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Stream Conditions after 18 Years of Passive Riparian Restoration in Small Fish-bearing Watersheds

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Abstract
Many of the ecological processes in the riparian forests and streams across the Pacific Northwest have become impaired through production forestry practices common prior to the 1990s. Some of these practices included forest harvest without stream buffers, removal of instream wood, road construction and use, and harvesting large proportions of watersheds. Passive ecological restoration (the use of natural processes of succession and disturbance to alleviate anthropogenic impacts over time) is a common practice used in the management of riparian forests previously subjected to production forestry. Eighteen years after the implementation of passive restoration of riparian forests, we used four common stream indicators (stream temperature, canopy closure, instream wood, and salmonid densities) to assess the effects of restoration in small fish-bearing streams. Summer stream temperatures have decreased below unmanaged reference levels, whereas riparian forest canopy closure has increased beyond that in reference watersheds. Instream wood and age-1 or older salmonids appear to be either stable at reduced levels or declining, compared with production forestry and unmanaged reference watersheds. Overall, second-growth riparian forests need more time to develop allowing more light into streams (increasing primary productivity), while also allowing for the continuous recruitment of larger pieces of instream wood (improving habitat for salmonids). Using only passive restoration, stream conditions in second-growth forests are unlikely to increase salmonid production in the near future.

Keywords Forest recovery · Riparian · Olympic Peninsula · Second-growth forests · Passive restoration · Active restoration

Introduction

Forest management in the Pacific Northwest region of North America changed in the 1990s with the realization that historic timber harvest and road construction practices were negatively influencing fish and wildlife populations (Naiman 1992; WADNR 1997; Thomas et al. 2006). Forest management in the Pacific Northwest was identified as a major factor leading to population declines of northern spotted owl (*Strix occidentalis caurina*), marbled murrelet (*Brachyramphus marmoratus*), and numerous salmon stocks (Cederholm and Reid 1987; Marshall 1988; Doak 1989; USFWS 1990, 1992; Nehlsen et al. 1991; FEMAT 1993). Many of these declines resulted in subsequent listings of species under the federal Endangered Species Act (ESA). The extent of management disturbance was evident in the state lands of western Washington managed by the Washington Department of Natural Resources (WADNR), where most of the forests were harvested at least once by 1995, leaving only 6% of the forests older than 160 years (WADNR 1997). In particular, past forest harvest practices in riparian forests had negative impacts on riparian and stream biota, including salmonids (Newbold et al. 1980; Johnson et al. 1986; Corn and Bury 1989; Young et al. 1999). These harvest practices included clear-cutting without buffers (Heifetz et al. 1986; Connolly and Hall 1999; Richardson and Béraud 2014), removal of instream wood (Mellina and Hinch 2009), road construction (Sheer and Steel 2006), and working on unstable banks (Cederholm and Reid 1987). A number of conservation measures were put in place on private and public lands in the 1990s to minimize negative ecological impacts of forest management

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and to restore riparian and aquatic habitat (Bilby and Wasserman 1989; USDA and USDI 1994; WADNR 1997). For state lands of western Washington, this resulted in increased riparian protections that allowed most of these forests to recover through natural processes.

From an ecological perspective, riparian forests stabilize stream banks and influence the quality of spawning sediment and channel formations (Beschta and Platts 1986). The amount of forest canopy controls the shading along streams that affects air and water temperatures as well as primary productivity (Warren et al. 2013). Stream temperature is a primary factor regulating the survival and productivity of aquatic communities (Vannote and Sweeney 1980; Taniguchi et al. 1998). Both vegetation composition and densities of riparian forests are important for providing allochthonous inputs that structure aquatic food webs (Allan et al. 2003; England and Rosemond 2004; Richardson et al. 2005). In addition, riparian forests provide a source of instream wood that increases stream complexity (especially pool formation) and sediment storage (Bilby and Ward 1989; Wood-Smith and Buffington 1996; Pollock and Beechie 2014). Instream wood is important for fish abundance (Fausch and Northcote 1992; Rosenfeld et al. 2000; Giannico and Hinch 2003; Johnson et al. 2005). Salmonids typically show an initial positive response to forest harvest, associated with increased primary productivity due to canopy removal (Bisson and Sedell 1984; Murphy et al. 1986; Holtby 1988). However, long-term decreases in salmonid production can occur from dense early-seral regrowth affecting stream shading and wood recruitment (Connolly and Hall 1999; Kaylor et al. 2017; Tschaplinski and Pike 2017).

The goal of ecological restoration is to reestablish the natural path of historic succession in forests and streams (Balaguer et al. 2014). Christensen (2014) suggested that ecological restoration goals should have (1) a full range of variation in species diversity associated with disturbance and forest succession; (2) pattern, scale, and context influenced by both disturbance and succession; and (3) restoration designed to meet the desired trajectories and succession of the ecosystem. Therefore, ecological restoration should capture the natural complexity of the landscape so that ecosystems can better withstand degradation in present and future conditions (Halme et al. 2013). Two strategies (active and passive) exist for restoring ecological processes of degraded aquatic ecosystems. Active restoration involves human interventions such as riparian silviculture to promote the development of complex forest structure, placement of large wood in streams, dam removal, and removal of invasive species. Passive restoration allows for natural processes of forest succession and disturbance to alleviate anthropogenic impacts over time (Wissmar and Bechta 1998). When implemented deliberately, passive restoration is an effective, cost-efficient, and sustainable strategy (Kauffman et al. 1995; Wissmar and Bechta 1998; Montgomery and Bolton 2003). The use of passive restoration avoids the risk of creating conditions outside of the natural range, which could create counterproductive stages in habitat development (Pollock et al. 2005). However, passive restoration may take too long to help threatened and endangered species (Montgomery and Bolton 2003).

The recovery rates and trajectories achieved through passive restoration vary by habitat attribute (Ziemer et al. 1991; Hartman et al. 1996). In the dense coniferous forests of the coastal Pacific Northwest, stream temperatures are one of the first habitat attributes to recover after forest canopy removal. Temperatures typically return to pre-harvest levels within 5–15 years (Moore et al. 2005), yet canopy closure, instream wood, and fish populations take considerably longer to recover to pre-disturbance levels. Kaylor et al. (2017) found that riparian canopies in Pacific Northwest forests had an initial recovery period within 30 years of harvest, followed by a period of maximum canopy density and shade (30–100 years), and a final period (≥300 years old) where canopy structure diversified. Connolly and Hall (1999) speculated that trout production might continue to decrease for 70 or more years after riparian harvest due to lack of instream wood and heavy canopy shading.

In this study, we distinguish the ecological conditions resulting from the two management approaches by categorizing them as production forestry or ecological forestry. Production forestry, which focuses on the efficient extraction of timber, is characterized by clear-cuts and short rotations of even-aged single-species plantations resulting in low structural complexity of forest stands and low spatial heterogeneity within stands and across landscapes (Franklin et al. 2007; Becknell et al. 2015). Ecological forestry, which seeks to sustain all ecological functions, is based on the understanding of natural disturbances and stand development processes (Franklin and Forman 1987; Franklin et al. 2007; Puettmann et al. 2015). It uses silviculture practices such as thinning and retention of biological legacies to produce stands with higher species diversity and greater structural complexity. In our comparative analyses, we use a third management category (reference) to describe ecological conditions in unharvested watersheds that are primarily influenced by natural disturbances (e.g., wind throw and debris flows).

As our knowledge of riparian forests and their connection with streams advances, questions remain whether streams and riparian forests should be managed to recover passively or if active management is needed to restore ecosystem processes. In this analysis, we use four key stream and riparian forest indicators to assess the outcome of 18 years of passive riparian restoration. We compare indicator values for sites passively restored under ecological forestry to values for unharvested reference sites and values for managed forests prior to the riparian conservation measures implemented in the 1990s (i.e., production forestry). We use the results to make recommendations regarding passive forest restoration in the Pacific Northwest.

Study Area, Management, and Monitoring

The study took place in the Olympic Experimental State Forest (OESF), a 523,000-ha planning area that contains 110,000 ha of state trust lands on the western Olympic Peninsula of Washington State, USA. All sites used in this study are on, or in the vicinity of, state lands within the OESF (Fig. 1). These state lands are located in the temperate rainforest zone of the Pacific Coastal Ecoregion (Naiman et al. 2000); elevation ranges from sea level to 1,155 m. The maritime climate receives heavy precipitation ranging from 203 to 355 cm per year with the majority falling as rain during the winter. The state lands of the OESF contain over 4,300 km of streams including portions of several major rivers such as the Queets, Clearwater, Hoh, Bogachiel, Calawah, Sol Duc, Dickey, Hoko, and Clallam (WADNR 2013). The smallest fish-bearing streams (stream order 1–3; Strahler 1957) contain various populations of juvenile coho salmon (*Oncorhynchus kisutch*), rainbow trout/steelhead (*Oncorhynchus mykiss*), coastal cutthroat trout (*Oncorhynchus clarkii clarkii*), lampreys (*Lampetra spp.*), and/or sculpins (*Cottus spp.*; Martens 2016).

Forest management and regulations in the OESF have varied historically through time and land ownership. The first forest practice rules for state and private lands were adopted in 1976 and significantly updated in 1988 to include riparian buffers as well as other regulations designed to protect and restore damaged riparian forests and streams (Bilby and Wasserman 1989; Tyler and Peterson 2004). Using production forestry, between 1970 and 1990, over half of the state forestlands in the OESF were clear-cut (WADNR 2016). Due to declining environmental conditions, the state's forest management approach transitioned to ecological forestry in 1997 (effectively implemented by 1999) guided by a Habitat Conservation Plan (HCP) negotiated under the ESA (WADNR 1997). The HCP riparian conservation strategy aims to maintain and aid restoration of riparian functions important to salmonid habitat by retaining riparian buffers, extending buffers in wind-prone areas, avoiding harvests in areas susceptible to landslides, protecting wetlands, and improving road conditions. Since its implementation 18 years ago, the riparian forests have mostly been left to recover through passive restoration, with the exceptions of a major effort to repair road crossings identified as potential fish-passage barriers and pre-commercial thinning in some of the young conifer stands (WADNR 1997). Riparian forest conditions on state lands in the OESF are mostly in the earlier stages of forest development (less than 80 years), with 70% of riparian areas dominated by hardwoods or young conifers (WADNR 1997).

WADNR has monitored aquatic and riparian habitat conditions (hereafter, "WADNR habitat monitoring") in 50 ecological forestry watersheds since 2012 and salmonids (hereafter, "WADNR salmonid monitoring") since 2016 to assess the effectiveness of the HCP. This monitoring takes place in WADNR Type-3 watersheds (stream order 1–3) with greater than 50% state ownership. In addition, four ecologically similar but unharvested reference watersheds are monitored in the adjacent Olympic National Park. Within each monitored watershed, the lowest 100 m of the stream is periodically sampled (Minkova and Foster 2017; Martens 2016).

Methods

To evaluate the effects of 18 years of passive restoration, we compared values of four ecological indicators (stream temperature, canopy closure, instream wood, and salmonid density) for watersheds passively restored under ecological forestry to values for watersheds managed under production forestry and to reference watersheds (Tables 1 and 2). For ecological forestry watersheds, we measured indicators through WADNR habitat monitoring and WADNR salmonid monitoring. For production forestry watersheds, we extracted indicator values from published data collected between 1973 and 1999. For the reference watersheds, we used indicator values from WADNR habitat monitoring as well as previously published data. The previously published data used in this analysis were collected on state, federal, and private lands.

To minimize potential influences of differences in underlying geophysical, biological, and climatic conditions, we used only data collected within the confines of the OESF planning area (Fig. 1). Additionally, we only selected published data reported at the individual watershed level (i.e., not averaged across watersheds) with overlapping ranges of watershed areas for analysis. Ralph et al. (1994) recommended the use of watershed area for comparing watersheds, because other metrics (e.g., bankfull width) may be influenced by management practices, and watershed area was a good indicator of stream size.

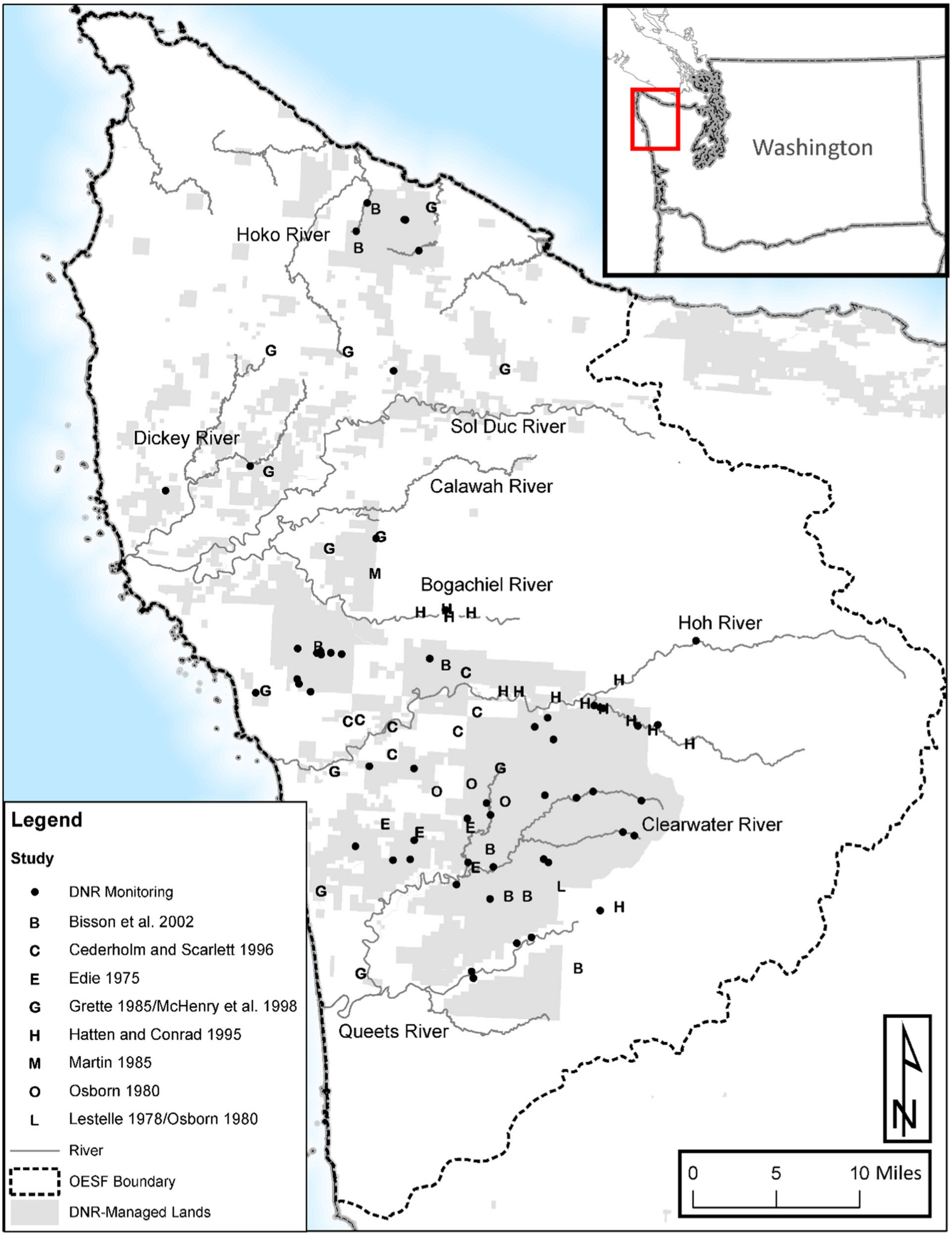

Fig. 1 Map of the OESF and locations of studies used for assessing passive riparian forest restoration

Table 1 Summary of the management history and overall stream and forest conditions of all analyzed sites

Study	Number of watersheds	Management history of watersheds	Watershed area (km^2)	Stream		Riparian forest	
				Bankfull width (m)	Gradient (%)	Species composition	Age
Production forestry							
Bisson et al. (2002)	8 of 62	Varying levels of harvest and road construction; harvest with no riparian buffers or with 10–30 m riparian buffers	1.5 (1.1)	–	5.2 (2.8)	–	Varies among watersheds: either young forest from harvest without buffers or 10–30 m buffers of second-growth or old-growth
Cederholm and Scarlett (1996)	7 of 10	Varying levels of harvest and road construction	3.9 (2.2)	–	3.3 (2.4)	–	Varies among watersheds: on average 28% young, 46% second-growth, 26% old-growth
Edie (1975)	4 of 15	13–31% of each watershed harvested	5.0 (1.7)	–	2.1 (0.4)	Primarily conifers, alder common	–
Grette (1985)	8 of 27	Logged 10–62 years prior	5.9 (1.4)	10.4 (2.9)	1.2 (0.4)	Primarily conifers, alder common	–
Hatten and Conrad (1995)	7 of 28	≥15% of watershed harvested or riparian corridor of sample reach harvested	2.4 (0.9)	8.4 (2.0)	14.7 (8.8)	Varies among watersheds: on average: 42% of riparian stands were coniferous; 33% deciduous, 25% mixed	Varies among watersheds: on average 33% young, 42% second-growth, and 25% old-growth
McHenry et al. (1998)	8 of 27	Extensive harvest in all watersheds; no logging in riparian forest for 10 years prior sampling	5.9 (1.4)	10.4 (2.9)	1.2 (0.4)	–	1 site with young and 6 sites with second-growth
Osborn (1980)	3[a]	Two sites harvested 5–16 prior sampling; one basin with only wood removed from stream	5.1 (0.9)		1.9 (0.9)	Primarily conifers	
Ecological forestry							
WADNR monitoring (habitat and salmonid)	50	Varying levels of harvest and road construction; at least 30-m riparian buffers applied since 1999	2.2 (1.7)	4.9 (1.9)	5.4 (3.9)	Varies among watersheds: on average, using BA/ha: 84% conifers and 16% deciduous	Varies among watersheds: on average: 5% young, 66% second-growth, 29% old-growth
Reference							
Grette (1985)	4 of 27	No direct harvest effects on sample sites	6.2 (1.0)	10.4 (0.2)	1.2 (0.3)	Primarily conifers	–
Hatten and Conrad (1995)	7 of 28	No harvest in riparian corridor of sample reach and <15% of watershed harvested	3.7 (1.9)	8.5 (2.5)	7.5 (5.5)	Varies among watersheds: on average: 52% of riparian stands coniferous; 13%, deciduous 35% mixed	Varies among watersheds: on average 10% young, 13% second- growth, 77% old-growth
Lestelle (1978)	1[a] of 2	No roads or harvest	0.7	–	6.3	"Principal species" are conifers	Old-growth
Martin (1985)	1[a]	No roads or harvest	2.3	6–10	2.1	Conifers "with frequent clumps of alder"	Old-growth
WADNR monitoring (habitat and salmonid)	4	No roads or harvest	1.5 (0.7)	4.5 (1.0)	11.3 (6.9)	Varies among watersheds: on average, using Ba/ha: 89% conifers and 11% deciduous	100% old-growth

[a]Sites were sampled for fish over multiple years

Tree species composition is reported only as proportion of deciduous and conifer trees because the availability of tree species data varied greatly among the literature sources. The data of riparian forest age were standardized to three categories: "young" is second-growth forest less than 30 years old, "second-growth" is 30–100 years old second-growth, and "old-growth" includes all forest older than 100 years. The number of analyzed watersheds are reported as a subset of all watersheds described in the respective study. The values of stream characteristics are reported as mean with SD in parentheses

Table 2 Summary of data sources, including the years sampled (with time period name where applicable), and number of watersheds analyzed

Management approach and indicator	Study	Years sampled	*n*
Production forestry			
Stream temperature	Hatten and Conrad (1995)	1992	7
Canopy closure	Hatten and Conrad (1995)	1992	7
	Cederholm and Scarlett (1996)	1992–1993	7
Instream wood	Grette (1985)	1982 (1980s)	8
	McHenry et al. (1998)	1993 (1990s)	8
Salmonid density	Edie (1975)	1973–1974 (1970s)	6[a]
	Osborn (1980)	1978–1979 (1970s)	6[a]
	Bisson et al. (2002)	1996–1999 (1990s)	8
Ecological forestry			
Stream temperature	WADNR habitat monitoring	2013–2016	48
Canopy closure	WADNR habitat monitoring	2013–2015	50
Instream wood	WADNR habitat monitoring	2013–2015 (2010s)	12
Salmonid density	WADNR salmonid monitoring	2016 (2010s)	17
Reference			
Stream temperature	Hatten and Conrad (1995)	1992	7
	WADNR habitat monitoring	2013–2016	4
Canopy closure	Hatten and Conrad (1995)	1992	7
	WADNR habitat monitoring	2013–2015	3
Instream wood	Grette (1985)	1982 (1980s)	4
	WADNR habitat monitoring	2013–2015 (2010s)	4
Salmonid density	Lestelle (1978)	1972–1973 (1970s)	2[a]
	Martin (1985)	1977–1980 (1970s)	4[a]

[a]Salmonid densities estimates are a combination of sites and years. See Methods or Table 6 for more information

Stream Temperature

We analyzed stream temperature data from Hatten and Conrad (1995), and WADNR habitat monitoring (Fig. 1, Table 3). Hatten and Conrad (1995) measured summer stream temperature at 60-min intervals in shaded pools. WADNR habitat monitoring measured stream temperature year-round at 60-min intervals, within 100 m of the watershed outlet (Minkova and Foster 2017). To facilitate valid comparisons between studies, we extracted WADNR habitat monitoring temperature data for the same date range (9 July–16 August), averaged across the four monitoring years. Within that date range, we analyzed mean water temperature and mean daily maximum water temperature.

Canopy Closure

We used canopy closure data from Hatten and Conrad (1995), Cederholm and Scarlett (1996), and WADNR habitat monitoring (Fig. 1, Table 4). Hatten and Conrad (1995) and Cederholm and Scarlett (1996) used spherical densiometers to measure percent canopy closure; the studies made 20 and 3 stream-center densiometer measurements per sample reach, respectively. WADNR habitat monitoring measured percent canopy closure using hemispherical photos taken at six stream-center locations, spaced evenly within each 100-m sample reach (refer to Minkova and Foster (2017) for details). To facilitate canopy closure comparisons among studies that used different instruments, we used the methodology of Englund et al. (2000) to develop an equation to relate canopy closure values from a spherical densiometer to those calculated from hemispherical photos. We derived the equation from 137 hemispherical photos with accompanying densiometer canopy closure values measured at the same locations across a variety of western Washington conifer stands.

The resulting relationship was:

$$CC_D = 0.9651 CC_{HP} + 21.829 \qquad \left(R^2 = 0.86\right)$$

where CC_D is percent canopy closure measured by densiometer and CC_{HP} is percent canopy closure measured by hemispherical photography, each with an 82.7° field of view.

Instream Wood

Our assessment of instream wood used data from Grette (1985), McHenry et al. (1998), and WADNR habitat monitoring (Fig. 1, Table 5). All three studies surveyed instream wood using 100-m sample reaches. McHenry et al. (1998) resurveyed the same reaches as Grette (1985) but 11 years later and following extensive additional logging within the watersheds. We quantified instream wood using density (piece count/100 m) and estimated volume (m^3/100 m). WADNR habitat monitoring recorded all individual instream wood pieces (≥0.1 m diameter and ≥2 m length) and logjams within a 100-m sample reach (Minkova and Foster 2017). Because Grette (1985) and McHenry et al. (1998) used a 3-m minimum piece length, we calculated density and volume values from WADNR habitat monitoring using both a 2-m and a 3-m minimum piece length to facilitate comparisons.

Salmonids

We assessed salmonid density using data from five previous studies (Edie 1975; Lestelle 1978; Osborn 1980; Martin 1985; Bisson et al. 2002) and from WADNR salmonid monitoring (Fig. 1, Table 6). The two production forestry studies from the 1970s (Edie 1975; Osborn 1980) were grouped, as were the reference studies from the 1970s (Lestelle 1978; Martin 1985). The Bisson and WADNR studies used a form of multiple-pass removal to estimate abundance, while the Edie, Lestelle, Osborn, and Martin studies used mark-recapture. Some methods for multiple-pass removal have been found to underestimate fish populations when compared to mark-recapture techniques (Rosenberger and Dunham 2005; Peterson et al. 2004). This underestimation can be minimized, with one study finding an average difference of less than one fish between the two techniques (Saunders et al. 2011). WADNR used a variable-pass form of multiple-pass removal to reduce this underestimation (Martens 2016), and so we did not adjust fish density estimates. As salmonid populations show a delayed response to past logging, we grouped the studies by time for analysis (Tschaplinski and Pike 2017).

The studies in the 1970s contained multiple years of sampling, so we combined fish densities for all watersheds and years to help account for both spatial and temporal variability. Timber harvest can initially cause increased productivity because of canopy removal, so we only used data collected at least two years after harvest to account for potential spikes in production after harvest (Bisson and Sedell 1984; Johnson et al. 1986). Bisson et al. (2002) surveyed many small watersheds without fish, so we used only watersheds with documented fish presence for this analysis. The age-0 trout numbers from WADNR salmonid monitoring may be exaggerated due to: 1) the presence of steelhead/rainbow trout in seven of the watersheds (the presence of these fish was not reported in other studies) and 2) our inability to definitively separate them from cutthroat trout at that stage of their maturation. WADNR salmonid monitoring and Bisson et al. (2002) only sampled fish one time, and thus the watersheds did not capture temporal variability in population size that can be high in salmonids (Bayley 2002).

Analysis

We analyzed differences in indicator values among management approaches (production forestry, ecological forestry, and reference) and, for instream wood and salmonids, by time period using analysis of variance (ANOVA) or a *t*-test (Table 2). Owing to the large number of data sources, we simply identify studies according to management approach and time period (e.g., "1970s production forestry"), wherever practical. For normally distributed data with more than two groups (i.e., stream temperature, canopy closure, and wood volume), we used parametric ANOVA; pairwise post-hoc comparisons were carried out using Tukey's HSD test. For comparison between only two groups (e.g., coho density) we used a *t*-test. For data that were not normally distributed (e.g., instream wood densities and salmonid data other than coho density), we used Kruskal–Wallis ANOVA followed by post-hoc comparisons using Dunn's Multiple Comparison test. Because canopy closure data were recorded as a percentage and did not meet the homoscedasticity assumption of parametric ANOVA, data were transformed prior to analysis using an arcsine-square root transformation. All statistical analyses were performed using R (R Core Team 2013). All statistical tests used an alpha level of 0.05.

Results

Stream Temperature

Overall, the production forestry watersheds had the highest stream temperatures (9 July–16 August) with a mean temperature of 14.4 °C ($n = 7$, SD 1.6) and mean daily maximum of 15.8 °C ($n = 7$, SD 2.4; Fig. 2). For the same seasonal interval, the ecological forestry watersheds had lower stream temperatures: 12.8 °C mean ($n = 48$, SD 1.1) and 13.4 °C mean daily maximum ($n = 48$, SD 1.1). Both mean ($F = 10.471$, df $= 2$, $P = 0.003$) and mean daily maximum ($F = 6.503$, df $= 2$, $P < 0.001$) stream temperatures were significantly different between the management approaches. The production forestry watersheds had significantly higher temperatures than both the reference (mean temperature, $P = 0.042$; mean daily maximum temperature, $P = 0.008$) and ecological forestry watersheds (mean temperature, $P = 0.002$; mean daily maximum temperature, $P < 0.001$; Fig. 2). Temperatures in ecological forestry watersheds were not significantly different from those in the reference watersheds (mean temperature, $P = 0.721$; mean daily maximum temperature, $P = 0.554$).

Canopy Closure

Canopy closure was highest (93%; $n = 50$, SD 5) in the ecological forestry watersheds (Fig. 3). The studies conducted during the period of production forestry averaged the lowest canopy closure (73%; $n = 14$, SD 22), and reference watershed studies averaged lower canopy closure than the ecological forestry watersheds (79%; $n = 10$ SD 10). Canopy closure ($F = 25.668$, df = 2, $P < 0.001$) was significantly different between management approaches. Overall, the ecological forestry watersheds had significantly higher canopy closures than then both the production forestry ($P < 0.001$) and reference watersheds ($P < 0.001$; Fig. 3).

Instream Wood

The WADNR-monitored ecological forestry watersheds had the lowest density of instream wood (35.2 pieces/100 m; $n = 12$, SD 24.8) while the reference watersheds assessed by Grette (1985) had the highest density (63.5 pieces per 100 m; $n = 4$, SD 22.3) and volume (67.4 m^3/100 m; $n = 4$, SD 22.5) of instream wood (Fig. 4). The WADNR-monitored ecological forestry watersheds had the second lowest volume of wood (22.2 m^3/100 m; $n = 12$, SD 15.1), with the lowest in the WADNR-monitored reference watersheds (18.0 m^3/100 m; $n = 4$, SD 3.6). Significant differences of wood densities ($H = 9.55$, df = 4, $P = 0.049$) and volumes ($F = 5.831$, df = 4, $P = 0.001$) were detected between management approaches. The WADNR-monitored ecological forestry watersheds of the 2010s had significantly lower densities and volumes of wood compared to the reference conditions of the 1980s (density, $P = 0.018$; volume, $P = 0.002$). In addition, ecological forestry densities but not volumes were significantly lower in than the production forestry of the 1980s (density, $P = 0.022$; volume, $P = 0.080$) and the re-measurement of the production forestry sites in the 1990s (density, $P = 0.037$; volume, $P = 0.293$). The WADNR reference watersheds density and volumes of wood were not significantly different from WADNR ecological forestry watersheds (density, $P = 0.574$; volume, $P = 0.704$).

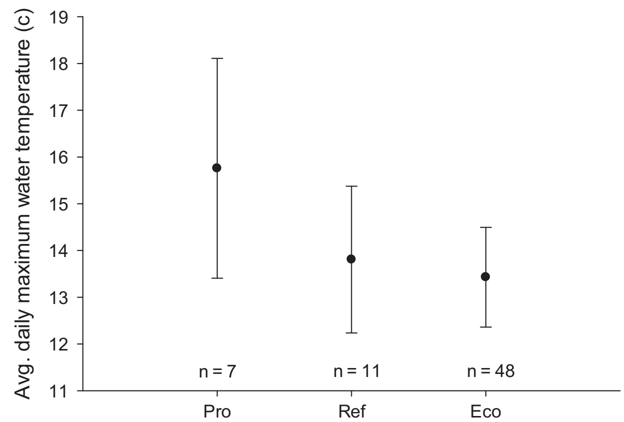

Fig. 2 Mean average daily maximum temperatures and standard deviation (9 July–16 August) for production forestry watersheds (Pro), reference watersheds (Ref), and ecological forestry watersheds (Eco)

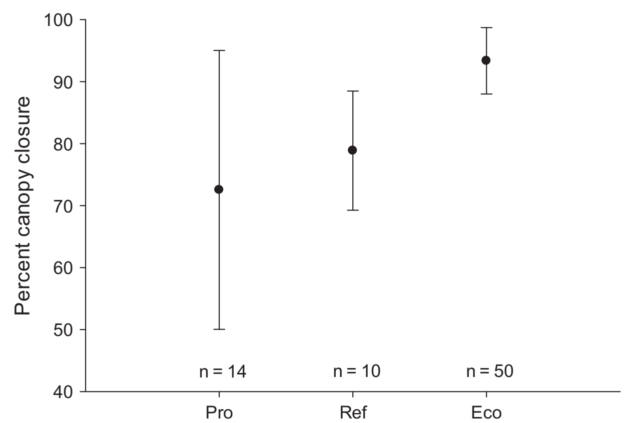

Fig. 3 Mean percent canopy closure and standard deviation for production forestry watersheds (Pro), reference watersheds (Ref), and ecological forestry watersheds (Eco)

Salmonids

The age-0 trout densities of the OESF were highest in the production forestry watersheds of the 1970s (49.3 fish per 100 m; $n = 12$, SD 42.5) with the second highest densities found in the ecological forestry watersheds (35.8 fish per 100 m; $n = 17$, SD 53.4; Fig. 5). Age-0 densities were lowest from the production forestry watersheds in the 1990s (10.3 fish per 100 m; $n = 8$, SD 9.1). Significant differences of age-0 trout densities ($H = 6.309$, df = 3, $P = 0.097$) were not detected between management approaches and time periods.

Coho were in the ecological forestry watersheds and in one study of production forestry watersheds of the 1970s. There were over three times as many fish in the production forestry watersheds in the 1970s (95.4 fish per 100 m; $n = 4$,

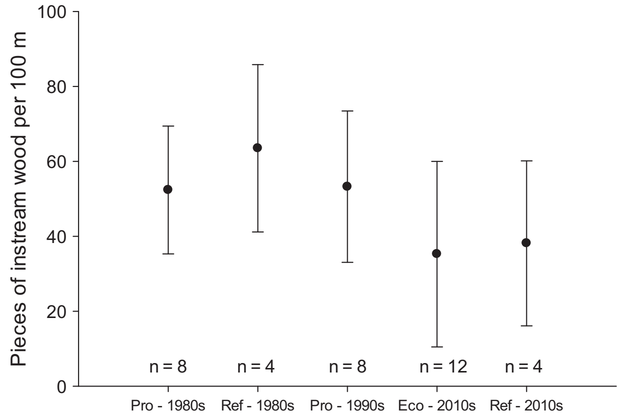

Fig. 4 Mean number of instream wood pieces per 100 (pieces 3 m and longer) and standard deviation for different times and types of management. Production forestry watersheds = Pro, reference watersheds = Ref, ecological forestry watersheds = Eco

SD 62.7) as the ecological forestry watersheds (29 fish per 100 m; $n = 9$, SD 22.3; Fig. 5). Coho abundance was significantly lower in ecological forestry watersheds than in the production forestry watersheds of the 1970s ($t = -2.708$, $P = 0.018$).

Age-1 or older cutthroat densities were highest in the reference watersheds in the 1970s through 1980 (49.0 fish per 100 m; $n = 6$, SD 20.3) followed by the production forestry watersheds of the 1970s (30.9 fish per 100 m; $n = 12$, SD 14.2). This was followed by the production forestry watersheds of the 1990s (11.3 fish per 100 m; $n = 8$, SD 8.8) and the ecological forestry watersheds in 2016 (9.9 fish per 100 m; $n = 17$, SD 12.0; Fig. 5). Significant differences of age-1 or older cutthroat trout densities ($H = 21.531$, df = 3, $P < 0.001$) were detected between management approaches and time periods. The production forestry watersheds of the 1990s were significantly lower than production forestry watersheds of the 1970s ($P = 0.015$) and reference watersheds of the 1970s–1980s ($P = 0.003$). The ecological forestry sites were significantly lower than the production forestry watersheds of the 1970s ($P = 0.001$) and reference watersheds of the 1970s–1980s ($P < 0.001$), but not significantly different from the production forestry sites of the 1990s ($P = 0.693$).

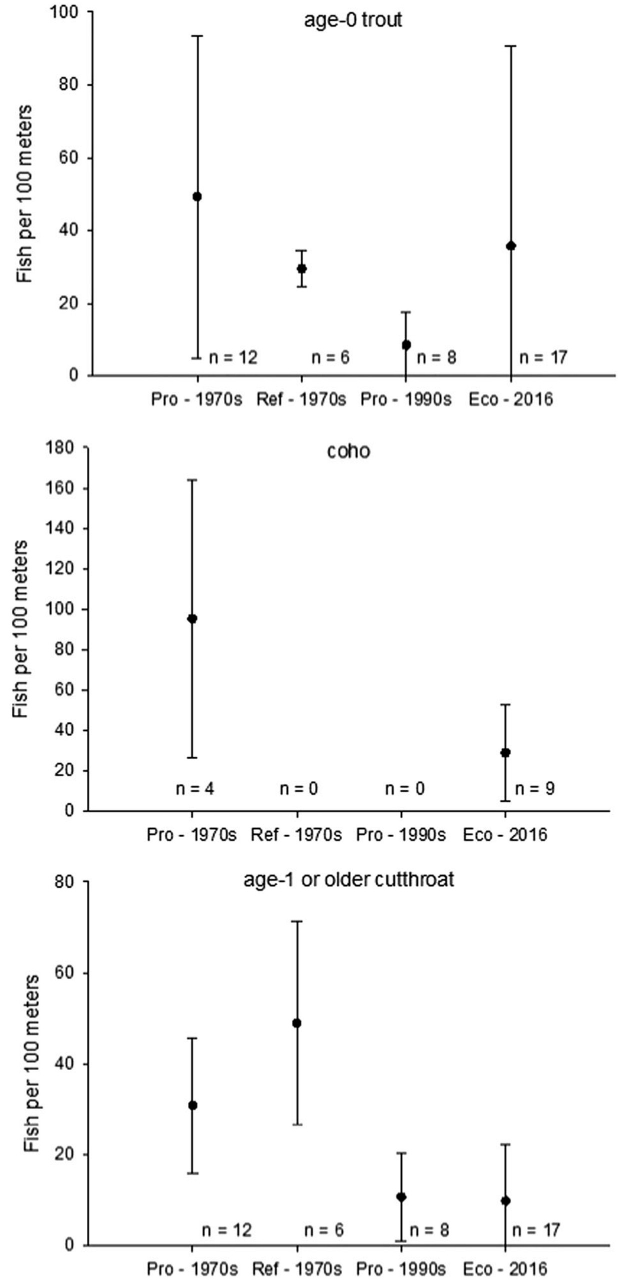

Fig. 5 Mean number of fish per 100 meters and standard deviation of three groups of fish (age-0 trout, coho, and age-1 or older cutthroat trout) for different time and management approaches. Production forestry watersheds = Pro, reference watersheds = Ref, ecological forestry watersheds = Eco

Discussion

Stream Temperature

Summer stream temperatures in the ecological forestry watersheds appear to have recovered and are now lower than the reference watersheds (Fig. 2). This follows the Moore et al. (2005) review that found stream temperatures typically recover 5–15 years after forest harvest. In addition, the stream temperature response was similar to findings of the Aquatic and Riparian Effectiveness Monitoring Program (AREMP) where region-wide reductions in stream temperatures were reported 20 years after implementation of the

Northwest Forest Plan's aquatic conservation strategy (Miller et al. 2015). Currently, the stream temperatures in the ecological forestry watersheds are within the optimal growth range for juvenile rearing of salmon and trout (10–16 °C) reported by the U.S. Environmental Protection Agency (USEPA 2003). The temperature differences between the three management approaches may be due to differences in riparian forest age (Franklin et al. 2002; Kaylor et al. 2017). Production forestry in riparian areas (clearcutting without buffers) initially created areas with reduced canopies and then the subsequent regrowth through passive restoration has created younger forests with dense canopies, as revealed in the canopy closure data, that limits sun exposure to streams and likely led to reductions in stream temperatures (Johnson and Jones 2000; Moore et al. 2005). The reference areas were in areas of older forests (>100 years), that typically contain more gaps in their canopies than younger forests (30–100 years), which would allow for more sun exposure and relatively higher temperatures than the streams in ecological forestry watersheds (Franklin et al. 2002; Kaylor et al. 2017).

Steam temperatures in this study describe the response of passive restoration and not cause and effect relationships between any specific forest management practice, habitat, or salmonid responses. Multiple studies (e.g., Osborne and Kovacic 1993; Moore et al. 2005) have shown that applying riparian buffers, similar to the ones in our study area, reduce the effects of timber harvest on stream temperatures. In a study of headwater streams in nearby coastal British Columbia, Gomi et al. (2006) found less than 2 °C increases in summer daily maximum temperature using 30 m buffers with timber harvests, but documented 2–8° increases when no buffers were used. This study shows that buffers can minimize temperature changes from timber harvest. In addition, Pollock et al. (2009) found stream temperatures in the OESF, shortly after the implementation of the HCP, were generally warmer than an extensive set of unharvested reference sites. They concluded that stream temperatures were affected by previous timber harvests and hypothesized that the amount of temperature change was due to the amount of the watershed harvested.

Canopy Closure

The almost 30% difference between the two production forestry studies (Table 4) may be due to the time lag between sampling and the preceding harvest. Hatten and Conrad (1995) sampled recently harvested watersheds, while the Cederholm and Scarlett (1996) assessed previously managed watersheds regardless of when they had been harvested. Overall, canopy closure in the ecological forestry watersheds was significantly higher than both the reference watersheds and the production forestry watersheds. This outcome likely represents the rapid regrowth of riparian vegetation from the complete removal of the canopy (e.g., clear cutting of riparian forests) and then successional forest development into dense stands where the canopy is blocking most of the light to the streams (Franklin et al. 2002). The dense vegetation is likely the result of passive restoration that has left much of the riparian forests in younger stages of forest development and falls within the range of regional riparian forests 30–100 years after harvest (Kaylor et al. 2017).

High amounts of canopy closure can limit primary production and nutrient cycling within streams (Kiffney et al. 2004; Warren et al. 2017). Kiffney and Roni (2007) found that moderate changes in light from forest canopies could lead to large changes in fish production in streams (these streams included both coho and cutthroat trout), while Wilzbach et al. (2005) found that opening riparian canopies could increase the densities and biomass of age-1 or older juvenile salmonids. Lack of fish productivity in freshwater habitats may be a factor in declining salmon populations (Bradford and Irvine 2000). Accordingly, the likely current light-limited phase in riparian forest canopies as a result of passive restoration may be limiting salmonid production and potentially their recovery.

Instream Wood

The lower density of instream wood in the WADNR reference watersheds may be due to the large difference in stream gradient between those watersheds (11.3%) and all of the other study groups (3.1% or less). Gradient has been found to be negatively associated with instream wood accumulations (Fox and Bolton 2007; Wohl and Cadol 2011). Given the steep gradient of the WADNR-monitored reference watersheds, we believe the wood indicators (density and volume) reported by Grette (1985) provides the more appropriate comparison for reference conditions of instream wood in low-gradient streams.

The lower amount of instream wood in the ecological forestry watersheds may be due to the past instream wood removal practices, the mobility of wood in the stream, lack of recruitment due to younger trees, size of trees in the riparian forest, and (or) species of wood (Hassan et al. 2005). Unfortunately, no information is available on the amount of wood removed from individual streams during the production forestry period, though the practice of removing wood from streams historically occurred on the OESF (Cederholm and Reid 1987).

The amount and size of wood recruited into streams from second-growth forests will be less than the pre-harvest levels until the species composition and tree size of the riparian forest fully recover to pre-harvest conditions (Gregory et al. 1987; Murphy and Koski 1989). Owing to the widespread implementation of production forestry in the OESF during the 20th century, riparian stands today are relatively young and thus contributing smaller pieces of wood to streams. Large pieces of wood can act as anchors to help accumulate smaller pieces of wood (Lienkaemper and Swanson 1987; May and Gresswell 2003). Without these anchors, smaller pieces are more likely to move downstream away from the area of recruitment (Lassettre and Harris 2000). The current low recruitment potential may partially explain the lower quantities of instream wood despite 18 years of passive restoration.

Another possible explanation for the reduced density and volume of instream wood after passive restoration in the ecological forestry watersheds may be the type of wood in the streams. Hardwoods (predominantly red alder in the coastal Pacific Northwest) establish in riparian forests after harvest or disturbance, often changing the composition of riparian forests from conifer-dominated to hardwood-dominated for a period of several decades (Gregory et al. 1987; Giordano and Hibbs 1996). This change would eventually lead to changes in the type and size of wood recruited into streams. Because hardwood trees decay more rapidly than conifer trees (Bilby et al. 1999), this may lead to a delayed decrease of instream wood. In addition, smaller diameter pieces of instream wood would probably break down quicker than larger diameter trees under higher stream velocities.

Salmonids

The increased mean number of age-0 trout per meter in ecological forestry watersheds from the studies conducted in the 1990s to levels similar to those in the 1970s, and the lack of significance between management types and years could be a sign of improvement (Fig. 5). The potential age-0 trout recovery before other salmonids may be due to stream habitat preferences. Coho and age-1 or older cutthroat are typically larger due to their emergence times (age-0 trout are spring spawners while coho are fall spawners) with age-0 trout preferring shallower habitats compared to larger salmonids (Mellina and Hinch 2009). With this preference, reduced levels of instream wood after passive restoration may not be as impactful to age-0 trout as other salmonid species or life stages. Overall, the age-0 trout numbers in ecological forestry watersheds may be showing an improvement over conditions in the 1990s, but additional years of age-0 abundance estimates would be required to make any more definite statements on their recovery.

Coho were only in production forestry watersheds in the 1970s and ecological forestry watersheds (Fig. 5). During exploratory sampling of a representative proportion of the smallest fish-bearing streams of the OESF (stream order 1–3) in 2015, Martens (2016) found that cutthroat trout were in 82% of the watersheds, while coho and steelhead/rainbow trout were in 62% and 23%, respectively. This corroborates the lack of coho salmon in some of the other studies previously conducted within the OESF (i.e., Edie 1975; Lestelle 1978; Osborn 1980; Martin 1985; Bisson et al. 2002). It is currently unknown if this lack of juvenile coho presence is due to reduced coho habitat, higher stream gradients (coho prefer stream gradients less than 3%), and/or limited adult returns (Reeves et al. 1989). Overall, we do not feel that there is currently enough data from the OESF to separate any range-wide declines of coho from habitat-specific changes after passive restoration.

The reduced level of age-1 or older cutthroat (despite potentially improved levels of age-0 trout) may be a result of degraded habitat within the ecological forestry watersheds after passive restoration. With no winter sampling or fish tagging, age-1 or older cutthroat trout provide our only indication of year-round conditions within the ecological forestry watersheds. As such, these populations represent some combination of immigration, emigration, and survival within the watersheds over a given year. The accumulation of information from our comparisons further supports other studies that have documented both salmonid fall parr migrations (Roni et al. 2012; Hall et al. 2016) and limited winter habitat (Tschaplinski and Hartman 1983; Heifetz et al. 1986) that would result in lower age-1 or older salmonids. While it is currently unknown whether salmonids are experiencing higher mortalities or increased movement, the small streams of the OESF appear to be expressing decreased densities of age-1 or older fish.

Instream wood has a positive effect on pool creation, frequency, and volume. Reductions of both instream wood and pool habitat can negatively affect age-1 or older salmonid densities (Murphy et al. 1986; Rosenfeld and Huato 2003). Riparian canopy coverage regulates food availability in forested streams and can also limit fish production in streams with higher canopy densities (Kaylor and Warren 2017). Accordingly, the combination of increased riparian canopy coverage and reduced level and size of instream wood after passive restoration may be negatively influencing age-1 or older salmonids. The lack of diversity in age classes within these smaller streams may be reducing the stability and resilience in the anadromous populations of the OESF and therefore slowing the recovery process (Waldman et al. 2016).

In this study, we used the best available data within the borders of the OESF within similar-sized watersheds. While the material is informative, some of the analyses did suffer from lower power due to smaller samples sizes and are best suited for developing hypotheses and directing further investigations. As such, we have more confidence in our conclusions with greater samples sizes and differences among management approaches such as for canopy closure and age-1 or older cutthroat, than results with smaller sample sizes or fewer differences like instream wood. Nevertheless, our findings follow similar patterns of other studies conducted on forest development and the effects of past timber harvest on streams in the Pacific Northwest (McHenry et al. 1998; Connolly and Hall 1999; Kaylor et al. 2017).

Conclusions

Stream conditions are continuing to change both positively and negatively in regard to salmonids after 18 years of passive restoration. Stream temperatures have recovered to, or fallen below, pre-disturbance levels. Riparian forest canopy closure has increased beyond the levels in unmanaged watersheds to the point where it is potentially limiting primary productivity in streams. Instream wood and age-1 and older cutthroat appear to be declining or stable at reduced levels compared to past or reference conditions. We hypothesize that salmonids and salmonid habitats in smaller watersheds (stream order 1–3) are unlikely to recover until second-growth riparian forests develop larger diameter trees and more canopy diversity. This would allow for the continued recruitment of large pieces of instream wood, developing pools and increasing habitat for salmonids while allowing more light into streams creating increased primary and secondary productivity.

Our results add to the current scientific literature that has found passive restoration of salmonid habitat in the Pacific Northwest is a slow process, which could take an additional 12–70 years for riparian forests and over 50 years for instream wood to accumulate (McHenry et al. 1998; Connolly and Hall 1999; Kaylor et al. 2017). Due to the slow pace of passive restoration, the stochasticity of natural disturbances, and rate of declines in listed species, selective use of active restoration may help increase productivity and habitat complexity in riparian forests and streams. Some potential examples of active restoration include riparian forest thinning and instream wood additions. Riparian forest thinning could create gaps in canopies and provide more production in streams while hastening the growth of larger conifer trees. Instream wood additions could restore the ecological functions in streams until the riparian forests could restore natural recruitment levels and maintain higher densities and volumes.

Recommendations for active restoration should be site-specific and consider information on the effectiveness of the restoration techniques as well as the current state and desired conditions. Most active restoration in the Pacific Northwest is undertaken with the assumption that it will increase freshwater salmonid production, however despite numerous evaluations of restoration effectiveness, strong evidence validating this basic assumption remains elusive (Bennett et al. 2016). Project-level monitoring would be more powerful if combined with landscape-level monitoring such as the WADNR habitat and salmonid monitoring programs. This would lead to a greater understanding of the successional and natural disturbance history of an area as well as the desired future conditions essential for restoration success. Otherwise, land manager's risk creating stream conditions outside of the natural range of variability.

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Compliance with ethical standards

Conflict of Interest: The authors declare that they have no conflicts of interest.

Appendix

See Tables 3–6.

Table 3 Summary of stream temperature indicator (mean with SD in parentheses), as reported by various studies conducted within the OESF where n represents the number of sample watersheds from each study summarized here

Study	Management history	n	Watershed area (km^2)	Reach gradient (%)	Year sampled	Stream temperature (°C)[a]			
						9 July–16 August		1–31 August	16 July–31 August
						Mean	Mean daily maximum	Mean	Mean daily maximum
Production forestry									
Hatten and Conrad (1995)	≥15% of watershed harvested or riparian corridor of sample reach harvested	7	2.4 (0.9)	14.7 (8.8)	1992	14.4 (1.6)	15.8 (2.4)	–	–
Ecological forestry									
WADNR monitoring[b]	Varying levels of harvest; riparian buffers used in recent decades	43	2.0 (1.6)	5.6 (4.1)	2013	12.2 (0.9)	12.8 (1.0)	12.7 (0.7)	13.0 (1.0)
		38	2.1 (1.7)	5.4 (3.9)	2014	12.9 (0.9)	13.6 (1.1)	13.2 (0.8)	13.6 (1.0)
		39	2.2 (1.8)	5.1 (4.0)	2015	13.7 (1.0)	14.2 (1.1)	13.5 (0.9)	14.1 (1.0)
		38	1.9 (1.7)	5.1 (4.0)	2016	12.6 (0.8)	13.1 (0.9)	13.1 (0.8)	13.5 (0.9)
					2013–2016	**12.8 (1.1)**	**13.4 (1.1)**	**13.1 (0.9)**	**13.5 (1.0)**
Reference									
Hatten and Conrad (1995)	No harvest in riparian corridor of sample reach and <15% of watershed harvested	7	3.7 (1.9)	7.5 (5.5)	1992	12.8 (1.7)	13.5 (1.7)	–	–
Martin (1985)	No harvests	1	2.3	2.1	1977	–	–	13.2	14.9
		1	2.3	2.1	1978			12.1	14.6
		1	2.3	2.1	1979			12.2	13.7
		1	2.3	2.1	1980			11.7	13.0
					1977–1980			**12.3 (0.6)**	**14.1 (0.9)**
WADNR monitoring[b]	No harvests	3	1.1 (0.2)	9.7 (7.5)	2013	12.2 (1.2)	13.1 (1.5)	13.2 (1.6)	13.8 (2.0)
		3	1.7 (0.8)	11.5 (8.4)	2014	14.1 (1.3)	14.8 (1.5)	15.2 (0.2)	14.8 (1.5)
		4	1.5 (0.7)	11.3 (6.9)	2015	14.5 (1.2)	15.1 (1.4)	14.4 (1.2)	15.0 (1.4)
		3	1.7 (0.8)	11.5 (8.4)	2016	13.4 (1.2)	13.9 (1.4)	14.0 (1.4)	14.4 (1.5)
					2013–2016	**13.6 (1.4)**	**14.3 (1.5)**	**14.2 (1.2)**	**14.6 (1.4)**

[a]Time periods for indicators are those reported by Hatten and Conrad (1995) and Martin (1985)

[b]Sample size in each year is typically fewer than 50 ecological forestry watersheds and 4 reference watersheds as a result of data loggers that became dewatered during low flows or were lost during flood events

Table 4 Summary of canopy closure (mean with SD in parentheses), as reported by various studies conducted within the OESF where n represents the number of sample watersheds from each study summarized here

Study	Management history	n	Year sampled	Reach gradient (%)	Bankfull width (m)	Watershed area (km^2)	Canopy closure (%)
Production forestry							
Hatten and Conrad (1995)[a]	≥15% of watershed harvested or riparian corridor of sample reach harvested	7	1992	14.7 (8.8)	8.4 (2.0)	2.4 (0.9)	58 (23)
Cederholm and Scarlett (1996)[a]	Riparian forest averages 64% immature (<30 years old) trees	7	1992–1993	3.3 (2.4)	–	3.9 (2.2)	87 (8)
Ecological forestry							
WADNR monitoring[b]	Varying levels of harvest; riparian buffers used in recent decades	50	2013–2015	5.4 (3.9)	4.9 (1.9)	2.2 (1.7)	93 (5)
Reference							
Hatten and Conrad (1995)[a]	No harvest in riparian corridor of sample reach and <15% of watershed harvested	7	1992	7.5 (5.5)	8.5 (2.5)	3.7 (1.9)	74 (7)
WADNR monitoring[b]	No management	3	2013–2015	11.5 (8.4)	4.9 (0.7)	1.7 (0.8)	90 (5)

[a]Canopy closure measured using a densiometer

[b]Canopy closure measured using hemispherical photography and subsequently converted to densiometer-equivalent values

Table 5 Summary of instream large wood indicators (mean with SD in parentheses), as reported by various studies conducted within the OESF where n represents the number of sample watersheds from each study summarized here

Study	Year sampled	Management history	n	Reach gradient (%)	Bankfull width (m)	Watershed area (km^2)	Instream large wood		
							Density (2 m min. length) pieces/100 m	Density (3 m min. length)	Volume (3 m min. length) m^3/100 m
Production forestry									
Grette (1985)[a]	1982	Logged 10–62 years prior	8	1.2 (0.4)	10.4 (2.9)	5.9 (1.4)	–	52.4 (17.1)	45.7 (20.0)
Cederholm and Scarlett (1996)	1992–1993	Riparian forest averages 64% immature (<30 years old) trees	6	2.7 (2.0)	–	3.8 (2.4)	60.3 (30.0)	–	–
McHenry et al. (1998)[a]	1993	All watersheds harvested or watersheds indirectly affected by harvests	8	1.2 (0.4)	10.4 (2.9)	5.9 (1.4)	–	53.3 (20.2)	37.6 (24.3)
Ecological forestry									
WADNR monitoring[b]	2013–2015	Varying levels of harvest; riparian buffers used in recent decades	12	3.1 (1.9)	7.3 (1.3)	4.8 (1.4)	41.5 (24.8)	35.2 (24.8)	22.2 (15.1)
Reference									
Grette (1985)	1982	No direct effects of logging	4	1.2 (0.3)	10.4 (0.2)	6.2 (1.0)	–	63.5 (22.3)	67.4 (22.5)
WADNR monitoring	2013–2015	No management	4	11.3 (6.9)	4.5 (1.0)	1.5 (0.7)	47.3 (21.2)	38.1 (22.0)	18.0 (3.6)

[a]Of 28 watersheds sampled, 8 managed watersheds were selected for this subset because watershed area was within the range of watershed sizes in DNR monitoring

[b]Of 50 watersheds sampled, 12 were selected for this subset because watershed area was within the range of watershed sizes in Grette (1985) and McHenry et al. (1998)

Table 6 Summary of salmonid densities (mean fish per 100 m, with SD in parentheses), as reported by various studies conducted within the OESF where *n* represents the number of sample watersheds from each study summarized here

Study	Management history	*n*	Watershed area (km^2)	Reach gradient (%)	Year Sampled	Salmonids		
						Age-0 trout[a] fish/100 m	Coho	Age-1 or older cutthroat
Production forestry								
Edie (1975)	Various levels of past management	2	6.5	2.0	1973	16.4	45.2	23.9
		4	5.0 (1.7)	2.1 (0.4)	1974	19.1 (24.6)	120.5 (61.9)	26.3 (16.1)
Osborn (1980)	1 year after harvest	2	0.9	–	1978	133.6	NP	44.6
	2 years after harvest	2			1979	145.0	NP	50.3
	5–16 years after harvest	3	18.9 (1.0)	–	1978	78.0 (39.7)	NP	34.8 (11.4)
		3			1979	83.0 (29.4)	NP	37.8 (11.1)
Average (1970s)	>2 years after harvest	**12**				**49.3 (42.5)**	**95.4 (62.7)**	**30.9 (14.2)**
Bisson et al. (2002)	Various levels of past management	**8**	1.5 (1.1)	5.2 (2.8)	1996–99	**10.3 (9.1)**	**NR**[b]	**11.3 (8.8)**
Ecological forestry								
WADNR Monitoring	Varying levels of harvest; riparian buffers used in recent decades	**17**[c]	2.3 (2.0)	5.3 (4.3)	2016	**35.8 (53.4)**	**29.0 (22.3)**	**9.9 (12.0)**
Reference								
Lestelle (1978)	No harvest	1	0.9	–	1972	30.5	NP	31.4
		1			1973	26.2	NP	28.4
Martin (1985)		1	2.3	2.1	1977	34.0	NP	57.0
		1			1978	36.0	NP	47.0
		1			1979	28.0	NP	40.5
		1			1980	22.5	NP	89.5
Average (1972–1980)		**6**				**29.5 (4.6)**	**NP**	**49.0 (20.5)**

NP not present

[a]Age-0 trout contain both steelhead and cutthroat

[b]Not reported—only two watersheds contained coho

[c]Not all species were in all watersheds and zeros were not averaged for coho

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