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Fish response to contemporary timber harvest practices in a second-growth forest from the central Coast Range of Oregon

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ABSTRACT

We used a paired-watershed approach to investigate the effects of contemporary logging practices on headwater populations of coastal cutthroat trout (*Oncorhynchus clarkii clarkii*) and juvenile coho salmon (*Oncorhynchus kisutch*) in a second-growth Douglas-fir forested catchment in Oregon. Stream habitat and fish population characteristics, including biomass, abundance, growth, size, and movement, were assessed over a 9-year period (4 years pre- and 5 years postlogging). The logged catchment was located on private industrial forestland and had been previously logged in 1966. The reference catchment was covered by an unharvested, fire-regenerated forest approximately 150–160 years old, which was unroaded and managed as a Research Natural Area by the USDA Forest Service. A single clearcut harvest unit of the upper 40% of the treatment catchment was implemented following current forest practice regulations, including the retention of riparian buffer of standing trees adjacent to fish bearing channels. No statistically significant negative effects on coastal cutthroat trout or coho salmon occurred following logging, and in fact, both late-summer density and total biomass of age-1+ coastal cutthroat trout increased in the logged catchment following logging. Increases in age-1+ coastal cutthroat were greatest closest to the harvest area and declined downstream as distance from the logged area increased. In contrast to the previous timber harvest in the catchment when few logging regulations existed, current forest practice regulations and logging techniques appear to have reduced acute negative effects on coastal cutthroat trout.

1. Introduction

Response of aquatic systems to disturbance (e.g. fire, flood, and timber harvest) is context dependent (Resh et al., 1988; Detenbeck et al., 1992; Gresswell, 1999). Strong linkages between terrestrial and aquatic systems result in a complex pattern of effects related to the interaction among physical, chemical, and biological characteristics of these systems (Gregory et al., 1991), and effects are often propagated downstream (Hicks et al., 1991; Gomi et al., 2002; Richardson and Danehy, 2007). Prior conditions of the system interact with the type, timing, and intensity of the disturbance to alter the system at a variety of spatial scales, and biota respond to these changes (Hartman and Scrivener, 1990; Andrew and Wulder, 2011). For example, the effects of timber harvest are contingent on the bedrock geology and geomorphic characteristics of the system, the stand age, and the methods used to harvest the timber (Hartman et al., 1987; Mellina and Hinch, 2009; Valdal and Quinn, 2010).

Effects of timber harvest on previously unharvested forests have been studied for decades (Murphy and Hall, 1981; Duncan and Brusven, 1985; Bilby and Bisson, 1987). Although biotic responses vary, effects can have substantial negative consequences for aquatic habitat and vertebrates (Hartman et al., 1987; Reeves et al., 1993; Mellina and Hinch, 2009). In some cases, effects related to changes (both negative and positive) in light flux may be apparent at various points in time

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(Kaylor and Warren, 2017; Connolly and Hall, 1999); however, persistent alterations are often related to the construction of roads and methods used to yard timber (Trombulak and Frissell, 2000; Valdal and Quinn, 2010; Richardson et al., 2012). Indeed, harvests that occurred in the first half of the 20th century included the use of small stream channels as roadbeds and movement corridors for yarding timber (Richardson et al., 2012).

Prior research played an important role in the development of forest management regulations that were intended to safeguard aquatic resources while facilitating timber harvest (Tschaplinski et al., 2004; Stednick, 2008a). Contemporary forest practices have advanced considerably in recent decades (Richardson et al., 2012). Currently timber harvest occurs primarily on private industrial timberlands and has shifted from harvesting old growth or naturally-regenerated mature timber, to logging previously-harvested stands on shorter stand rotation intervals, using pre-existing road networks (Bateman et al., 2016). All of these activities occur in concordance with forest practice regulations (e.g., Oregon Department of Forestry, 2006) developed in response to prior research (Ice et al., 2010; Richardson et al., 2012), and regulations generally require standing tree riparian buffers when timber harvest is adjacent to streams where fish are present (Lee et al., 2004).

Effects of harvest in second-growth forests on fish are not as well-documented (De Groot et al., 2007), but there is some evidence that many of the negative consequences reported with the harvest of previously unharvested forests have not occurred during subsequent logging activities (Mellina and Hinch, 2009). In fact, in some cases the removal of thick closed-canopy, young- to middle-aged forests can increase aquatic productivity by increasing light availability to the stream benthos (Ambrose et al., 2004), and if water temperatures do not exceed recognized optimums, growth and total biomass of stream-dwelling salmonids may increase (Murphy and Hall, 1981; Connolly and Hall, 1999; Wilzbach et al., 2005). Although conceptual models suggest changing trends in fish abundance and forest stand development through time (Warren et al., 2016), the long-term effects of second-growth timber harvests (with standing tree riparian buffers) on persistence of salmonid populations has not been investigated with empirical field studies.

Whether muted responses of aquatic systems to timber harvest of second-growth forests is the result of improved management practices or related to diminished system capacity or some combination of the two is not well understood. For example, the installation of infrastructure (e.g., roads and landings) associated with removal and transport of downed trees often caused press disturbances (Lake, 2000) that persisted long after the harvest of primeval forests (Sedell et al., 1991). Because additional roads and infrastructure development are frequently unnecessary during second-growth harvest, physical alterations of the watershed associated with erosion and subsequent streambed sedimentation may not occur, or may be substantially reduced during these secondary perturbations (Bateman et al., 2016). Furthermore, research focused on reducing the negative consequences of logging have resulted in substantial changes in forest-harvest practices, and contemporary forest practices are intended to reduce the negative effects of harvest. Alternatively, it has been argued that effects on physical and biotic components of some systems following old-growth timber harvest persist. Because these second-growth systems have never fully recovered, they no longer have the capacity to respond to disturbance associated with timber harvest (sensu Frissell et al., 1997).

Second-growth forests now being subjected to harvest (50–60 years of regrowth) provide the opportunity to investigate the effects of logging on fish populations in adjacent channels (De Groot et al., 2007) or in channels downstream of harvest (Bateman et al., 2016). These recent studies provided examples that stream adjacent logging could occur without negatively affecting fish abundance when bank and streambed disturbance was avoided and large wood was left in the channel; however; capacity of these systems prior to logging was unknown.

Plans to commercially harvest portions of the Alsea watershed in the Coast Range of western Oregon provided a unique opportunity to revisit an area where a paired-watershed study conducted in the 1950s–1960s documented the influence of the harvest of mature forests on headwater watersheds and the populations of coastal cutthroat trout and coho salmon in those systems (Stednick, 2008a). This seminal study was a primary impetus for changes in forest practices throughout the Pacific Northwest. We took advantage of this opportunity to revisit the Alsea Watershed Study during timber harvest of second-growth coniferous forest using contemporary forest practices, including standing tree buffers in the fish bearing portions of the logged catchment. Our goal was to document the effects of the harvest on stream physical habitat and stream salmonids using the paired-watershed approach and to compare these effects to those documented in the original Alsea Paired Watershed Study (Moring and Lantz, 1975, Hall and Stednick, 2008).

Examining the effects of contemporary forest harvest in paired catchments of the Alsea River watershed allows us to place current research in an important historical context because the Alsea Watershed Study, initiated in 1959, also used this approach and provides some indication of the pre-harvest capacity of Needle Branch. Given documented sensitivity to historic forest management, this system was ideal for evaluating the response of fish populations to harvest of second-growth forest under contemporary regulations. Although replication would be required to assess the influence the relative proportion of response related to diminished capacity or improved logging practices, Needle Branch provides a unique opportunity to observe the response of a previously harvested system to a subsequent harvest after water quality and fish abundance parameters have returned pre-logging conditions. Furthermore, an evaluation of fish response to upslope clearcut logging in the presence of a standing tree buffer has not been conducted in the Pacific Northwest since the Alsea Watershed Study, but recent studies evaluating effects of contemporary logging practices on fish under a range of logging treatments (De Groot et al., 2007; Olson et al., 2013; Bateman et al., 2016) have not documented negative effects on headwater fish populations. We hypothesized that in the presence of a standing tree buffer and contemporary upslope clearcut logging, effects in parameters commonly used to evaluate the status of fish populations and habitat quality would not be biologically significant.

2. Methods

2.1. Study location and background

Needle Branch and Flynn Creek are small headwater catchments (85 and 212 ha respectively) that flow into Drift Creek, approximately 16 km inland from the Pacific Ocean in the Alsea River watershed of the Oregon Coast Range (Fig. 1). Elevations range from 140 to 590 m (Hall and Stednick, 2008). The maritime climate is characterized by mild wet winters and dry summers. Most of the annual precipitation (mean = 250 cm) occurs as rain falling between October and March (Hall and Stednick, 2008). Bedrock is primarily sandstone of the Tyee formation (Corliss and Dyrness, 1965). Douglas-fir (*Pseudotsuga menziesii*) in plantation and fire-regenerated forests dominates Needle Branch and Flynn Creek, and red alder (*Alnus rubra*) is common in riparian areas (Moring and Lantz, 1975). Salmonberry (*Rubus spectabilis*), skunk cabbage (*Lysichitum americanum*), sword fern (*Polystichum munitum*), and vine maple (*Acer circinatum*) are common in the understory (Moring and Lantz, 1975). Coastal cutthroat trout, coho salmon, reticulate sculpin (*Cottus perplexus*), western brook lamprey (*Lampetra richardsoni*), and Pacific lamprey (*L. tridentata*) comprise the fish community (Hall and Stednick, 2008). Steelhead trout (*Oncorhynchus mykiss irideus)* are occasionally collected in the study catchments. Additional vertebrates found in the study catchments include the coastal giant salamander (*Dicamptodon tenebrosus*) and the coastal tailed frog (*Ascaphus truei*).

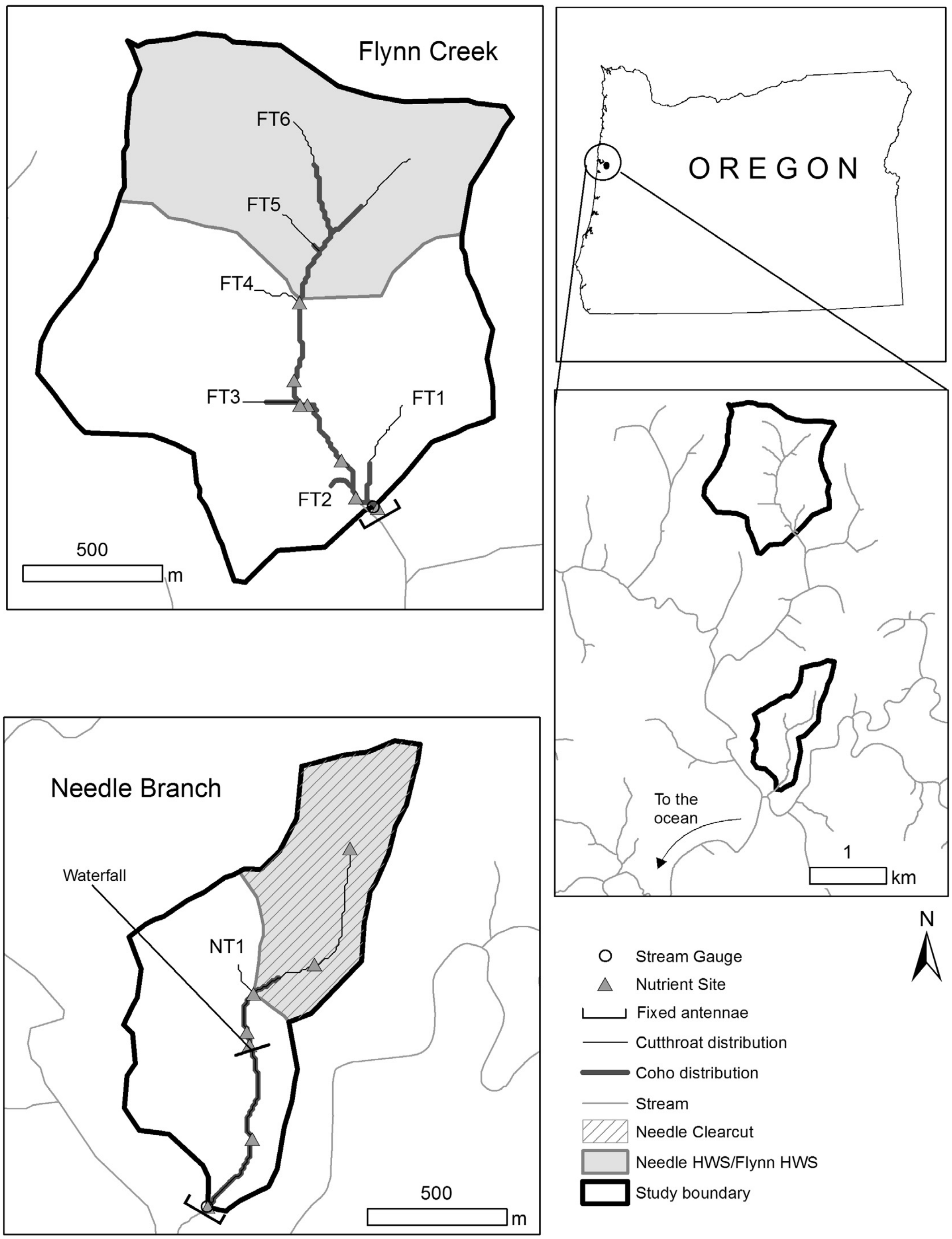

Fig. 1. Study catchments tributary to Drift Creek and the Alsea River, Lincoln County, Oregon. USA. Locations of stream gauges, nutrient sites, fish-bearing channel, and fixed antennae in each catchment, including the distributional extent of coastal cutthroat trout (*Oncorhynchus clarkii clarkii*) and juvenile coho (*Oncorhynchus kisutch*) and the portion of each catchment comprising the headwater assessment scale (HWS) in Needle Branch and Flynn Creek. Alphanumeric labels (e.g. NT1 and FT1) indicate fish-bearing tributaries number sequentially from downstream to upstream in Needle Branch and Flynn Creek.

The Alsea Watershed Study was designed to evaluate the effects of logging in previously unharvested forests on water quality and fish populations. Eighty-two percent of the Needle Branch catchment was clearcut in 1966 using harvest practices typical of the period (Stednick, 2008a). Harvest occurred adjacent to the stream without riparian forest buffers, and logs were yarded through the stream. The stream channel was subsequently cleared of logging slash and naturally occurring wood (i.e., stream cleaning; Hall and Stednick, 2008), and the watershed was burned. Flynn Creek was an unharvested reference catchment during the Alsea Watershed Study. It has been managed as a research natural area by the U. S. D. Forest Service since 1977 and is one of the few unlogged and unroaded catchments remaining in the Oregon Coast Range (McKee and Greene, 2008).

Since the completion of the Alsea Watershed Study, management activity typical of industrial forests in western Oregon has continued in Needle Branch. In 1981, the lower 75% of the catchment was pre-commercially thinned; and approximately 40% of the area was commercially thinned in 1997, removing approximately 14,000 m^3 or 5.8 $m^3 \cdot ha^{-1}$ of timber (Steve Gravelle, Plum Creek Timber Company, personal communication, 2015). The following year, fertilizer (225 kg urea$\cdot ha^{-1}$) was applied to the commercially thinned area (Steve Gravelle, Plum Creek Timber Company, personal communication, 2015).

2.2. Study design and harvest treatment

A Before-After-Control-Impact (BACI) study design (Stewart-Oaten et al., 1986; Stewart-Oaten and Bence, 2001) was used to determine fish population responses to contemporary logging (Oregon Department of Forestry, 2006) in the Alsea watershed. As in the initial study, Needle Branch served as the "impact" site and Flynn Creek served as the reference catchment.

Beginning in 2006, we collected data annually characterizing stream habitat and fish populations in both catchments. Logging occurred in the Needle Branch catchment between June 2009 and September 2009, and postlogging data were collected from September 2009 through August 2014. Forest harvest consisted of a single clearcut unit that encompassed the upper 40% (34 ha) of the Needle Branch catchment, including 800 m of fish-bearing channel (Fig. 1). In accordance with Oregon Forest Practice Rules, a 15-m wide (minimum) riparian buffer was retained on each side of the channel in this section of Needle Branch, but in the fishless upstream portion of the channel network, no standing trees were retained adjacent to the channel (Oregon Department of Forestry, 2006).

An existing road network (mostly along ridgetops) provided access for equipment, and no roads crossed the channel in the fish-bearing portion of the catchment. New road construction was limited to a few short rocked spurs (for landings). Some of the native-surfaced road system was rocked to enable wet-weather access. All roadwork (except maintenance) occurred prior to logging.

Logs were yarded by cable (at least one end suspended) from the streamside buffers to landings. Large wood was left in the stream. Logging slash was piled and burned near landings. Aerial application of herbicides (i.e., glyphosate, imazapyr, sulfometuron methyl, and metsulfuron methyl) for site preparation occurred during August 2010. An 18 m (horizontal distance) spray buffer was maintained on each side of the stream where fish were present, and a minimum 3 m spray buffer was maintained along the channel in the fishless portion of the catchment (Louch et al., 2016). Approximately 24 ha on south slopes of the upper catchment were burned in December 2010. The entire harvested unit was replanted with Douglas-fir seedlings during the winter of 2010–2011

Western redcedar (*Thuja plicata*) seedlings were planted among the Douglas-fir along stream channels in concordance with company policy to replant with a mix of native species when conditions are appropriate (i.e., riparian areas).

3. Data collection

3.1. Stream habitat

Stream habitat surveys were conducted annually (2006–2014) in late July to early August over the entire fish-bearing portions of both Needle Branch and Flynn Creek. Prior to the initial surveys, catchments were divided into stream segments based on barriers to upstream fish movement and junctions with major fish-bearing tributaries (Moore et al., 1997). Channel-unit types (pool, riffle-rapid, cascade, and vertical step) were classified in each stream segment (Bisson et al., 1982), except for the requirement that all pools needed to have a maximum depth ≥ 15 cm. Sections without flowing water were designated as dry channel. Length and wetted width, streambed composition (i.e., percent of surface area in bedrock, boulder, cobble, gravel, sand, and silt for the three most abundant types), active channel width, and valley floor width were visually estimated in each channel unit. Correction factors for each surveyor were developed by measuring wetted width, length, and active channel and valley floor widths with a meter tape at every tenth habitat unit and comparing results to the visual estimates (Hankin and Reeves, 1988). A meter stick was used to measure maximum depth of all pool and riffle-rapid habitats.

A visual estimate of the percentage of the active channel area shaded by tree and shrub canopies was used to estimate the amount of light reaching the active channel. Overstory tree and understory shrub shade were estimated separately, and therefore, combined shade estimates exceeded 100% for some channel units.

Instream cover for fish provided by aggregations of small wood (pieces of wood < 2 m in length, and pieces of wood > 2 m in length but < 10 cm in diameter 2 m in for the end with the largest diameter) and large wood (Cederholm and Peterson, 1985), and near-stream vegetation was visually estimated (to nearest 10% for items occupying ≥10% of unit area and to the nearest 5% for items occupying < 10% of unit area) in all pool and cascade habitats. The proportion of pool and cascade surface area concealed by each cover item was estimated separately for zones 1 (wetted cross-section during summer low flow) and 2 (wetted cross-section between summer low flow and bankfull flow) (Robison and Beschta, 1990). Ultimately, all hiding cover for fish created by small and large wood and vegetation were combined to provide a single estimate of organic cover for each pool.

Physical cover was also assessed for each pool and cascade. If undercut bank length, and vertical and horizontal depth were ≥10 cm, the area of undercut bank was estimated from measurements of average horizontal depth and length of the bank. Unembedded boulders > 0.5 m long axis were also counted. In 2007, 2011, and 2013 the number of pieces and volume of large wood were tallied by spatial reference tag interval. Only wood with some portion in zones 1, 2, or 3 (above active channel height but over the active channel) (Robison and Beschta, 1990) with a diameter ≥10 cm two meters in from the large end of the piece and ≥2 m length were included. Total length and mid-point diameter were estimated for each piece and every tenth piece was measured and compared to visual estimates for each observer.

Stream discharge was measured continuously (10-min intervals) at the downstream end of each catchment (Fig. 1). Discharge at the mouth of Flynn was measured using a broad crested compound V-notch concrete weir. A smaller compound V-notch crest with vertical concrete walls was used in Needle Branch (Stednick, 2008b). Water samples for chemical analysis were collected monthly from January 2006 through September 2014 from seven sites in Needle Branch and seven sites in Flynn Creek (Fig. 1). Water samples were analyzed for nitrite-, nitrate-, and ammonia-nitrogen, and ortho- and total-phosphorus. Water temperature was monitored at three sites in each catchment from July through September before (2006–2008) and after (2010–2012) timber harvest (Bladon et al., 2016). The 7-day moving averages of daily maximum, daily mean, and diel fluctuation of stream temperatures were assessed and analyzed for changes related to logging in Needle Branch (Bladon et al., 2016).

3.2. Fish capture

Fish sampling occurred annually in early to mid-August immediately following the annual habitat survey. Estimates of relative abundance for age-1+ coastal cutthroat trout and juvenile coho salmon were derived from a census of pool and cascade habitats throughout the fish-bearing portion of each catchment using single-pass electrofishing (Bateman et al., 2005). We chose to exclude age-0 cutthroat trout because of difficulty in obtaining precise estimates of abundance for this age group (Thompson and Rahel, 1996; Peterson et al., 2004) and high probability of injuring larger fish at voltages necessary to collect fish < 80 mm effectively (Hollender and Carline, 1994; Dalbey et al., 1996; Thompson et al., 1997). Single-pass electrofishing was assumed to provide an unbiased relative measure of salmonid abundance during our study because the number of fish captured with single-pass electrofishing is strongly correlated with multiple-depletion (Bateman et al., 2005) and mark-recapture population estimates (Foley et al., 2015) in streams similar to Needle Branch and Flynn Creek. In addition, we had the same crew leaders present for all electrofishing, which

Table 1

Catchment	Scale	Drainage area (ha)	Gradient (%)	Mean wetted width (m)	Mean active channel width (m)	Mean valley floor width (m)	Length fish-bearing channel (m)	Logged (%)
Flynn Creek	Catchment	212	7.3	1.34 (0.11)	3.14 (0.26)	15.05	4276	0
	Headwater	85	8.1	1.12 (0.13)	2.68 (0.20)	12.62	2082	0
Needle Branch	Catchment	85	5.7	1.11 (0.15)	2.29 (0.25)	15.84	2078	40
	Headwater	34	5.5	1.00 (0.21)	2.14 (0.34)	5.81	785	100

Channel characteristics with associated standard deviation in parentheses by stream for the catchment and headwater assessment scales for Flynn Creek and Needle Branch. Drainage area is from the downstream pour point of each scale (see Fig. 1).

helped to ensure more consistent effort and capture efficiency.

Electrofishing effort was limited to pools and cascades because these channel units provide the primary feeding and survival habitats in the study streams during summer when sampling occurred, and age-1+ coastal cutthroat trout and juvenile coho salmon were less commonly encountered in other habitat types (Bateman et al., 2016).

Single-pass electrofishing began at the downstream end of the main stem or tributary channel and progressed upstream until all pool habitats in the fish-bearing portion of the channel were sampled. A fixed set of markers attached to riparian trees or shrubs approximately every 15 m (measured along the thalweg using a hip chain) provided spatial reference (distance upstream from the V-notch weir) for each sampled habitat unit.

Fish from each habitat unit were processed separately. Captured fish were anesthetized in a solution of buffered MS-222 (40 mg·L^{-1}). Fork length (nearest 1 mm) and wet weight (nearest 0.1 g) were recorded for each individual. Age-0 and age-1+ coastal cutthroat trout were differentiated using length-frequency histograms; individuals ≥80 mm fork length (FL) were considered age-1+.

Following measurements, individually identifiable passive integrated transponder tags (PIT; Texas Instruments, Inc., Dallas, Texas) were surgically implanted in all coastal cutthroat trout ≥100 mm (Bateman and Gresswell, 2006). The half-duplex tags were glass-encapsulated and weighed 0.6 g (air). Fish were allowed to recover (defined by upright swimming) in an aerated bucket of stream water prior to release near the location of their capture.

3.3. PIT-tagged fish detection

Fixed and portable antennas were used to detect PIT-tagged fish. Fixed PIT tag antennas (Babin-Zydlewski et al., 2001) were installed immediately upstream of the V-notch weirs at the downstream boundary of each study catchment during the summer of 2006 (Fig. 1). Two antennas were installed at each site so that direction of travel could be determined, and antennas were operated continuously year-round. Fish detection data were uploaded at bi-weekly intervals. Antennas were tuned at the same time to maintain a minimum read range of 25 cm in any direction; at this range, 100% of the test PIT tags were generally detected when floating through antennas.

Beginning in October 2006, portable PIT tag antennas (Baben Zydlewski et al., 2001) were used to obtain spatially precise locations of tagged coastal cutthroat trout. Typically, portable PIT tag antenna surveys occurred in October, December, February, April, and June. Because discharge is usually greater during these months, age-1+ cutthroat trout are less likely to be restricted to pool habitats. Therefore, the entire wetted area of the fish-bearing portion of each study stream network was scanned during portable antenna surveys.

Because portable PIT tag antennae have a maximum detection range of < 1 m, two portable antennas were always used in the main stem of Flynn Creek up to the junction with tributary 6 (Fig. 1). The number of antennas used in the main stem of Needle Branch was discharge dependent (i.e., when flows were high two antennas were used; only one was used during periods of low flow). Additional information on methods relative to PIT tag detections is available in Bateman et al. (2016).

Tag number, time, habitat characteristics, and location in relation to the nearest distance marker, were recorded for each detection. Because detection of a PIT tag with a portable antenna does not always signify a live fish (Hill et al., 2006; Bateman et al., 2009), we used protocols developed by Bateman et al. (2016) to qualitatively categorize tag status. Categories included live trout (tag location changed), possible live trout (tag detected in location likely to support trout), possible shed tag (tag detected in location unlikely to accommodate trout), and shed tag (tag detected in location that would not accommodate trout, i.e., outside the stream channel, in a dry channel or in shallow water with very fine substrate). Categories were determined in the field by experienced observers. In cases where tag location did not change, observers would typically disturb the area in an attempt to encourage fish movement.

4. Data analysis

Because the effects of logging may vary with proximity to the harvest unit, stream habitat and fish population response variables were analyzed at two different spatial scales: (1) the entire catchment (i.e., main stem and tributaries combined); and (2) the headwaters alone (i.e., upstream from tributary 1 in Needle and upstream of tributary 4 in Flynn (Table 1)).

Stream habitat response variables included channel-unit substrate composition (e.g., percent fines), channel-unit composition (e.g., percent pool habitat), maximum depth of pools, deep-pool threshold (90th percentile of maximum depth), riffle-rapid maximum depth, and nutrients (nitrite-nitrogen, nitrate-nitrogen, and ammonia-nitrogen and *ortho*-phosphate and total-phosphorus). Total catch (numbers and mass), density (fish·m^{-2}), biomass density (g·m^{-2}), size-at-age, body condition, growth, and movement comprised the fish population variables, but growth and movement were evaluated only for age-1+ coastal cutthroat trout. Densities were calculated as the total number of individuals, or total grams captured, divided by the total surface area of pool and cascade habitat sampled. Within-catchment comparisons of nutrient concentrations were assessed statistically using a Wilcoxon Rank Sum Test.

Analyses at the headwater scale were focused only on age-1+ coastal cutthroat trout because juvenile coho salmon were not present in all fish-bearing portions of the stream network in all years (Moore and Gregory, 1988). Age-1+ coastal cutthroat trout were found throughout the entire fish-bearing stream network and, therefore, provided a more sensitive response variable for investigating potential effects of headwater logging in the study catchments (Reeves et al., 1997; Gresswell et al., 2006; Hall, 2008).

Fish size was compared using the mean and 90th percentile of fish length separately for age-1+ coastal cutthroat trout and juvenile coho salmon. The 90th percentile was used to identify the threshold for large fish. Late-summer fish body condition (i.e., body weight at a given length) was evaluated for juvenile coho salmon and age-1+ coastal cutthroat trout using Fulton's condition factor K, calculated as: $K = (W/L^3) \times 100{,}000$ where W is fish weight in grams and L is FL in millimeters Anderson and Neumann (1996). Fish growth analysis was

Table 2

Variable	Scale	Mean prelogging (CV)		Mean postlogging (CV)		Change in prelogging and postlogging difference (CV)	LCL	UCL
		Needle Branch	Flynn Creek					
Bedrock (%)	Catchment	4 (28)	4 (8)	2 (22)	5 (19)	2 (39)	1	4
	Headwater	2 (92)	6 (22)	1 (40)	8 (26)	3 (93)	−1	7
Boulder (%)	Catchment	1 (76)	4 (32)	1 (44)	3 (17)	−1 (11)	−2	1
	Headwater	1 (64)	3 (26)	1 (65)	3 (47)	−1 (89)	−2	0
Cobble (%)	Catchment	5 (60)	13 (26)	7 (20)	16 (6)	1 (407)	−5	7
	Headwater	5 (66)	7 (53)	8 (23)	8 (27)	−1 (380)	−6	4
Gravel (%)	Catchment	43 (27)	47 (10)	46 (19)	39 (20)	−10 (79)	−23	3
	Headwater	49 (32)	24 (25)	51 (19)	30 (27)	5 (378)	−23	32
Fines (%)	Catchment	34 (26)	21 (13)	25 (17)	18 (23)	7 (163)	−10	23
	Headwater	25 (21)	40 (19)	21 (28)	31 (46)	−5 (197)	−19	10
Channel length in pool habitat (%)	Catchment	32 (31)	23 (22)	34 (14)	18 (10)	−6 (105)	−17	4
	Headwater	20 (40)	13 (29)	29 (18)	9 (21)	−13 (2189)	−23	−4
Pool maximum depth (cm)	Catchment	22 (9)	23 (7)	27 (7)	27 (11)	−2 (100)	−6	1
	Headwater	20 (12)	21 (12)	27 (8)	25 (10)	−3 (81)	−6	1
Deep pools (cm)	Catchment	35 (6)	37 (6)	41 (7)	38 (9)	−6 (60)	−12	0
	Headwater	29 (11)	31 (5)	38 (13)	35 (9)	−5 (85)	−11	2
Riffle-rapid maximum depth (cm)	Catchment	5 (14)	7 (6)	9 (9)	11 (15)	1 (169)	−1	2
	Headwater	4 (39)	6 (7)	8 (11)	10 (18)	1 (54)	−2	4
In-stream cover (%)	Catchment	62 (27)	35 (20)	88 (43)	68 (60)	7 (132)	−8	22
	Headwater	54 (17)	36 (26)	64 (29)	34 (27)	−13 (61)	−31	5
Pools with undercut bank (%)	Catchment	21 (13)	8 (39)	42 (14)	20 (20)	−8 (60)	−17	0
	Headwater	13 (23)	0 (0)	31 (30)	7 (30)	*		
Shrub cover (%)	Catchment	36 (37)	66 (8)	57 (13)	77 (5)	−11 (101)	−30	9
	Headwater	32 (69)	74 (12)	68 (7)	79 (7)	−32 (42)	−56	−8
Canopy cover (%)	Catchment	60 (25)	13 (83)	64 (4)	11 (13)	−6 (201)	−35	22
	Headwater	63 (18)	10 (146)	64 (5)	9 (24)	−2 (418)	−14	11

Mean annual values of habitat variables for the prelogging and postlogging periods and the mean of the prelogging and postlogging difference with the associated coefficient of variation (100·SD/mean) in parentheses and lower (LCL) and upper (UCL) 95% confidence limits for Needle Branch and Flynn Creek at the catchment and the headwater assessment scales. Values for substrate categories represent the percentage of the summer wetted stream channel occupied by substrate size class and pool length represents the percentage of fish-bearing channel length occupied by pool habitat. The category "fines" includes both sand and silt sized particles.

* Assumption of additivity and/or trend violated.

restricted to individuals that had been at large for 330–402 days between captures. Annual relative growth rate (mm·mm^{-1}·year^{-1}) was estimated by: RGR = $((L_2 - L_1)\cdot L_1{}^{-1})\cdot(365\cdot(t_2 - t_1)^{-1})$, where L_2 is the recapture length, L_1 is the initial length, t_1 is the day of initial capture and t_2 the day of recapture (Busacker et al., 1990). Relative growth rates were not included in the BACI analysis because no PIT-tagged fish captured during 2007 in Needle Branch were recaptured in 2008, and therefore, only 2 years of prelogging growth data were available.

Fish movement was evaluated on an annual basis. Movement was defined by relocation ≥50 m from the last known point of detection within the same channel (i.e., main stem or tributary) or relocation in a different channel regardless of distance. Based on assessment at single year time steps, fish were placed in one of four different movement categories (Bateman et al., 2016): (*i*) upstream movement only; (*ii*) downstream movement only; (*iii*) both upstream and downstream movement within a main stem or tributary; or (*iv*) complex, if fish moved between main stems and tributaries or between catchments.

The influence of logging on each response variable, except relative growth rate, was analyzed by first calculating the annual difference between the logged catchment (Needle) and the reference catchment (Flynn) in the response variable mean. Subsequently, *t*-tests were used to assess a statistically significant change in this difference from the prelogging to the postlogging period (Stewart-Oaten et al., 1986; Stewart-Oaten and Bence, 2001). With this approach, differences between Needle Branch and Flynn Creek for each year are used as replicates to compare the prelogging period to the postlogging period (Stewart-Oaten et al., 1986). Prior to statistical analysis, we examined data from the prelogging period for temporal trends and additivity (i.e., parallel trajectories between the manipulated and reference catchment; Stewart-Oaten et al., 1986; Stewart-Oaten, 2003). Additionally, we visually assessed the data for distribution symmetry (normality) and non-constant variance. This approach is common in large-scale field manipulations where spatial replication is not feasible (e.g.,Taylor et al., 2006; Greenwood et al., 2007; Tiegs et al., 2011). We used two-sample Student's *t*-tests when variances during prelogging and postlogging periods were approximately equal and Welch's *t*-tests when these variances were unequal (Bateman et al., 2016). Variables that met the assumptions of the BACI were also normally distributed. Log transformations were applied to variables that failed to meet the assumption of no trend in the pretreatment period, (Stewart-Oaten et al., 1986), but conversions universally failed to eliminate trends. The NCSS software (Hintz, 2007) was used for all statistical analyses.

5. Results

During the study period, pools composed a mean of 33% of the 2078 m of fish-bearing stream channel (range 16–42%) in Needle Branch. Minimum pool habitat occurred in 2006 during late summer drought conditions, when 1249 m of the channel of Needle Branch was dry. In other years, dry channel ranged from 0 to 34 m. In Flynn Creek, a mean of 20% of the 4276 m of fish-bearing stream channel was classified as pools (range 16–29%) during the study period. The maximum length of dry channel in Flynn Creek was 150 m in 2007; in other years, dry channels ranged from 0 to 53 m.

Large wood was the most common primary pool-forming agent in both Needle Branch (mean = 59% of pools; range = 50–73%) and Flynn Creek (mean = 41% of pools; range = 29–47%) during the entire study. More pieces of large wood were documented in Needle Branch (mean = 34, 34, and 38 pieces per 100 m of stream channel for 2007, 2011, and 2013, respectively) than in Flynn Creek (mean = 22, 23, and 23 pieces per 100 m for 2007, 2011, and 2013). However, pieces of large wood were generally smaller in Needle Branch (mean volume = 12.3, 10.4, and 15.8 m^3 per 100 m of stream channel for 2007, 2011, and 2013, respectively) than in Flynn Creek (mean volume = 23.5, 23.6, 37.9 m^3 per 100 m of stream channel for 2007, 2011, and 2013, respectively). Mean maximum pool depth ranged from

19 to 31 cm and 22 to 32 cm in Needle Branch and Flynn Creek, respectively. Mean maximum depth in riffle-rapid habitats ranged from 5 to 10 cm and 6 to 14 cm in Needle Branch and Flynn Creek, respectively. Gravel was the most common substrate in both Needle Branch and Flynn Creek (mean = 46% and 44%, respectively). Fines (sand and silt combined; mean 29% and 19% of wetted streambed in Needle Branch and Flynn Creek, respectively) were more common than boulder and bedrock substrates in both catchments (mean < 5% of the wetted streambed in both catchments).

The dominant source of shade differed among catchments. In Needle Branch, the canopy layer provided almost two-thirds of the shade at both the catchment and headwater spatial scales, but shrubs provided the greatest amount of shade in Flynn Creek, regardless of spatial scale or period (mean = 66% and 74% at the catchment and headwater scale, respectively; Table 2). Deciduous shrubs and non-woody vegetation (e.g., grass, sedge, and ferns) comprise > 99% of vegetation included in the shrub layer in both catchments throughout the study. Small conifers accounted for < 1% of the shrub layer in both catchments. Deciduous trees adjacent to the channel were the most prevalent component of canopy shade in Needle Branch (mean = 95%), but in Flynn Creek, conifers were much more abundant (mean = 63% of canopy shade).

Abundance of juvenile coho salmon varied in Needle Branch throughout the study. Total catch of coho ranged from 330 in 2010 to 1515 in 2007 (mean = 756; coefficient of variation (CV) = 45%). Juvenile coho salmon abundance was less variable in Flynn Creek where annual total catch ranged from 839 in 2010 to 1558 in 2009 (mean = 1202; CV = 21%). Similarly, abundance of age-1+ coastal cutthroat trout was more variable in Needle Branch with annual total catch ranging from 41 in 2007, the summer after the 2006 drought, to 328 in 2012 (mean = 204; CV = 43%). In Flynn Creek, annual catch of age-1+ coastal cutthroat trout ranged from 205 in 2012 to 397 in 2013 (mean = 302; CV = 22%). The largest coastal cutthroat trout captured were 225 and 201 mm in Needle Branch and Flynn Creek, respectively. In general, fish ≥200 mm were rare in both streams; only three were captured in each stream during the study.

5.1. Headwater scale comparisons

There were few statistically significant differences detected in habitat between the pre- and postlogging periods at the headwater scale (Table 2). Statistically significant differences were observed in the percent of pool habitat and the percentage of shade from shrubs (Table 3, Fig. 2). The percentage of channel length in pool habitat increased from a mean of 20% to 29% from the pre- to postlogging period in Needle Branch, but in Flynn Creek, the percentage declined from 13% to 9% between periods (Table 2). Meanshade from shrubs increased from 32% to 68% in Needle Branch between periods, but mean estimates for the prelogging and postlogging periods were similar (74% and 79%, respectively) in Flynn Creek (Table 2).

At the headwater scale, grams from total catch (Fig. 3) and density (fish·m^{-2}) (Fig. 4, C and D) of age-1+ coastal cutthroat trout differed significantly between the pre- and postlogging periods (Table 3). Mean mass of age-1+ coastal cutthroat trout from total catch during the prelogging period was 270 and 693 g for Needle Branch and Flynn Creek, respectively. The mean postlogging mass from total catch increased to 1770 g in Needle Branch but decreased to 589 g in Flynn Creek (Table 4). Mean density of age-1+ coastal cutthroat trout was 0.1 and 0.3 fish·m^{-2} in Needle Branch and Flynn Creek, respectively, during the prelogging period. Mean density increased in Needle Branch during the postlogging period to 0.4 fish·m^{-2}, but in Flynn Creek, density of age-1+ coastal cutthroat trout was unchanged from the prelogging period (mean = 0.3 fish·m^{-2}; Table 4).

A statistically significant increase in the large fish threshold (Table 3) was observed following the logging in Needle Branch. Differences in the large fish threshold were similar in Needle Branch and Flynn Creek during the prelogging period, but postlogging, the large fish threshold increased slightly in Needle Branch but declined in Flynn Creek (Fig. 4 E and F, Table 4). Differences in total catch, mean length, and condition of age-1+ coastal cutthroat trout were not statistically significant. Estimates of age-1+ coastal cutthroat trout biomass density (g·m^{-2}) failed to meet the assumptions of the BACI, and therefore, statistical analysis was not possible (Table 4). Juvenile coho salmon were uncommon in the headwaters portion of Needle Branch; 47 juvenile coho salmon were captured in 2007, and 1 was captured in 2008.

5.2. Catchment scale comparisons

There were few statistically significant differences detected in habitat between the pre- and postlogging periods at the catchment scale (Table 2). At the catchment scale, the percentage of pools with undercut banks increased in Needle Branch relative to Flynn Creek following logging, and the difference was statistically significant (Table 3). Although the percentage of pools with undercut banks increased in both catchments during the postlogging period, changes were greater in Needle Branch (Fig. 5A and B; Table 2). A decline in the percent bedrock substrate was observed in Needle Branch relative to Flynn Creek from pre- to postlogging (Fig. 5, C and D, Table 2); the difference was statistically significant (Table 3). Although the deep-pool threshold in Flynn Creek was greater than or similar to Needle Branch in the prelogging period, it was greater in Needle Branch during four of five years of the postlogging period (Fig. 5, E and F Tables 2 and 3).

At the catchment scale, differences in total biomass (Fig. 3), total catch, and biomass density (Fig. 4A and B) of age-1+ coastal cutthroat trout differed significantly from the pre- to postlogging periods (Table 3). Mean total biomass of age-1+ coastal cutthroat trout

Table 3

Scale	Variable	*P*-value	T-value	Degrees of freedom
Catchment scale	Age-1+ coastal cutthroat trout total catch	0.02	3.12	7
	Age-1+ coastal cutthroat trout total biomass (g)	0.01*	4.44	4.4
	Age-1+ coastal cutthroat trout biomass density (g·m^{-2})	0.05	2.59	7
	Pools with undercut banks (%)	0.04	2.50	7
	Bedrock substrate (%)	0.01	3.73	7
	Deep pools	0.04	2.49	7
Headwater scale	Age-1+ coastal cutthroat trout total biomass (g)	0.01*	4.44	4.3
	Age-1+ coastal cutthroat trout density (trout·m^{-2})	0.01	3.28	7
	Large Age-1+ coastal cutthroat trout	0.03	2.66	7
	Pool habitat (%)	0.01	3.29	7
	Shrub cover (%)	0.05*	2.79	3.9

Statistically significant comparisons of habitat and biological variables between the prelogging and postlogging periods in Needle Branch (treatment) and Flynn Creek (reference) at the catchment and the headwater assessment scales.

* Denotes use of Welch's *t*-test due to unequal variance.

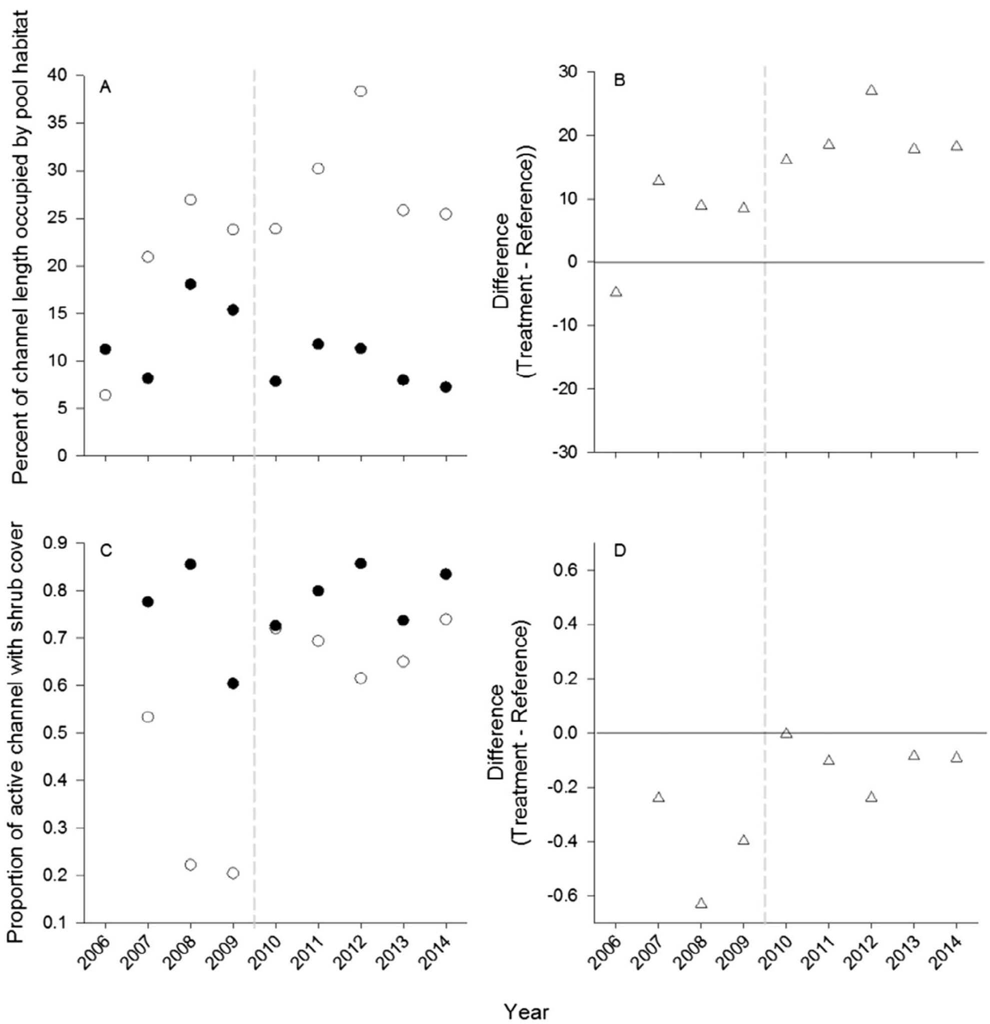

Fig. 2. Headwater scale annual estimates of the percent of channel length occupied by pool habitat (A), and the proportion of the active channel area with shade from shrubs (C) with associated annual difference in measured values (B and D) in Needle Branch and Flynn Creek, Lincoln County, Oregon, USA. Estimates in the treatment (Needle Branch; open circles) and reference (Flynn Creek; solid circles) catchments before and after logging (vertical dashed line), which occurred during the summer and fall of 2009.

increased from 1634 to 4092 g in Needle Branch while declining from 4533 to 3714 g in Flynn Creek from the pre- to postlogging period (Table 4). Mean total catch of age-1+ coastal cutthroat trout increased from 123 to 268 individuals in Needle Branch while declining from 316 to 290 individuals in Flynn Creek from pre- to postlogging periods (Table 4). Biomass density increased from 2.0 to 3.8 $g{\cdot}m^{-2}$ in Needle Branch while declining from 3.3 to 3.1 $g{\cdot}m^{-2}$ in Flynn Creek from the pre- to postlogging periods. There was no evidence for statistically significant changes due to treatment in age-1+ coastal cutthroat trout density ($fish{\cdot}m^{-2}$), condition, mean length, or threshold for large fish (Table 4). We failed to detect statistically significant changes in any of the biotic parameters measured from pre- to postlogging periods for juvenile coho salmon (Table 4).

5.3. Fish growth and movement

Relative growth rate was estimated for 146 PIT-tagged coastal cutthroat trout that were recaptured during the study period. Mean length at initial tagging increased from 107 to 116 mm in Needle Branch between the prelogging and postlogging periods, but mean length declined from 124 to 116 mm in Flynn Creek during the same time period. Mean relative growth rates were lower in Needle Branch (0.21 $mm{\cdot}mm^{-1}{\cdot}year^{-1}$, CV = 0.21) than Flynn Creek (0.28 $mm{\cdot}mm^{-1}{\cdot}year^{-1}$, CV = 40), but differences between catchments in growth were generally consistent through time (Fig. 6). No PIT tagged fish were recaptured in Needle Branch during water year 2008, and we could not test for trend in relative growth rates between the catchments during the pretreatment period. Therefore, it was not possible to assess whether growth data met the assumption of the BACI, and we did not statistically assess the effects of logging on growth. Mean relative growth rate in Needle Branch was relatively consistent through time, however, despite increases in abundance and biomass during the period (Fig. 6).

Movement of PIT-tagged coastal cutthroat trout was common in the study catchments, and mean proportion of tagged fish classified as movers was 29% and 37% for Needle Branch and Flynn Creek, respectively. For all years combined, coastal cutthroat trout in Needle Branch were most frequently moving downstream-only (75% of coastal cutthroat that moved). In Flynn Creek, the downstream-only movement pattern was also common but accounted for only 42% of the detected movers; 39% exhibited the complex movement pattern. The upstream-only movement pattern was the least common, accounting for 9 and 10% of the coastal cutthroat trout that moved in Needle Branch and Flynn Creek, respectively. The percentage of tagged individuals detected moving annually ranged from 16% to 42% in Needle Branch and from 24% to 48% in Flynn Creek. Differences from the pre-to postlogging periods were not statistically significant for any of the movement patterns ($|t| \leq 1.869$; $P \geq 0.111$).

Movement appeared to have minimal effect on coastal cutthroat trout abundance estimates based on electrofishing surveys. None of the age-1+ coastal cutthroat trout tagged below the first falls in Needle Branch were ever recaptured upstream during electrofishing surveys. However, five coastal cutthroat trout tagged above the falls but downstream of the headwater assessment area were subsequently

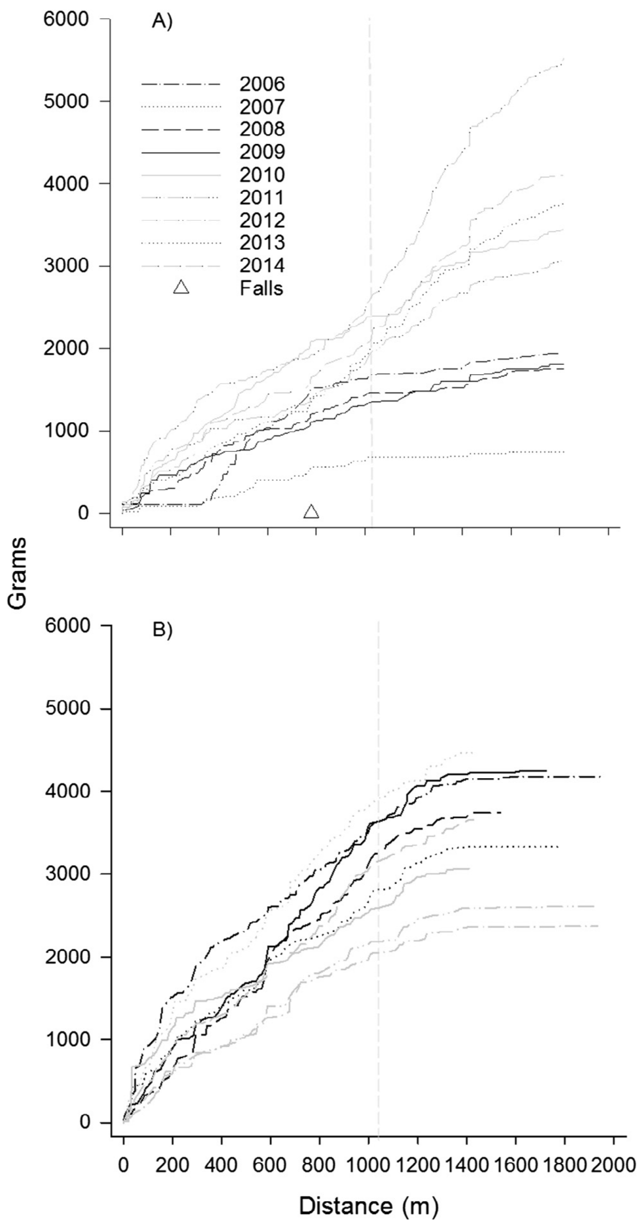

Fig. 3. Cumulative catch in grams of age-1+ coastal cutthroat trout from annual censuses of pool and cascade habitats for the main stem channels of Needle Branch (A) and Flynn Creek (B). Black lines are years before logging, grey lines are years after logging. Dashed vertical grey line indicates the downstream end of the headwater assessment scale which in Needle Branch coincides with the downstream extent of the clearcut harvest unit and the junction of tributary 1 with the main stem in Needle Branch and tributary 4 and the main stem in Flynn Creek. The triangle in figure A represents the most downstream falls in Needle Branch. The x-axis displays the distance upstream from the weirs located at the downstream end of each sample catchment. In most years, pools were not present to the determined upper extent of fish in the main stem of Flynn Creek.

recaptured upstream of tributary 1 in the headwater assessment area. These individuals represented 0.4% of the total number (all years combined) of tagged coastal cutthroat trout downstream of the headwater assessment area. Three of these coastal cutthroat trout were originally tagged in 2013. Fourteen coastal cutthroat trout initially tagged in the headwater assessment area were recaptured downstream during electrofishing surveys. These individuals represented 1.9% of all coastal cutthroat trout initially tagged in the headwater assessment area. The greatest proportion of coastal cutthroat trout recaptured downstream of the headwater assessment area (6 of 59) were tagged in 2010.

5.4. Stream nutrients and low flows

In Flynn Creek, differences in ammonia-nitrogen between prelogging (0.010 $mg \cdot L^{-1}$) and postlogging (0.013 $mg \cdot L^{-1}$) periods were statistically significant ($P \leq .05$). Differences in ortho-phosphorus concentrations between prelogging (0.029 $mg \cdot L^{-1}$) and postlogging (0.034 $mg \cdot L^{-1}$) periods were also statistically significant ($P \leq .05$). Similarly, total phosphorus increased between prelogging (0.028 $mg \cdot L^{-1}$) and postlogging (0.036 $mg \cdot L^{-1}$), and differences were statistically significant ($P \leq .05$). Because statistically significant differences occurred in the control catchment before and after logging, changes of these nutrients in Needle Branch (treatment catchment) could not be evaluated in relation to timber harvest. Nitrate- and total-nitrogen were statistically consistent between pre- and postlogging periods in Flynn Creek, however, and therefore, values of these two nutrients represented an unbiased statistical control for assessing the effects of logging in Needle Branch.

At the headwater scale, differences in nitrate-nitrogen (NO3-N) between prelogging (0.99 $mg \cdot L^{-1}$) and postlogging (1.47 $mg \cdot L^{-1}$) periods in Needle Branch were statistically significant ($P \leq .05$). Differences in total nitrogen between prelogging (1.20 $mg \cdot L^{-1}$) and postlogging (1.61 $mg \cdot L^{-1}$) at the headwater scale were also statistically significant in Needle Branch ($P \leq .05$). The high N:P ratio did not suggest a nitrogen limitation in stream productivity in either sample period. A more detailed water quality analysis is under review (Harbin and Stednick, in review).

Instantaneous daily low stream flows annually normalized by area ($L \cdot sec^{-1} \cdot km^{-2}$) were compiled for all study years (Table 5). Annual daily low flows for Flynn Creek were similar during pre- and postlogging periods. After harvesting, annual daily lowest flows increased by over 100% in Needle Branch (Table 5).

6. Discussion

In this study, we evaluated the effects of timber harvest in a second-growth forest conducted with contemporary forest management techniques. Specifically we evaluated a clearcut logging treatment with a standing tree riparian buffer in the fish bearing portion of the logged catchment. Our approach incorporated both sampling intensity and spatial and temporal extent in order to maximize inference at the catchment scale (Gresswell et al., 2006; Bateman et al., 2016). The Alsea Watershed Study provided the opportunity to compare our results to those associated with previous timber harvest in the Needle Branch catchment. None of the habitat and biological variables responded negatively to logging, and indeed, individual characteristics either did not differ statistically or increased in a positive direction for the duration of the study. For example, significant changes in habitat included a decline in bedrock substrate, and an increase in percent pools, the deep pool threshold, and the number of pools with undercut banks observed in Needle Branch relative to Flynn Creek. All of these changes appear to be directly related to increased low-flow discharge. Statistically significant responses observed in age-1+ coastal cutthroat trout were generally positive and included an increase in total catch, density ($fish \cdot m^{-2}$), total biomass, biomass density ($g \cdot m^{-2}$), and an increase in larger fish. Failure to detect a treatment effect in juvenile coho salmon for any of the response variables may be related to the distribution of coho within the treated catchment. Juvenile coho were primarily found downstream from the harvest area (Fig. 1) where the greatest changes in age 1+ coastal cutthroat trout occurred. In addition, coho are probably affected by factors outside the study catchments (e.g., commercial and sport fisheries harvest, and mortality in migration and rearing areas) that were not influenced by the current study.

Despite rigor in the study design and sampling strategies, the scope of inference of these results is limited to Needle Branch at the time of the study. Paired-catchment studies provide insights that can improve understanding of mechanisms associated with experimental

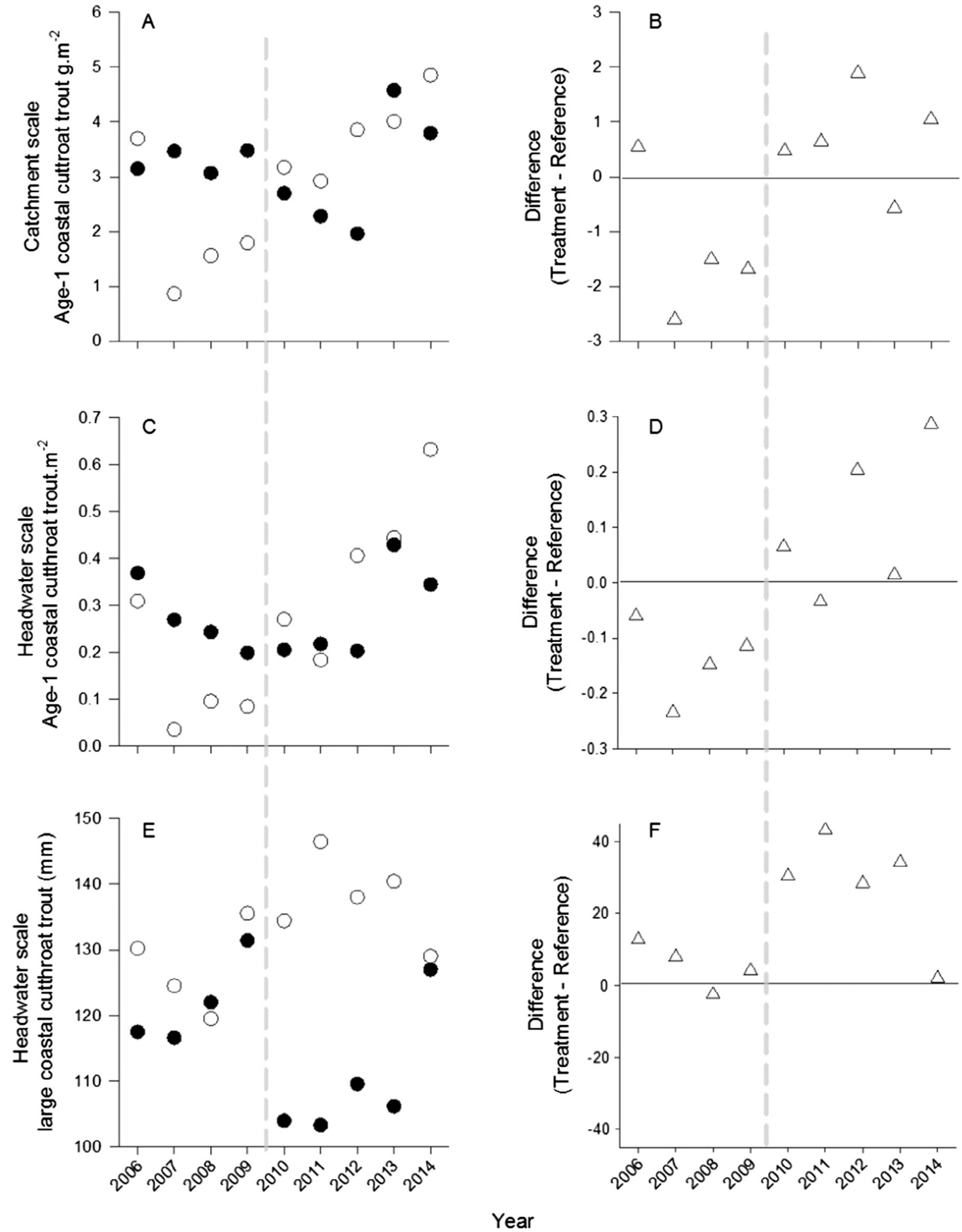

Fig. 4. Catchment scale annual mean relative biomass density of age-1+ coastal cutthroat trout g•m^{-2} (A), headwater scale relative density age-1+ coastal cutthroat trout•m^{-2} (C), and the headwater scale threshold of length for classifying large fish (E) in Needle Branch and Flynn Creek, Lincoln County, Oregon, USA. Estimates in the treatment (Needle Branch; open circles) and reference (Flynn Creek; solid circles) catchments before and after logging (vertical dashed line), which occurred during the summer and fall of 2009. (B, D, F) Differences between measured values of the treatment and reference catchments, before and after logging in Needle Branch.

disturbances; however, replication in other catchments and regions is the key to increasing the inferential scope beyond the individual case study. Concomitantly, recent studies (e.g., De Groot et al., 2007; Bateman et al., 2016) where clearcut logging treatments were applied in small, high-gradient catchments using contemporary logging practices (e.g., existing road networks were revitalized instead of building new roads and large wood was not removed from streams), yielded no detectable acute negative responses in local salmonid populations Although results of this study provide support for the perception that regulations developed for current best management practices have substantially improved outcomes for stream biota relative to unregulated forest harvest (Blinn and Kilgore, 2001; Ice et al., 2010), it is important to note that Needle Branch experienced no episodes of mass wasting and standing tree buffers remained intact (i.e., only minor loss from windfall) during the postlogging period. The response of coastal cutthroat trout to a second timber harvest in Needle Branch was

Table 4

	Variable		Mean prelogging (CV)		Mean postlogging (CV)		Prelogging – postlogging difference (CV)	LCL	UCL
Species		Scale	Needle	Flynn	Needle	Flynn			
Age-1+ coastal cutthroat trout	Total biomass (g)	Catchment	1634 (32)	4533 (8)	4092 (20)	3714 (22)	−3276 (50)	−5044	−1508
		Headwater	270 (41)	693 (10)	1770 (43)	589 (13)	−1604 (49)	−2562	−647
	Total catch (n)	Catchment	123 (39)	316 (12)	268 (19)	290 (28)	−171 (48)	−300	−41
		Headwater	18 (41)	55 (26)	111 (43)	53 (15)	*		
	Density (trout·m^{-2})	Catchment	0.1 (53)	0.2 (8)	0.3 (18)	0.2 (27)	−0.1 (1 0 0)	−0.2	0.0
		Headwater	0.1 (80)	0.3 (26)	0.4 (40)	0.3 (32)	−0.3 (33)	−0.4	−0.1
	Biomass density (g·m^{-2})	Catchment	2.0 (53)	3.3 (6)	3.8 (18)	3.1(32)	−2.0 (50)	−3.8	−0.3
		Headwater	1.8 (79)	3.3 (7)	6.1 (28)	3.1 (30)	*		
	Fulton's K	Catchment	1.01 (6)	1.04 (4)	1.08 (1)	1.08 (3)	0.03 (2)	−0.07	0.14
		Headwater	1.02 (10)	1.05 (5)	1.11 (0.03)	1.09 (2)	−0.05 (2)	−0.15	0.05
	Mean length (mm)	Catchment	108 (6)	107 (3)	109(3)	104 (3)	−5 (1 2 0)	−14	5
		Headwater	106 (6)	101 (5)	111 (5)	95 (3)	−11 (77)	−24	3
	Mean length large fish (≥ 90th percentile of length)	Catchment	132 (8)	135 (4)	135 (5)	126 (3)	−13 (81)	−29	4
		Headwater	127 (5)	122 (5)	138 (4)	110 (8)	−22 (56)	−42	−2
Juvenile coho salmon	Density (fish·m^{-2})	Catchment	0.9 (53)	0.9 (17)	0.7 (35)	0.9 (19)	0.2 (3 0 0)	−0.7	0.9
	Biomass Density (g·m^{-2})	Catchment	1.90 (44)	2.2 (20)	1.6 (32)	2.3(24)	0.5 (1 6 0)	−0.8	1.8
	Fulton's K	Catchment	1.13 (6)	1.20 (5)	1.20 (5)	1.23 (4)	−0.05 (80)	−0.12	0.02
	Mean length	Catchment	56 (4)	56 (2)	57 (2.9)	58 (5)	−0 (1148)	−5	5
	Mean length large fish (≥ 90th percentile of length)	Catchment	67 (3)	67 (2)	72 (3.9)	69 (5)	−3 (1 4 0)	−9	4

Mean annual values of fish response variables for the prelogging and postlogging periods and the mean of the prelogging and postlogging difference with associated coefficient of variation (100·SD/mean) in parentheses and lower (LCL) and upper (UCL) 95% confidence limits for Needle Branch and Flynn Creek at the catchment and headwater scales for age 1+ coastal cutthroat trout and for catchment scale for juvenile coho salmon.

* Assumption of additivity and/or trend violated.

materially different from that observed in the early Alsea Watershed Study (Moring and Lantz, 1975) and surprisingly more robust than the responses reported by De Groot et al. (2007) and Bateman et al. (2016). Indeed, in this contemporary study, total biomass and density of age-1+ coastal cutthroat trout increased in Needle Branch relative to Flynn Creek following harvest, and this was the most negatively affected species/age group during the initial study (Moring and Lantz, 1975). Furthermore, the positive response of age-1+ coastal cutthroat trout was most apparent at the headwater scale where almost 100% of the drainage area was logged (i.e., a riparian buffer was retained along the fish-bearing channel). Although an increase in biomass of age-1+ coastal cutthroat trout was also apparent at the catchment scale (40% of area logged), the response was actually muted when compared to the headwater scale.

Recaptures of PIT-tagged coastal cutthroat trout suggest that changes in abundance at both the headwater and catchment scales in this contemporary forest management study were not the result of individuals moving from one portion of the catchment to another (Gowan and Fausch, 2002). In fact, although many cutthroat trout in this study could be individually identified with PIT tags, annual electrofishing assessments and bimonthly mobile-tracking surveys, which occurred during the 9 years of this study, failed to detect any fish from lower reaches that moved upstream past both waterfalls in the middle section of Needle Branch. This result is not unprecedented. Gresswell and Hendricks (2007) reported that coastal cutthroat trout in another headwater stream of the Oregon Coast Range Mountains moved frequently, but the extent of movements was usually within several habitat units; few cutthroat trout in their study moved downstream over a waterfall that formed an upstream barrier to fish movement. Therefore, we have a high degree of confidence in our conclusion that the increases in biomass observed in the upstream reaches of Needle Branch in the years following harvest are related to increased fish production. In contrast, Hall (2008) reported that large numbers of coastal cutthroat trout moved downstream over the falls in Needle Branch during the original Alsea Watershed Study, and apparently, downstream movement was associated with increased abundance in a portion of the catchment during the postlogging period. Regardless, the elimination of immigration as a potential explanation of population increase suggests that production of coastal cutthroat trout increased in Needle Branch during the postlogging portion of the current study and that the behavioral response of coastal cutthroat trout differed from results observed in Needle Branch during the original Alsea Watershed Study.

In contrast to outcomes from the historical timber harvest methods, current practices produced only small changes in stream temperatures during the warmest portion of the summer (Bladon et al., 2016). There was a statistically significant increase (0.6 °C) in the 7-day maximum temperature for the July 15-August 15 period after logging in Needle Branch, but no differences were detected in other metrics for this period when water temperatures are usually highest or when viewed over the full summer (mid-June to mid-September) (Bladon et al., 2016). The water temperature criterion designed to protect core coldwater rearing environments for salmonids, 16 °C (Buckhouse et al., 2004), was not exceeded in the treatment catchment during this study (Bladon et al., 2016), suggesting fish did not experience thermal stress subsequent to logging in the Needle Branch catchment. In fact, water temperatures were probably more favorable to growth of coastal cutthroat trout in Needle Branch after logging. In addition, summer low-flows increased by approximately 100% in Needle Branch in the postlogging period, and increased flow is often associated with increased food availability (Harvey et al., 2006).

Nitrate concentrations increased significantly at all stations following the harvest and were similar to the results obtained in the original Alsea Watershed study (Harbin and Stednick, in review). This result is not uncommon in streams following disturbances that reduce the standing biomass in forested landscapes (Gresswell, 1999; Bernhardt et al., 2004), but it is important to reiterate that effects can vary substantially in relation to local context (Kreutzweiser et al., 2008; Wootton, 2012). For example, the presence of nitrogen-fixing plants such as alder can affect nitrogen concentrations in adjacent streams (Compton et al., 2003). In this study, nitrogen did not appear to be a limiting nutrient either before or after logging.

Ultimately, however, primary production in headwater streams is limited by a number of variables, including the stream flow and the availability of light and nutrients (Wilzbach et al., 2005; Hill et al., 2011; Warren et al., 2013), that vary through space and time (sensu Warren and Liss, 1980). Although benthic primary production was not

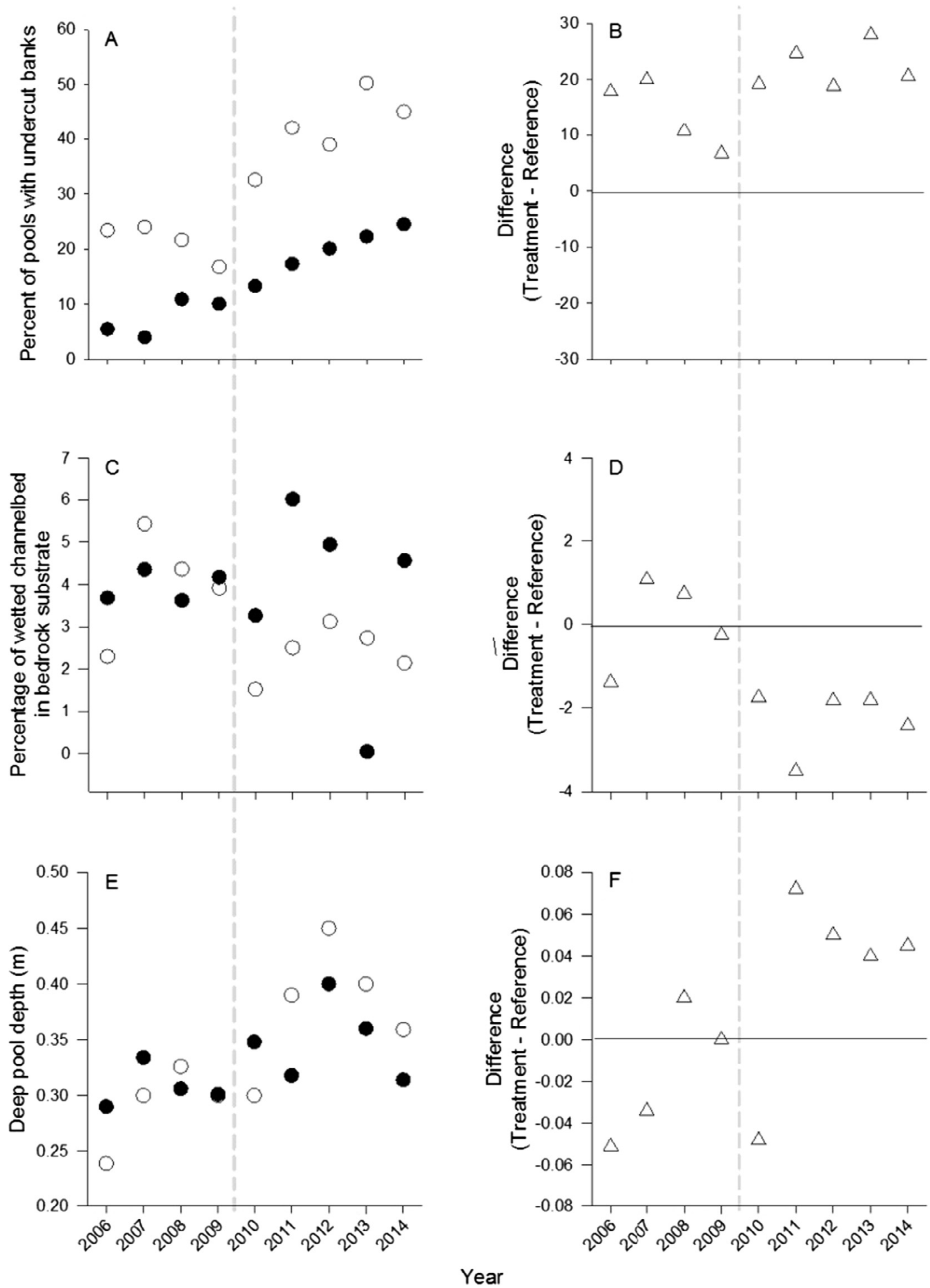

Fig. 5. Catchment scale annual estimates of the percent of pools with undercut banks present (A), the percent of wetted channel area occupied by bedrock substrate (C), and the annual estimate of the threshold for deep pools (E) with associated annual difference in measured values (B, D and F) in Needle Branch and Flynn Creek, Lincoln County, Oregon, USA. Estimates in the treatment (Needle Branch; open circles) and reference (Flynn Creek; solid circles) catchments before and after logging (vertical dashed line), which occurred during the summer and fall of 2009.

quantified directly in this study, given likely increases in insolation and the documented changes in nutrient availability in Needle Branch, we assume that autotrophic production increased in the stream, especially in sections adjacent to the harvest unit. A localized response in primary production is also consistent with the more localized increase of age-1+ cutthroat trout in areas adjacent to the harvest unit, suggesting that increases in primary production where light and nutrients were both elevated may contribute to increase in fish biomass.

Light availability commonly increases in streams following timber harvest, even when buffers are left along the riparian zone (Fuchs et al., 2003; Kiffney et al., 2003; Wilzbach et al., 2005), and light is also a limiting factor for primary production in some headwater streams (Hill and Knight, 1988; Hill et al., 1995; Hetrick et al., 1998). Indeed, increases in light availability are often associated with disturbance that

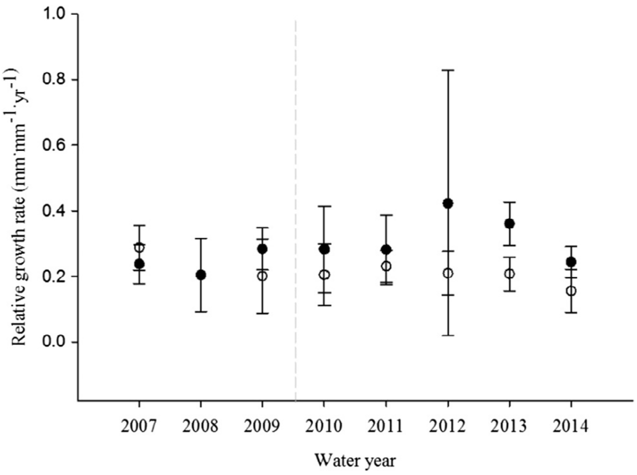

Fig. 6. Mean annual relative growth rate (mm·mm^{-1}·year^{-1}) of PIT-tagged coastal cutthroat trout ≥ 100 mm FL at time of tagging with 95% confidence interval in Needle Branch and Flynn Creek, Lincoln County, Oregon. Estimates in the treatment (Needle Branch; open circles) and reference (Flynn Creek; solid circles) catchments before and after logging (vertical dashed line), which occurred during the summer and fall of 2009.

Table 5

Period	Water year	Flynn Creek	Needle Branch
Prelogging	2006	4.24	0.28
	2007	3.68	0.28
	2008	6.23	0.56
	2009	6.51	0.56
	Mean	5.16	0.42
Postlogging	2010	6.79	0.85
	2011	7.07	0.85
	2012	6.51	1.42
	2013	7.08	0.85
	2014	2.55	0.57
	Mean	6.00	0.91

The annual daily instantaneous low flow normalized by area (L·sec^{-1}·km^{-2}) for each gauging station and year with treatment period means.

removes or damages the riparian canopy, such as logging or fire (Gresswell, 1999; Mellina and Hinch, 2009), and these changes are often accompanied by increases in production. In contrast, we observed a shift toward greater shade from riparian shrubs in the headwaters of Needle Branch following forest harvest. The transition to shade from deciduous riparian shrubs and deciduous riparian red alder from upslope second-growth conifers in Needle Branch has the potential to actually decrease the amount of light available during the time when deciduous shrubs and trees are covered with leaves. In fact, blooms of benthic periphyton are common during the spring in systems dominated by deciduous riparian trees and shrubs before leaves grow and canopy closure has occurred (Hill et al., 1995; Roberts et al., 2007). An increase in benthic primary production may be particularly relevant to coastal cutthroat trout in Needle Branch because late winter/early spring is the period of fastest growth of coastal cutthroat trout in the Oregon Coast Range (Connolly, 1996). Increases in riparian shrubs in the logged catchment may also provide additional terrestrial invertebrate resources for coastal cutthroat trout and juvenile coho salmon during the low-flow periods in mid to late summer (Romero et al., 2005).

Finally, the influence of enhanced summer stream flows following harvest may have contributed to positive changes in density and biomass of coastal cutthroat trout in Needle Branch by increasing available pool habitat. In the decades since the original Alsea Watershed Study, late summer flows in this small headwater system have typically been very low, and substantial lengths of stream can become desiccated in late August-October (Gregory et al., 2008, current study). Moreover, Gregory et al. (2008) suggested that desiccation of substantial portions of Needle Branch during summer droughts might lead to increased interannual variability in coastal cutthroat trout demographics and impede return to prelogging levels. Although fish survival was not measured in the current study, previous research suggests that mortality for coastal cutthroat trout in headwater streams is greatest during low-flow periods in late summer (Berger and Gresswell, 2009). Increases in water yield generally attenuate over a relatively short period (5–10 years; Moore and Wondzell, 2005).

Although all of these factors undoubtedly had some influence on the observed outcome of this study, determining which factor, or group of interacting factors was the primary mechanism of change is far beyond the scope of this study. Indeed, these factors comprise a mosiac of causal mechanisms, the effects of which vary across space and through time. The goal of our study was to document the fish and habitat response to upslope clearcut logging in a second-growth forest, where there was a standing tree riparian buffer. Although the results were encouraging, it is doubtful that short-term increase in coastal cutthroat trout density and biomass following logging will influence persistence of the species in Needle Branch. Indeed, as noted above, press disturbances such as climate change (including persistent long-term changes in flow and water temperature) or catchment conversion to other land uses (e.g., agriculture or residential development) are more likely to affect persistence of coastal cutthroat trout in Needle Branch.

6.1. Implications for forest management

The objective of stream protection rules in forestry (Best Management Practices) has been to prevent or reduce acute negative effects of forest management. Much of the debate on the adequacy of current protection rules is focused on whether the capacity of systems to respond and adapt to disturbance is maintained in managed catchments. To evaluate the effectiveness of current rules in managed second-growth systems, it is important to understand the constraint previous management activities have already imposed on the ability of a system to respond (Ebersole et al., 1997). Needle Branch represents an example of a stream where stream flow characteristics (e.g., annual runoff volume, peak flow discharges, and number of low-flow days), annual sediment yield, and summer maximum stream temperatures were modified by initial timber harvest but have subsequently returned to historic levels (Hale, 2007). Similarly, coastal cutthroat trout biomass was substantially reduced following the previous timber harvest in the catchment, and it remained low for decades, even after total abundance and density estimates had returned to prelogging levels (Gregory et al., 2008). Results obtained during the pretreatment period of the current study were similar to historic prelogging levels, however. Moreover, 31 years following the initial harvest, peak and low-flow metrics in Needle Branch did not differ statistically from the original pretreatment (Stednick, 2008b). It appears that for these variables, Needle Branch retained the capacity to respond to timber harvest following the initial Alsea Watershed Study, and the results of the current study further suggest substantial resilience in this system to the negative consequences of the historic timber management.

Our results are concordant with previous research suggesting that current rules have reduced acute negative effects from logging in small high-gradient catchments of the maritime Pacific Northwest where 20–40% of the catchment has been clearcut (De Groot et al., 2007; Bateman et al., 2016); however, the response to logging, like all disturbances, is context dependent, and caution should be used in generalizing these results to the population of catchments in this region, especially with regard to questions of resilience. Nevertheless, small headwater catchments and the biota that evolved in these systems experience frequent and sometimes deleterious perturbations (Frissell et al., 1986), and therefore, they may have substantial capacity to respond to intermittent short-term (pulse) disturbance. In this study, the focus was on fish and fish habitat because current regulatory restrictions on forest management practices are primarily driven by concerns about fish (Lee et al., 2004). Although interest in the effects of forest management on other aquatic vertebrates is growing and results from recent manipulative studies are equivocal (Leuthold et al., 2012; Olson et al., 2013; Chelgren and Adams, 2017), speculation on the effects treatments applied in this study on those species is beyond the scope of this paper. Concomitantly, we expect that results would be much different in systems where perturbations (e.g., climate change or the conversion of forested catchments to agricultural or urban uses) persist through time (press disturbance).

Our paired-watershed study in the Alsea River Basin was designed to evaluate acute effects of contemporary forest management practices, including standing tree riparian buffers, on fish in headwater streams. We evaluated the effects of stream-adjacent clearcut logging of the upper 40% of the Needle Branch catchment following contemporary forest practice regulations (Oregon Department of Forestry, 2006). Most importantly, we did not observe any statistically significant declines in stream habitat characteristics or metrics used to describe the salmonid community (coastal cutthroat trout and juvenile coho salmon). Moreover, there was a statistically significant increase in the total biomass of age-1+ coastal cutthroat trout at the catchment scale, and the density and mean length of larger age-1+ coastal cutthroat trout at the headwater scale. Furthermore, relative growth rate of PIT-tagged coastal cutthroat trout in Needle Branch remained similar following logging.

Although results of our study support those observed in Deer Creek where standing tree buffers were used during the original Alsea Watershed Study (Hall and Stednick, 2008), there are surprisingly few more recent studies with which to compare. Our study and those of De Groot et al. (2007) and Bateman et al. (2016) document a gradient of treatments all suggesting that negative effects to fish and fish habitat are difficult to detect in high-gradient headwater streams during the first 5 years or less following logging conducted in compliance with current forest practice rules, including leaving large wood in the stream channel. These results also seem compatible with the findings of Mellina and Hinch (2009), who reported that negative responses timber harvest were primarily associated with logging in conjunction with stream cleaning. However, questions remain about the capacity of second-growth catchments to respond to harvest. Our study suggests that in at least this instance, where water quality and fish abundance were similar to pre-management values, the response to logging using contemporary practices and following current Oregon regulations was neutral to generally positive. Context is important, however, and the same study conducted at the same site at a different time could yield different results. Concomitantly, as forestry best management practices continue to evolve broadly, and especially as they are adapted to address local conditions, it appears that the larger, more acute, negative effects of past logging practices are less likely to occur.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.foreco.2018.01.030.

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