plant communities typical of Olympic rain forests normally impart a high level of resistance to fluvial erosion via two primary mechanisms: (1) a relatively large hydraulic roughness from any submerged stems and branches that reduces shear stress on river banks and (2) extensive root systems that physically bind soil particles (Gore, 1985; Sedell and Beschta, 1991). This capability of riparian vegetation to help maintain bank stability, as well as significantly influence the quality of aquatic ecosystems, became increasingly recognized in the late 1900s (e.g. Johnson *et al.*, 1985; Salo and Cundy, 1987; Meehan, 1991). Indeed, Washington and many other western states have established buffer zones along forested rivers and streams for the express purpose of providing special levels of protection to riparian plant communities.

With perhaps the exception of red alder (*Alnus rubra*) and sitka spruce (*Picea sitchenis*), the vast majority of young shrub and tree species that occur along rivers within Olympic National Park are highly palatable to ungulates (Schwartz and Mitchell, 1945; Moorehead, 1994). Thus, following the extirpation of gray wolves in the early 1900s (Ratti *et al.*, 2004), biologists began to consistently report that palatable woody species were experiencing increased herbivory by elk (e.g. Bailey, 1918; Webster, 1922; Murie, 1935; Sumner, 1938). In more recent decades, various vegetation studies have also indicated woody species have been generally unable to successfully establish and grow (e.g. Fonda, 1974; McKee *et al.*, 1982; Harmon and Franklin, 1983; Van Pelt *et al.*, 2006).

In 2005 and 2006 we undertook field measurements aimed at assessing the long-term status of riparian plant communities along three major rivers draining the western portion of Olympic National Park, namely the Hoh, Queets, and Quinault Rivers, reporting those results in Beschta and Ripple (2008). The resulting age structure data (i.e. frequency of stems by date of establishment) for black cottonwood (*Populus trichocarpa*) and bigleaf maple (*Acer macrophyllum*), long-lived deciduous woody species with vastly different autecologies, confirmed that their recruitment (i.e. growth above the browse level of elk) began a major decline in the early 1900s, a decline associated with increased elk herbivory after wolf extirpation (Beschta and Ripple, 2008). Furthermore, recruitment levels of black cottonwood during the 1940s, 1950s, and 1960s were only 1–3% of expected and ceased entirely after the 1960s (Figure 1a). Similarly, Jenkins (1981, p. 32) indicated that 'persistent browsing by deer and elk may inhibit the establishment and survival of cottonwood, bigleaf maple, elderberry and willow.' These results, along with those of other riparian vegetation studies conducted in Olympic National Park, strongly suggest that the long-term deterioration in woody plant communities has been a primary cause of the decline in elk numbers over a 65-year period ($\bar{x} = -0.5\%\ \mathrm{yr}^{-1}$) and that the inverse association of both channel width and braiding with slowly declining elk abundance, which East *et al.* (2017) identify as 'counterintuitive,' appears to represent a spurious correlation.

In their discussion, East *et al.* (2017) indicated that the effects of elk browsing and trampling were greatest some 80 to 90 years ago, a conclusion not supported by age structure data for black cottonwood, bigleaf maple, or the results of other vegetation studies in western Olympic National Park. Nevertheless, they used this faulty assumption as a basis for generally dismissing the importance of top-down trophic cascades upon plant communities and rivers. We would instead suggest that the increasingly severe impacts of elk on riparian vegetation throughout most of the 1900s (e.g. Figure 1a) has been a major contributor to increased channel widening and braiding over time for Olympic National Park rivers and explains the large differences in channel geometry associated with rivers inside and outside the park (Beschta and Ripple, 2008).

We should also note that, following large carnivore loss, major temporal declines in woody species recruitment have been observed in other national parks of western North America, including Wind Cave, Yellowstone, Yosemite, and Zion in the United States (Barmore, 2003; Beschta and Ripple, 2009) as well as Banff, Jasper, Kootenay, and YoHo in Canada (White *et al.*, 1998; Beschta and Ripple, 2007; Hebblewhite and Smith, 2010). Even though Yellowstone and Zion National Parks utilized culling practices to reduce ungulate numbers by as much as 50%, extensive impacts to riparian and upland plant communities continued to become more severe over time (Ripple and Larsen, 2000; Ripple and Beschta, 2004, 2006; Beschta, 2005; Halofsky and Ripple, 2008). In addition to rivers of Olympic National Park, increased river bank erosion and channel widening following large predator loss has been documented for the Gallatin River in Yellowstone National Park (Beschta and Ripple, 2006) as well the North Fork of the Virgin River in Zion National Park (Ripple and Beschta, 2006), areas representing different (1) ecoregions and (2) peakflow regimes (i.e. rain-on-snow peakflows in Olympic National Park, snowmelt peakflows in Yellowstone National Park, and flash flooding from summer thunderstorms in Zion National Park).

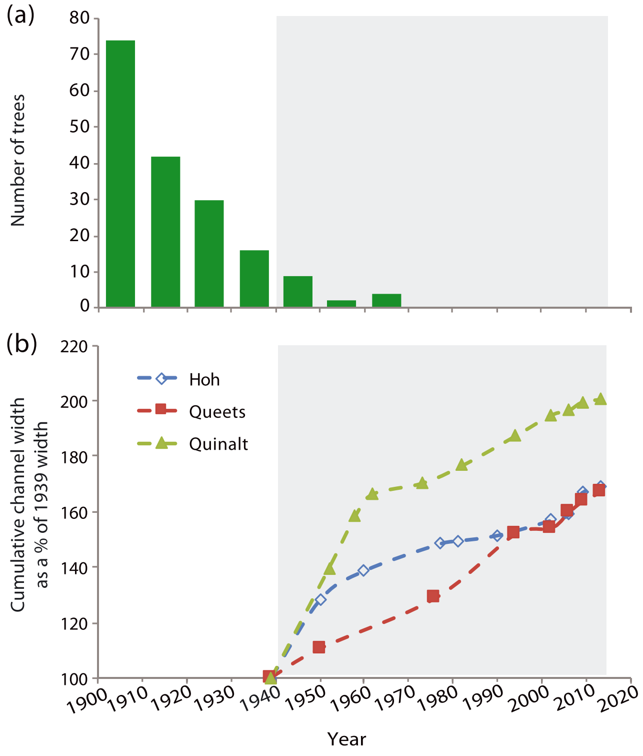

Figure 1. (a) Age structure of black cottonwood (i.e. number of trees by decade of establishment) since 1900 for riparian areas along the Hoh, Queets, and Quinault Rivers (adapted from Beschta and Ripple, 2008) and (b) cumulative active channel footprint for the Hoh, Queets, and Quinault Rivers as a percentage of the 1939 active channel (adapted from East *et al.*, 2017); data for (a) and (b) are for study areas inside Olympic National Park. The shaded portion of each panel emphasizes the years 1939–2013 when channel planform measurements were performed by East *et al.* (2017); note the exceptionally low levels of cottonwood recruitment in the 1940s, 1950s, and 1960s and the complete lack of recruitment in subsequent decades. Overall, these results are consistent with the hypothesis that decreased recruitment of woody plants, due to high levels of elk herbivory following the extirpation of wolves, were a major contributor to the cumulative increase in active channel widths measured between 1939 and 2013. [Colour figure can be viewed at wileyonlinelibrary.com]

Flow Hypothesis

In the Beschta and Ripple (2008) study, we utilized 1994 orthophotos for channel morphology measurements (i.e. active channel width and braiding) along sections of western Olympic rivers both inside versus outside the park (i.e. spatial controls). Without a chronosequence of aerial photographs we were unable determine temporal patterns of channel change over time, such as was accomplished by East *et al*. (2017). However, our spatial comparison of active channel width and braiding for the Hoh, Queets, and Quinault Rivers inside the park, where elk have been protected from hunting since 1905 and plant communities extensively degraded by elk herbivory (Figures 2a–2c), were significantly greater than for the Quinault and Clearwater Rivers outside the park, where ungulate hunting and recruitment of deciduous woody species have continued to occur (Figure 2d). Active channel widths were 2.8–3.5 and 1.5–1.6 times larger than wetted widths for rivers inside and outside the park, respectively. Similarly, the percentage of reach length with channel braiding averaged between 19% and 70% for rivers inside the park, but only 0% to 3% for rivers outside the park. These differences were consistent with highly degraded riparian plant communities inside the park contributing to accelerated rates of riverbank erosion and channel widening. Had East *et al*. (2017) undertaken a chonosequence assessment of channel morphology for rivers outside the park it would have provided an important perspective regarding how these channels have responded to peakflows over time for comparison with those inside the park, but unfortunately they did not.

The analysis of peakflow discharge records by East *et al*. (2017) indicated that the two-year recurrence interval discharge (i.e. Q2) during the most recent approximately four decades was somewhat higher, +11%, +12%, and +36% for the Hoh, Queets, and Quinault Rivers, respectively, than for the entire period of record (approximately seven decades). Results of their Akaike information criterion (AIC) analysis indicated wider channels and increased braiding were positively associated with flow magnitude (FLOOD), thus signifying planform morphology was likely to be more strongly influenced by the relatively high flows in recent decades. We would also suggest that the greater channel widths and braiding observed by East *et al*. (2017) in recent decades for these Olympic Park rivers are entirely consistent with the consequences of long-term degradation of riparian plant communities, a degradation strongly etched in the age structure of black cottonwood during the twentieth century (Figure 1a), as well as effects to other vegetation species (e.g. Sumner, 1938; Jenkins, 1981; Van Pelt *et al*., 2006; Beschta and Ripple, 2008). It is important to note that the larger channel planform measurements of East *et al*. (2017) occurred during the most recent decades, the exact period of time when recruitment of deciduous woody species in riparian areas had essentially ceased (Figure 1a; Beschta and Ripple, 2008) and thus had little capability of retarding river bank erosion during high flows.

To further pursue the topic of temporal trends in channel morphology, we utilized the digitized margins of unvegetated channels for reaches along the Hoh, Queets, and Quinault Rivers that were quantified by East *et al*. (2017, appendix). Starting with the digitized margins of each river in 1939 as the initial condition, we sequentially overlaid each successive margin to determine any additional unvegetated channel width that had occurred since the previous aerial photography. Any cumulative increase (%) in unvegetated channel width for each set of photographs was expressed relative to the 1939 unvegetated channel width (set at 100%). If we assume that

Figure 2. Examples of continuing floodplain and terrace erosion along the (a) Hoh, (b) Queets and (c) Quinault Rivers, inside Olympic National Park, and (d) a multi-aged stand of black cottonwood on a point bar of the lower Quinault River and outside of the Olympic National Park, indicating ongoing cottonwood recruitment over time (adapted from Beschta and Ripple, 2009). The dark textured soils and large trees, particularly evident in (a) and (b), indicate floodplains and terraces that have existed for long periods of time. In (a), (b), and (c) cottonwood, willows, and various deciduous woody shrub species that would normally be widespread along riverbanks and under forest canopies have been nearly entirely suppressed by elk browsing since the mid-1900s (e.g. Van Pelt *et al*., 2006; Beschta and Ripple, 2008), thus contributing to accelerated bank erosion along the park's rivers. The ongoing recruitment of cottonwood shown in (d) no longer occurs along rivers inside the park. [Colour figure can be viewed at wileyonlinelibrary.com]

the 1939 measurements represent an approximation of active channel widths before extensive changes in planform morphology had begun to occur from increased elk herbivory in riparian areas, then any increase in the channel-width 'footprint' would potentially indicate additional loss over time of formerly stable floodplain and terrace alluvium adjacent to each river, as well as the removal of old-growth forests (Balian and Naiman, 2005; Van Pelt *et al.*, 2006) normally found on these geomorphic surfaces.

Results indicate a continuously expanding footprint of cumulative active channel width from 1939 to 2013 (Figure 1b), signifying a persistently increasing volume of sediment entering each river. Since 1939, the cumulative footprint for these rivers increased some 67–101% (i.e. 175 to 296 m [69%], 165 to 275 m [67%], and 270 to 542 m [101%] for the Hoh, Queets, and Quinault, respectively), increases entirely consistent with the fact that riparian plant communities had been decimated by elk herbivory. Furthermore, an expanding footprint of active channel width over time indicates additional volumes of large wood have entered these channels, increasing the potential for major in-channel debris accumulations. Yet, in 2005 and 2006 such accumulations were noticeably absent due to the relatively wide channels of the park's rivers (Beschta and Ripple, 2008), a situation quite different than was observed by Euro-Americans over a century ago when the Press Expedition of 1890 found accumulations of large wood along the Quinault River were of sufficient size and frequency that it could not be floated (Lien, 2001).

During our 2005 and 2006 field study we noted that vertical eroding banks were common along the Hoh, Queets, and Quinault Rivers (Figure 2; see also Beschta and Ripple, 2008), thus making significant volumes of sediment locally available to river channels. For example, each 1-m increase in active channel width from fluvial erosion that removes a river bank 2-m in height (bulk density of ~2 g cm^{-3}; Brye *et al.*, 2004) represents a local addition of 4000 tonnes of sediment per kilometer of river length, as well as appreciable volumes of large wood. East *et al.* (2017) essentially ignore any effects of near-channel sources of sediment in their study reaches as a potential mechanism for augmenting ongoing increases in channel widths and braiding. Instead, they speculated that increased sediment might have become available from glacial ice retreat in headwater areas.

In 1980 the Hoh watershed contained an estimated 65% of parks glacial ice volume and since then ~34% of the park's glacial ice area has been lost (Riedel *et al.*, 2015). With regard to the 'strong temporal trend toward greater braiding' observed for the Hoh River study reach, East *et al.* (2017) 'surmised' that this braiding may have been a result of 'increased sediment supply due to glacial retreat in the upper watershed,' a sediment supply of unspecified magnitude, unknown particle size distribution, and requiring a 30-km transport distance before arriving at the study reach. Based on this presumed increase in sediment availability, they concluded that 'the downstream propagation of climate-driven geomorphic perturbations shows how effectively river systems with high connectivity in the sediment routing system can transmit environmental signals.'

Even if one accepts the conceptual argument that glacial retreat may have augmented sediment loads along the upper reaches of the Hoh River, we would suggest any ensuing effects on channel morphology at their study reach would be considerably less than the effects of locally derived increases in sediment (i.e. erosion of historical floodplains and terraces from a widening river footprint over time). With regard to the Queets and Quinault watersheds, they had only 9% and < 5% of the Olympic Mountain's ice volume, respectively, in 1980. Glacial retreat in their headwaters is likely to have been of little, if any, importance for affecting channel morphology along the Queets and Quinault study reaches of East *et al.* (2017) because (a) ice volume losses in their headwaters were small and (b) any potential sediment increase would again have a long transport distance, ~30 km. All major rivers draining the western portion of Olympic National Park, regardless of the presence or absence of glacial ice in their headwater reaches, demonstrate a pattern of relatively wide channels as indicated in the 1996 satellite imagery (Beschta and Ripple, 2008), a situation consistent with widespread impacts to riparian areas from intensive ungulate herbivory.

Overall, the relatively wide and braided channels of the Hoh, Queets, and Quinault Rivers in recent decades that was found by East *et al.* (2017) are consistent with the long-term effects of herbivory degraded riparian plant communities (Beschta and Ripple, 2008) whereby (1) unimpeded elk herbivory since the early 1900s impaired the capability of palatable woody species in riparian areas to become established and (2) these highly altered riparian plant communities allowed for accelerated riverbank erosion, thus significantly affecting long-term channel dynamics and contributing to wider channels and increased braiding. This intensive browsing-suppression of woody plant communities by elk has allowed high flows to increasingly erode floodplain and terrace alluvial deposits (Figure 1b), as well as remove their ancient forests, resulting in river channels today within Olympic National Park that are wider, more braided, and contain fewer debris accumulations than historically. Spatial comparisons of channel morphology for rivers inside and outside the park (Beschta and Ripple, 2008) further emphasize the importance and magnitude of these trophic cascade effects.

The Future of Olympic National Park's Riparian Systems and Rivers

Approximately two decades ago, gray wolves were reintroduced into Yellowstone National Park, as well as central Idaho, and subsequently they have re-colonized additional areas of the western United States. Gray wolves have also reoccupied portions of several Lake States in recent years. Where studies have addressed vegetation responses following the return of this apex predator, plant communities that formerly were being negatively affected by high levels of ungulate herbivory are beginning to recover. Woody plants previously suppressed by elk herbivory when wolves were absent are now increasingly able to establish and grow above the upper browse level of elk in the northern winter range of Yellowstone National Park (Beschta and Ripple, 2010, 2016; Painter *et al.* 2014), as well as in Banff National Park (Hebblewhite and Smith, 2010; White *et al.*, 2015), Jasper National Park (Beschta and Ripple, 2007), and northern Wisconsin (Bouchard *et al.*, 2013; Callan *et al.*, 2013; Flagel *et al.*, 2016). Collectively, various studies have found that (1) the removal of large mammalian predators allowed ungulates to significantly degrade plant communities and (2) the return of these predators has initiated recovery of herbivory-impacted plant communities.

If our hypothesis regarding the cascading trophic effects of gray wolf extirpation upon the riparian areas and rivers of Olympic National Park (Beschta and Ripple, 2008) is correct, this would seem to provide a compelling ecological argument for the return of this apex predator. The restoration of gray wolves could have substantial indirect benefits to riparian plant communities and food webs within the park's highly productive forest ecosystems, benefits that have been increasingly documented in Yellowstone's riparian areas following the 1995–1996 re-introduction of wolves (e.g. Ripple *et al.*, 2014; Beschta and Ripple, 2016).

Without wolves, all evidence indicates riparian vegetation along rivers of western Olympic National Park will continue to lose composition and structure (i.e. a loss of biodiversity) and remain functionally ineffective (e.g. a lack of hydraulic roughness and root strength) due to intensive elk herbivory, a circumstance that has been ongoing for decades. This situation is counter to one of the main goals of 1938 enabling legislation which emphasized the park's importance for preserving 'the finest sample of primeval forests in the entire United States.' The continued effects of intensive ungulate herbivory, in the absence of wolves, are also likely to insure that the park's western rivers will continue to transition toward an alternative state, a state characterized by ever greater active channel widths and braiding.

In order to restore the composition, structure, and function of riparian plant communities, as well as the long-term morphology of the Hoh, Queets, and Quinault River channels, it would seem imperative that the Park Service initiate a process leading to the re-introduction of gray wolves in Olympic National Park. If wolf restoration is accomplished, future scientists will be able to evaluate the effectiveness of this apex predator in the recovery of riparian plant communities, as well as assess the capability of such vegetation for stabilizing river banks, contributing to narrower and less braided channels, insuring a long-term supply of large wood, and improving instream habitat for aquatic species.

References

Bailey V. 1918. *Report on Investigation of Elk Herds in the Olympic Mountains, Washington*. Olympic National Forest: Olympia, WA.

Balian EV, Naiman RJ. 2005. Abundance and production of riparian trees in the lowland floodplain of the Queets River, Washington. *Ecosystems* **8**: 841–861.

Barmore WJ, Jr. 2003. *Ecology of Ungulates and their Winter Range in Northern Yellowstone National Park*. Yellowstone Center for Resources: Yellowstone National Park, WY.

Bennett SJ, Simon A (eds). 2004. *Riparian Vegetation and Channel Morphology, Water Science and Application 8*. American Geophysical Union: Washington, DC.

Beschta RL. 2005. Reduced cottonwood recruitment following extirpation of wolves in Yellowstone's Northern Range. *Ecology* **86**: 391–403.

Beschta RL, Ripple WJ. 2006. River channel dynamics following extirpation of wolves in northwestern Yellowstone National Park, USA. *Earth Surface Processes and Landforms* **31**: 1525–1539.

Beschta RL, Ripple WJ. 2007. Wolves, elk, and aspen in the winter range of Jasper National Park, Canada. *Canadian Journal of Forest Research* **37**: 1–13.

Beschta RL, Ripple WJ. 2008. Wolves, trophic cascades, and rivers in the Olympic National Park, USA. *Ecohydrology* **1**: 118–130.

Beschta RL, Ripple WJ. 2009. Large predators and trophic cascades in terrestrial ecosystems of the western United States. *Biological Conservation* **142**: 2401–2414.

Beschta RL, Ripple WJ. 2010. Recovering riparian plant communities with wolves in northern Yellowstone, USA. *Restoration Ecology* **18**: 380–389.

Beschta RL, Ripple WJ. 2016. Riparian vegetation recovery in Yellowstone: the first two decades after wolf reintroduction. *Biological Conservation* **198**: 93–103.

Bouchard K, Wiedenhoeft JE, Wydeven AP, Rooney TP. 2013. Wolves facilitate the recovery of browse-sensitive understory herbs in Wisconsin forests. *Boreal Environment Research* **18**(Supplement A): 43–49.

Brye KR, Morris TL, Miller DM, Formica SJ, van Eps MA. 2004. Estimating bulk density in vertically exposed stoney alluvium using a modified excavation method. *Journal of Environmental Quality* **33**: 1937–1942.

Callan R, Nibbelink RTP, Wiedenhoeft JE, Wydeven AP. 2013. Recolonizing wolves trigger a trophic cascade in Wisconsin (USA). *Journal of Ecology* **101**: 837–845.

East AE, Jenkins KJ, Happe PJ, Bountry JA, Beechie TJ, Mastin MC, Sankey JB, Randle TJ. 2017. Channel-planform evolution in four rivers of Olympic national Park, Washington, USA: the role of physical drivers and trophic cascades. *Earth Surface Processes and Landforms* **42**(7): 1011–1032. https://doi.org/10.1002/esp.4048.

Flagel DG, Belovsky GE, Beyer DE, Jr. 2016. Natural and experimental tests of trophic cascades: gray wolves and white-tailed deer in a Great Lakes forest. *Ocelogia* **180**: 1183–1194.

Fonda RW. 1974. Forest succession in relation to river terrace development in Olympic National Park, Washington. *Ecology* **55**: 927–942.

Franklin JF, Dyrness CT. 1973. *Natural Vegetation of Oregon and Washington*, USDA Forest Service, General Technical Report PNW-8. USDA Forest Service: Portland, OR.

Gore JA (ed). 1985. *The Restoration of Streams and Rivers: Theories and Experience*. Butterworth Publishers: Boston, MA.

Grier CC. 1979. Productivity assessment of pacific northwest forests. In *Forest Fertilization Conference, Institute of Forest Research Contribution 40*, Gessel SP, Kenady RM, Atkinson WA (eds). Seattle: University of Washington; 23–28.

Halofsky J, Ripple WJ. 2008. Linkages between wolf presence and aspen recruitment in the Gallatin elk winter range of southwestern Montana, USA. *Forestry: An International Journal of Forest Research* **81**: 195–207.

Harmon M, Franklin JF. 1983. Age distribution of western hemlock and its relation to Roosevelt elk populations in the South Fork Hoh River Valley, Washington. *Northwest Science* **57**: 249–255.

Hebblewhite M, Smith DW. 2010. Wolf community ecology: ecosystem effects of recovering wolves in Banff and Yellowstone National Parks. In *The World of Wolves: New Perspectives on Ecology, Behaviour and Management*, Boitani L, Musiani M (eds). University of Calgary Press: Calgary, AL.

Hudiburg T, Law B, Turner DP, Campbell J, Donao D, Duane M. 2009. Carbon dynamics of Oregon and northern California forests and potential land-based carbon storage. *Ecological Applications* **19**: 163–180.

Hupp CR, Bornette G. 2003. Vegetation as a tool in the interpretation of fluvial geomorphic processes and landforms in humid temperate areas. In *Tools in Fluvial Geomorphology*, Kondolf GM, Piegay H (eds). John Wiley & Sons: San Francisco, CA.

Jackson RB, Canadell J, Ehleringer JR, Mooney HA, Sala OE, Schulze ED. 1996. A global analysis of root distributions for terrestrial biomes. *Oecologia* **108**: 389–411.

Jenkins K. 1981. *Status of Elk Populations and Lowland Habitats in Western Olympic National Park*, Oregon Cooperative Park Studies Unit, Report 81-1. Oregon State University: Corvallis, OR.

Johnson RR, Ziebell CD, Patton DR, Ffolliott PF, Hamre RH (eds). 1985. *Riparian Ecosystems and their Management: Reconciling Conflicting Uses*, USDA Forest Service, General Technical Report RM-120. USDA Forest Service: Fort Collins, CO.

Lien C. 2001. *Exploring the Olympic Mountains: Accounts of the Earliest Expeditions, 1878–1890*. Mountaineers Books: Seattle, WA.

McKee A, LaRoi G, Franklin JF. 1982. Structure, composition, and reproductive behavior of terrace forests, South Fork Hoh River, Olympic National Park. In *Ecological Research in National Parks of the Pacific Northwest: Proceedings*, Franklin JF, Matthews JW (eds). Forest Research Laboratory, Oregon State University: Corvallis, OR.

Meehan WR (ed). 1991. *Influences of Forest and Range Management on Salmonid Fishes and their Habitats*, American Fisheries Society Special Publication 19. American Fisheries Society: Bethesda, MD.

Moorehead BB. 1994. *The Forest Elk; Roosevelt Elk of the Olympic National Park*. Northwest Interpretive Association: Seattle, WA.

Murie OJ. 1935. *Report on the Elk of the Olympic Peninsula*. US Biological Survey: Jackson, WY.

National Research Council. 2002. *Riparian Areas: Functions and Strategies for Management*. National Academy Press: Washington, DC.

Painter LE, Beschta RL, Larsen EJ, Ripple WJ. 2014. After long-term decline, are aspen recovering in northern Yellowstone? *Forest Ecology and Management* **329**: 108–117.

Ratti JT, Scott JM, Wiseman PA, Miller CA, Szepanski MM, Svancara LK. 2004. Feasibility of wolf reintroduction to Olympic Peninsula, Washington. *Northwest Science* **78**(special issue): 1–76.

Riedel JL, Wilson S, Baccus W, Larrabee M, Fudge TJ, Fountain A. 2015. Glacier status and contribution to streamflow in the Olympic Mountains, Washington, USA. *Journal of Glaciology* **61**: 8–16.

Ripple WJ, Beschta RL. 2004. Wolves, elk, willows, and trophic cascades in the upper Gallatin Range of southwestern Montana, USA. *Forest Ecology and Management* **200**: 161–181.

Ripple WJ, Beschta RL. 2006. Linking a cougar decline, trophic cascade, and catastrophic regime shift in Zion National Park. *Biological Conservation* **133**: 397–408.

Ripple WJ, Estes JA, Beschta RL, Wilmers CC, Ritchie EG, Hebblewhite M, Berger J, Elmhagen B, Letnic M, Nelson MP, Schmitz OJ, Smith DW, Wallach AD, Wirsing AJ. 2014. Status and ecological effects of the world's largest carnivores. *Science* **343**(6167) 1241484. DOI: https://doi.org/10.1126/science.1241484.

Ripple WJ, Larsen EJ. 2000. Historic aspen recruitment, elk, and wolves in northern Yellowstone National Park, USA. *Biological Conservation* **95**: 361–370.

Salo EO, Cundy TW (eds). 1987. *Streamside Management: Forestry and Fisheries Interactions*, Contribution No. 57. College of Forest Resources, University of Washington: Seattle, WA.

Schwartz JE, Mitchell GE. 1945. The Roosevelt Elk on the Olympic Peninsula, Washington. *Journal of Wildlife Management* **9**: 295–319.

Sedell JR, Beschta RL. 1991. Bringing back the 'Bio' in bioengineering. *American Fisheries Society Symposium* **10**: 160–175.

Sumner EL. 1938. *Special Report of Elk in Olympic National Park*. USDI National Park Service: San Francisco, CA.

Van Pelt R, O'Keefe TC, Latterell JJ, Naiman van Pelt R, O'Keefe TC, Latterell JJ, Naiman RJ. 2006. Riparian forest stand development along the Queets River in Olympic National Park, Washington. *Ecological Monographs* **76**: 277–298.

Webster EB. 1922. *State and Forest Officials Make Study of Conditions of Elk Herds of the Hoh*. Olympia, WA: Olympic National Forest.

White CA, Buckingham A, Hebblewhite M. 2015. *'Shrubwatch' Monitoring Bow Valley, Banff National Park*, Report to Parks Canada. Summit Environmental Consulting: Canmore, AL.

White CA, Olmsted CE, Kay CE. 1998. Aspen, elk, and fire in the Rocky Mountain national parks of North America. *Wildlife Society Bulletin* **26**: 449–462.