

## Fish response to successive clearcuts in a second-growth forest from the central Coast range of Oregon

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### ABSTRACT

Research dating back to the 1950 s has documented negative effects from harvesting of primeval forests on stream ecosystems of the Pacific Northwest. By the early 1990 s, state and federal forest practice rules governing timber harvest were modified throughout North America to better protect aquatic habitats and biotic resources, principally salmonids. These rules inspired a generation of studies using a before-after-control-impact (BACI) design to document the capacity of contemporary timber harvest rules to protect salmonids in headwater streams of second-growth forests. One important unanswered question concerns the potential effects of successive clearcuts in second growth forests. Consequently, we used a paired watershed approach to evaluate the effects of two successive clearcut harvests in the Alsea Watershed, site of the seminal Alsea Watershed Study that was conducted from 1958 to 1973, on relative biomass, movement, survival, and distribution of coastal cutthroat trout (*Oncorhynchus clarkii clarkii*) and three physical habitat characteristics (pool area and depth, and water temperature). Although the total clearcut harvest encompassed 87% of the treatment catchment in six years, no negative effects of logging were detected for either age-1 + coastal cutthroat trout or habitat variables. Comparisons between the harvested and reference catchments suggested the survival of coastal cutthroat trout (>94 mm fork length) and total catchment relative biomass of age-1+ (i.e., > 80 mm) exhibited similar patterns, increasing from the pre-logging period (2006–2009) through the Phase I post-logging period (2009–2014), and decreasing to levels observed in the pre-logging period during the Phase II post-logging period (2014–2017). Additionally, there was no evidence for differences in movement of coastal cutthroat trout related to the harvesting treatment. In terms of habitat variables, there was a relative increase in annual total pool area in the harvested catchment during the Phase II post-logging period, but there was no evidence the 7-day moving mean maximum stream temperature changed after the Phase I and Phase II harvests. Moreover, stream water temperatures never exceeded the criterion designed to protect core coldwater habitat for salmonids (16 °C). As such, it is unlikely that cutthroat trout experienced thermal stress following either harvest. More generally, results from this and other recent studies suggest that forest practice rules developed in conjunction with current best management practices for logging in headwater catchments have substantially improved outcomes for stream biota relative to unregulated forest harvest, at least for short periods of time after logging (i.e., ≤ 8 years).

### 1. Introduction

Research on the effects of forest management on stream ecosystems in the Pacific Northwest extends back to the 1950 s (Northcote and Hartman, 2008). For example, long-term monitoring associated with studies such as the Alsea Watershed Study, Oregon, USA (1959–1974;

Stednick, 2008) and the Carnation Creek and Queen Charlotte Islands studies, British Columbia, Canada (1970–present; Tschaplinski et al., 2004) provided valuable information on the negative effects of historical forest harvest practices to streams and fish. In the following decades a number of studies on fluvial systems in the Pacific Northwest provided important insights into the effects of timber harvest on the ecological

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processes of these systems, including alterations to stream hydrology and temperature, organic debris delivery and accumulation, sedimentation, channel morphology, habitat complexity, and ultimately the fishes supported by the streams (e.g., Brown and Krygier 1970; Gregory et al. 1987; Bilby and Ward 1991; Reeves et al. 1993). Findings from these studies have resulted in a series of state and federal forest practice rules intended to provide improved protection of aquatic habitats and biotic resources, principally salmonids (Hall et al., 2004; Ice et al., 2010).

Contemporary forest harvest practices differ substantially from what were largely unregulated historical practices. For example, contemporary forest harvests primarily occur in previously harvested stands (second-growth forests), rather than unharvested forests (often old-growth or late-seral forests; Franklin and Van Pelt, 2004; Brockerhoff et al., 2008). Additionally, road systems remaining from past harvest activities reduce the need for new road construction. Contemporary forest management activities are also guided by a complex array of “best management practices” to minimize undesirable effects (Ice et al., 2010; Cristan et al., 2016). Indeed, there is substantial interest in the effects of these new practices on stream ecosystems, particularly smaller (lower-order) streams located on private forestlands where the majority of timber harvest currently occurs (Johnson et al., 2007). In mountainous catchments of the Pacific Northwest, USA, these lower-order systems represent the majority of total channel length (Leopold, 1964), and because lower-order streams are closely linked to riparian and upslope conditions, they are often directly affected by anthropogenic activities, such as forest harvest (Vannote et al., 1980; Gomi et al., 2002; Richardson and Danehy, 2007).

Since 1997, four case studies in the Pacific Northwest have incorporated before-after-control-impact (BACI) designs to evaluate the effects of contemporary logging practices in second-growth forests on fishes in headwater streams (De Groot et al., 2007; Bateman et al., 2016; Jensen, 2017; Bateman et al., 2018). In the context of the paired watershed design, the BACI approach accounts for environmental variability that inevitably occurs among treatment periods. While a diverse array of riparian and upslope treatments and sampling strategies was applied among these previous studies (see Supplemental Table S1 for details), all of them occurred in second-growth forests. Furthermore, none of these studies incorporated any form of stream “cleaning,” a largely historical practice whereby large wood was removed from the stream, typically with strong negative effects on salmonid abundance (Mellina and Hinch, 2009). Two of the recent studies evaluated downstream effects on fish from logging that occurred upstream in fishless areas (Bateman et al., 2016; Jensen, 2017). Comparatively, the other two studies evaluated treatments where at least some portion of the harvest unit was adjacent to a fish bearing channel (De Groot et al., 2007; Bateman et al., 2018). In contrast to research associated with historical forest harvests, results of these recent studies revealed no statistical evidence of short-term ( $\leq 5$  years post logging) negative effects to salmonid fish populations.

Although these studies suggested that contemporary forest harvest practices can protect streams and fish, some questions remain unanswered. For example, contemporary forest practice rules restrict the size of individual harvest units and establish standards for the minimum elapsed time prior to additional harvest adjacent to existing units (Oregon Department of Forestry, 2006). But, under this policy it is possible that large areas in a single catchment can be converted to early seral-stage plantations within 10–20 years. Concomitantly, information concerning fish response to timber harvest has generally focused on the first phase of the overall harvest and thus, a small proportion of the total watershed. The potential effects of successive clearcuts in second growth forests, including the response of fish populations in lower-order streams, have not been reported.

Here, we evaluated survival, relative biomass, movement, and distribution of coastal cutthroat trout (*Oncorhynchus clarkii clarkii*), and three physical habitat characteristics (pool area, pool depth, and water

temperature) after a secondary clearcut harvest occurred in the Coast Range, Oregon immediately after “green up” requirements were met following the initial harvest (Bateman et al., 2018). Specifically, the harvest occurred in Needle Branch, a watershed that was clearcut harvested during the original Alsea Watershed Study in 1966. In our study Needle Branch was harvested with two successive smaller harvests, due to clearcut size limits; however, the combined clearcut harvest area was equivalent to the total area harvested in the original study.

Because evaluating the effects of habitat changes on fish abundance through time is often confounded by the interactions among mortality, emigration, immigration, and production (Gowan et al., 1994; Roni et al., 2015), we chose to assess both intra-catchment movement and survival in addition to coastal cutthroat trout relative biomass and physical habitat metrics. Given the results of recent studies of logging in second-growth forests (De Groot et al., 2007; Bateman et al., 2016; Bateman et al., 2018), we hypothesized that statistically significant decreases in survival and relative biomass of age-1+ (i.e.,  $\geq 80$  mm, fork length) coastal cutthroat trout would not be detectable following the second harvest in Needle Branch and movement patterns would remain similarly unchanged.

## 2. Methods

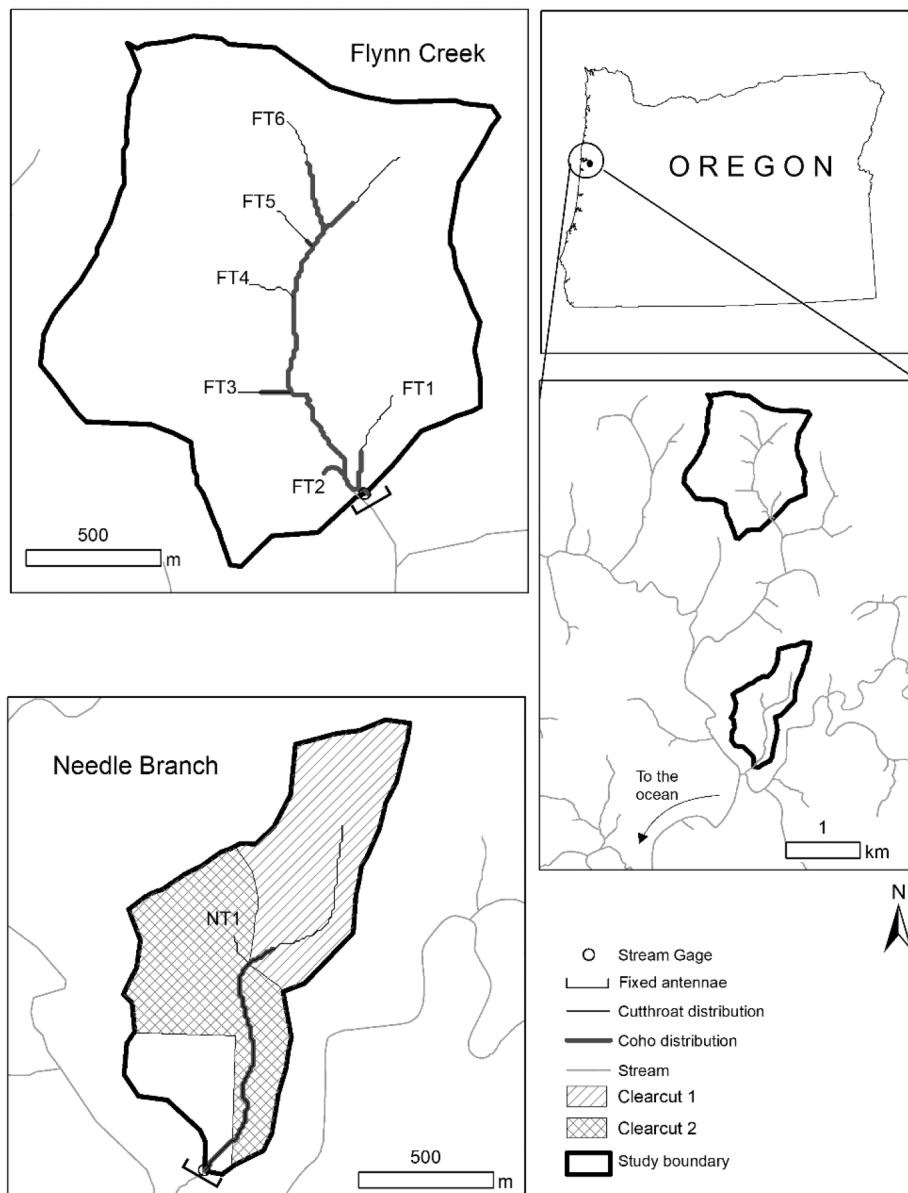
### 2.1. Study sites

The original and revisited Alsea Watershed Studies have focused primarily on two headwater catchments, Needle Branch and Flynn Creek, which are small tributaries (catchment area = 85 and 212 ha, respectively) of Drift Creek in the Alsea River watershed of western Oregon (Fig. 1). Needle Branch was designated as the clearcut-harvest treatment catchment in both the original and current Alsea Watershed studies, and Flynn Creek was the reference. Flynn Creek is steeper with a stream gradient of 7.3%, compared to 5.7% in Needle Branch. The mean wetted width in the fish bearing portion of the stream is 1.3 m in Flynn Creek and 1.1 m Needle Branch. The length of fish bearing channel is 4,276 m in Flynn Creek and 2,078 m in Needle Branch (see Table 1 in Bateman et al., 2018).

Although the Alsea Watershed is relatively low elevation (140–590 m; Hall and Stednick, 2008), hillslopes are steep and prone to landslides (May and Gresswell, 2004). Mild, wet winters and dry summers reflect the maritime climate—mean annual precipitation ( $\approx 250$  cm) is primarily rain that is concentrated between October and March (Hall and Stednick, 2008). Douglas- fir (*Pseudotsuga menziesii*) dominates the forests of Needle Branch and Flynn Creek; red alder (*Alnus rubra*) is common in riparian areas (Moring and Lantz, 1975). Understory vegetation primarily consists of salmonberry (*Rubus spectabilis*), skunk cabbage (*Lysichiton americanum*), sword fern (*Polystichum munitum*), and vine maple (*Acer circinatum*) (Moring and Lantz, 1975). The catchments are underlain by sandstone of the Tyee formation (Corliss and Dyrness, 1965).

The fish assemblage includes coastal cutthroat trout, coho salmon (*Oncorhynchus kisutch*), reticulate sculpin (*Cottus perplexus*), western brook lamprey (*Lampetra richardsoni*), and Pacific lamprey (*Entosphenus tridentatus*) (Hall and Stednick, 2008). Although steelhead trout (*Oncorhynchus mykiss irideus*) are found in the study catchments, they are uncommon. Coastal giant salamander (*Dicamptodon tenebrosus*) and the coastal tailed frog (*Ascaphus truei*) are additional members of the vertebrate assemblage in the study area (Bateman et al., 2018).

During the original Alsea Watershed study, harvest practices typical of the period (1966) were used to clearcut 82% of the Needle Branch catchment. Stream adjacent harvest did not include riparian buffers and felled logs were yarded through the stream. Following harvest, logging slash and naturally occurring woody debris were removed from the stream channel (i.e., stream cleaning; Hall and Stednick, 2008), and the watershed was subsequently burned. Flynn Creek, the unharvested reference catchment during the Alsea Watershed Study, has been



**Fig. 1.** Maps showing the location of study catchments, Flynn Creek and Needle Branch, in the state of Oregon, their relative proximity to each other and drainage to Drift Creek and the Alsea River, as well as the harvest area and locations of stationary antennas in each catchment. Detailed maps also indicate the distributional extent of salmonid fishes (coastal cutthroat trout [*Oncorhynchus clarkii clarkii*] and coho salmon [*Oncorhynchus kisutch*]). Alphanumeric labels (e.g., NT1 and FT1) indicate fish-bearing tributaries, which were numbered sequentially from downstream to upstream.

managed as a research natural area by the U.S. Forest Service since 1977.

Trees harvested in Needle Branch during the current study have grown since the 1966 harvest. To comply with the current forest practice rules, two separate clearcut harvest events were required, which resulted in a harvest area that was the same as the original Alsea Watershed Study. The Phase I clearcut harvest of second-growth forest in Needle Branch occurred June–September 2009 (Bateman et al., 2018). The second clearcut harvest unit (Phase II, 2014) occurred as soon as permitted under current forest practice rules (Oregon Department of Forestry, 2006).

There were three distinct data collection periods: (1) prelogging, July 2006 through August 2009, (2) Phase I postlogging, 2010 through August 2014, and (3) Phase II postlogging, 2015 through August 2017. The Phase I clearcut in Needle Branch was 34 ha, accounting for 40% of catchment area. The Phase II clearcut was approximately 40 ha and covered 47% of the Needle Branch catchment area. The balance of the Needle Branch catchment area (11 ha or 13%) was in non-industrial forestland and was not harvested (Fig. 1).

## 2.2. Study design and harvest treatment

Contemporary logging practices (as defined by Oregon Forest Practice Rules; Oregon Department of Forestry, 2006) were used during both Phase I and Phase II clearcut harvests. In fish-bearing channels of Needle Branch, an unharvested riparian buffer (15 m minimum width) was retained along both streambanks, but no standing trees were retained adjacent to the channel in the upstream portion of the catchment where fish were not present (Oregon Department of Forestry, 2006). Cable-yarding with a tower was used to transport logs away from the streamside buffers to landings. At least one end of yarded logs was suspended to minimize soil disturbance. A tracked log loader was used on some small relatively flat portions of upslope area that were not cable logged. Landings were located adjacent to ridgetop roads. All wood (existing pre-harvest and resulting from harvest operations) within the active channel and extending into the surrounding buffer strips was not removed in accordance with the current regulations. Logging slash was piled and burned near landings, and approximately 24 ha (28% of the catchment) was broadcast burned following the Phase I clearcut harvest. Each harvest unit was replanted, primarily with Douglas-fir seedlings,

**Table 1**

Treatment effects and trend for response variables between treatment periods in the treated catchment relative (Needle Branch) to the reference catchment (Flynn Creek). For total pool area, pool maximum depth, the three movement categories (upstream only, downstream only, and complex), and total annual relative biomass, the initial *F*-ratio and *p*-value are from a one-way analysis of variance. Subsequent *p*-values are from the associated multiple comparison test of individual variables among the treatment periods. The 7-day moving mean of daily maximum water temperature (7-day moving max) was analyzed statistically using the restricted maximum likelihood estimation (REML) in a random-intercept, linear mixed-effects model (Bladon, et al. 2016), and each comparison between treatment periods generated an *F*-ratio and associated *p*-value. Effect size for all response variables was estimated using Hedges' *g* (Hedges 1981), as modified by Hedges and Olkin (1981). Effect size classes (small: 0.2–0.5, medium: 0.5–0.8, and large: >0.8) refer to Cohen (1988).

Variable	<i>F</i> -ratio	<i>p</i> value	Prelogging vs Phase I			Prelogging vs Phase II			Phase I vs Phase II			Mean difference		
			<i>p</i> value	Relative trend	Effect size (g)	<i>p</i> value	Relative trend	Effect size (g)	<i>p</i> value	Relative trend	Effect size (g)	Pre-logging	Phase I	Phase II
Total pool area	8.43*	0.02	0.07	>	Large (1.15)	0.02	>	Large (1.49)	0.26	>	Large (1.42)	-429	-146	33
Pool maximum depth	1.81	0.22	NA	>	Large (0.96)	NA	>	Medium (0.79)	NA	=	Small (0.17)	-0.01	0.01	0.01
7-day moving max	<0.38	NA	0.97	>	Large (1.89)	0.72	>	Large (2.23)	0.88	>	Medium (0.52)	-0.81	-0.15	<-0.00
Movement														
Upstream only	1.79	0.23	NA	=	Small (0.36)	NA	>	Large (1.06)	NA	>	Large (0.86)	-0.01	-0.01	0.01
Downstream only	2.45	0.15	NA	>	Large (1.22)	NA	>	Small (0.16)	NA	<	Large (1.11)	0.04	0.12	0.05
Complex	2.14	0.18	NA	<	Large (0.87)	NA	<	Large (1.57)	NA	<	Medium (0.50)	-0.03	-0.15	-0.23
Total relative biomass	9.82	0.01	0	>	Large (2.62)	0.07	>	Large (1.67)	0.09	<	Large (1.01)	-2898	377	-1165

\* This value is the test statistic *H* for a Kruskal-Wallis one-way ANOVA on ranks.

but western hemlock (*Tsuga heterophylla*) and western red cedar (*Thuja plicata*) were also included in some riparian areas. Aerial and backpack application of herbicides (glyphosate, imazapyr, sulfometuron methyl, and metsulfuron methyl) were used to suppress forbs, shrubs, and commercially undesirable tree species after both Phase I and Phase II clearcut harvest events and prior to replanting. A spray buffer of 18 m (horizontal distance) was maintained along fish-bearing channels, and in the fishless portion of the catchment the minimum spray buffer was 3 m (Louch et al., 2016).

Site access used an existing road network that primarily followed ridgetops, and roads did not cross fish-bearing channels. Limited new road construction was necessary, except for a few short, rocked spurs (for landings), and crushed rock that was added to portions of the native-surfaced road system to facilitate wet-weather access. All road work was accomplished in accordance with current forest practice rules and, except for maintenance, was completed before logging was initiated.

### 3. Data collection

Data collection methods developed during Phase II of the Alsea Watershed Study, remained unchanged from Phase I (Bateman et al., 2018) to facilitate direct comparisons across the study time periods. However, to enable reader comprehension, we have summarized our methods below.

#### 3.1. Stream habitat

Habitat in the fish-bearing channels in Needle Branch and Flynn Creek was surveyed annually (2006–2017) in late July to early August. Prior to the initial surveys, barriers to upstream fish movement and junctions with major fish-bearing tributaries were used to divide both catchments into stream segments (Moore et al., 1997). Individual channel-unit types (pool, riffle-rapid, cascade, and vertical step) were classified (Bisson et al., 1982) in each stream segment. A maximum depth of  $\geq 15$  cm was required for units to be classified as pools. Streambeds without flowing water were defined as dry channels. Visual estimates of channel-unit length, wetted width, and active-channel width were recorded for each channel unit. Additionally, channel-unit length, wetted width, and active-channel width were measured with a tape (m) at every tenth habitat unit, and correction factors were

developed for each surveyor by comparing results to the visual estimates (Hankin and Reeves, 1988). Maximum depth of all habitat units was measured with a meter stick.

Stream temperatures were quantified (30 min intervals) at the outlet of each of the study catchments from June through September using Onset Tidbit water temperature data loggers (UTBI-001, Onset Corporation, Bourne, MA). Data loggers were shielded from direct solar radiation by placing them in rock cairns with the case ends open parallel to stream flow to ensure thorough mixing.

#### 3.2. Fish capture

Each year fish sampling followed the annual habitat survey in early to mid-August. A single-pass electrofishing census of pool and cascade habitats was used to estimate relative abundance for age-1 + coastal cutthroat trout and juvenile coho salmon throughout the fish-bearing portion of each catchment (Bateman et al., 2005). Age-0 cutthroat trout were excluded because obtaining precise estimates of abundance for this age group is often problematic (Thompson and Rahel, 1996; Peterson et al., 2004) and larger fish are susceptible to injury at voltages necessary to effectively capture trout and charr < 80 mm (Hollender and Carline, 1994; Dalbey et al., 1996; Thompson et al., 1997). 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, we assumed single-pass electrofishing would provide an unbiased measure of relative salmonid abundance and relative biomass. To ensure consistent effort and capture efficiency, the same individuals functioned as electrofishing crew leaders throughout the study.

All pools and cascades were sampled by single-pass electrofishing because these channel units represented primary feeding and survival habitats for age-1 + coastal cutthroat trout in headwater streams and were the habitat units most commonly occupied during low flow periods when sampling occurred (Bateman et al., 2016). We began electrofishing at the downstream end of the main stem or tributary channel and continued upstream until all pool habitat units in the fish-bearing channel were sampled. Markers were attached to riparian trees or shrubs approximately every 15 m (measured along the thalweg) to act as spatial reference (distance upstream from gauging station) for each



habitat unit. Captured fish from each habitat unit were anesthetized in a solution of buffered MS-222 (40 mg L<sup>-1</sup>), and each individual was measured (fork length, nearest 1 mm) and weighed (wet weight, nearest 0.1 g). Length-frequency histograms were used to distinguish age-0 and age-1 + coastal cutthroat trout; individuals  $\geq 80$  mm were considered age-1 + . Subsequently, a 23 mm passive integrated transponder tag (PIT; Texas Instruments, Inc., Dallas, Texas) with a unique identification number was surgically implanted in all coastal cutthroat trout  $\geq 100$  mm (Bateman and Gresswell, 2006). Beginning in 2010, a syringe was used to inject a 12 mm PIT tag (Prentice et al., 1990) into coastal cutthroat trout  $\geq 65$  mm and  $< 100$  mm fork-length. The 23 mm and 12 mm half-duplex glass encapsulated tags (Texas Instruments, Inc., Dallas, Texas) weighed 0.6 and 0.1 g (air), respectively. Following recovery (defined by upright swimming) in an aerated bucket of stream water, fish were released into the habitat unit where they were initially captured.

### 3.3. PIT-tagged fish detection

Tagged fish were relocated during the study with stationary and portable antennas. Beginning in 2006, two stationary swim-through PIT-tag antennas (Babin-Zydlowski et al., 2001) were installed upstream of the weirs at the downstream boundary of each study catchment. The two antennas were connected to a multiplexor and data were processed through a single reader, which enabled us to determine the direction of travel of tagged fish (Fig. 1). Antennas operated continuously throughout the year, and fish detection data were uploaded every two weeks. Concomitantly, antennas were adjusted to maintain a minimum read range of 25 cm in any direction, a distance at which virtually all the test PIT-tags could be detected when floating through antennas.

Beginning in October 2006, portable PIT-tag antennas (Babin-Zydlowski et al., 2001) provided spatially explicit locations of tagged coastal cutthroat trout. Surveys with the portable antennas usually occurred in October, December, February, April, and June, but exact timing varied depending on the occurrence of extreme high-water events that precluded instream work. Because greater discharge during these sampling periods facilitated movement of age-1 + cutthroat trout and obscured the distinction among habitat units, it was necessary to sample the entire wetted area of the fish-bearing portion of each study stream network during the portable-antenna surveys.

Portable PIT-tag antennas have a maximum detection range of  $< 1$  m; therefore, two portable antennas were used in the main stem of Flynn Creek from the weir up to the junction with tributary 6 (Fig. 1). The number of portable antennas required for the main stem of Needle Branch varied with discharge (i.e., two antennas were used when discharge was high; only one was necessary during periods of low discharge). Additional information concerning PIT-tag-detection methods and determining fish locations was documented by Bateman et al. (2016 and 2018).

## 4. Data analysis

### 4.1. Stream habitat

Stream habitat response variables were total pool area (estimated during low-flow), mean maximum depth of pools (measured during low-flow), and water temperature (7-day moving mean of the daily maximum water temperature, hereafter 7-day moving max). Pool area was the sum of the surface area of all pools within the fish bearing portion of each catchment. Similarly, the mean pool depth metric was the sum of the maximum depth of all pools divided by the number of pools within the fish bearing portion of each catchment. Both pool area and mean pool depth were summarized and compared by year (see 4.3. *Before-After-Control-Impact Analysis*). Annual cumulative pool area (m<sup>2</sup>) and cumulative maximum pool depth (m) were plotted by distance from the downstream terminus of the main-stem channels of each catchment

to visually assess the spatial distribution of these variables through time. To detect changes in water temperature related to clearcut harvests in Needle Branch relative to Flynn Creel, data from data loggers were summarized as the 7-day moving max water temperature and compared for each catchment during the prelogging period and Phase I and Phase II postlogging periods (see 4.3. *Before-After-Control-Impact Analysis*).

### 4.2. Coastal cutthroat trout Movement, Distribution, and biomass

Movement, distribution, and relative biomass were assessed annually for age-1 + coastal cutthroat trout. Fish movement was defined as any relocation  $\geq 50$  m from the last known point of detection within the same channel (i.e., main stem or tributary) or a relocation in a different channel regardless of distance. Fish movements were subsequently categorized as: (a) upstream movement only; (b) downstream movement only; (c) both upstream and downstream movement within a main stem or tributary; or (d) complex movement (if fish moved between main stems and tributaries or between catchments; Young, 2011; Bateman et al., 2016). The proportion of tagged fish displaying one of the four movement types was calculated for each year for each catchment. The difference in the proportion of fish displaying a movement type was calculated annually by subtracting the proportion of fish exhibiting that movement in the reference catchment (Flynn Creek) from the proportion of fish exhibiting the same movement type in the harvested catchment (Needle Branch). The annual differences were then compared by treatment period using a one-way analysis of variance.

Distribution was assessed using cumulative relative biomass (total weight, g) of age-1 + coastal cutthroat trout collected from pool and cascade habitat units in the main-stem channel of each catchment by year. Cumulative relative biomass for each year was plotted by distance from the downstream terminus of main-stem channels of each catchment to upper most sample unit (i.e., pool or cascade), immediately downstream from the point where fish were no longer present in the main-stem.

### 4.3. Before-After-Control-Impact analysis

Stream habitat and fish population responses to contemporary logging in the Alsea watershed were evaluated using a Before-After-Control-Impact (BACI) study design (Stewart-Oaten et al., 1986; Stewart-Oaten and Bence 2001). The influence of logging on each response variable (except water temperature and age-1 + coastal cutthroat trout survival, site fidelity, and distribution) was analyzed by initially calculating the difference between variable means for the harvested catchment (Needle Branch) and the reference catchment (Flynn Creek) by year (i.e., harvested minus reference; 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 and Bence, 2001). For each response variable, the differences were also evaluated for normality and constant variance. If the assumptions of normality and constant variance were met, a one-way analysis of variance was used to assess statistically significant differences among means. If the assumptions of normality or equal variance were not met, we used a Kruskal-Wallis one-way analysis of variance on ranks. If the *F*-test suggested significant differences in treatment means ( $p \leq 0.05$ ), a Tukey-Kramer multiple comparison procedure was used to evaluate a set of planned comparisons (i.e., prelogging to Phase I, prelogging to Phase II, and Phase I to Phase II). This approach is common in large-scale field manipulations where spatial replication is not feasible (Taylor et al., 2006; Greenwood et al., 2007; Tiegs et al., 2011). All statistical analyses except those related to survival and site fidelity were performed with NCSS software (Hintz, 2007).

Analysis of 7-day moving max was focused on assessing whether it changed due to time (pre-harvest vs. post-harvest Phase I vs. post-

harvest Phase II), treatment (reference catchment vs. harvested catchment), and the interaction between time and treatment. Data were analyzed statistically using the restricted maximum likelihood estimation (REML) in a random-intercept, linear mixed-effects model with the nlme package (Pinheiro et al., 2021) in R (R Core Team, 2021) following the approach of Bladon et al. (2016).

#### 4.4. Survival analysis

A multistate capture-recapture model (Arnason, 1972; Hestbeck et al., 1991; Brownie et al., 1993) was used to estimate survival (S) and site fidelity (F) to study catchments of three length-groups of coastal cutthroat trout: 65–93 mm, 94–110 mm, and > 110 mm. Transitions between multiple states applied to length group transitions and also fidelity to, or emigration from study catchments. We used a Bayesian state-space or full data likelihood modeling approach in this effort (Schofield et al., 2009; King, 2012). We multiplied interval-specific survival and site fidelity to obtain estimates of annual rates.

The sampling scenario and capture-recapture model that we employed largely followed protocols of Horton et al. (2011), where fish were observed to be alive during discrete-time sampling occasions within the study areas using portable antennas or electrofishing, and stationary PIT-tag detection antenna systems recorded marked individuals that emigrated the study areas between discrete-time sampling occasions, or moved near enough to the boundary of the area to be detected by a stationary antenna. In our application, portable antennas were used for relocating coastal cutthroat trout within study catchments in all but an annual electrofishing event that occurred in August. Therefore, no modification to the structure of the multistate model was necessary except to represent capture or detection probability, in the case of portable antenna surveys, with two different parameters. We limited the portable-antenna relocation data used in the survival analysis to those detections categorized as ‘live trout’ (see Section 3.3. PIT-tagged fish detection).

Our capture-recapture model differed from that of Horton et al. (2011) in two respects. First, Horton et al. (2011) estimated stationary-antenna detection probabilities by using multiple antennas at the boundary of the study area and strict assumptions about fish movement near the antennas. Instead, we assumed 100% detection probability at the study area boundaries by each pair of stationary antennas. Fish that were detected at a study catchment boundary (i.e., emigrated) were censored from the data and did not subsequently contribute to parameter estimation unless they were later detected inside the study area (i.e., reentrant), either by electrofishing or mobile antenna tracking. Once detected, reentrants were characterized as present and alive, and they once again became part of the group of individuals considered for survival.

The second departure we made from Horton et al. (2011) was to correct survival estimates for emigration occurring at known times between the capture-recapture or portable-antenna surveys. To avoid bias in estimates of survival and site fidelity, detections at stationary antennas, which typically occurred between electrofishing and portable antenna surveys (i.e. did not represent a full survival interval), were treated differently from detections by electrofishing or portable antenna surveys (Barker, 1997). Because stationary antennas recorded exact emigration time, we considered individuals as having survived only up to the time of the observed emigration with probability  $S^t(1-F)$ , where  $t$  is the fraction of the survival interval when the emigration was detected, at which time the fish was censored (Joe and Pollock, 2002).

Transitions among the three length groups were modeled using a state-space approach (Schofield et al., 2009; King, 2012). However, because fish were not measured during the mobile tracking occasions, when they were not handled, length transitions in the model were restricted to annual intervals, immediately prior to the electrofishing sampling in August. Survival and emigration throughout the year were therefore functions of length as measured or assigned (for individuals

not captured) during the most recent electrofishing sampling occasion. Length group transition probabilities representing negative growth were set to zero in the model.

Modeling fidelity to study catchments facilitated the incorporation of large numbers of fish detections at the stationary antennas. Unbiased estimates of survival from these data required accounting for fidelity to the study catchments (Horton et al. 2011). We distinguish fidelity to the study catchments from more detailed trout ‘movement’ that occurred within the study catchments; these more specific spatial data on fish locations were obtained during portable antenna surveys.

#### 4.5. Effect size

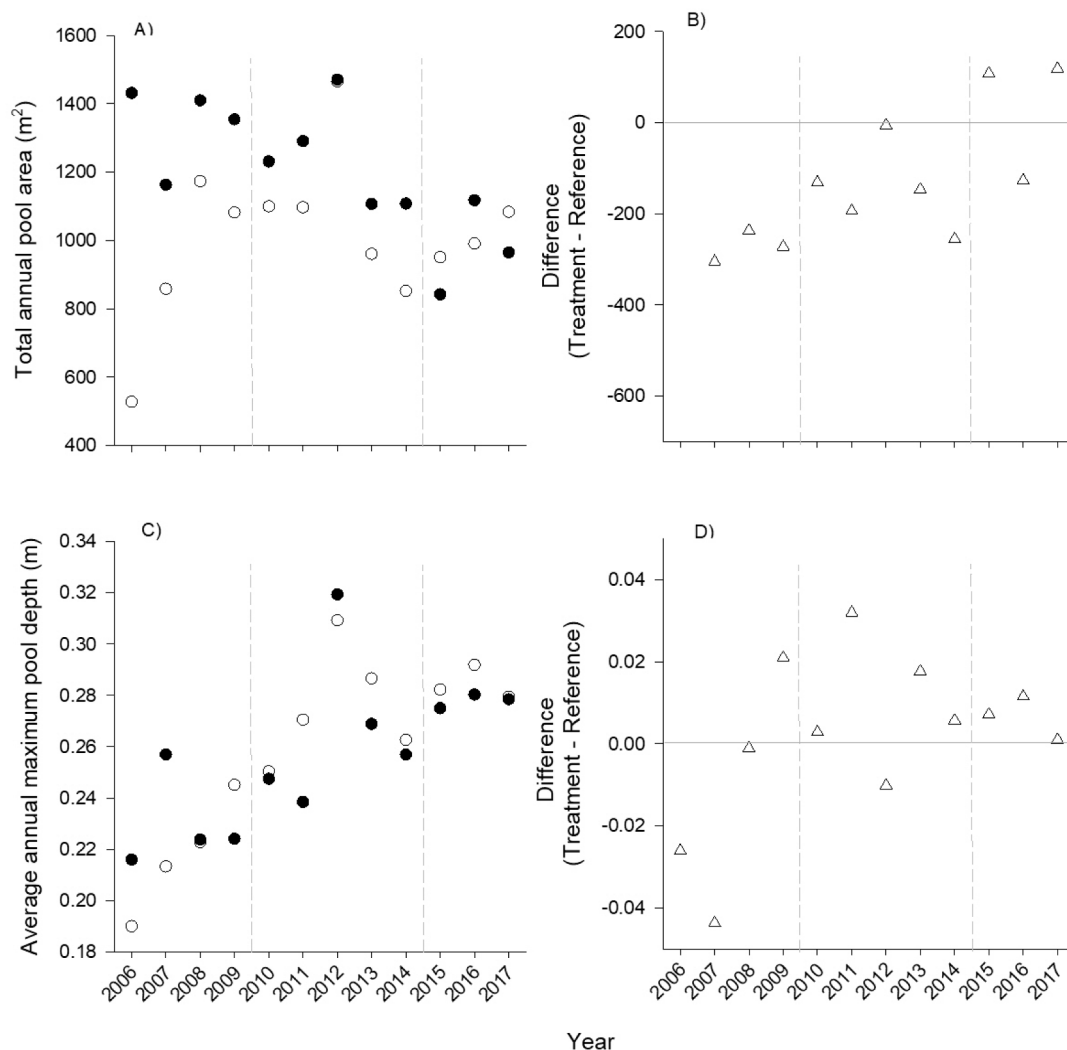
Stewart-Oaten (1996) has argued that hypothesis testing is not adequate for decision making related to intervention assessments because it only provides a limited amount of information. Furthermore, statistical analyses used with BACI study designs are often perturbed by small sample sizes and high variability, and the power to detect statistically significant change, especially at the conventional level of Type I error rate ( $\alpha = 0.05$ ), is limited (Mapstone 1996). In contrast, estimates of effect size (differences in means of two groups of observations in relation to their pooled standard deviation; Ellis 2010) can facilitate the understanding of practical significance of study results. Therefore, we estimated effect size for all response variables (except survival) using Hedges’  $g$  (Hedges 1981), as modified by Hedges and Olkin (1981) to remove small positive bias associated with calculation of Cohen’s  $d_s$  (Cohen 1988) when sample size is low. We used effect size classes (small: 0.2–0.5, medium: 0.5–0.8, and large: >0.8) developed by Cohen (1988) to aid in interpretation of these statistical analyses.

## 5. Results

### 5.1. Stream habitat

Annual total pool area during the low-flow period in Needle Branch was exceeded by the total pool area in Flynn Creek in 9 of the 11 years of the study (Fig. 2, A). Annual total pool area in Flynn Creek gradually declined over the course of the study, but annual total pool area increased in Needle Branch during the first two years of the prelogging period and then remained relatively constant through Phase I and Phase II (Fig. 2, A). The difference in annual total pool area among treatment periods was statistically significant (Kruskal-Wallis one way ANOVA on ranks;  $H = 8.43$ ,  $df = 2$ ,  $p = 0.01$ ). Specific results from the multiple means comparisons suggested that changes in annual total pool area between prelogging and Phase I (Table 1) and Phase I and Phase II (Table 1) were not statistically significant, but differences in annual total pool area (Needle Branch – Flynn Creek) between prelogging and Phase II were statistically significant (Table 1; Fig. 2, B). In contrast to the lack of statistical significance, the effect sizes for the comparisons between prelogging and both Phase I and Phase II, and between Phase I and Phase II were large (Table 1), suggesting substantial changes in mean difference between Needle Branch and Flynn Creek relative to the pooled SD associated with these changes. In fact, the difference in mean annual total pool area decreased almost 66% between prelogging and Phase I, and Phase II, mean total annual pool area in Needle Branch relative to Flynn Creek continued to increase, resulting in a 108% decline in the difference during the study period.

Cumulative pool area was variable among years in Needle Branch, and 2012 (Phase I) exhibited the highest rate, apparent throughout the catchment (Fig. 3, C). Cumulative pool area was also high in 2008, but the difference was most apparent in the upper portion of the catchment. The lowest cumulative pool area occurred throughout the catchment in 2006. In general, cumulative pool area was also variable in Flynn Creek and was greatest in prelogging years and lowest during Phase II; accumulation rates were most variable in the upper portion of the watershed (Fig. 3, D).



**Fig. 2.** Annual total pool area for Needle Branch (treated, open circles) and Flynn Creek (reference, closed circles) (A) and mean maximum pool depth (C) with associated annual difference (Needle Branch – Flynn Creek) in total pool area (B) and mean maximum pool depth (D), Lincoln County, Oregon, USA. The horizontal solid line in B and D denotes zero or no difference between treated and reference catchment values. Logging occurred in Needle Branch in 2009 and 2014 as denoted by vertical dashed grey lines.

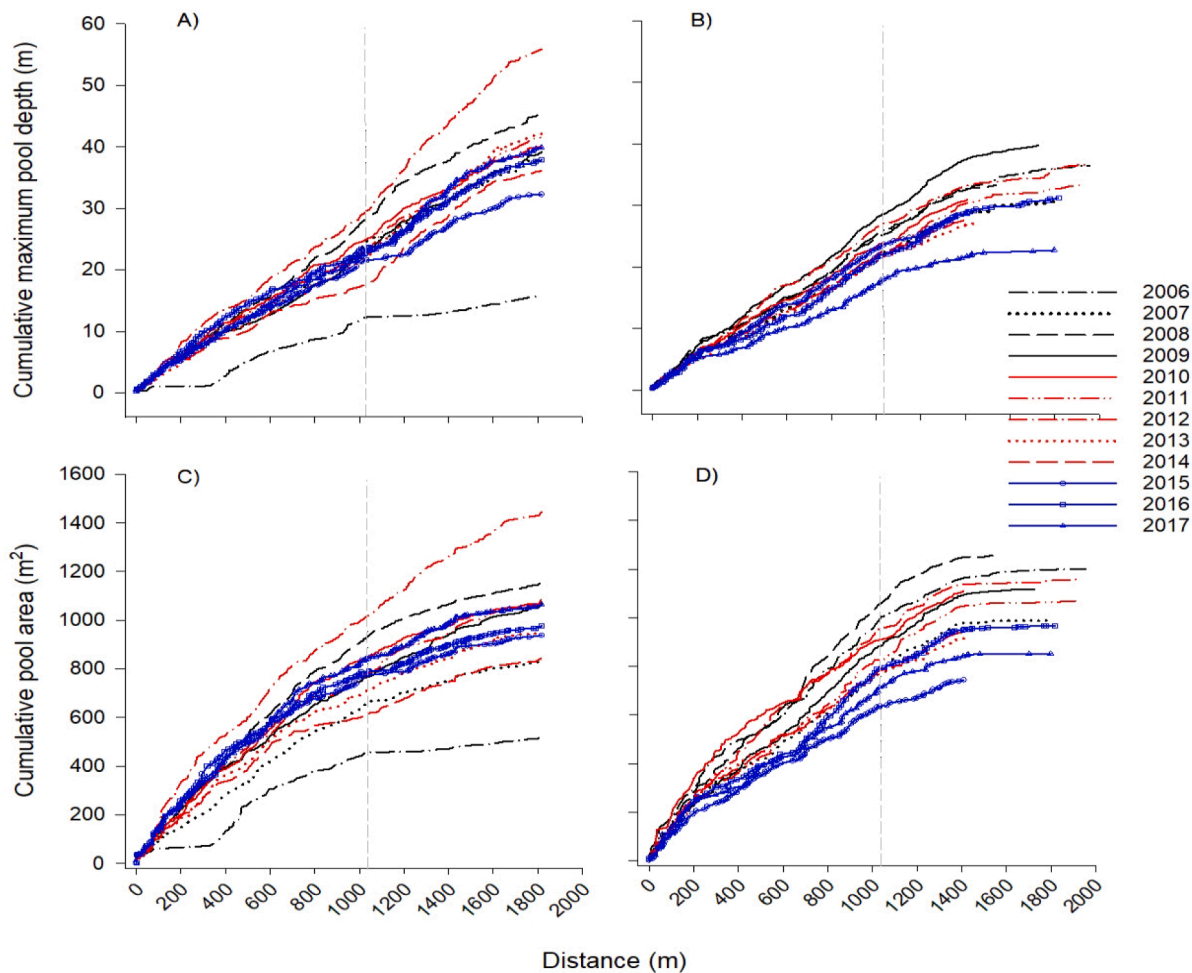
Annual mean pool maximum depth in Needle Branch during low-flow periods exceeded that of Flynn Creek in 8 of the 12 years of the study (Fig. 2, C). Estimates generally increased in both catchments through time (Fig. 2, C), but there was no clear temporal trend in the difference (Needle Branch–Flynn Creek) in annual mean low-flow maximum pool depths (Fig. 2, D). Furthermore, changes among sample periods in mean maximum low-flow pool depths in Needle Branch relative to Flynn Creek were not statistically significant ( $F = 1.53$ ,  $df = 2, 9$ ,  $p = 0.27$ ). The effect size for differences in annual mean pool maximum depth between the two catchments was large between pre-logging and both Phase I and Phase II, but effect size was small between the two postlogging phases (Table 1).

Inspection of cumulative maximum pool depth in Needle Branch suggested a relatively consistent pattern among years with the exception of 2006 (prelogging), which was anomalously low. Additionally, in 2008 (prelogging) and 2012 (Phase I post-harvest) the cumulative maximum pool depth was anomalously greater in the upper portions of the catchment (Fig. 3, A). Cumulative maximum pool depth in Flynn Creek was even less variable among years; however, 2009 exhibited a higher cumulative total, and 2017 was substantially lower (Fig. 3, B). Changes were greatest in the upper portion of the catchment in 2009, but declines in 2017 appeared to be more consistent throughout the catchment.

The 7-day moving max water temperatures at the outlet of Flynn Creek were 13.4 °C in the pre-harvest period, 13.2 °C during Phase I, and 13.4 °C during Phase II. Comparatively, the 7-day moving max water temperatures at the downstream terminus of Needle Branch were 12.6 °C in the pre-harvest period, 13.1 °C during Phase I, and 13.4 °C during Phase II. Statistically, however, there was no evidence that differences in 7-day moving max water temperatures between Needle Branch and Flynn Creek changed after the Phase I or Phase II harvest (Table 1, Fig. 4). Conversely, effect size was large for the difference between Needle Branch and Flynn Creek in moving 7-day moving max water temperature between prelogging and both Phase I and Phase 2 (Table 1); effect size was moderate between Phase I and Phase II (Table 1). Despite the apparent increase in water temperature in Needle Branch relative to Flynn Creek in both postlogging periods, stream temperatures never exceeded the Oregon regulatory standard of a 7-day moving max of 16.0 °C for core cold water habitat (IMST 2004).

## 5.2. Fish movement, biomass, and distribution

For years 2007 to 2017, the proportion of tagged coastal cutthroat trout classified as movers was 28% in Needle Branch and 33% in Flynn Creek. The smallest proportion of tagged individuals classified as movers



**Fig. 3.** Cumulative maximum pool depth for main-stem channels of Needle Branch (treatment) (A) and Flynn Creek (reference) (B) and cumulative pool area for main stems of Needle Branch (C) and Flynn Creek (D) by year. Black lines without symbols are prelogging, the red lines are Phase I (postlogging of upper 40% of Needle Branch), and blue lines with symbols are Phase II (postlogging lower 47% of Needle Branch). Dashed vertical grey line indicates the downstream boundary of the Phase I clearcut and the upstream boundary of the Phase II clearcut in Needle Branch and an analogous location in Flynn Creek (see Bateman et al., 2018). The x-axis displays the distance upstream from the weirs located at the downstream terminus of each catchment. In most years neither pools nor cascades were present over the entire distance to the upstream extent of fish in Flynn Creek. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

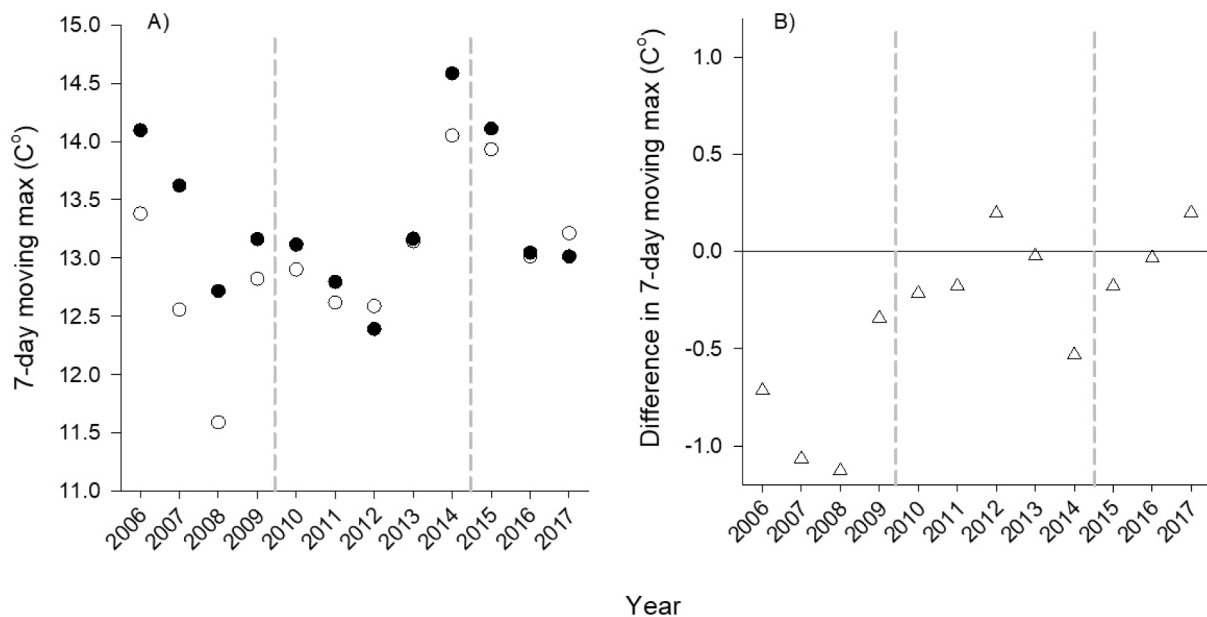
occurred in 2017 in both catchments, when only 12% of tagged coastal cutthroat trout “moved” in Needle Branch and 17% “moved” in Flynn Creek. The maximum proportion of tagged fish classified as movers was 44% in Needle Branch and occurred in 2011, and the maximum proportion in Flynn Creek was 46% and occurred in 2013. There was no evidence of statistically significant changes in the differences in the proportion PIT-tagged coastal cutthroat trout exhibiting any of the four movement strategies (upstream only, downstream only, both upstream and downstream, and complex) among treatment periods ( $F < 2.45$ ,  $df = 2, 8$ ,  $p > 0.15$ ).

The relative proportion of tagged coastal cutthroat trout moving both upstream and downstream was very consistent among prelogging and the two postlogging periods; however, patterns in the other movement types were apparent (Fig. 5). For example, although the proportion of individuals displaying a complex movement pattern was consistent in Needle Branch (mean = 0.5%) throughout the study period (Fig. 5, A), the proportion of individuals from Flynn Creek exhibiting complex movement increased substantially during Phase I and remained high during Phase II. A large effect size was observed in the differences in the proportion of complex movement exhibited in Needle Branch and Flynn Creek between prelogging and Phase I and Phase II (Table 1, Fig. 5, B), but these changes occurred only in Flynn Creek, and therefore, are

probably not related to logging in Needle Branch. Although upstream movement was highly variable in Needle Branch during all periods (Fig. 5, C), no distinct trend was apparent through time. Furthermore, decreases in upstream movement of coastal cutthroat trout in Flynn Creek during Phase II resulted in large effect sizes in the differences between the two catchments from prelogging to Phase II, and between the two postlogging periods (Table 1; Fig. 5, D). The proportion of tagged coastal cutthroat trout moving downstream in Needle Branch increased during the prelogging period and the first two years of Phase I but then declined during remainder of Phase I and stayed low during Phase II (Fig. 5, E). Resulting effect sizes in the difference between the two catchments in the proportion of coastal cutthroat trout exhibiting downstream movement were large between prelogging and Phase I and between Phase I and Phase II, but small between prelogging and Phase II (Table 1; Fig. 5, F).

Differences among treatment periods in total relative biomass of age-1 + coastal cutthroat trout in Needle Branch relative to Flynn Creek were statistically significant ( $F = 9.8$ ,  $df = 2, 9$ ,  $p = 0.01$ ). Specific results from the multiple means comparisons indicated that differences in relative biomass between prelogging and Phase I ( $p = 0.02$ ) were statistically significant. Results for comparisons between prelogging and Phase II ( $p = 0.07$ ), and postlogging Phase I and Phase II ( $p = 0.09$ ) were





**Fig. 4.** The 7-day moving mean of the daily maximum water temperatures (7-day moving max) in Needle Branch (harvested, open circles) and Flynn Creek (reference, closed circles) (A) and the difference in 7-day moving max water temperature (Needle Branch – Flynn Creek) (B), Lincoln County, Oregon, USA. The horizontal line in B denotes zero or no difference between the harvested and reference catchment values. Logging occurred in Needle Branch in 2009 and 2014, which is denoted by vertical grey dashed lines.

more ambiguous; however, effect sizes were large for comparisons between prelogging and both Phase I and Phase II, and between the two postlogging periods (Table 1; Fig. 6). The decline in relative biomass of age-1 + coastal cutthroat trout in Needle Branch from Phase I to Phase II was greatest in the upper (higher elevation) portion of the catchment in the area adjacent to the initial clearcut harvest unit (Fig. 7).

Spatial patterns (downstream terminus of study to upper most sample unit) of annual cumulative relative biomass curves for age-1 + coastal cutthroat trout in the main stems of Needle Branch and Flynn Creek suggest that the increase in relative biomass that occurred during Phase I relative to prelogging or Phase II was strongly influenced by individuals captured in upper Needle Branch (Fig. 7, A). Additionally, the decline in biomass from Phase I to Phase II, also was affected by declines in biomass in upper Needle Branch (Fig. 7, A). The slopes of the lines (accumulation rate) for the portion of Needle Branch that was clearcut harvested during Phase I (just over 1 km upstream of the catchment terminus) were steeper (four of the five years) than in the prelogging period or Phase II (Fig. 7, A). This pattern was not observed in Flynn Creek (Fig. 7, B).

### 5.3. Survival

Survival of PIT-tagged coastal cutthroat trout varied among years, time periods, and length groups (Fig. 8). Survival of coastal cutthroat trout > 110 mm and those 94–110 mm was similar between Needle Branch and Flynn Creek during the prelogging period. For the coastal cutthroat trout > 110 mm, annual survival rate in Needle Branch equaled or exceeded the survival rate in Flynn Creek in every year during the Phase I period, but survival of this length group during Phase II was lower in Needle Branch than Flynn Creek in two of the three years (Fig. 8, B). Although patterns of survival for the 94–110 mm coastal cutthroat trout was similar to those > 110 mm, survival in Needle Branch only exceeded Flynn Creek in three of the five years of Phase I. Annual survival rates of the 94–110 mm length group in Needle Branch were lower than those of Flynn Creek fish in two of the three years during Phase II (Fig. 8, D). Survival of individuals in the 65–93 mm length group could only be evaluated for the two postlogging periods, and despite some variation, survival rates were similar between these

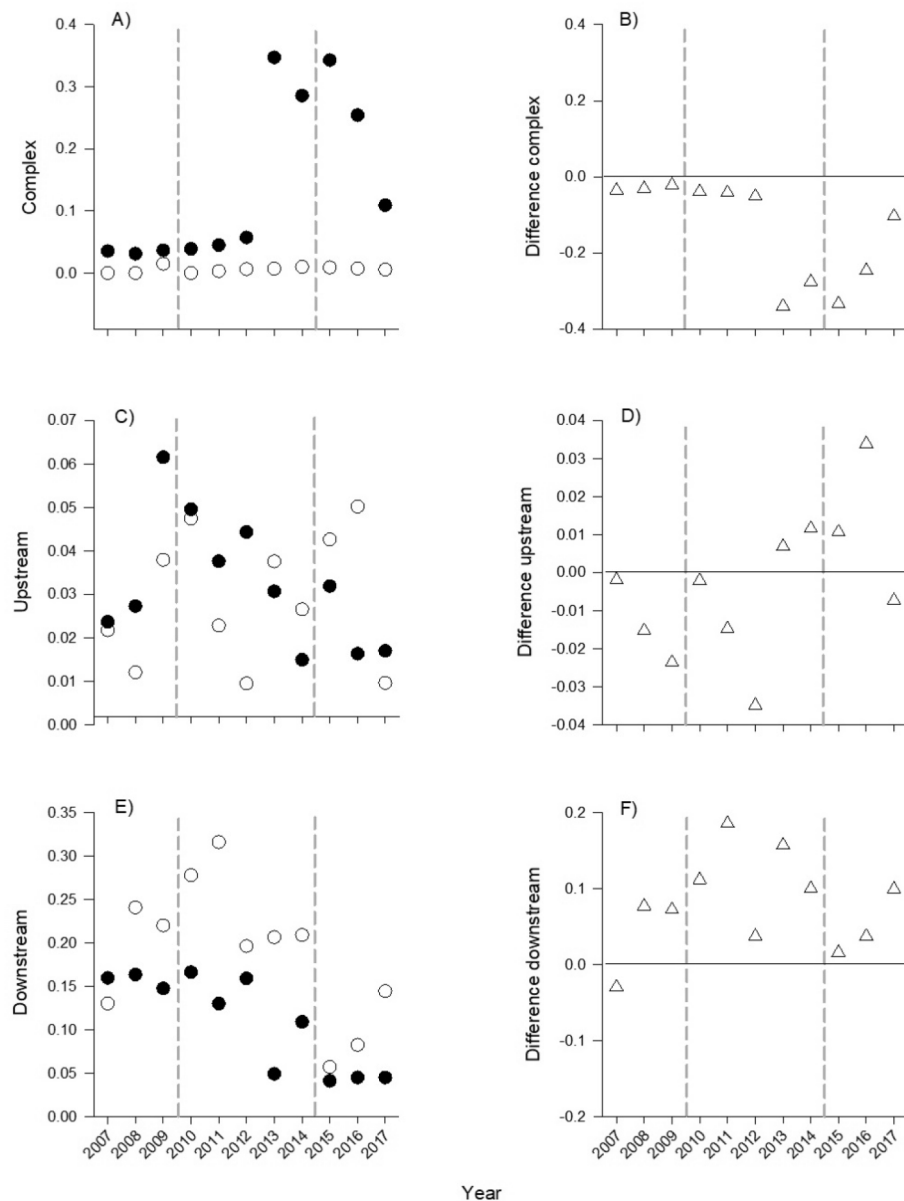
periods (Fig. 8, E and F); effect size of differences between the Phase I and Phase II was low (Table 1). Generally, annual survival rates in Needle Branch increased in the two larger length groups of coastal cutthroat trout during Phase I and then declined during Phase II (Fig. 8).

## 6. Discussion

In this study, we employed a BACI study design to assess the effects timber harvest in a second-growth forest, conducted with contemporary forest management techniques, on coastal cutthroat trout. In many respects, our study complements our initial research of a second-growth timber harvest in the Alsea watershed (Bateman et al. 2018), but uniquely, we examined the potential effects of consecutive clearcut harvest units in this regenerated forest on the response of fish populations in the headwater streams. Another unique aspect of our study was the consideration of survival, movement, distribution and biomass of coastal cutthroat trout as response variables to provide a more complete assessment of the effects of timber harvest treatments (Bateman et al., 2018).

Generally, survival for the two larger length groups of coastal cutthroat trout increased in Needle Branch during Phase I and then declined during Phase II to levels similar to the prelogging period. This response was strongest in the > 110 mm length group. There was considerable variability among years and treatments in all length groups, and overall there appears to be little evidence of a negative effect of logging on survival of coastal cutthroat trout in Needle Branch. Patterns in annual estimates of relative biomass of age 1 + coastal cutthroat trout followed a pattern similar to that of survival of the two larger length groups with biomass increasing in Needle Branch relative to Flynn Creek during Phase I then declining in Phase II. Although survival of the coastal cutthroat trout in smallest length group (65–93 mm) was somewhat lower than individuals in the other length groups, interannual patterns of survival were actually quite similar to the larger length groups in Phase I and Phase II (Fig. 8).

In addition to survival and biomass, we were able to account for the potential influences of individual movement on observed responses. Movement is a common behavior exhibited by fish when conditions for survival, growth, and reproduction are not favorable (Warren and Liss,



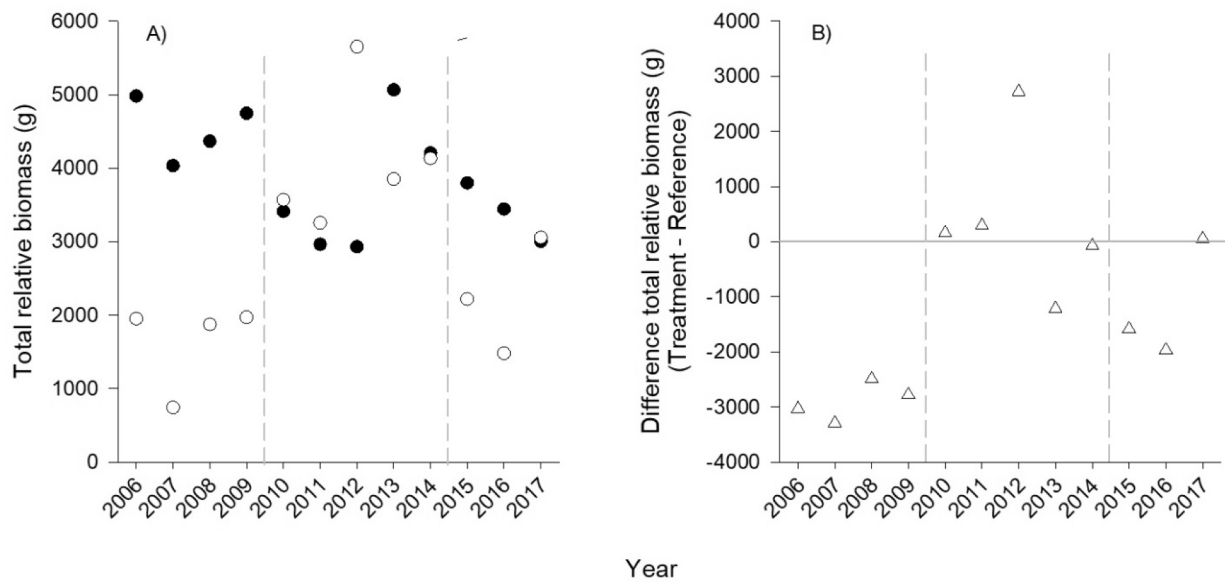
**Fig. 5.** Annual proportion of tagged coastal cutthroat that exhibited complex (A), upstream only (C), and downstream only (E) movement in Needle Branch (treatment, open circles) and Flynn Creek (reference, closed circles), with associated annual differences (Needle Branch – Flynn Creek) in the proportion of tagged coastal cutthroat trout exhibiting complex (B), upstream only (D), and downstream only (F) movements, Lincoln County, Oregon, USA.. The horizontal line in B, D, and F denotes zero or no difference between treated and reference catchment values. Logging occurred in Needle Branch in 2009 and 2014 denoted by vertical grey dashed lines.

1980). In fact, salmonids regularly move within stream networks in response to changes in physical habitat and food resources (Gowan and Fausch, 2002). Direct tracking of coastal cutthroat trout indicates movement patterns of populations can vary seasonally, and that some fish can move extensively (>250 m; Gresswell and Hendricks, 2007). In the current study, a substantial proportion of the coastal cutthroat population was PIT tagged annually and sampling occurred in a spatially continuous manner over the entire study area, but despite some variation among years, movement patterns in Needle Branch did not appear to be related to forest harvest. In fact, differences in movement between the treatment and reference catchments that occurred among prelogging and postlogging periods appeared to be most strongly influenced movement trends in Flynn Creek. Furthermore, none of the coastal cutthroat trout tagged below the lower falls in Needle Branch were ever captured upstream during summer electrofishing surveys (Bateman et al., 2018).

Collectively, these results suggest that the observed changes in relative biomass were not related to a simple redistribution of individuals, but in fact might be related to local changes in habitat and productivity. Fish distribution, as indicated by annual cumulative

relative biomass curves, suggest that although declines in biomass of age-1 + coastal cutthroat trout in Needle Branch relative to Flynn Creek during Phase II occurred throughout Needle Branch, the vast majority of the decline occurred in the upper portion of the catchment (Fig. 6, A; Fig. 7, A), where biomass had increased following logging in that portion of the catchment.

Evidence for an increase in age-1 + coastal cutthroat trout biomass in Needle Branch relative to Flynn Creek was documented by Bateman et al. (2018) for the Phase I harvest. The reported increase was greatest in the fish bearing portion of Needle Branch immediately adjacent to the clearcut harvest unit (Bateman et al., 2018). In the current study, we evaluated the relative biomass with an additional 3 years of data collected during Phase II, where the second clearcut harvest unit was conducted in Needle Branch immediately downstream from the Phase I clearcut harvest unit (Fig. 1). Because Phase II harvest unit was located downstream of the Phase I unit, it is unlikely that the more recent harvest had substantial effects on habitat conditions upstream (adjacent to the Phase I harvest unit) where increases in annual biomass were greatest during Phase I (Fig. 6, panel A to the right of vertical grey dashed line; Fig. 7, A). Data collected over the entire study period



**Fig. 6.** Annual total relative biomass (g) of age-1 + coastal cutthroat trout from censuses of pool and cascade habitats in Needle Branch (treatment, open circles) and Flynn Creek (reference, closed circles) (A) and differences in annual total relative biomass (Needle Branch – Flynn Creek) (B), Lincoln County, Oregon, USA. The horizontal line in B denotes zero or no difference between treated and reference catchment values. Logging occurred in Needle Branch in 2009 and 2014 as denoted by vertical dashed grey lines.

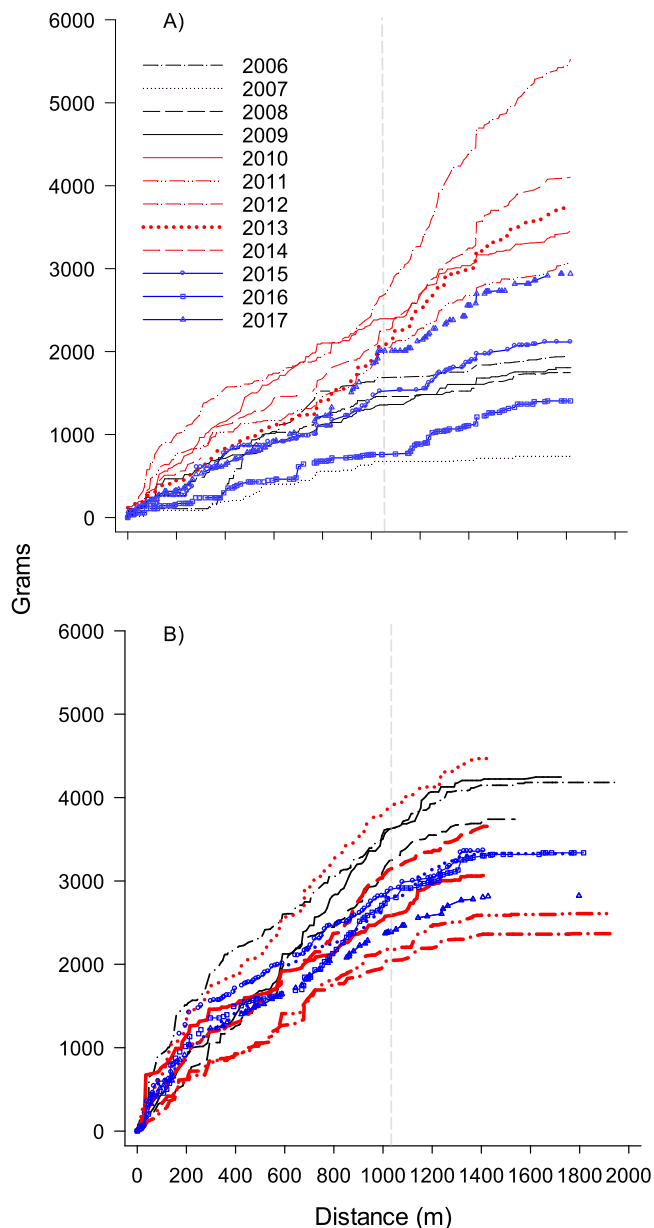
suggested that the increase in biomass reported by Bateman et al., (2018) for the upper portion of Needle Branch had dissipated to levels approaching prelogging levels eight years after the Phase I clearcut harvest (Fig. 6). Additionally, the effect sizes (increases in Phase I and decreases in Phase II) were large (Table 1).

We assessed pool area and maximum depth, two variables known to be generally important for coastal cutthroat trout in small streams (e.g., Railsback et al. 2009), and in our study streams specifically (Penaluna et al., 2021). Cumulative curves of total pool area and maximum depth varied somewhat over the study period (Fig. 3), but general patterns did not track changes in age-1 + coastal cutthroat trout biomass observed in either Needle Branch or Flynn Creek. Total pool area in Needle Branch increased relative to Flynn Creek during the Phase II period, but relative biomass (Fig. 6 and Fig. 7) was declining in Needle Branch during this period. Furthermore, with the exceptions of 2006 and 2012, total pool area was similar among years in Needle Branch but in Flynn Creek total pool area generally declined through time (Fig. 2, A). The lack of concordance between our metrics of relative biomass of age-1 + coastal cutthroat trout and total pool area and mean maximum depth suggest that changes in relative biomass among treatment periods was not strongly related to either variable, but more likely were related to changes in productivity or some other environmental factor.

Curiously, there was little evidence for interactions between factors commonly associated with logging and negative responses of salmonids (Chamberlin et al., 1991) following the harvesting activity in the Alsea Watershed Study. In our study, all wood was left in the stream (Mellina and Hinch, 2009), and although the percentage change in discharge in Needle Branch relative to Flynn Creek was positive for both the Phase I and Phase II compared to the prelogging period (Segura et al., 2020), there was no evidence of increased suspended sediment concentration or yield associated with logging in Needle Branch during either Phase I or Phase II (Hatten et al., 2018). Additionally, there were no statistically significant changes in the 7-day moving max water temperature related to the Phase I (Bladon et al. 2016) or Phase II harvests. While our analysis indicated increases in the 7-day moving max with large effect sizes between prelogging and both postlogging periods in Needle Branch relative to Flynn Creek, the annual mean estimates in the harvested catchment only exceeded the reference catchment in two of the 12 study years. Moreover, our results indicated that stream temperatures never

exceeded the criterion designed to protect core coldwater habitat for salmonids (16 °C; IMST, 2004), which suggests that it was unlikely that cutthroat trout experienced thermal stress following either harvest. Furthermore, 7-day moving max temperatures in Needle Branch (Fig. 4) were well within the optimum growth temperature range (13–15 °C) reported for westslope cutthroat trout (*Oncorhynchus clarkii lewisi*), another headwater subspecies of cutthroat trout (*Oncorhynchus clarkii*). Our findings are similar to other studies that have reported small increases in water temperatures following logging associated with current forest practice rules for private lands in Oregon (Oregon Department of Forestry, 2006; Groom et al., 2011; Bladon et al., 2018).

Although it is important to understand how and why fish populations respond to environmental changes, our study was not specifically designed to assess the plethora of processes potentially related environmental variation. This qualification can apply to many studies of influences of forestry on fish. For example, short-term increases in abundance have been reported for headwater populations of coastal cutthroat trout and rainbow trout *Oncorhynchus mykiss* in other second growth systems following the removal of riparian vegetation and subsequent increases in light availability (Murphy and Hall, 1981; Wilzbach and Cummins, 1986; Wilzbach et al., 2005; Kaylor and Warren, 2017). Indeed, if increased light does not result in excessive water temperatures (Brown and Krygier, 1970; Beschta et al., 1987; Johnson and Jones, 2000) or if other detrimental effects of logging, (e.g., wood removal, increased sedimentation, and loss of pools) are avoided (Murphy and Hall, 1981; Mellina and Hinch, 2009; Rachels et al., 2020), the removal of the riparian forest canopy can stimulate primary and secondary production, and subsequent increases in salmonid biomass (Warren et al. 2016). Conversely, De Groot et al. (2007) observed no change in coastal cutthroat trout abundance after clearcut logging in areas where no standing tree buffer was retained, suggesting that increasing light does not always result in increased production. Additionally, the rapid response of coastal cutthroat trout after logging reported by Bateman et al. (2016) and in this study between prelogging and Phase I are not consistent with what would be expected from a bottom-up process. Ultimately, the relationship between light and nutrients is complicated and spatially dependent (Wilzbach et al., 2005). Although this does not preclude increased light as an important factor with regard to increased coastal cutthroat trout production, it does suggest other factors are likely



**Fig. 7.** Cumulative relative biomass (g) of age-1 + coastal cutthroat trout from annual census of pool and cascade habitats from the main-stem channels of Needle Branch (treatment) (A) and Flynn Creek (reference) (B). Black lines without symbols are prelogging, the red lines are Phase I (postlogging of upper 40% of Needle Branch), and blue lines with symbols are Phase II (postlogging lower 47% of Needle Branch). Dashed vertical grey line indicates the downstream boundary of the Phase I clearcut and the upstream boundary of the Phase II clearcut in Needle Branch and an analogous location in Flynn Creek (see Bateman et al., 2018). The x-axis displays the distance upstream from the weirs located at the downstream terminus of each catchment. In most years neither pools nor cascades were present over the entire distance to the upstream extent of fish in Flynn Creek. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

involved.

For example, changes in discharge can result in changes in food availability, and Harvey et al. (2006) reported increases in summer growth rates of rainbow trout with increasing discharge even though pool areas and depth remained unchanged, a finding similar to the results in Phase I of our study (Bateman et al., 2018). In addition, fishless headwaters are known to transport invertebrate prey into downstream fish-bearing segments in some instances (Wipfli et al., 2007) and boost

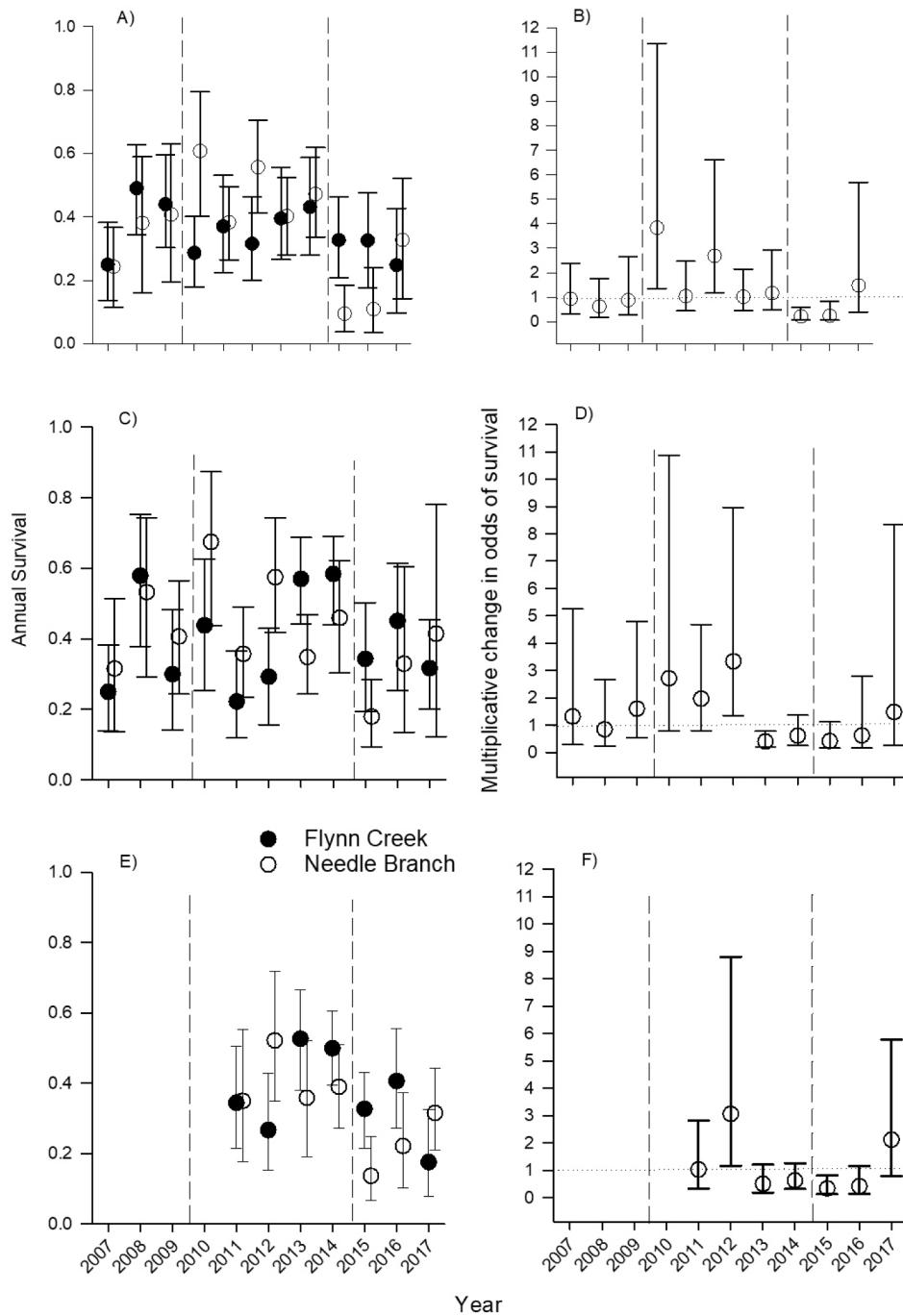
fish growth and abundance in food-limited streams (Wipfli and Gregovich, 2002). Although invertebrate drift distances in small headwater streams appear to be short (Danehy et al., 2011), increases in minimum discharge could increase food availability even if food production remained unchanged. For example, Bateman et al. (2016) reported increases in age-1 + coastal cutthroat trout after clearcut harvesting occurred upstream from fish bearing channels in Hinkle Creek (Cascade Mountains of Oregon, USA), and in Phase I of the current study, density and total biomass of age-1 + coastal cutthroat increased following clearcut logging that included the fishless headwaters of Needle Branch (Bateman et al., 2018).

Conversely it is difficult to predict how food production and availability may change in relation to altered minimum discharge after logging in headwater streams (Moore and Wondzell, 2005). A primary objective of this study was to document the response of coastal cutthroat trout to contemporary logging practices in western Oregon in a catchment that had been previously harvested. No negative effects of logging were detected for either age-1 + coastal cutthroat trout or habitat variables, even after Phase II logging that increased the total clearcut harvest to 87% of the Needle Branch catchment in six years. Results from this study support the perception that forest practice rules 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). For example, coastal cutthroat trout in Needle Branch were not negatively affected by current forest harvests under contemporary forest management guidelines; a major difference considering the adverse effects on coastal cutthroat trout observed when a similar portion of the catchment was clearcut harvested in the original Alsea Watershed Study prior to forest practice rules designed to protect aquatic habitats and biotic resources (Hall, 2008). Results here are consistent with those reported in other recent studies of logging in second growth forests in the Pacific Northwest (De Groot et al., 2007, Bateman et al., 2016; Jensen, 2017; Bateman et al., 2018). Although these studies cannot be considered replicates (Supplemental Table S1), it is important to note that even with a wide array of treatments and sampling methodologies, what might be considered adverse effects have not been detected. Two factors were consistent among the studies: (1) large wood was left in streams and (2) stream banks were carefully protected regardless of the presence of standing tree buffers. In a meta-analysis of past logging studies, Mellina and Hinch (2009) identified the removal of large wood from stream channels as the one factor most commonly associated with negative effects on salmonids.

Although there are issues with the BACI approach (Murtaugh, 2002; Stewart-Oaten, 2003), paired-catchment studies are useful because it is possible account for environmental variables that are not easily controlled by researchers. However, replication is the key to increasing the inferential scope beyond the individual case study, and the only recent study using a replicated design is that of De Groot et al. (2007). Furthermore, paired-catchment studies represent relatively large spatial and temporal scales, but many questions remain because the scales at which forests are managed are much greater in both time and space than those included in this or other recent studies (e.g., De Groot et al. 2007; Bateman et al., 2016, Jensen, 2017, Bateman et al., 2018). Furthermore, the current study does not address lagged effects, such as those occurring 30–40 years post logging (Zhang et al., 2009; Ross et al. 2019) or persistent changes in discharge (Segura et al., 2020). Although clearcut harvest-unit size is often restricted, and requirements such as Oregon's "green up" rules constrain the rate at which contiguous catchments can be harvested, large areas can be logged over relatively short periods of time (i.e., 10–15 years). Ultimately, new tools will be required to allow evaluations of land management practices at these larger scales (Richardson et al., 2012; Coble et al., 2019).

Results from recent studies imply that in the short term, contemporary logging practices can provide ample protection for native populations of coastal cutthroat trout in headwater catchments, even when a substantial proportion of the catchment is harvested in a relatively





**Fig. 8.** Annual probability of coastal cutthroat trout survival by length group (fork length; >110 mm (A), 94–110 mm (C), and 64–93 mm (E)) for Needle Branch (treated, open circles) and Flynn Creek (reference, closed circles) with 95% credible intervals, and the ratio of annual survival odds (Needle Branch/Flynn Creek) by length group (>110 mm (B), 94–110 (D), and 64–93 mm (F)). The horizontal solid line in B and D denotes zero or no difference between treated and reference catchment values. Data on survival of 64–93 mm coastal cutthroat trout were only available for 2011–2017. Logging occurred in Needle Branch in 2009 and 2014 as denoted by vertical dashed lines.

short period (i.e., six years). Concomitantly, there is a growing body of evidence suggesting that fixed riparian management zones or buffers, common to many contemporary timber harvest rules and regulations, do not provide adequate protection for some small mammals (Cockle and Richardson 2003), amphibians (Marczak et al, 2010), and aquatic macroinvertebrate and periphyton communities (Kiffney et. al, 2003; Zhang et. al, 2009). Furthermore, when preventing loss of merchantable timber by windthrow is an objective, a 30 m buffer may be sufficient, but selective logging in a wider buffer strip may be preferable (Mäenpää et al., 2020). In the end it is apparent that large-scale experiments and subsequent long-term monitoring are needed to develop management strategies for maintaining ecological capacity of aquatic and riparian ecosystems (Richardson et.al, 2012), and it is critical to recognize that local context at a variety of spatial scales and variation in species pools

will have a major effect on ultimate outcomes.

**CRedit authorship contribution statement**

**D.S. Bateman:** Conceptualization, Methodology, Investigation, Supervision, Project administration. **N.D. Chelgren:** Methodology. **R.E. Gresswell:** Conceptualization, Methodology, Writing - review & editing. **J.B. Dunham:** . **D.P. Hockman-Wert:** Data curation, Visualization. **D.W. Leer:** Investigation, Supervision, Methodology. **K.D. Bladon:** .

**Declaration of Competing Interest**

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence

the work reported in this paper.

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## Appendix A. Supplementary material

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.foreco.2021.119447>.

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