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Western forests have become increasingly fragmented landscapes dominated by young stands. Given that western Oregon forests largely consist of headwater systems, there is a need to better understand how headwater forest taxa and their habitats are impacted by forest management practices. Several amphibian species associated with forested headwater systems have emerged as management concerns. Forest management strategies, such as harvests that remove only part of the canopy and retention of riparian buffer strips, may help ameliorate some of the negative effects on amphibians in managed forests. Pre-existing site conditions, such as legacy downed wood, also may play a role in buffering the impacts of silvicultural practices on terrestrial amphibians. Downed wood is an important habitat component for many amphibians, because the cool, moist microclimates of downed wood can provide refugia for terrestrial amphibians during warmer summer months. However, downed wood habitat suitability is another emerging concern as the rate of input and size of downed wood declines in managed forests.

As part of the USDI Bureau of Land Managements Density Management Study, we investigated how untreated streamside buffers modify impacts of upland thinning on headwater forest terrestrial amphibians and their habitat at three sites in the Oregon Coast Range. To further assess habitat associations of these animals, we conducted a field experiment to address amphibian cover use, including downed wood, moss and coarse and fine substrates. In addition, we examined how temperature profiles inside small- and large-diameter downed wood and soil temperatures differed from ambient air temperatures. Temperatures of wood and soil were monitored at different slope positions (near streams and upslope) and overstory regimes (thinned and unthinned stands) to assess potential habitat suitability and buffering capabilities against seasonal temperature extremes for plethodontid salamanders.

Our results suggest that pre-existing site conditions (e.g., amount of rocky or fine substrate) play an important role in determining the response of terrestrial amphibians to upland forest thinning. However, retention of stream buffers is important in maintaining unaltered stream and riparian conditions. Moderate thinning and preservation of vital habitat in riparian and nearby upland areas by way of variable-width buffers (15 m minimum width) may be sufficient in maintaining suitable habitat and microclimatic conditions vital to amphibian assemblages in managed headwater forests.

Additionally, logs of a wide size range and soils may provide sufficient protection against thermal extremes harmful to plethodontid salamanders in thinned stands with limited overstory. However, this alone cannot support plethodontid salamanders. These salamanders require exposed areas (e.g., leaf litter, soil surface, rock faces) where much of their foraging and well as courtship occurs. Partial retention of the canopy through moderate thinning coupled with variable-width riparain buffers that may increase in width when suitable terrestrial habitat is encountered, may provide sufficient microhabitat, microclimate, and protection in maintaining terrestrial amphibian assemblages in managed headwater forests.

Terrestrial Amphibian Distribution, Habitat Associations and Downed Wood Temperature Profiles in Managed Headwater Forests with Riparian Buffers in the Oregon Coast Range

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CHAPTER 1

TERRESTRIAL AMPHIBIAN DISTRIBUTION, HABITAT ASSOCIATIONS AND DOWNED WOOD TEMPERATURE PROFILES IN MANAGED HEADWATER FORESTS WITH RIPARIAN BUFFERS IN THE OREGON COAST RANGE

General Introduction

Headwater streams, the channels that occur at the fringe of fluvial networks (Richardson and Danehy, 2007), comprise an estimated 80% of the length of flowing waterways in the Oregon Coast Range (Bury, 1988). There is a need to understand how forest management activities affect these areas and the taxa and habitats associated with them (Bury, 1988; Meyer and Wallace, 2001).

Terrestrial amphibians associated with headwater forests generally require cool, moist habitats due to physiological (e.g., high body to surface ratio and thin, permeable skin) and embryological (e.g., anamniotic embryo) characteristics (Pough et al., 2004). The cool, moist microclimates in moderate to well-decayed downed wood provide shelter and relief from heat stress during warmer summer months, and nesting and foraging sites (Maser and Trappe, 1984; Aubry et al., 1988; Bury et al., 1991; Blessing et al., 1999; Rose et al., 2001).

The riparian ecotone between aquatic and terrestrial environments also provides unique habitat for terrestrial amphibians (Davic and Welsh, 2004). For example, Dunn's salamanders (*Plethodon dunni*) are more abundant in riparian than in upslope habitats (Dumas, 1956; Bury, 1988; Corn and Bury, 1991; Gomez and Anthony, 1996) and are often associated with rocky substrates (Dumas, 1956; Blaustein et al., 1995; Sheridan and Olson, 2003). Western red-backed salamanders (*Plethodon vehiculum*) are found in riparian and upslope areas (Gomez and Anthony, 1996), but often in lower numbers at greater distances from streams (Vesely and McComb, 2002; Rundio and Olson, 2007). *Plethodon vehiculum* are commonly associated with rocky substrates and downed wood (Dumas, 1956; Ovaska and Gregory, 1989; Blaustein et al., 1995; Dupuis et al., 1995). Ensatina (*Ensatina eschscholtzii*) are often detected further upslope from headwater streams (Gomez and Anthony, 1996; Vesely and McComb, 2002; Rundio and Olson, 2007) or in dry stream channels that may have conditions similar to upslope forests (Olson and Weaver, 2007). *Ensatina eschscholtzii* are often associated with surface debris, decaying logs, and moss (Blaustein et al., 1995; Rundio and Olson, 2007).

Riparian buffers play a multifaceted role along headwater streams. The retention of forest vegetation in riparian buffers provides numerous ecological functions including shading that dampens microclimate extremes, improved water quality, a source of downed wood input, a source of leaf litter and organic matter inputs, rooting strength for soil stability (USDA and USDI, 1994; Naiman et al., 2000), a source of terrestrial prey for aquatic predators (Wipfli, 1997), and dispersal habitat for terrestrial and aquatic-dependent species of concern (USDA and USDI, 1994; Olson et al., 2007). Additionally, thinning treatments that remove only part of the canopy and maintain riparian buffer strips may help sustain suitable habitat conditions for amphibians and allow for a faster recovery from disturbances related to timber harvests (Petranka et al., 1994; Harpole and Haas, 1999; Perkins and Hunter, 2006; Vesely and McComb, 2002).

As concerns increase regarding degradation of forests and lotic ecosystems in the Pacific Northwest, laws regulating timber harvest activities have been enacted providing more restrictive guidelines for forest harvest practices (Young, 2000).

However, riparian management approaches differ across jurisdictions and land ownerships. Policies often differ for fishbearing and non-fishbearing streams, and for perennial and ephemeral streams (Olson et al., 2007). In headwaters, these differences range from no protection to a two site-potential tree height no-cut buffer (Sheridan and Olson, 2003; Olson et al., 2007).

No regulations are in place on private timber lands in western Oregon for the retention of no-cut riparian reserves along small perennial non-fishbearing streams (average water flow $< 0.6\ m^3/s$). However, riparian management zones (0-3 m) are designated along these streams in which limited timber harvest is allowed (e.g., limitations of equipment use or levels of tree removal; Young, 2000). Additionally, there are no regulations in place for small ephemeral non-fishbearing streams on private timber lands in western Oregon (Oregon Administrative Rules, 2006). No-cut and management zones are also designated along headwater streams by the Oregon Department of Forestry. Small perennial and ephemeral non-fishbearing streams on lands managed by the State of Oregon receive 8 m-wide no-cut reserves applied to at least 75% of the reach, and a 21-23 m management zone in which limited harvest is allowed (Oregon Department of Forestry, 2001).

Under the Northwest Forest Plan, interim riparian buffers along small perennial non-fishbearing streams on federal forest lands in western Oregon receive 46-76 m-wide riparian buffers, while small ephemeral non-fishbearing streams receive 30-76 m-wide riparian buffers. Within these buffers, selected forest management activities may be permitted, including tree harvest for density control or salvage, but only after extensive assessment of potential impacts of these activities is completed (USDA and USDI, 1994). When riparian management rules were developed in the Northwest Forest Plan, a broad range of values, processes and mechanisms of impact were considered in developing conservation measures on federal forest lands. These management plans considered more than shade retention and sediment and nutrient filtration, ranging from long-term plans for recruitment of downed wood to channels and soil surfaces, downstream transport of both wood and sediment to off-site areas, and trophic sources from riparian habitats to aquatic food webs, to the effects of vegetation and vegetation management on riparian microclimate (Olson et al., 2007). The interim riparian reserves implemented under the Northwest forest plan allow for the possibility of future changes in buffer widths and silvicultural practices within said buffers subsequent to site analysis of harvest impacts.

Whereas most studies support an adverse effect on amphibians of clearcut harvest (Petranka et al., 1994; deMaynadier and Hunter, 1998; Grialou et al., 2000; Karraker and Welsh, 2006), various responses of terrestrial salamanders to thinning treatments have been reported. The suite of responses range from: 1) no reduction in relative abundance (Karraker and Welsh, 2006); 2) short-term declines in relative abundance (Harpole and Haas, 1999; Grialou et al., 2000; Suzuki and Hayes, 2007); 3) short-term increase in captures (Suzuki and Hayes, 2007); 4) little or no effect on amphibian assemblages in upland sites (Perkins and Hunter, 2006); and 5) site-specific patterns potentially driven by available habitat (McKenny et al., 2006; Olson et al., 2006; Rundio and Olson, 2007). Amphibian response to thinning appears to be influenced by a number of interacting factors and may be at least partially explained by site conditions (e.g., availability of suitable habitat such as downed wood or rocky substrates).

Distributions of amphibian species and site conditions may play a large role in the effectiveness of riparian buffers (Rundio and Olson, 2007). Narrow riparian buffers may provide source populations to recolonize upslope areas that have been harvested (Petranka and Smith, 2005). However, salamander species associated primarily with upslope areas, such as *E. eschscholtzii* (Gomez and Anthony, 1996; Vesely and McComb, 2002; Rundio and Olson, 2007) and clouded salamanders (*Aneides ferreus*) (Vesely and McComb, 2002), may not receive sufficient protections from these smaller riparian buffers (Vesely and McComb, 2002). Riparian buffers coupled with sufficient upslope habitat components, such as downed wood (Rundio and Olson, 2007), may serve a dual purpose in protecting both streamside and upslope amphibian populations. However, downed wood habitat suitability as refugia for these species is an emerging concern as downed wood size and input decrease in managed forests (Aubry et al., 1988; Spies and Cline, 1988; USDA and USDI, 1993; Hayes, 2001; Rose et al., 2001; Olson et al., 2006). In addition, the reduced canopy cover from timber harvest activities may result in increased temperatures within downed wood (Marra and Edmonds, 1996), which could make them unsuitable habitat for terrestrial amphibians.

The US Department of Interior Bureau of Land Management's Density Management Study (DMS) was established in 1994 to develop and test alternative forest thinning treatments for managing young (40-70 year-old) stands (Cissel et al., 2006). The primary objective of the upland forest treatments was to assess approaches to accelerate development of late-successional characteristics, while still supplying timber for revenue. The riparian buffer component of the study assesses alternative buffer widths, coupled with upland thinning, in terms of their efficacy to retain key forest values, particularly species and habitats in headwaters. Forest density management treatments and riparian buffer treatments have been monitored to determine how these alternative management strategies affect stand microclimate and habitat characteristics, and to determine treatment effects on selected animal taxa (Cissel et al., 2006; Wessell, 2006; Anderson et al., 2007; Olson and Rugger, 2007; Rundio and Olson, 2007).

As a component of the DMS, this study is one of the first to address effects of upland forest thinning coupled with riparian buffer treatments on terrestrial headwater forest amphibians and their habitats. Specifically, Chapter 2 examines amphibian and habitat distributions, habitat associations, and amphibian species and habitat composition between narrow and wide buffer treatments and how these relationships differ with distance from stream. In addition, to experimentally address amphibian cover preference, enclosures with controlled cover options and terrestrial salamanders were deployed and monitored at one site.

To further address terrestrial amphibian habitat suitability, Chapter 3 compares microclimate differences between substrates (smaller- and larger-diameter downed wood and soil) and ambient air temperatures to: 1) determine how these substrates differ in protecting against temperatures extremes that are harmful to terrestrial amphibians, specifically plethodontid salamanders; and 2) determine if the buffering capabilities of these substrates differ with slope position or canopy cover.

My thesis will provide insight into impacts of thinning treatments and riparian buffers on amphibian assemblages and their habitats in headwater forests. To understand these interactions, we investigated specific aspects of amphibian habitat that are, and are not directly impacted by silvicultural management (e.g., temperature in downed wood and substrate preference, respectively). The work presented in this thesis advances the knowledge on how to manage for microhabitat critical to sustaining terrestrial amphibian assemblages in managed forests of western Oregon.

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CHAPTER 2

TERRESTRIAL AMPHIBIAN DISTRIBUTIONS IN RIPARIAN AND UPSLOPE AREAS AND THEIR HABITAT ASSOCIATIONS ON MANAGED FOREST LANDSCAPES IN THE OREGON COAST RANGE

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ABSTRACT

Over the past fifty years western forests have become increasingly fragmented landscapes dominated by young stands. Given that headwater streams comprise a large portion of the length of flowing waterways in western Oregon forests, there is a need to better understand how forest management impacts headwater taxa and their habitats. Several amphibian species associated with forested headwater systems have emerged as management concerns, especially after clearcutting. Mitigation strategies include alternatives to clearcutting, such as harvests that remove only part of the canopy and maintaining riparian buffer strips. Our study investigates effects of upland forest thinning coupled with riparian buffer treatments on terrestrial headwater forest amphibians. To address how untreated buffers modify impacts of upland thinning on headwater forest amphibians, we examined terrestrial amphibian distribution and habitat associations 5 to 6 years post-thinning in near-stream and upslope positions, within three forest management regimes (streamside-retention buffers and variable-width buffers with upland thinning, and unthinned reference stands). To further address amphibian habitat associations, we conducted a field experiment to address amphibian cover use. Our results suggest that pre-existing site conditions (e.g., amount of rocky or fine substrate) play a significant role in determining the response of terrestrial amphibians to upland forest thinning. However, retention of stream buffers is important in maintaining unaltered stream and riparian vegetation conditions, as microclimates and microhabitats needed for amphibian surface activities. Moderate thinning, and preservation of vital habitat in riparian and nearby upland areas by way

of variable-width buffers (15 m minimum width) may be sufficient in maintaining suitable habitat and microclimatic conditions vital to amphibian assemblages in managed headwater forests.

1. Introduction

Between 1950 and 1995 approximately 80-90% of old-growth forest stands in Oregon and Washington were harvested (Spies and Franklin, 1988; Smith et al., 1998) resulting in an increasingly fragmented landscape (Bury and Pearl, 1999; Biek et al., 2002; Bury, 2004) dominated by young stands (Franklin et al., 2002). Given that headwater streams comprise an estimated 80% of the length of flowing waterways in the Oregon Coast Range (Bury, 1988) there is a need to better understand how forest management activities affect headwater forest taxa and their habitats (Bury, 1988; Meyer and Wallace, 2001). A variety of approaches to managing forests along headwater streams have been applied ranging from no protection to a one or two site-potential tree height buffer (USDA and USDI, 1994; Sheridan and Olson, 2003; Olson et al., 2007). However, the efficacy of buffers for protecting terrestrial headwater species against disturbances experienced during thinning is not well known.

Several amphibian species have been associated with forested headwater systems (Sheridan and Olson, 2003; Olson and Weaver, 2007) including stream-associated species, such as tailed frogs (*Ascaphus truei*) and torrent salamanders (*Rhyacotriton spp.*; Olson et al., 2000, 2007; Bisson et al., 2002; Raphael et al., 2002; Stoddard and Hayes, 2005), and terrestrial species, such as western red-backed salamanders (*Plethodon vehiculum*; Wilkins and Peterson, 2000; Sheridan and Olson, 2003) and Dunn's salamanders (*Plethodon dunni*; Bury et al., 1991; Wilkins and Peterson, 2000; Sheridan and Olson, 2003). Amphibians may play important roles in functions of headwater ecosystems because they provide a central link in food webs as predators and prey (Davic and Welsh, 2004). In particular, amphibians in headwater forests may be pivotal in the exchange of nutrients between streams and uplands (Davic and Welsh, 2004). Disruption of these processes by forest management activities could affect aquatic and terrestrial headwater fauna.

Many plethodontid salamanders are long-lived (e.g., *P. vehiculum* may live up to 10 years) and do not reproduce annually (Ovaska and Davis, 2005). Therefore effects of silvicultural treatments on terrestrial amphibian populations may not be fully realized for many years after timber harvest (Ash, 1988; Petranka et al., 1993). As an alternative to clearcutting, thinning treatments that remove only part of the canopy and maintain riparian buffer strips may help sustain microclimate and habitat conditions suitable for amphibians, allowing for a quicker recovery from disturbances (Petranka et al., 1994; Harpole and Haas, 1999; Vesely and McComb, 2002; Perkins and Hunter, 2006).

Effects of thinning on terrestrial amphibians are beginning to emerge (Harpole and Haas, 1999; Grialou et al., 2000; Perkins and Hunter, 2006; Suzuki and Hayes, 2007). Two years post-thinning, Rundio and Olson (2007) found a negative effect on terrestrial amphibian abundance in response to thinning with riparian buffers at one of two case study sites in western Oregon. Our study is one of the first to investigate effects of upland forest thinning coupled with riparian buffer treatments on terrestrial headwater forest amphibians. In this study we further investigate the site with the negative treatment effect and two additional sites at a later time period, 5 to 6 years post-thinning.

Our primary objectives for this study were to: 1) examine effects of upland thinning and riparian stream buffers on terrestrial amphibian abundance and species composition; 2) investigate effects of upland thinning and riparian stream buffers on habitat distribution and composition; 3) explore amphibian-habitat associations on managed landscapes; and 4) assess how these associations differed with lateral distance from stream. This latter objective is particularly relevant as alternative riparian buffer widths are considered for forested headwaters.

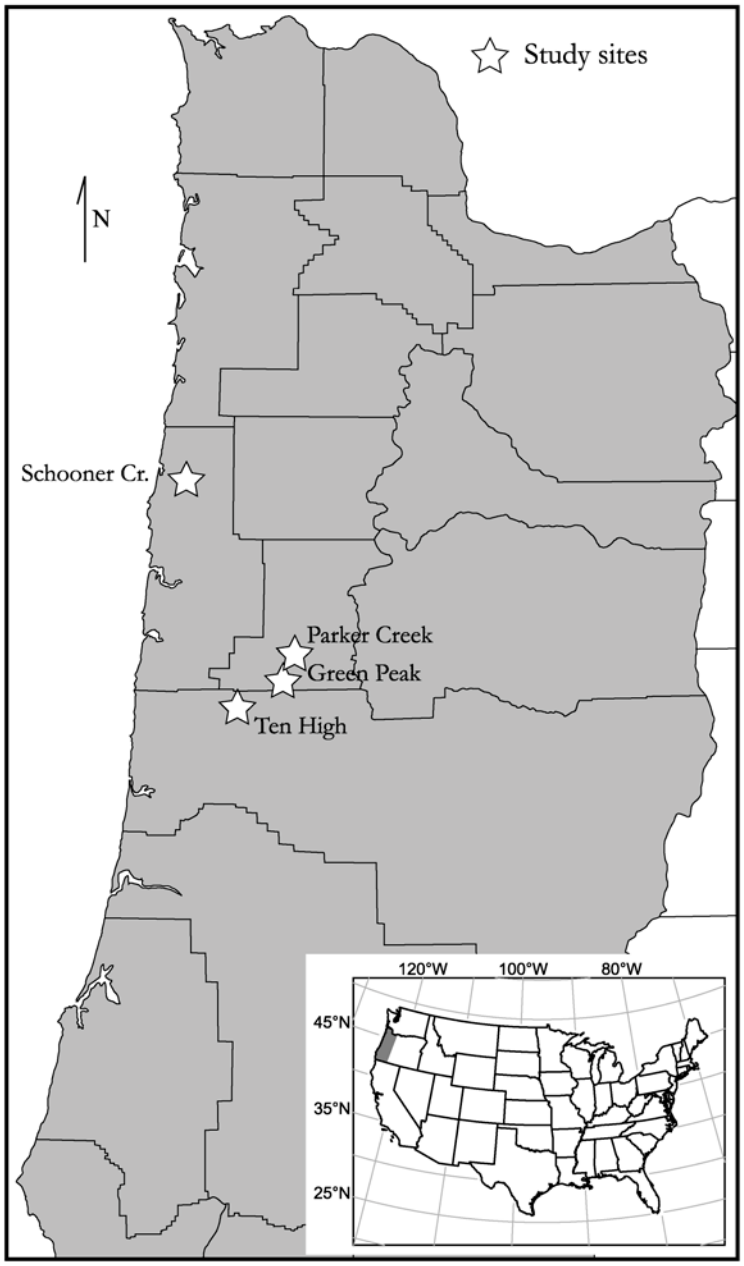
We expected areas that experienced greater canopy cover and fewer disturbances to result in more favorable microhabitat conditions for terrestrial amphibians. As a result, we predicted amphibian captures to be greater in wider stream buffers and unthinned areas compared to narrower buffers and thinned uplands.

2. Methods

2.1 Field Study

Our study was conducted in the central Oregon Coast Range within the western hemlock (*Tsuga heterophylla*) vegetation zone, consisting of wet, mild maritime conditions (Franklin and Dyrness, 1988). For our field study, three sites were selected from U.S. Bureau of Land Management and U.S. Forest Service lands (Figure 2.1). Criteria for site selection included location in the Oregon Coast Range, implementation of thinning and buffer treatments, generally as per the U.S. Bureau of Land Management Density Management Study protocol (Cissel et al., 2006), a minimum of 50 m of upland perpendicular to streams before reaching a ridgeline or entering into the next sub-drainage, and a minimum of 100 m of riparian and upslope area parallel to streams. Two study sites were managed by the Bureau of Land Management (Green Peak, BLM, Salem District; Benton Co., OR; 44˚ 22’ 00”N, 123˚ 27’ 30”W, and Ten High, BLM, Eugene District; Lane Co., Benton Co., OR; 44˚ 16’ 50”N, 123˚ 31’ 06”W) and one study site was managed by the US Forest Service (Schooner Creek, USFS, Siuslaw National Forest; Lincoln Co., OR; 44˚ 55’ 45”N, 123˚ 59’ 18”W). Elevation of the sites ranged from 384 to 870 m.

Figure 2.1. Location of our study sites within the study area of western Oregon showing field study sites (Green Peak, Ten High, Schooner Creek), and enclosure experiment site (Parker Creek).



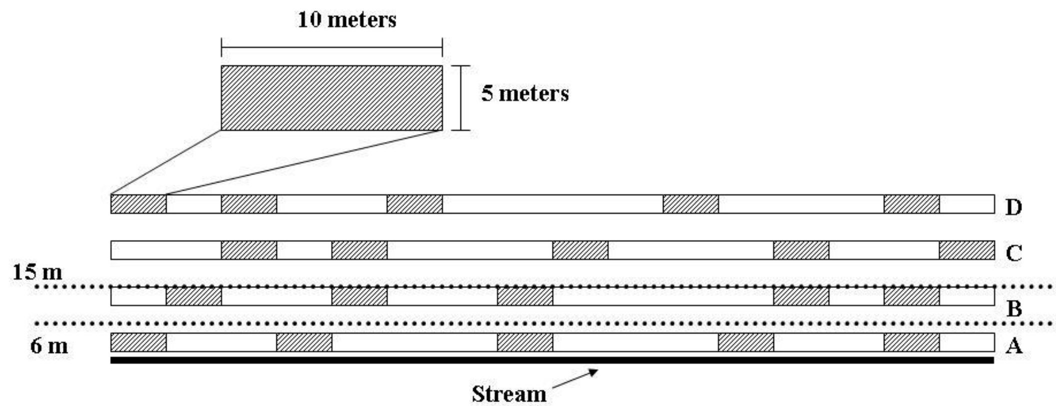
Each site consisted of two streams with buffer treatments and upland thinning and one reference stream with no upslope thinning. The stream reaches were perennial (with the exception of the streamside-retention buffer stream at Ten High, which was intermittent), ranging in width from 0.5 m to 1.5 m, and non-fishbearing. Sites ranged from 12-24 ha in size and consisted of previously unthinned 40-60 year old stands dominated by Douglas-fir (*Pseudotsuga menziesii*), naturally regenerated after clearcut harvests. In 1999 and 2000, thinning occurred at each site as part of a study examining approaches to develop late-successional habitat, such as accelerating development of understory and midstory canopies and increasing spatial heterogeneity of trees and understory vegetation (Cissel et al., 2006). All sites received density management prescriptions, which reduced tree density from about 600 trees per ha (tph) to about 200 tph. An unthinned reference stand was retained at each site.

Riparian buffers were deployed along headwater streams within thinned treatments (Cissel et al., 2006). Our study was conducted along streamside-retention and variable-width buffers. The streamside-retention buffers were about 6 m wide, and were designed to retain trees along stream banks that likely contributed to bank stability and allowed for overhead shading of streams by their crowns extending over the channel. To ensure a higher degree of stream and riparian shading, as well as litter and wood inputs, the variable-width buffers had a minimum slope distance of 15 m from stream edges on both sides of the stream. Widths were increased for unique riparian vegetation, as well as breaks in slope character such as steep slopes, slumps, and surface seeps. In unthinned reference stands, no harvesting was conducted upslope or adjacent to streams.

During the spring of 2005, amphibian and habitat sampling transects (hereafter bands) were deployed at all sites, at four distances from each stream, each band extending parallel to streams (Figure 2.2). Bands were 5 m wide and from 100 to 360 m long, depending on the amount of suitable upland available. Band *A* extended from stream edges and continued upslope 5 m (0-5 m from the stream); Band *B* covered the area 10-15 m from stream edges; Band *C* extended 20-25 m upslope; and band *D* extended 30-35 m upslope (Figure 2.2).

Within bands, amphibians and habitat conditions were sampled by randomly placing five 5x10 m sub-sample units for each of the 4 distance categories (resulting in 20 sub-sample units per treatment, 60 per site, 180 total sub-sample units; Figure 2.2). Sub-sample unit within a band were at least 10 m apart. To avoid edge effects, sub-sample units had to be >20 m from roads, leave islands, and gap cuts. Additionally, due to the semi-destructive nature of the amphibian sampling method, we avoided plots of other studies being conducted on the sites. In order to accommodate all of these concerns, sub-sample units had to be placed on both side of streams for each distance category.

Figure 2.2. Schematic diagram of sub-sample units within bands. A-band is 0-5 m from stream, B-band is 10-15 m from stream, C-band is 20-25 m from stream, D-band is 30-35 m from stream. Dotted line at 6 m indicates streamside-retention buffer width. Dotted line at 15 m indicates minimum width of variable-width buffer.



Amphibian sampling was limited to one site visit during one sampling season between 4 April and 7 June 2005. Sampling was area-constrained (Olson, 1999) and followed a 1 m wide zigzag path within sub-sample units. All moveable cover objects (e.g., rocks, small pieces of wood, moss) were lifted and replaced, any moveable downed wood was turned, decaying logs were dismantled, but not totally destroyed (Olson, 1999), duff and litter were searched, and substrates were searched to maximum depth of 20 cm using a hand tool. No more than 5 minutes were spent searching any cover object. Bark and dismantled logs were replaced as best as possible. When amphibians were captured, species was recorded, as well as position of capture including vertical position (e.g., on or under litter; on, under or in log) and identification of cover object (Bury and Corn, 1988). To estimate amphibian occurrence per distance from stream, amphibian captures in the five sub-sample units within each band were averaged within treatments.

Ten habitat variables were measured or estimated (Table 2.1). Visual estimates were used to determine percent cover for 9 habitat variables within sub-sample units. Percent canopy cover for overstory species was measured in the center of each sub-sample unit using a spherical densiometer (Lemmon, 1956). Substrate composition of each sub-sample unit was determined by digging three 0.10 m x 0.10 m x 0.20 m holes into the substrate. Percentages of fine and rocky substrates were estimated for each hole and averaged for each sub-sample unit. Percent cover of habitat variables within bands was aggregated by averaging values collected across sub-sample units to estimate percent cover. All estimates of cover and substrates were rounded to the nearest five percent. Due to the ability of terrestrial amphibians to use habitat in layers, habitat variables were recorded as layers. For example, a single location within a sub-sample unit may have included moss on downed wood on either fine or rocky substrates. Therefore total percent cover of habitat variables within a sample unit may have exceeded 100%.

Table 2.1. Habitat variables for which percent cover was measured or estimated at Green Peak, Ten High and Schooner Creek study sites in the Oregon Coast Range. Visual estimation of percent cover within sub-sample unit was used for all habitat variables except canopy cover, which was measured using a spherical densiometer.

Variable	**Description**
Fine substrate	Substrate < 3 cm diameter
Rocky substrate (coarse substrate)	Substrate > 3 cm diameter
Litter and duff	Twigs, dead foliage, branches < 10 cm diameter, organic detritus
Shrub cover	Woody plants < 3 m in height
Forb cover	Herbaceous plants (including graminoids)
Moss	Bryophytes
Miscellaneous wood	Chips, chunks, slabs, stumps, loose bark on ground
Downed wood	Downed wood > 10 cm diameter and > 1 m in length
Midstory cover	Foliage of trees < 10 m in height
Canopy cover	Foliage of trees > 10 m in height

For our field study we tested for differences in amphibian abundance relative to distance from stream and treatment using ANOVA with repeated measures of distance (PROC MIXED) in SAS v. 9.1 statistical software (SAS Institute, 2004). Captures of all amphibian species and captures of the most abundant amphibian species were modeled as a randomized complete block (by site) with three treatments. Distance from stream was treated as a repeated measure factor with four levels (i.e., bands). The Tukey-Kramer adjustment was applied to accommodate multiple comparisons. A Treatment x Distance interaction was used to determine if effects of distance from stream was similar among treatments. After viewing residual plots, logarithmic transformations were performed on amphibian captures to meet model assumptions of normality and constant variance. We analyzed whether distance and treatment affected distributions of habitat variables using the same approach. The significance level for all analyses was set at $\alpha \leq 0.10$.

To examine amphibian assemblage and habitat composition, multivariate analysis was conducted using PC-ORD software version 4.25 (McCune and Mefford, 1999). Nonmetric multidimensional scaling (NMS; Kruskal, 1964; Mather, 1976) was used to examine correlations between amphibian assemblages and habitat variables and whether buffer or thinning treatments and lateral distance from stream were related to amphibian and habitat compositions. NMS ordinated plots (bands) in species space using Euclidian distance measures provided a graphical representation of amphibian assemblage relationships with habitat variables. Euclidian distance was used for ordination of plots in species space because it is capable of calculating distances or similarities with bands having 0 captures. To ordinate plots in habitat space, the Sørensen distance measure was used because it works well for non-normal distributions of data (McCune and Grace, 2002). Ordinations were performed for plots in species space and in habitat space with all sites combined, as well as for individual sites to determine site-specific trends. All ordinations used the "slow and thorough" setting of autopilot (maximum number of iterations = 400; 40 runs with real data; stability criterion = 0.00001). For plots in species space, southern torrent salamanders (*Rhyacotriton variegatus*) were deleted as a rare species, as this species was captured in only one band. Data transformation was performed by relativizing by species maxima (McCune and Grace, 2002). Data were not transformed for ordination of plots in habitat space. Outlier analysis with a cutoff of 2 standard deviations indicated no significant outliers.

Differences in amphibian assemblages and habitat composition between distance from stream and buffer treatments were further tested using blocked (for all sites combined) and one-way (for individual sites) multi-response permutation procedures (MRBP and MRPP, respectively; Mielke, 1979). These procedures test for differences among groups by providing a p-value and an effect size (A) which measures within group agreement compared to random expectation (McCune and Grace, 2002).

Indicator species analysis (ISA; Dufrêne and Legendre, 1997) was used to identify amphibian species and habitat attributes indicative of bands and treatments. ISA determines an indicator value (IV), which is based on relative abundance and relative frequency of each species or habitat attribute. A Monte Carlo test of significance provides standard deviations for IV's, along with p-values for the hypothesis of no difference between IVs.

The complex, layered nature of microhabitats available to amphibians presented difficulties in accurately assessing habitat use versus availability. Our sampling of animals and habitat further complicated this assessment because animals were sub-sampled in plots (zig-zag path) while habitat attributes were either fully sampled (e.g., moss, downed wood) or sub-sampled using different methods (e.g., rocky and fine substrates). Additionally, not all habitat variables were encountered within all sub-sample units. As a result, direct estimates of amphibian use of habitat versus habitat availability were unreliable. To gain insight as to how amphibians were using habitat at our sites, we constructed matrices comprised of the microhabitats each amphibian was associated with at the time of capture, allowing us to subjectively assess cover-substrate combinations used by different species.

2.2 Enclosure experiment

Our field experiment to further investigate amphibian cover preferences was conducted near Parker Creek, which originates on Marys Peak in Benton County, Oregon, in the Oregon Coast Range (Figure 2.1). The study site had an elevation of approximately 700 m and was covered by an 80-year-old unthinned stand regenerated after a 1930's clearcut. The dominant overstory species was *T. heterophylla* with *P. menziesii* as a subdominant species..

Six lidded Rubbermaid© containers (0.60 m x 0.45 m x 0.40 m) were used as experimental field enclosures. A 0.30 m x 0.40 m rectangular hole was cut in each lid and replaced with screen to allow air-flow and prevent salamanders from escaping. Holes were drilled in the bottom of enclosures to allow for water drainage and screens were placed over the holes. Half-inch foam weather seal was placed around rims of enclosures to provide a seal between enclosures and lids. Enclosures were set into the ground, approximately 0.20 m deep.

Enclosures were divided into quarters and different substrates and cover options were installed in each quarter. Based on previous literature of salamander cover associations and results from the observational portion of our study, we selected four types of cover-substrate combinations: 1) wood for cover, soil for substrate; 2) moss cover, soil substrate; 3) rock cover, rock substrate; and 4) no cover, soil substrate (resulting in 3 degrees of freedom). Substrates in enclosures were about 0.20 m deep. In an attempt to keep conditions comparable, pieces of wood of similar size, species, and decay class, and chunks of moss (primarily *Eurynchium oreganum*) of similar size were placed in the enclosures. The six enclosures were deployed and experimental trials were conducted once a week for 6 weeks between 14 April and 26 May 2006 ($N = 36$). After each trial, enclosures were washed with a mild soap solution and rinsed. New substrate and cover items were placed in enclosures for each experimental trial.

Western red-backed salamanders (*Plethodon vehiculum*) were used in this experiment because they were most abundant at our three study sites. Adults (SVL > 35 mm) were captured in surrounding sub-drainages each day experiments were conducted. One salamander was placed in the middle of each enclosure and lids were closed and secured with a bungee cord. The enclosures were checked the next morning and position and cover of salamanders were recorded. Salamanders were then returned to their place of capture. For each experimental trial, a new sub-drainage was sampled for salamanders to avoid using the same salamanders twice. A χ^2 test of association was used to determine if *P. vehiculum* cover use differed between cover and substrate provided in our field enclosure experiment.

3. Results

3.1 Field study-Animal analyses

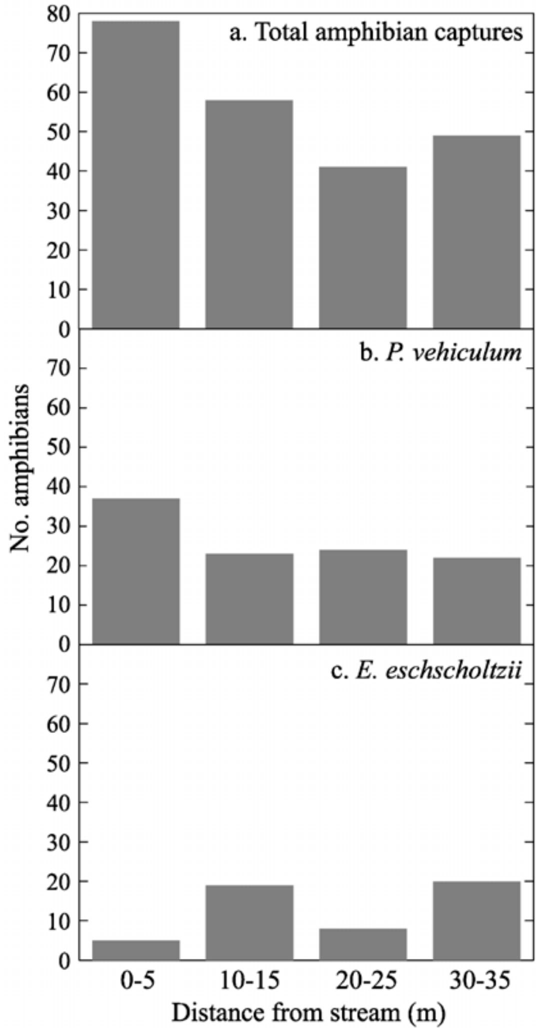
We captured 225 amphibians of 8 species (Table 2.2). Western red-backed salamander (*Plethodon vehiculum*), ensatina (*Ensatina eschscholtzii*) and Dunn's salamander (*Plethodon dunni*) were the most abundant species. We encountered few captures of 4 species: coastal giant salamander (*Dicamptodon tenebrosus*); tailed frog (*Ascaphus truei*); rough-skinned newt (*Taricha granulosa*); and southern torrent salamander (*Rhyacotriton variegatus*). The more abundant amphibians were terrestrially associated species, while less abundant amphibians were generally aquatic-associated species.

Table 2.2. Amphibian species capture by site. *Plethodon spp.* is comprised of both juvenile *Plethodon vehiculum* and *Plethodon dunni* that were too small (SVL < 35mm) to positively identify to species.

Species	**N**	**Green Peak**	**Ten High**	**Schooner**
Plethodon vehiculum	105	78	12	15
Ensatina eschscholtzii	52	30	16	6
Plethodon spp.	32	16	7	9
Plethodon dunni	18	4	8	6
Dicamptodon tenebrosus	7	6	0	1
Ascaphus truei	6	3	0	3
Taricha granulosa	4	3	0	1
Rhyacotriton variegatus	1	1	0	0
Totals	225	141	43	41

ANOVA revealed that amphibian captures were highest near streams and lower in upland plots ($p = 0.019$; Figure 2.3a). Buffer width did not influence this pattern. Amphibian occurrence was 50% greater in the 0-5 m bands compared to the 20-25 m bands ($p = 0.014$) and 30% greater in the 0-5 m bands compared to the 30-35 m bands ($p = 0.09$; Figure 2.3 a). Buffer width also did not influence captures of *P. vehiculum* or *E. eschscholtzii*. There were no differences observed in captures between treatments and no effect of distance x treatment interactions. However, captures differed with distance for *P. vehiculum* ($p = 0.062$). *Plethodon vehiculum* occurrence was 30% greater in the 0-5 m bands compared to the 10-20 m bands ($p = 0.06$) (Figure 2.3 b). Distance effects were again observed with *E. eschscholtzii* captures ($p = 0.006$). Approximately 20% more *E. eschscholtzii* were captured in the 10-15 m ($p = 0.016$) and 30-35 m ($p = 0.02$) bands than in the 0-5 m band, where there were 22% more *E. eschscholtzii* captures in the 10-15 m band ($p = 0.088$) than in the 20-25 m band (Figure 2.3 c).

Figure 2.3. Amphibian distribution by distance from stream (all sites combined) for a) total amphibian captures, b) *Plethodon vehiculum* captures, and c) *Ensatina eschscholtzii* captures.



With all sites combined, NMS ordination showed how habitat variables related to patterns in amphibian species composition, showing subtle grouping by site (Figure 2.4). NMS selected a 2-dimensional solution (Table 2.3). The two axes represented 87.2% of the variation in the original data (Axis 1 = 68.4%; Axis 2 = 18.8%). *Plethodon vehiculum, P. dunni,* and *D. tenebrosus* showed positive correlations with rocky substrates and negative correlations with fine substrates. Conversely, *E. eschscholtzii* and *T. granulosa* had positive correlations with fine substrates and negative correlations with rocky substrates (Table 2.4).

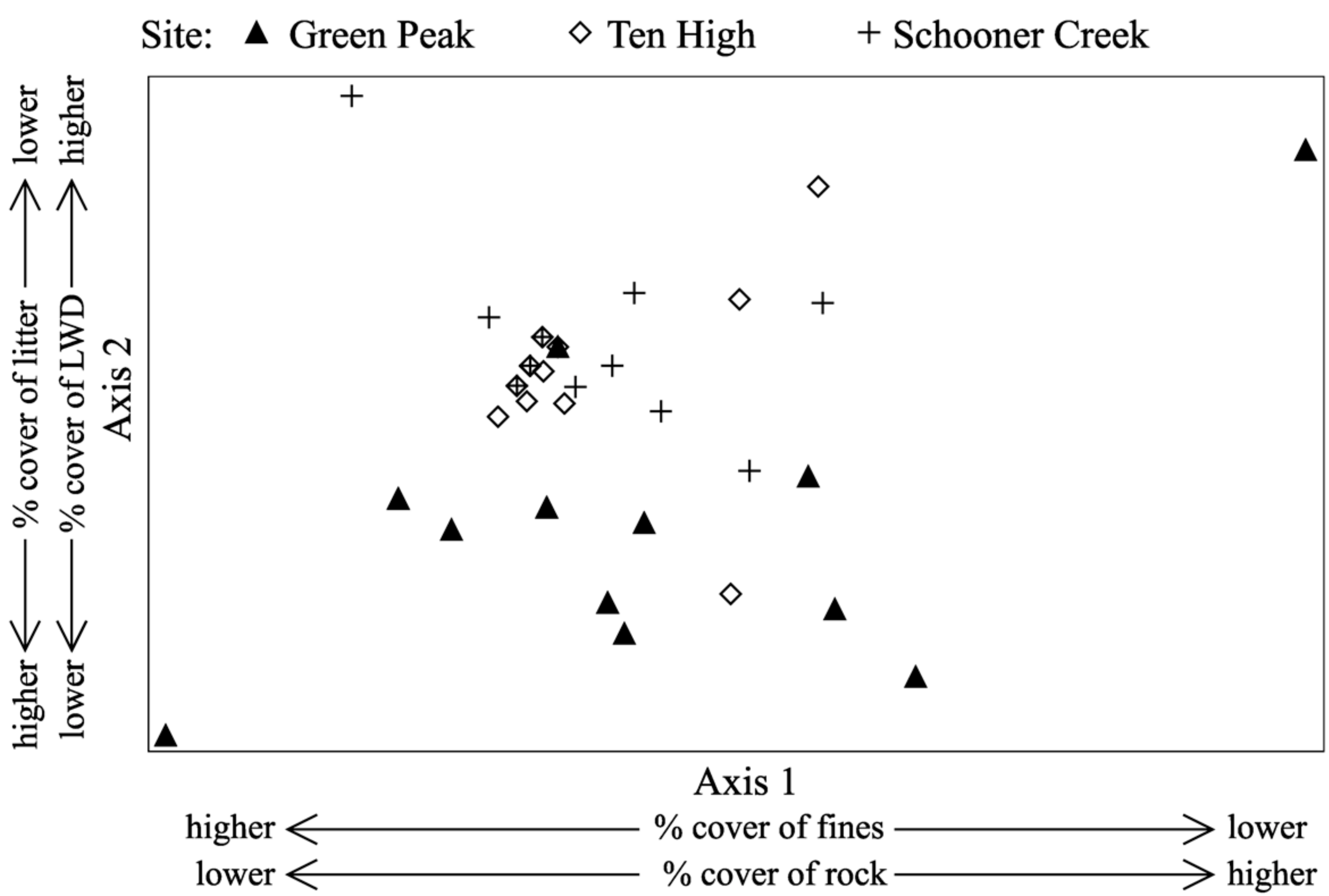
Table 2.3. Summary of NMS ordination output for plots in species space for: all sites combined, Green Peak, Ten High and Schooner Creek.

Site	p-value	Final stress	Final Instability	Iterations
All Sites	0.0196	13.81	0.00571	400
Green Peak	0.0196	6.713	<0.0001	59
Ten High	0.0392	1.26644	<0.0001	51
Schooner Creek	0.0392	4.254	<0.0001	42

Table 2.4. Selected amphibian species and habitat variables and their strongest correlation (Pearson's r) with one of the two axes displayed in the NMS ordinations of plots on species space. Species and habitat variables listed only if correlation is > 0.350.

Species	**r (axis)**	**Habitat variable**	**r (axis)**
All sites			
Plethodon vehiculum	0.504 (1)	Rocky substrate	0.373 (1)
Dicamptodon tenebrosus	0.601 (1)	Fine substrate	-0.379 (1)
Plethodon dunni	0.605 (1)		
Ensatina eschscholtzii	-0.439 (1)		
Taricha granulosa	-0.448 (1)		
Green Peak			
Plethodon vehiculum	0.627 (1)	Rocky substrate	0.632 (1)
Plethodon dunni	0.712 (1)	Downed wood	0.431 (1)
Ensatina eschscholtzii	-0.752 (1)	Litter/duff	-0.615 (1)
		Fine substrate	-0.650 (1)
Ten High			
Plethodon vehiculum	0.896 (2)	Rocky substrate	0.717 (2)
Plethodon dunni	0.902 (2)	Shrub cover	0.714 (2)
		Moss cover	0.402 (2)
		Fine substrate	-0.781 (2)
Schooner Creek			
Ensatina eschscholtzii	0.692 (1)	Shrub cover	0.717 (1)
Plethodon dunni	-0.654 (1)	Miscellaneous wood	0.763 (1)
Plethodon vehiculum	-0.787 (1)		

Figure 2.4. Axes 1 and 2 of a 2-dimensional NMS ordination solution for amphibian species data, all sites combined.



To avoid confounding factors due to site differences, individual sites also were examined using NMS. Grouping by treatment was seen at all three individual sites. At Green Peak, positive correlations were seen between *P. vehiculum* and *P. dunni* captures with rocky substrates and downed wood and negative correlations with litter coverage and fine substrates along axis 1 (Table 2.4), whereas *E. eschscholtzii* captures were positively correlated with litter and fine substrates and negatively correlated with rocky substrates and downed wood along axis 1. NMS selected a 2-dimensional solution (Table 2.3). This ordination was rotated to align the strongest variable from the habitat matrix (rocky substrates) with axis 1. The two axes represented 91.8% of the variation in the original data (Axis 1 = 74.7%; Axis 2 = 17.2%).

At Ten High, *P. vehiculum* and *P. dunni* captures were positively correlated with rocky substrates and shrub and moss coverage, and negatively correlated with fine substrates and miscellaneous wood along axis 2 (Table 2.4). NMS selected a 2-dimensional solution (Table 2.3). This ordination was rotated by a variable from the habitat matrix (downed wood) to allow for a better visual interpretation of the ordination. The two axes represented 96.8% of the variation in the original data, Axis 1 = 33.7%; Axis 2 = 63.2%)

At Schooner Creek, *E. eschscholtzii* captures were positively correlated with shrub cover and negatively correlated with miscellaneous wood, whereas *P. vehiculum* and *P. dunni* captures were negatively correlated with shrub cover and positively correlated with miscellaneous wood cover along axis 1 (Table 2.4). NMS selected a 2-

dimensional solution (Table 2.3). This ordination was rotated to align one of the stronger variables from the habitat matrix (shrub) with axis 1. The two axes represented 90.4% of the variation in the original data (Axis 1 = 49.1%; Axis 2 = 41.3%).

When investigating differences in species composition between buffer treatments and distance from stream for all sites combined, neither buffer width nor distance from stream resulted in a change of species composition (MRBP: treatment: $p = 0.25$; distance: $p = 0.39$), indicating more heterogeneity within treatments and distances than expected by chance alone. When examining sites individually, MRPP showed that amphibian species composition differed among treatments at Green Peak ($p = 0.046$, $A = 0.113$) and Ten High ($p = 0.004$, $A = 0.259$). Species composition did not differ among distances at either site. Additionally, species composition did not differ among treatments or distances at Schooner Creek.

With all sites combined, indicator species analysis identified *P. dunni* as being indicative of the variable-width buffer treatment (Table 2.5). *Ensatina eschscholtzii* was indicative of the 10-15 m band, whereas *D. tenebrosus* was indicative of the 0-5 m band (Table 2.5). *Plethodon vehiculum* and *E. eschscholtzii* were indicative of site conditions (both species for Green Peak; Table 2.5). When sites were examined separately, *P. vehiculum* was indicative of streamside-retention buffers at Green Peak and variable-width buffers at Ten High, whereas at Schooner Creek, *P. vehiculum* was indicative of the 0-5 m band.

Table 2.5. Summary of results for indicator species analysis (plots in species space), for: all sites combined; Green Peak; Ten High; and Schooner Creek. Indicator Value (IV, ranges from 1 to 100) for each species based on combined values of relative abundance and frequency. Only species with IV > 25 and $p \leq 0.1$ are shown.

Site	**Indicator Species**	**Group indicated**	**IV (p-value)**
All Sites	*Plethodon dunni*	Variable-width buffer trt.	38.9 (0.01)
	Plethodon vehiculum	Site (Green Peak)	67.4 (0.001)
	Ensatina eschscholtzii	Site (Green Peak)	47.7 (0.01)
	Ensatina eschscholtzii	10-15 m distance bands	36.6 (0.06)
	Dicamptodon tenebrosus	0-5 m distance bands	28.6 (0.09)
Green Peak	*Plethodon vehiculum*	Streamside-retention buffer trt.	46.0 (0.008)
Ten High	*Plethodon vehiculum*	Variable-width buffer trt.	69.2 (0.01)
Schooner Creek	*Plethodon vehiculum*	0-5 m distance bands	64.7 (0.02)

Visual inspection of amphibian microhabitat association matrices showed that amphibians were most commonly found associated with combinations of rocky cover and fine substrates, rocky cover and rocky substrates, and moss cover and fine substrates (Table 2.6a). *Plethodon vehiculum* captures showed similar associations as all species combined (Table 2.6b), which was expected, because *P. vehiculum* comprised the majority of amphibian captures. *Ensatina eschscholtzii* captures showed associations with moss cover and fine substrates (Table 2.6c).

Table 2.6. Amphibian–habitat associations at the time of capture for: a) all species combined; b) *Plethodon vehiculum*; and c) *Ensatina eschscholtzii*. Data are number of captures with a particular combination of cover and substrate. Columns describe habitat variables for cover used by each amphibian, whereas rows describe habitat variables for substrate on which the animal was captured on. The cover/substrate variables are: fines, rock = coarse substrate > 3 mm diameter, litter = litter or duff, moss, vegetation, misc. wood = miscellaneous wood (e.g., chips, chunks, slabs, bark, wood <10 cm diameter and <1 m in length), downed wood (wood > 10 cm diameter and > 1 m in length). 'Surface visible' indicates the animal was found on the surface, not under any kind of cover. Gray shaded areas indicate dominant habitat associations.

a. All amphibians

Substrate	**Cover Habitat Variables**								
	fines	rock	litter	moss	vegetation	misc. wood	large wood	surface visible	No. amph (%)
fines	13	26	10	43	3	8	0	2	105 (46)
rock	4	32	3	7	1	0	0	0	47 (22)
litter	na	2	8	5	4	2	3	4	28 (13)
moss	na	na	na	0	1	na	na	1	2 (1)
vegetation	na	na	na	na	0	1	na	0	1 (<1)
misc. wood	1	1	2	6	0	11	2	0	23 (10)
downed wood	0	0	0	2	0	0	17	0	19 (8)
No. amph (%)	18 (8)	61 (27)	23 (10)	63 (28)	9 (4)	22 (10)	22 (10)	7 (3)	225 (100)

Table 2.6. (Continued)

b. Western Red-backed Salamander (*Plethodon vehiculum*)

Substrate	**Cover Habitat Variables**								
	fines	rock	litter	moss	vegetation	misc. wood	large wood	surface visisble	No. amph (%)
fines	6	14	3	14	1	4	0	1	43 (41)
rock	3	24	2	4	0	0	0	0	33 (31)
litter	na	2	2	1	2	0	3	0	10 (10)
moss	na	na	na	0	1	na	na	0	1 (1)
vegetation	na	na	na	na	0	0	na	0	0 (0)
misc. wood	1	1	1	2	0	3	2	0	10 (9)
downed wood	0	0	0	1	0	0	7	0	8 (8)
No. amph (%)	10 (10)	41 (39)	8 (8)	22 (21)	4 (4)	7 (7)	12 (11)	1 (<1)	105 (100)

c. Ensatina (*Ensatina eschscholtzii*)

Substrate	Cover Habitat Variables								
	fines	rock	litter	moss	vegetation	misc. wood	large wood	surface visible	No. amph (%)
fines	4	5	4	12	0	3	0	0	28 (54)
rock	0	0	1	2	0	0	0	0	3 (6)
litter	na	0	2	5	0	0	0	0	7 (13)
moss	na	na	na	0	0	na	na	0	0 (0)
vegetation	na	na	na	0	0	1	na	0	1 (2)
misc. wood	0	0	1	2	0	6	0	0	9 (17)
downed wood	0	0	0	0	0	0	4	0	4 (8)
No. amph (%)	4 (8)	5 (10)	8 (15)	21 (40)	0 (0)	10 (19)	4 (8)	0 (0)	52 (100)

3.2 Field study-Habitat analysis

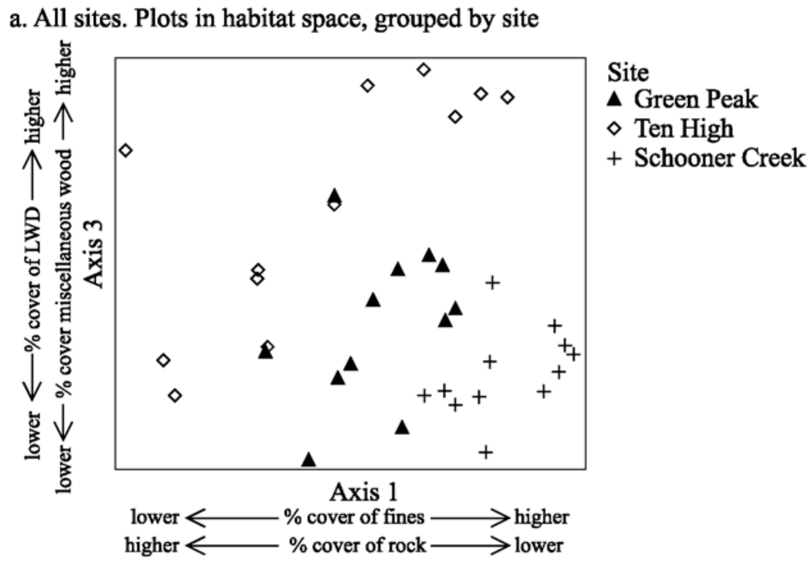
ANOVA revealed Treatment x Distance interactions with percent canopy cover in our analysis of distance from stream and treatment effects on habitat distribution ($p < 0.001$). As might be expected, there were no differences in canopy cover between treatments in the 0-5 m bands. However, within the 10-15 m, 20-25 m, and 30-35 m bands canopy cover was greater along reference and variable-width buffer streams than along streamside-retention buffer streams (Table 2.7).

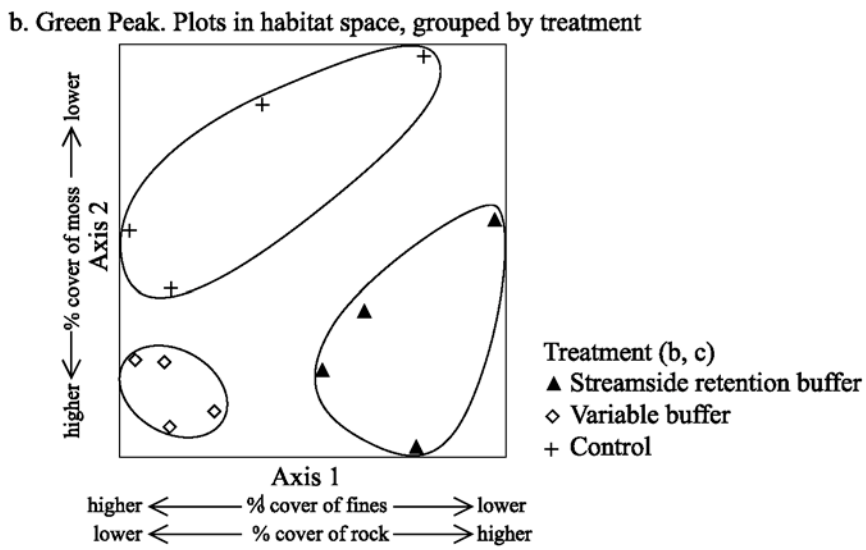
Percent cover of habitat was generally similar among treatments, with the exception of moss ($p = 0.03$). ANOVA indicated sub-sample units along variable-width buffer streams contained greater moss coverage than sub-sample units along reference streams (Table 2.7). Differences in percent cover of downed wood ($p = 0.0009$), litter/duff ($p = 0.009$), and forbs ($p = 0.003$) were observed with distance from stream. Downed wood abundance was greatest in the 0-5 m bands compared to the other three bands, showing that as distance from stream increases, abundance of downed wood decreases (Table 2.7). Litter/duff coverage in the 0-5 m bands was greater compared to both the 10-15 m bands and the 30-35 m bands (Table 2.7). No difference in coverage was observed between the 0-5 and 20-25 m bands. Coverage of forbs was greater in the 0-5 m bands compared to the 30-35 m bands and greater in the 10-15 m bands than in the 30-35m bands (Table 2.7), whereas no difference in forb coverage was detected between the 0-5, 10-15 and 20-25 m bands

Table 2.7. Significant differences (ANOVA, $p \leq 0.10$) in percent cover of habitat variables at three western Oregon study sites between: 1) buffer treatment and distance from stream (canopy cover); 2) distance from stream (downed wood, litter/duff, forbs), and; 3) buffer treatment (moss). "Difference" between contrasts in mean percent cover values is shown.

Habitat variable	Contrast (% cover)	Difference	p-value	95% CI	Comments
Canopy Cover	Streamside 10-15 m (80) vs. Reference 10-15 m (100)	-20.00	0.049	-39.99, -0.006	20% less canopy cover along streamside-retention buffer stream at 10-15 m
	Streamside 20-25 m (65) vs. Variable 20-25 m (86.67)	-21.67	0.027	-41.66, -1.67	25% less canopy cover along streamside-retention buffer stream at 20-25 m
	Streamside 20-25 m (65) vs. Reference 20-25 m (100)	-35.00	0.0002	-54.99, -15.00	35% less canopy cover along streamside-retention buffer stream at 20-25 m
	Streamside 30-35 m (55) vs. Variable 30-35 m (83.33)	-28.33	0.002	-48.32, -8.34	34% less canopy cover along streamside-retention buffer stream at 30-35 m
	Streamside 30-35 m (55) vs. Variable 30-35 m (98.33)	-43.33	< 0.0001	-63.32, -23.34	44% less canopy cover along streamside-retention buffer at 30-35 m
Downed wood	0-5 m (20.56) vs. 10-15 m (12.22)	8.33	0.032	0.59, 16.08	40% more downed wood at 0-5 m
	0-5 m (20.56) vs. 20-25 m (9.44)	11.11	0.004	3.37, 18.86	54% more downed wood at 0-5 m
	0-5 m (20.56) vs. 30-35 m (7.78)	12.78	0.001	5.03, 20.52	62% more downed wood at 0-5 m
Litter/duff	0-5 m (71.67) vs. 10-15 m (80)	-8.33	0.021	-15.60, -1.07	12% less litter/duff at 0-5 m
	0-5 m (71.67) vs. 30-35 m (80.56)	-8.89	0.014	-16.15, -1.62	12% less litter/duff at 0-5 m
Forbs	0-5 m (45) vs. 30-35 m (20)	25.00	0.0003	8.98, 41.02	55% more forbs at 0-5 m
	10-15 m (36.67) vs. 30-35 m (20)	16.67	0.009	0.64, 32.69	45% more forbs at 10-15 m
Moss	Variable (83.75) vs. Reference (55)	28.75	0.036	2.88, 54.61	34% more moss in variable buffer treatment

NMS ordination was used to examine habitat composition associations with treatments, distances from stream or sites. A 3-dimensional solution was selected by NMS ($p = 0.196$, final stress = 12.14, final instability = 0.0163, 400 iterations). The three axes represented 89.1% of the variation in the original data, (Axis 1 = 37.8%; Axis 2 = 11.3%; Axis 3 = 40%) representing patterns in habitat composition between buffer treatments and distance from stream and among sites. Axis 1 was strongly related to site conditions, with Ten High, Green Peak, and Schooner Creek showing significant grouping along axis 1 (Figure 2.5a).





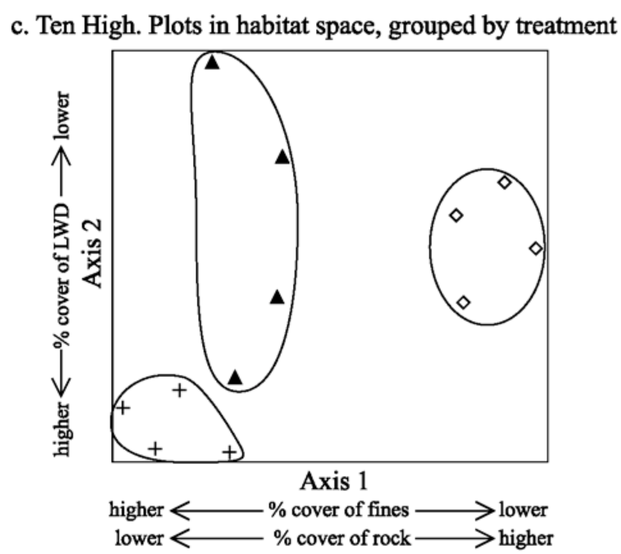


Figure 2.5. NMS ordinations for plots in habitat space for a) all sites combined, grouped by site, b) Green Peak, grouped by treatment, and c) Ten High, grouped by treatment.

Ordinations of habitat attributes for individual sites showed grouping by treatment at Green Peak (Figure 2.5b) and Ten High (Figure 2.5c), but not at Schooner Creek. Habitat attributes did not vary with distance from stream. NMS selected a 2-dimensional solution for Green Peak (Table 2.8). The two axes represented 92.3% of the variation in the original data (Axis 1 = 40.8%; Axis 2 = 51.5%). NMS selected a 2-dimensional solution for Ten High (Table 2.8). The two axes represented 96.0% of the variation in the original data (Axis 1 = 53.5%; Axis 2 = 42.5%). NMS selected a 3-dimensional solution for Schooner Creek (Table 2.8). The three axes represented 97.3% of the variation in the original data (Axis 1 = 42.1; Axis 2 = 35.0%; Axis 3 = 20.2%).

Table 2.8. Summary of NMS ordination output for bands in habitat space for: all sites combined, Green Peak, Ten High, and Schooner Creek.

Site	p-value	Final stress	Final Instability	Iterations
All Sites	0.0196	12.14	0.0163	400
Green Peak	0.0392	9.06	0.0004	400
Ten High	0.0196	4.578	<0.0001	58
Schooner Creek	0.0196	2.791	<0.0001	71

For all sites combined, differences in habitat composition were seen between distances from stream (MRBP; All sites: distance: $p = 0.009$, $A = 0.234$). When examining sites individually, differences in habitat composition among treatments was shown at all three sites (Green Peak: $p = 0.0006$, $A = 0.341$; Ten High: $p = 0.0005$, $A = 0.495$; Schooner Creek: $p = 0.047$, $A = 0.175$), whereas differences in habitat composition with distance from stream were evident only at Schooner Creek ($p = 0.08$, $A = 0.176$).

With all sites combined, indicator species analysis identified moss as indicative of the variable-width buffer treatment. Canopy cover was identified as being indicative of the reference units; whereas downed wood was identified as being indicative of the 0-5 m bands (Table 2.9). Fine substrates, shrubs and forbs were identified as being indicative of Schooner Creek; whereas rocky substrates, miscellaneous wood, and downed wood were identified as being indicative of Ten High (Table 2.9).

Table 2.9. Summary of results for indicator species analysis (plots in habitat space), for: all sites combined; Green Peak; Ten High; and Schooner Creek. Indicator Value (IV, ranges from 1 to 100) for each habitat variable based on combined values of relative abundance and frequency. Only species with IV > 25 and $p \leq 0.1$ are shown.

Site	Indicator Habitat	Group indicated	IV (p-value)
All Sites	Moss	Variable-width buffer trt.	35.2 (0.003)
	Canopy cover	Reference (no trt.)	37.8 (0.001)
	Downed wood	0-5 m distance bands	41.4 (0.01)
	Rocky substrates	Ten High	48.6 (0.03)
	Miscellaneous wood	Ten High	49.4 (0.001)
	Downed wood	Ten High	52.2 (0.006)
	Fine substrates	Schooner Creek	38.9 (0.001)
	Shrub cover	Schooner Creek	39.0 (0.006)
	Forbs	Schooner Creek	58.9 (0.001)
Green Peak	Moss	Variable-width buffer trt.	42.1 (0.03)
	Fine substrates	Variable-width buffer trt.	38.1 (0.05)
	Rocky substrates	Streamside-retention buffer trt.	47.3 (0.09)
Ten High	Canopy cover	Reference (no trt.)	38.1 (0.03)
	Fine substrates	Reference (no trt.)	42.1 (0.004)
	Rocky substrates	Variable-width buffer trt.	60.9 (0.004)
	Shrub cover	Variable-width buffer trt.	50.7 (0.004)
	Midstory cover	Variable-width buffer trt.	58.1 (0.04)
	Downed wood	0-5 m distance bands	40.4 (0.02)
Schooner Creek	Canopy cover	Reference (no trt.)	37.0 (0.03)
	Midstory cover	Reference (no trt.)	65.2 (0.05)
	Downed wood	0-5 m distance bands	47.8 (0.06)

Indicator species analysis conducted with habitat data on an individual site basis identified moss and fine substrates as being indicative of the variable-width buffer treatment; whereas rocky substrates were identified as being indicative of the streamside-retention buffer at Green Peak (Table 2.9). At Ten High, canopy cover and fine substrates were identified as being indicative of the reference unit, whereas rocky substrates, shrub and midstory cover were identified as being indicative of the variable-width buffer treatment (Table 2.9). At Schooner Creek, canopy cover and midstory cover were identified as being indicative of the reference unit (Table 2.9). Downed wood was identified as being indicative of the 0-5 m distance at both Ten High and Schooner Creek (Table 2.9).

3.3 Enclosure experiment

Four of 36 observations were dropped due to escape. In the remaining 32 observations, *Plethodon vehiculum* showed a clear preference for the rock cover-rock substrate treatment ($\chi2 = 20$, df = 3, s = < 0.001) and an avoidance of the open (no cover)-soil substrate treatment ($\chi2 = 20$, df = 3, p = < 0.001). Additionally, no preference or avoidance was shown for the moss cover and soil substrate or the wood cover and soil substrate options.

4. Discussion

Our results indicated that pre-existing site conditions (e.g., amount of rocky or fine substrate) play a significant role in the response of terrestrial amphibians to upland forest thinning. Rundio and Olson (2007) proposed this idea when they found short-term effects of thinning, 1-2 years post-harvest, on salamanders at one site (Green Peak, examined here 5-6 years post-thinning) but not at another site, Keel Mountain, which was not examined in our study. They suggested the large amount of down wood at their Keel Mountain site ameliorated effects of thinning. In our study, no thinning effects were evident at Green Peak after this longer time interval, and rocky substrates appeared to be the likely factor ameliorating negative effects of harvest. Site conditions may result in a lessened immediate effect and a quicker recovery of populations post-harvest.

Plethodon vehiculum is known as a species with an affinity towards rocky substrates (Dumas, 1956; Keen, 1985; Ovaska and Gregory, 1989; Corn and Bury, 1991), which was confirmed in our study. We found *Plethodon vehiculum* associated with rocky substrates, regardless of buffer or thinning treatments.

The importance of pre-existing site conditions may lead to interacting factors influencing results of thinning and buffer effects studies on amphibians and may be responsible for the variation in results of other studies. The suite of responses range from: 1) no reduction in captures (Karraker and Welsh, 2006); 2) short-term decline in captures (Harpole and Haas, 1999; Grialou et al., 2000); 3) increased decline in captures as thinning intensities increased (Suzuki and Hayes, 2007); 4) little or no effect on amphibian assemblages in upland sites (Perkins and Hunter, 2006); and 5) site-specific patterns driven by available habitat (McKenny et al., 2006; Olson et al., 2006; Rundio and Olson, 2007).

While thinning or riparian buffer width did not effect amphibian captures in our study, we found amphibian captures were greater near streams (within 0-5 m of stream edge) compared to upslope (10-35 m from stream edge) (see summary of key findings in Table 2.1). This suggests that amphibians may rely on the cooler temperatures and higher relative humidity, as well as physical habitat characteristics (e.g., downed wood) of near-stream environments for physiological or ecological functions (Petranka et al., 1993; Dupuis et al., 1995). Over half (60%) of our amphibian captures occurred within 15 m from the stream, suggesting that a variable-width buffer capable of including vital habitat (e.g., rocky outcrops, seeps, downed wood, unique riparian vegetation) with a minimum width of 15 m and moderate thinning in upland areas and may be able to provide suitable protection for amphibian populations on managed forests.

Our results further support the current understanding of northwestern terrestrial salamanders, that *P. vehiculum* numbers vary between riparian and upslope areas (Gomez and Anthony, 1996), but sometimes decrease in numbers at greater distances upslope from stream edges (Vesely and McComb, 2002; Rundio and Olson, 2007). On the other hand *E. eschscholtzii* are more often detected in upland areas compared to riparian areas (Gomez and Anthony, 1996; Vesely and McComb, 2002; Rundio and Olson, 2007).

Although amphibian habitat associations in our study were relatively consistent with our current understanding, our results revealed some notable differences. It has been well documented that *E. eschscholtzii* often are associated with downed wood (Blaustein et al., 1995; Butts and McComb, 2000; Biek et al., 2002). Percent cover of downed wood at our sites was greater near streams, whereas *E. eschscholtzii* occurrence was greater upslope. There have been variable results regarding *E. eschscholtzii* use of moss as habitat, ranging from moss use when downed wood volume is low (Rundio and Olson, 2007) to negative associations with the presence of moss (Gilbert and Allwine, 1991). We found *E. eschscholtzii* used a variety of habitats ranging from fine substrates, rocky substrates, litter, moss and wood, but were most often captured with moss as cover, showing that they can be flexible in their use of habitat. *Plethodon vehiculum* were captured most often using rock for cover, but were also associated with a number of other microhabitat conditions, ranging from fine substrates, rocky substrates, litter, moss and wood, similar to what researches have found in British Columbia (Ovaska and Gregory, 1989) and Washington (Dupuis et al., 1995). These amphibians seem to be flexible in using available habitat, but when given options for cover and substrate in a controlled experiment, *P. vehiculum* showed an affinity toward rock for cover and substrate.

Interpretation of our results should consider study limitations, including the small number of study sites used, with only three replications of each buffer treatment. Although our results are directly applicable to these sites only, forest stand conditions at these locations are reflective of managed stands of the Oregon Coast Range. Our findings are also limited by short-term data (one sampling visit in one season, 5 to 6 years post-thinning). Finally, as with any study, detectability of amphibians was of concern. Sampling was conducted during the spring rainy season in an attempt to capitalize on the increased surface activity of amphibians during this time (Ovaska and Gregory, 1989; Dupuis et al., 1995). Green Peak was sampled first, where 63% of our amphibian captures occurred. Warmer and drier conditions were experienced later in the sampling period when Ten High and Schooner Creek were sampled. During the time when these two sites were sampled, daily maximum temperatures reached 23.4°

C and 29.5 ° C, respectively. As temperatures increase and moisture decreases, terrestrial salamanders usually retreat to subsurface refugia (Brattstrom, 1963), reducing their surface activity and making it difficult to find them.

The ability to sustain amphibian assemblages in managed headwater forests appear to be influenced by pre-existing site characteristics and availability of inherently favorable habitat attributes, such as rocky substrates and downed wood. On our sites five to six years post thinning, differences in amphibian captures were not seen between buffer treatments if abundant coarse substrates were present. However, amphibian population densities were consistently higher near the stream than in upland areas, suggesting that riparian zones provide vital habitat through substrates and/or microclimate conditions. Although moist and thermally stable microhabitats are almost always available subsurface (e.g., burrows, downed wood, rock crevices), terrestrial amphibians require exposed microhabitats such as leaf litter, the soil surface, rock faces and the surface of vegetations, where most of their foraging (and probably courtship) occurs (Feder, 1983). The retention of stream buffers is important in maintaining unaltered stream and riparian conditions. Moderate thinning and preservation of vital habitat in riparian and nearby upland areas, for example with variable-width buffers (15 m minimum width), may be sufficient in maintaining suitable habitat and microclimatic conditions vital to amphibian assemblages in managed headwater forests.

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CHAPTER 3

THERMAL PROFILES OF DOWNED WOOD UNDER DIFFERENT FOREST MANAGEMENT REGIMES IN THE OREGON COAST RANGE AND THEIR POTENTIAL IMPACT ON PLETHODONTID SALAMANDER HABITAT

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ABSTRACT

Downed wood provides important faunal microhabitat in forests for numerous invertebrate taxa, small mammals and amphibians. Habitat suitability of downed wood as refugia is an increasing concern in managed forests of the US Pacific Northwest where overstory reduction may result in both reduced downed wood recruitment and increased temperatures within logs may make them unsuitable habitat refugia. We examined temperature regimes inside downed wood and soils to assess buffering capabilities against seasonal temperature extremes and potential habitat suitability for plethodontid salamanders. Temperature profiles of small- (0.3-0.45 m) and large- (0.7-1.0 m) diameter logs, as well as ambient soil and air temperatures, were measured in a 60-year-old forest stand under different slope positions (0-5 m and 35-40 m from stream edge) in three case studies: 1) along a headwater stream with streamside-retention buffer (6 m) and moderate upland thinning; 2) along a headwater stream with variable-width buffer (minimum 15 m) and moderate upland thinning; and 3) along a headwater stream with an unthinned upland. Streamside and upslope maximum air temperatures experienced on all three streams during our study were near or exceeded critical temperatures for western plethodontid salamanders (i.e., ~30°C). Streamside and upslope temperatures inside small logs, large logs and soils stayed below critical temperatures. Our results suggest that logs of a wide size range and soils may provide sufficient protection against thermal extremes in uncut forests and thinned stands with limited overstory.

1. Introduction

Microclimate regimes are an emerging concern in managed forest landscapes as it has become apparent that many organisms are dependent on late-successional and old-growth forest conditions (Franklin and Foreman, 1987; Lehmkuhl and Ruggiero, 1991; Lehmkuhl et al., 1991). In particular, some amphibian species are dependent on cool, moist conditions often found in older forests west of the Cascade Range in the US Pacific Northwest (Welsh, 1990). Impacts of forest management practices on microclimate regimes have been widely studied, ranging from thinning and riparian buffers (Anderson et al., 2007), clearcut and riparian buffers (Brosofske et al., 1997; Rykken, 2005) and structural retention (Heithecker and Halpern, 2006). However, information about microclimates of specific microhabitat refugia is largely lacking.

Over the past three decades, the importance of downed wood for many fish and wildlife species has been established (Maser et al., 1979; Maser and Trappe, 1984: Bull et al., 1997) and also its importance for overall ecosystem functioning (Walker, 1992). Downed wood provides faunal microhabitat for invertebrate taxa (Harmon et al., 1986; Hammond, 1997; Buddle, 2001; Koenings et al., 2002), small mammals (Bull, 2002; Maguire, 2002), and amphibians (Aubry et al., 1988; Butts and McComb, 2000; Bull, 2002). In western forests of the Pacific Northwest, several amphibian species are strongly associated with downed wood (e.g., Dupuis et al., 1995; Butts and McComb, 2000; Welsh and Droege, 2001; Martin and McComb, 2003). Specifically, moderate to well-decayed downed wood is an important habitat component for plethodontid salamanders, such as western red-back salamanders (*Plethodon* *vehiculum*; Aubry et al., 1988; Corn and Bury, 1991), clouded salamanders (*Aneides ferreus*; Bury and Corn, 1988; Corn and Bury, 1991; Butts and McComb, 2000; Bull, 2002), Oregon slender salamanders (*Batrachoseps wrighti*; Bury and Corn, 1988; Bull, 2002) and ensatina (*Ensatina eschscholtzii*; Aubry et al., 1988; Bury and Corn, 1988; Corn and Bury, 1991; Butts and McComb, 2000; Bull, 2002). Temperature regimes are especially critical for these salamanders because they are lungless, therefore gas exchange and water balance occurs through the permeable surface of their skin (Feder, 1983), making them highly susceptible to dehydration (Tracy, 1976).

Thermal tolerances have been documented for plethodontid salamanders, including several species found in the Oregon Coast Range: *P. vehiculum*, *P. dunni*, *A. ferreus* and *E. eschscholtzii*, generally concluding that critical thermal maxima for these species are ~ 30-31 °C (Brattstrom, 1963). Body temperatures of terrestrial salamanders are often the same temperature as the substrates they are occupying (Bogert, 1952; Dumas, 1956). As temperature and moisture limits are reached, these salamanders usually retreat to subsurface refugia (Brattstrom, 1963).

Recently, downed wood habitat suitability as refugia for these species has become an emerging management concern as downed wood is less available and of smaller size in managed forests (Aubry et al., 1988; Spies and Cline, 1988; USDA and USDI, 1993; Hayes, 2001; Rose et al., 2001; Olson et al., 2006). The cool and moist micro-environment provided within logs is an important habitat component for these salamanders (Bury and Corn, 1988; Corn and Bury, 1991; deMaynadier and Hunter, 1995; Maguire, 2002), providing relief from heat stress during warmer summer months, as well as nesting and foraging sites (Maser and Trappe, 1984; Aubry et al., 1988; Bury et al., 1991; Blessing et al., 1999; Rose et al., 2001). However, reduced canopy cover after timber harvests may result in increased temperatures within downed wood (Marra and Edmonds, 1996), potentially making them unsuitable cover for terrestrial amphibians.

This is one of the first studies to address comparisons among downed wood, soil, and air temperatures in relation to wildlife habitat in managed forests of the western Pacific Northwest. Our primary study objective was to assess an important aspect of plethodontid salamander habitat: how thermal environments within logs compare with air and soil temperatures. With the recruitment of large diameter downed wood declining on managed landscapes, our study was designed to determine whether smaller-diameter downed wood can provide similar thermal regimes as larger-diameter legacy downed wood. In addition, to address how lateral distance from stream affects downed wood and soil microclimates, we examined small and large logs, as well as soil and air temperatures near streams and upslope. Our predictions followed logical patterns of expected temperature gradients including: 1) logs, being surrounded by air, would track air temperatures more closely than soil; 2) temperature profiles of small logs would fluctuate more during periods of extreme air temperatures relative to profiles of large logs, as smaller logs are more influenced by daily air temperature fluctuations; and 3) near-stream temperature regimes would be less variable than upslope areas due to the moderating influence of the stream ("stream effect," e.g., Olson et al., 2007).

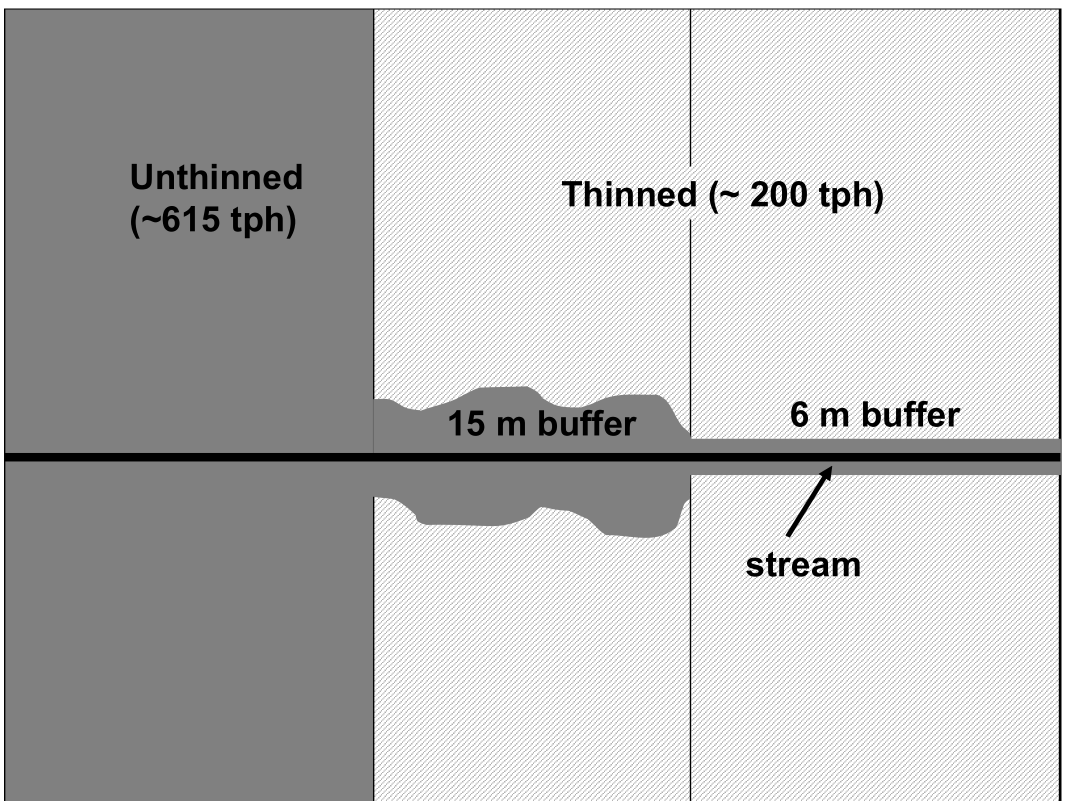
Furthermore, these temperature regimes were investigated along three streams with differing forest management histories. To gain initial insights into effects of thinning and buffers with varying widths, we examined the relationships described above in a narrow riparian buffer with upslope thinning, a wider riparian buffer with upslope thinning, and an unthinned stand, thereby evaluating the temperature regimes relative to temperature tolerances and habitat requirements for plethodontid salamanders.

2. Methods

Our study was conducted at an experimental forest stand in the Oregon Coast Range (Ten High study site: USDI Bureau of Land Management, Eugene District; Lane Co., Benton Co., OR; 44˚ 16’ 50”N, 123˚ 31’ 06”W, elevation 384-870 m). This site is part of the USDI Bureau of Land Management (BLM) Density Management and Riparian Buffer Study (Cissel et al., 2006) and is within the *Tsuga heterophylla* vegetation zone (Franklin and Dyrness, 1988). Douglas-fir (*Pseudotsuga menziesii*) dominated the stand that was naturally regenerated from clearcut harvest in 1946. Other less frequent conifer species included western hemlock (*Tsuga heterophylla*) and western redcedar (*Thuja plicata*). Hardwoods included red alder (*Alnus rubra*) and bigleaf maple (*Acer macrophyllum)*. Common understory species included sword fern (*Polystichum munitum*), Oregon grape (*Mahonia nervosa*), salal (*Gaultheria shallon*), Oregon oxalis (*Oxalis oregana*) and vine maple (*Acer circinatum*).

Characterization of microclimate regimes at different slope positions were attained through repeated sampling of downed wood, soil and air temperatures along three headwater streams with different management histories (Figure 3.1): 1) one stream with a streamside retention buffer (6 m) with moderate thinning to ~ 200 trees per hectare (tph) in the upland; 2) one stream with a variable-width buffer (15 m minimum) with moderate thinning to ~200 tph in the upland; and 3) one stream in an unthinned stand (615 tph). Each stream was treated as a separate case study due to a lack of replications (e.g., buffer treatments). Thinning at our study site occurred in 1999.

Figure 3.1. Schematic diagram of the 6 m and 15 m buffers (variable width buffer with a 15 m minimum width), and thinned and unthinned forest on our study site.



During the winter of 2005, temperature data-loggers (GPSE 301 203, A.R. Harris, Ltd., Christchurch, NZ) were deployed for continuous monitoring over a ten-month period, recording data hourly from December 2005 to October 2006. Eighteen data loggers were deployed at each of the three streams, nine placed streamside (0-5 m from the stream bank) and nine placed in uplands (35-40 m from the stream bank). Of these nine, three were placed in larger-diameter downed wood (0.7 to 1-m diameter), three in smaller-diameter downed wood (0.3 to 0.45-m diameter) and three in soil locations near downed wood, resulting in 54 data-loggers (sample units). Each data logger also monitored ambient air temperatures, allowing for paired substrate-air comparisons.

Following a downed wood survey, downed wood pieces (i.e., logs) were selected randomly from the pool of logs that fit following criteria. Downed wood could not be suspended in the air, thus a log had to be fully resting on the forest floor. Terrestrial amphibians associated with downed wood are most-often found with moderately decayed logs (Aubry et al., 1988; Corn and Bury, 1991; Butts and McComb, 2000). Hence only logs ranging between decay classes 3 and 4 (sensu Maser et al., 1979; Franklin et al., 1981) were chosen for our study. These were logs with bark beginning to slough off and decaying sapwood (decay class 3) and logs with a loss of most bark and decaying heartwood (decay class 4).

To install temperature sensors into downed wood, a drill was used to bore to the center of each log. Once sensors were in place, holes were sealed to prevent external influences. Near each log, soil temperatures were measured at a subsurface depth of 0.15 m, and air temperatures were measured 0.25 m above ground. Air temperature sensors were protected from rain, dew, and direct sunlight by an inverted plastic cup mounted on a fiberglass rod.

To examine temperatures that may be relevant to terrestrial amphibians, hourly temperature data were used to calculate daily maximum and minimum temperatures for substrates (small- and large-diameter downed wood, soil) and air temperatures, near stream and upslope. These daily values were then averaged per week, then per month to determine mean monthly maximum and minimum temperatures. Monthly means were compared among substrates and between substrates and air, within and between the two distances from stream (0-5 and 35-40 m) during February, the coldest month, and July, the hottest month. The coldest and hottest months were those with the lowest and highest mean monthly maximum temperatures, respectively.

To address whether small logs, large logs, and soil differ in their abilities to protect against air temperature extremes, and to investigate if these abilities differed with distance from stream, temperature data were analyzed using analysis of variance (ANOVA) modeled as a randomized split-plot design. In this design, distance from stream was considered the whole plot factor (two levels: streamside and upslope), whereas substrate type formed the first split-plot factor (three levels: small-diameter logs, large-diameter logs and soil) and temperature as the second split-plot factor (two levels: ambient air and inside substrate). Residual plots and histograms were used to check all data distributions for normality. Data were logarithmically transformed to meet assumptions of normality. Back-transformations to the original scale were performed for data presentation. All pairwise comparisons were tested for significance using the Tukey-Kramer procedure. The significance level for all analyses was $\alpha \leq 0.05$. All statistical analysis was done using SAS v. 9.1 statistical software (SAS Institute, 2004).

Daily maximum and minimum temperatures calculated from hourly temperature data were used in determining diel differences for streamside and upslope air temperatures and substrates. To further examine differences in air temperature fluctuation with distance from stream, daily differences were averaged per week and graphed for the month experiencing the lowest and highest mean monthly temperatures (February and July, respectively). Additionally, monthly means of diel differences among substrates (small- and large-diameter logs and soil) were examined during February and July to compare how temperatures of these different substrates were influenced by daily air temperature fluctuations.

To gain additional insight to how these different substrates may buffer against temperature extremes, temperature regimes of small and large logs, as well as ambient soil and air temperatures were plotted and visually inspected for patterns of temperature change for the entire ten-month sampling period.

3. Results

3.1 6 meter buffer

Several significant differences were seen between mean maximum and minimum air and substrate temperatures with distance from stream along the 6 m buffer stream (Tables 3.1 and 3.2). Both February and July mean maximum air temperatures were considerably cooler streamside when compared to upslope (Tables 3.1 and 3.2). When comparing diel temperature differences between streamside and upslope air temperatures, larger differences were seen in the upslope transects along the 6 m buffer stream during February (Figure 3.2a) and July (Figure 3.3a). Substrates (small- and large-diameter logs, soil) along the 6 m buffer stream did not reach the thermal extremes observed for air temperatures, either streamside or upslope (Tables 3.1 and 3.2). These differences in air temperature with distance from stream and differences among substrates and air temperatures also can be seen through visual inspection of temperatures plotted for the entire ten-month sampling period (Figure 3.4). Large and small logs reached similar mean maximum and minimum temperatures, with the exception of minimum temperatures in July where large logs were slightly warmer than small logs (Table 3.2). Although small and large logs reached similar maximum temperatures, larger logs took longer to reach maximum temperatures and longer to cool off, exhibiting a lag time of about two days behind small logs (e.g., July: Figure 3.5). Diurnal temperature fluctuations were obvious within small logs and soil, whereas internal temperatures of large logs were more stable, experienced little to no diurnal fluctuations (Figures 3.5 and 3.6). Small logs experienced greater fluctuations compared to soils, as can be seen during July, especially in the upslope outside the buffer (Figure 3.5), and to a lesser degree during February (Figure 3.6). Diel temperature patterns for substrates generally showed large logs experiencing the least temperature difference, followed by soil and small logs during both February and July (Table 3.3). Substrate temperature diel differences were considerably less than those for air (Table 3.3, Figure 3.2). These results further demonstrate the temperature stability within large logs relative to soil and small logs. Minimum temperatures of large logs were generally warmer than small logs in streamside and upslope positions during July. Additionally, small log minimum temperatures were cooler than soils during February, whereas both small and large logs were warmer than soils during July (Tables 3.1 and 3.2).

Table 3.1. Streamside and upslope mean monthly maximum and minimum temperature ranges for air, large logs, small logs, and soil during February and July along three headwater streams under different management regimes: 1) one stream with a 6 m stream buffer and upslope thinning; 2) one stream with a 15 m stream buffer and upslope thinning; and 3) one stream in an unthinned stand.

Stream	Time	Location	Variable	Maximum Temperature Range (°C)	Minimum Temperature Range (°C)
6-m buffer stream	February	Streamside	Air	7.0 - 9.2	1.9 - 2.7
			Large log	4.9 - 5.0	4.5 - 4.7
			Small log	4.1 - 4.8	3.2 - 3.8
			Soil	6.0 - 6.5	5.5 - 6.1
		Upslope	Air	10.7 - 14.5	1.2 - 2.0
			Large log	5.6 - 5.7	5.0 - 5.4
			Small log	5.3 - 6.3	3.6 - 5.3
			Soil	6.4 - 7.0	5.8 - 5.9
	July	Streamside	Air	22.6 - 30.6	13.2 - 13.9
			Large log	16.6 - 17.5	16.1 - 17.2
			Small log	17.5 - 18.4	16.2 - 16.9
			Soil	14.6 - 16.5	14.3 - 15.3
		Upslope	Air	26.4 - 36.6	12.1 - 13.2
			Large log	20.8 - 23.6	20.3 - 22.6
			Small log	17.8 - 21.6	15.5 - 19.7
			Soil	16.1 - 18.8	16.0 - 16.9
15-m buffer stream	February	Streamside	Air	7.1 - 8.9	2.3 - 2.7
			Large log	4.7 - 4.7	3.8 - 4.4
			Small log	4.9 - 5.3	4.2 - 4.6
			Soil	6.4 - 6.7	5.8 - 6.2
		Upslope	Air	91.- 13.8	1.8 - 2.2
			Large log	4.0 - 5.5	3.3 - 5.1
			Small log	5.6 - 5.8	4.6 - 4.8
			Soil	5.9 - 6.6	5.1 - 5.9
	July	Streamside	Air	21.7 - 23.6	13.1 - 13.6
			Large log	15.9 - 17.6	15.1 - 16.9
			Small log	15.6 - 16.2	14.7 - 15.4
			Soil	15.0 - 16.0	14.3 - 14.9
		Upslope	Air	24.2 - 33.4	12.8 - 13.8
			Large log	17.2 - 18.8	16.5 - 18.0
			Small log	18.8 - 19.3	16.8 - 17.8
			Soil	15.9 - 18.4	14.9 - 15.8

Table 3.1. (Continued)

Stream	Time	Location	Variable	Maximum Temperature Range (°C)	Minimum Temperature Range (°C)
Unthinned	February	Streamside	Air	5.4 - 6.6	1.6 - 2.4
			Large log	3.9 - 5.5	3.4 - 5.2
			Small log	3.4 - 3.6	2.8 - 2.9
			Soil	5.1 - 6.7	4.8 - 6.3
		Upslope	Air	6.0 - 7.7	1.6 - 3.3
			Large log	4.4 - 4.7	3.6 - 3.9
			Small log	3.7 - 3.8	2.3 - 2.3
			Soil	5.3 - 6.0	4.4 - 5.2
	July	Streamside	Air	20.2 - 22.7	12.1 - 12.6
			Large log	15.2 - 16.6	14.9 - 16.3
			Small log	14.8 - 15.9	14.2 - 14.8
			Soil	12.8 - 14.4	12.5 - 13.8
		Upslope	Air	22.3 - 23.3	12.4 - 12.9
			Large log	15.8 - 16.6	15.3 - 16.2
			Small log	16.2 - 16.5	14.6 - 15.6
			Soil	14.4 - 14.9	13.8 - 14.2

Table 3.2. Mean monthly maximum and minimum temperature comparisons (ANOVA) between air, soil, large logs and small logs along a headwater stream in the Oregon Coast Range with a 6 m buffer and upland thinning during February and July. Only findings where $p \leq 0.05$ are shown.

Comparison	Mean Temp Measure	Time	Estimate	95% CI	p-value	Interpretation
Air (streamside vs. upslope)	Maximum	February	0.61	0.55, 0.68	< 0.0001	Air 39% cooler streamside
Substrates (streamside vs. upslope)			0.86	0.77, 0.96	0.009	Substrates 14% cooler streamside
Air vs. substrates (streamside)			1.49	1.36, 1.63	< 0.0001	Air 49% warmer than substrates
Air vs. substrates (upslope)			2.09	1.91, 2.29	< 0.0001	Air 109% warmer than substrates
Air vs. small logs			1.92	1.70, 2.17	< 0.0001	Air 92% warmer than small logs
Air vs. large logs			1.81	1.60, 2.04	< 0.0001	Air 81% warmer than large logs
Air vs. soil			1.59	1.41, 1.80	< 0.0001	Air 59% warmer than soil
Small logs vs. soil			0.78	0.67, 0.91	0.002	Small logs 22% cooler than soil
Large logs vs. soil			0.81	0.69, 0.95	0.08	Large logs 19% cooler than soil
Air vs. small logs (streamside)	Minimum	February	0.61	0.51, 0.73	< 0.0001	Air 39% cooler than small logs
Air vs. small logs (upslope)			0.36	0.30, 0.43	< 0.0001	Air 64% cooler than small logs
Air vs. large logs (streamside)			0.56	0.47, 0.67	< 0.0001	Air 44% cooler than large logs
Air vs. large logs (upslope)			0.36	0.30, 0.43	< 0.0001	Air 64% cooler than large logs
Air vs. soil (streamside)			0.38	0.32, 0.45	< 0.0001	Air 62% cooler than soil
Air vs. soil (upslope)			0.30	0.25, 0.36	< 0.0001	Air 70% cooler than soil
Small logs vs. soil (streamside)			0.62	0.42, 0.90	0.009	Small logs 38% cooler than soil
Overall temperatures (streamside vs. upslope)	Maximum	July	0.81	0.75, 0.87	< 0.0001	Overall temperatures 19% cooler streamside
Air vs. small logs			1.56	1.39, 1.74	< 0.0001	Air 56% warmer than small logs
Air vs. large logs			1.48	1.32, 1.66	< 0.0001	Air 48% warmer than large logs
Air vs. soil			1.71	1.53, 1.91	< 0.0001	Air 71% warmer than soil
Air vs. substrates (streamside)	Minimum	July	0.83	0.79, 0.88	< 0.0001	Air 17% cooler than substrates
Air vs. substrates (upslope)			0.69	0.65, 0,72	<0.0001	Air 31% cooler than substrates
Substrates (streamside vs. upslope)			1.15	0.81, 0.92	0.0002	Substrates 14% cooler streamside
Air vs. small logs			0.75	0.69, 0.81	< 0.0001	Air 25% cooler than small logs

Table 3.2. (Continued)

Comparison	Mean Temp Measure	Time	Estimate	95% CI	p-value	Interpretation
Air vs. large logs			0.70	0.65, 0.76	< 0.0001	Air 30% cooler than large logs
Air vs. soil			0.82	0.76, 0.89	< 0.0001	Air 18% cooler than soil
Large logs vs. small logs	Minimum	July	1.09	1.00, 1.20	0.04	Large logs 9% warmer than small logs
Small logs vs. soil			1.09	1.00, 1.20	0.04	Small logs 9% warmer than soil
Large logs vs. soil			1.20	1.09, 1.31	0.0002	Large logs 20% warmer than soil

Figure 3.2. Weekly averages (SD) of daily diel differences in air temperatures (maximum minus minimum) during the month of February at streamside (diamond) and upland (triangle) transects along the: a) streamside retention buffer stream; b) variable width buffer stream; and c) unthinned upland stream.

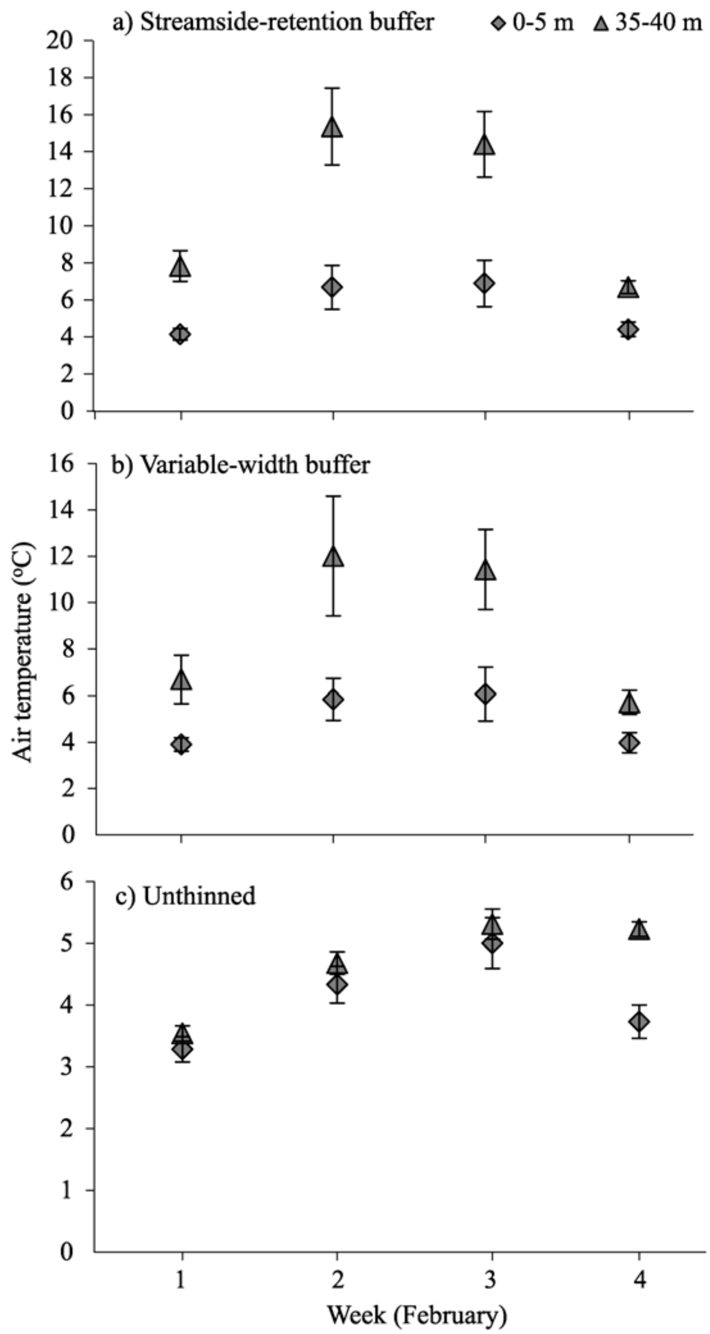


Figure 3.3. Weekly averages (SD) of daily diel differences in air temperatures (maximum minus minimum) during the month of July at streamside (diamond) and upland (triangle) transects along the: a) streamside retention buffer stream; b) variable width buffer stream; and c) unthinned upland stream.

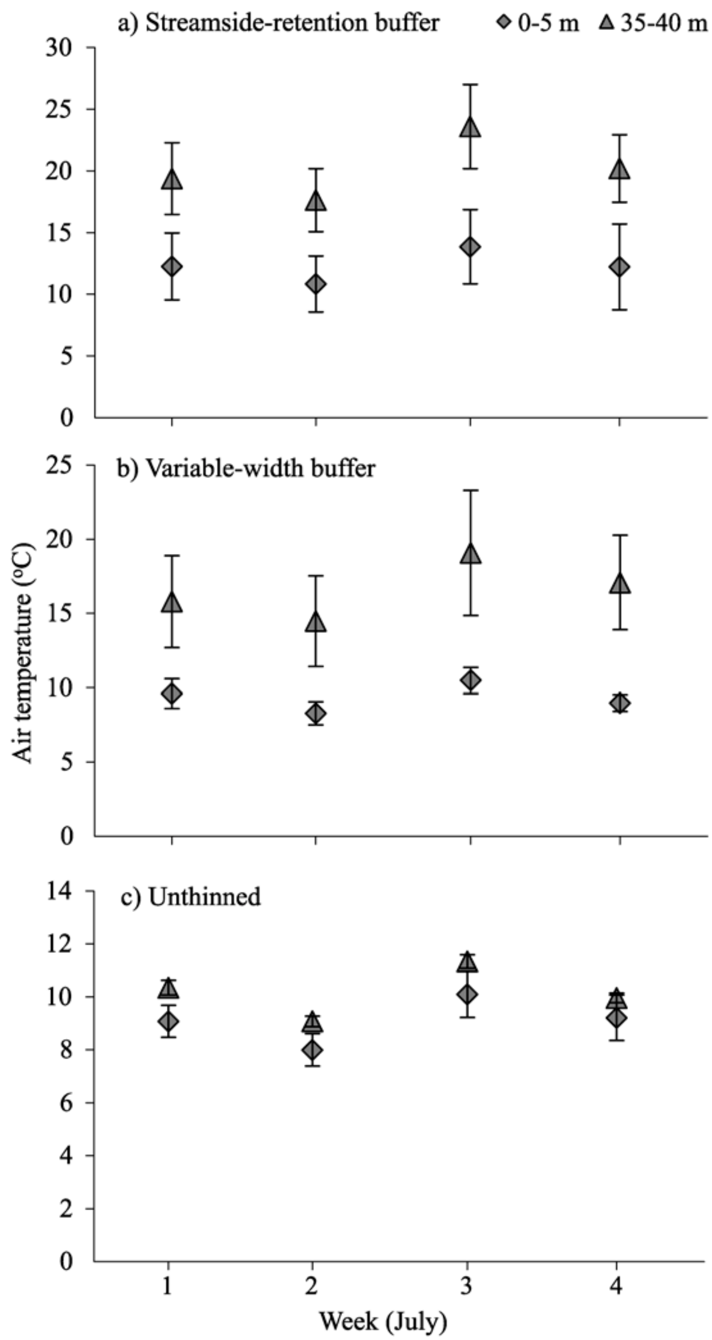


Table 3.3. Monthly means (n=28) of daily diel differences (SD) during February and July for large logs, small logs, and soil along three headwater streams under different management regimes: 1) one stream with a 6 m stream buffer and upslope thinning; 2) one stream with a 15 m stream buffer and upslope thinning; and 3) one stream in an unthinned stand.

Stream	Month	Distance (m)	Substrate	Mean	SD
6 m buffer stream	February	0-5	Large logs	0.39	0.09
			Small logs	0.96	0.04
			Soil	0.51	0.27
		35-40	Large logs	0.51	0.13
			Small logs	1.26	0.40
			Soil	0.91	0.34
	July	0-5	Large logs	0.42	0.07
			Small logs	1.22	0.34
			Soil	0.73	0.48
		35-40	Large logs	0.73	0.25
			Small logs	1.98	0.28
			Soil	1.36	0.58
15 m buffer stream	February	0-5	Large logs	0.58	0.29
			Small logs	0.67	0.03
			Soil	0.52	0.09
		35-40	Large logs	0.53	0.13
			Small logs	0.98	0.09
			Soil	0.78	0.10
	July	0-5	Large logs	0.80	0.48
			Small logs	0.84	0.07
			Soil	0.84	0.26
		35-40	Large logs	0.66	0.22
			Small logs	1.84	0.32
			Soil	1.55	0.89
Unthinned upslope stream	February	0-5	Large logs	0.35	0.15
			Small logs	0.68	0.06
			Soil	0.34	0.06
		35-40	Large logs	0.84	0.04
			Small logs	1.46	0.05
			Soil	0.87	0.07
	July	0-5	Large logs	0.38	0.17
			Small logs	0.86	0.28
			Soil	0.40	0.13
		35-40	Large logs	0.40	0.02
			Small logs	1.34	0.63
			Soil	0.68	0.07

Figure 3.4. Temperature profiles of large and small logs, soil and air for the 10-month sampling period at two distances from stream (0-5 m and 35-40 m) along a headwater stream with a 6 m buffer stream and upland thinning in western Oregon. Average critical thermal maximum (~ 30 °C) for western plethodontid salamanders (Brattstrom, 1963) shown by dotted line.

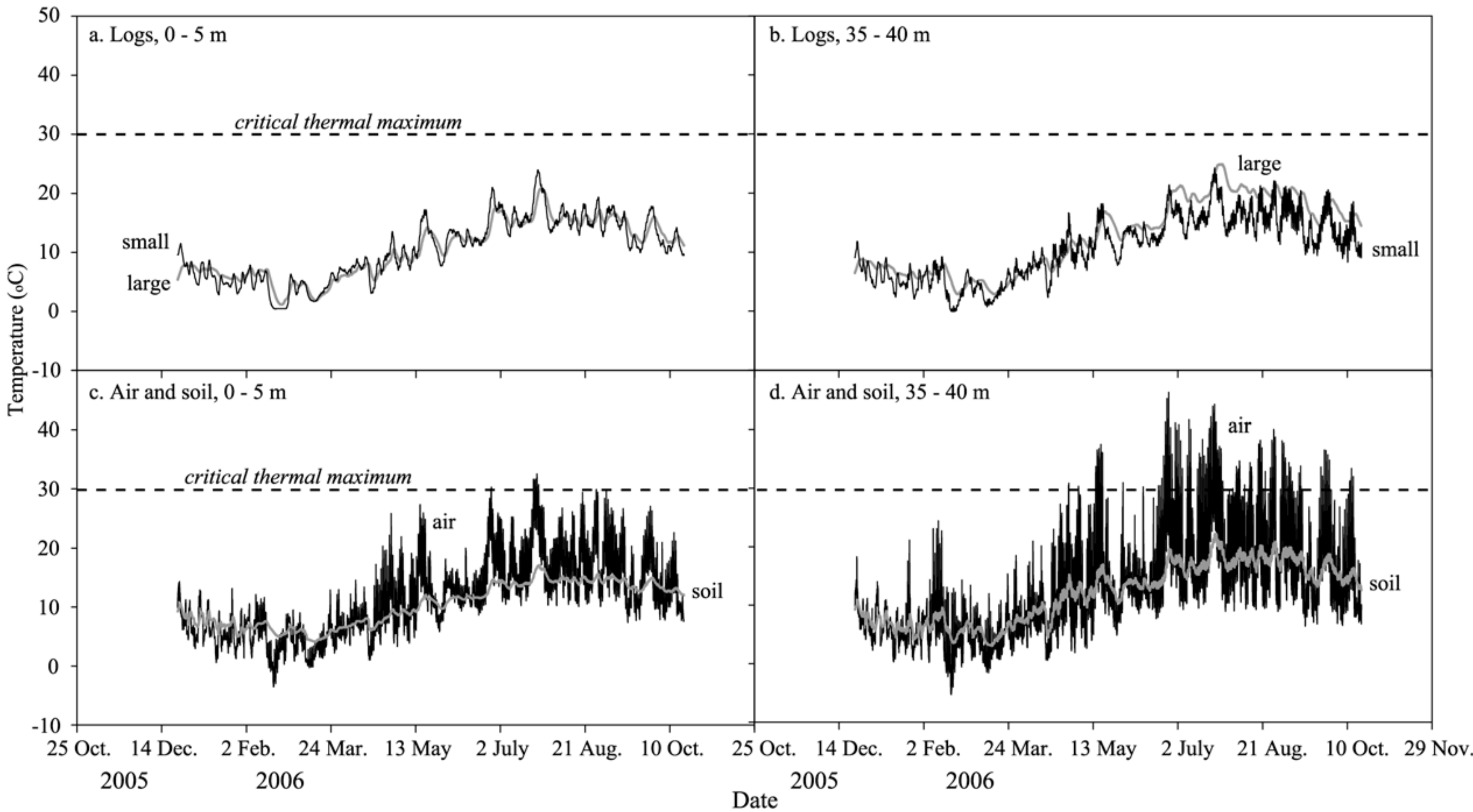


Figure 3.5. Temperature profiles of large and small logs, soil and air during July at two distances from stream (0-5 m and 35-40 m) along a headwater stream with a 6 m buffer and upland thinning in western Oregon. Average critical thermal maximum (~ 30 °C) for western plethodontid salamanders (Brattstrom, 1963) shown by dotted line.

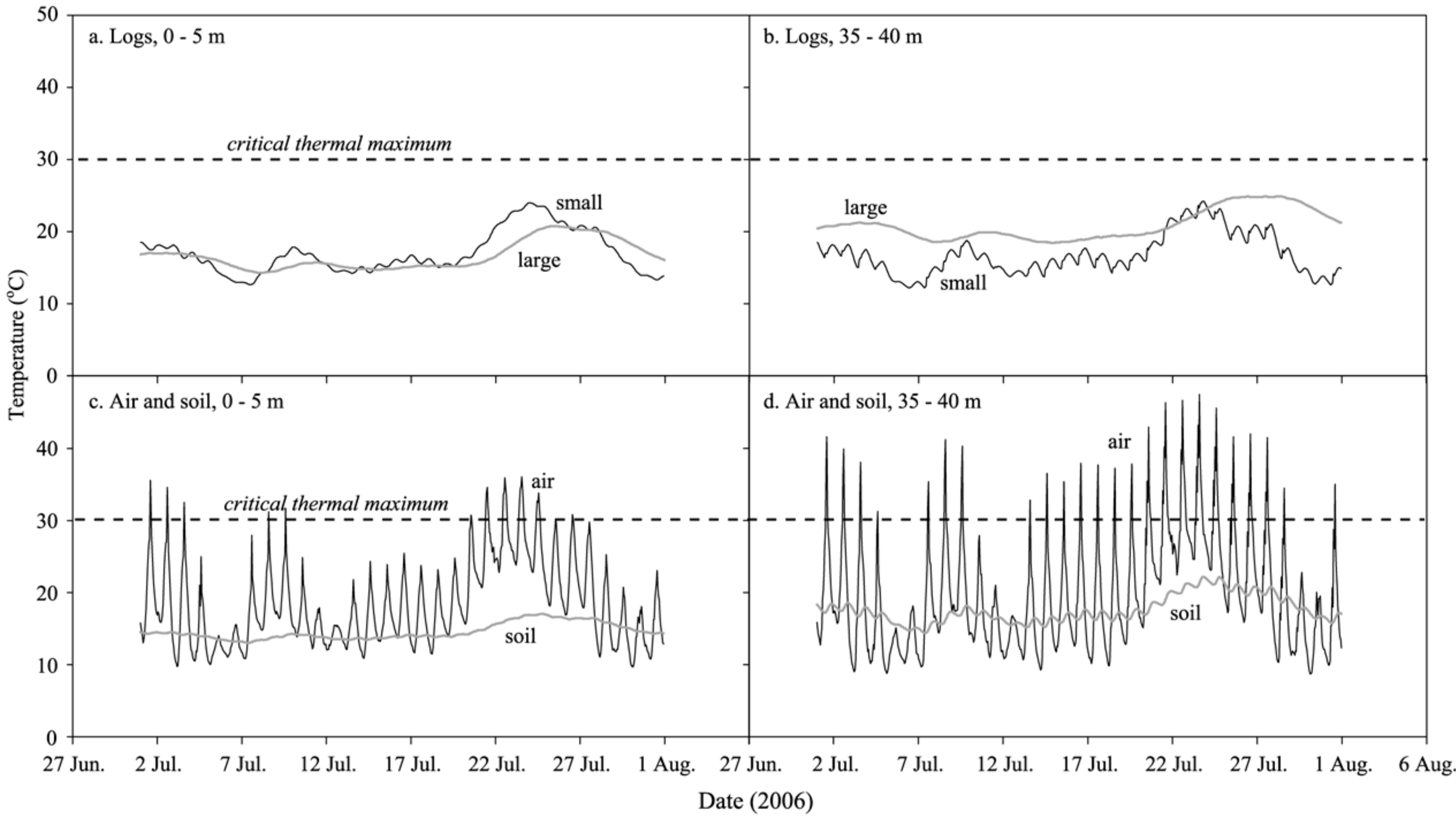
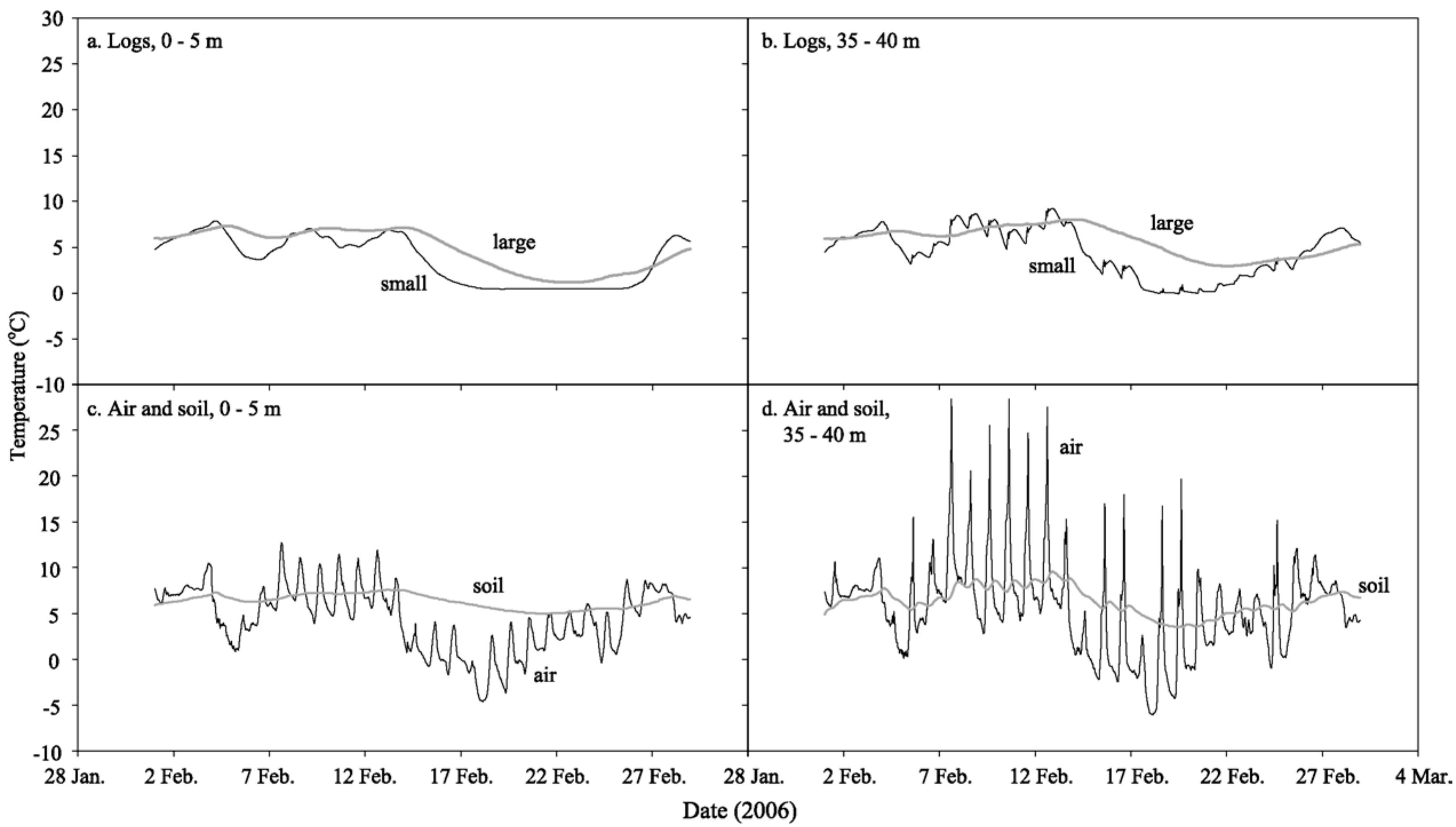


Figure 3.6. Temperature profiles of large and small logs, soil and air during February at two distances from stream (0-5 m and 35-40 m) along a headwater stream with a 6 m buffer and upland thinning in western Oregon.



3.2 15 m buffer

Many results along the 15 m buffer stream mirror those along the 6 m buffer stream (Tables 3.2 and 3.4). Air temperature extremes were greater upslope compared to streamside, and substrate temperatures did not reach the extremes of air temperatures (Tables 3.1 and 3.4; Figure 3.7). Diel differences in air temperatures were greatest in the upslope during February (Figure 3.2a) and July (Figure 3.3b) compared to streamside, although fewer differences were experienced relative to air temperatures along the 6 m buffer stream (Figures 3.2a, b and 3.3a, b). Mean maximum substrate temperatures were generally cooler than air temperatures, whereas mean minimum substrate temperatures were warmer, both streamside and upslope (Tables 3.1 and 3.4). Much less variation in temperature was seen during February when compared to July (Figures 3.8 and 3.9). Small logs experienced the greatest diurnal fluctuation, followed by soil, especially upslope during July (Figure 3.8). However, diel temperature differences among the substrates were relatively similar during both February and July along the 15 m buffer stream (Table 3.3). Large logs experienced a lag time before reaching their maximum and minimum temperatures relative to small logs (Figures 3.8 and 3.9). Interestingly, the temperature profiles of large logs more closely tracked that of small logs in the upslope along the 15 m buffer stream than upslope along the 6 m buffer stream (Figures 3.4b vs. 3.7b; Figures 3.5b vs. 3.8b; Figures 3.6b vs. 3.9b).

Table 3.4. Mean monthly maximum and minimum temperature comparisons (ANOVA) between air, soil, large logs and small logs along a headwater stream in the Oregon Coast Range with a 15 m buffer and upland thinning during February and July. Only findings where $p \leq 0.05$ are shown.

Comparison	Mean Temp Measure	Time	Estimate	95% CI	p-value	Interpretation
Air (streamside vs. upslope)	Maximum	February	0.69	0.61, 0.78	< 0.0001	Air 31% cooler streamside
Air vs. substrates (streamside)			1.39	1.29, 1.50	< 0.0001	Air 39% warmer than substrates
Air vs. substrates (upslope)			1.94	1.80, 2.09	< 0.0001	Air 94% warmer than substrates
Air vs. small logs			1.78	1.60, 1.98	< 0.0001	Air 78% warmer than small logs
Air vs. large logs			1.78	1.60, 1.98	< 0.0001	Air 78% warmer than large logs
Air vs. soil			1.39	1.25, 1.55	< 0.0001	Air 39% warmer than soil
Small logs vs. soil			0.84	0.71, 0.99	0.04	Small logs 16% cooler than soil
Large logs vs. soil			0.55	0.63, 0.88	0.001	Large logs 45% cooler than soil
Air (streamside vs. upslope)	Minimum	February	1.28	1.15, 1.43	< 0.0001	Air 28% warmer streamside
Air vs. substrates (streamside)			0.54	0.49, 0.60	< 0.0001	Air 46% cooler than substrates
Air vs. substrates (upslope)			0.42	0.37, 0.46	< 0.0001	Air 58% cooler than substrates
Air vs. small logs			0.48	0.42, 0.56	< 0.0001	Air 52% cooler than small logs
Air vs. large logs			0.55	0.48, 0.64	< 0.0001	Air 45% cooler than large logs
Air vs. soil			0.40	0.35, 0.46	< 0.0001	Air 60% cooler than soil
Small logs vs. soil			0.79	0.68, 0.92	0.003	Small logs 21% cooler than soil
Large logs vs. soil			0.73	0.63, 0.85	0.0002	Large logs 27% cooler than soil
Air (streamside vs. upslope)	Maximum	July	0.76	0.69, 0.84	< 0.0001	Air 24% cooler streamside
Substrates (streamside vs. upslope)			0.89	0.80, 0.98	0.02	Substrates 11% cooler streamside
Air vs. substrates (streamside)			1.41	1.30, 1.52	< 0.0001	Air 41% warmer than substrates
Air vs. substrates (upslope)			1.64	1.51, 1.77	< 0.0001	Air 64% warmer than substrates
Substrates (streamside vs. upslope)	Minimum	July	0.91	0.87, 0.94	< 0.0001	Substrates 9% cooler streamside
Air vs. substrates (streamside)			0.87	0.84, 0.91	< 0.0001	Air 13% cooler than substrates
Air vs. substrates (upslope)			0.78	0.76, 0.81	< 0.0001	Air 22% cooler than substrates
Air vs. small logs			0.81	0.77, 0.86	< 0.0001	Air 19% cooler than small logs
Air vs. large logs			0.80	0.76, 0.84	< 0.0001	Air 20% cooler than large logs
Air vs. soil			0.88	0.86, 0.92	< 0.0001	Air 12% cooler than soil
Small logs vs. soil			1.07	1.01, 1.13	0.01	Small logs 7% warmer than soil
Large logs vs. soil			1.10	1.04, 1.16	0.0005	Large logs 10% warmer than soil

Figure 3.7. Temperature profiles of large and small logs, soil and air for the 10-month sampling period at two distances from stream (0-5 m and 35-40 m) along a headwater stream with a 15 m buffer and upland thinning in western Oregon. Average critical thermal maximum (~ 30 °C) for western plethodontid salamanders (Brattstrom, 1963) shown by dotted line.

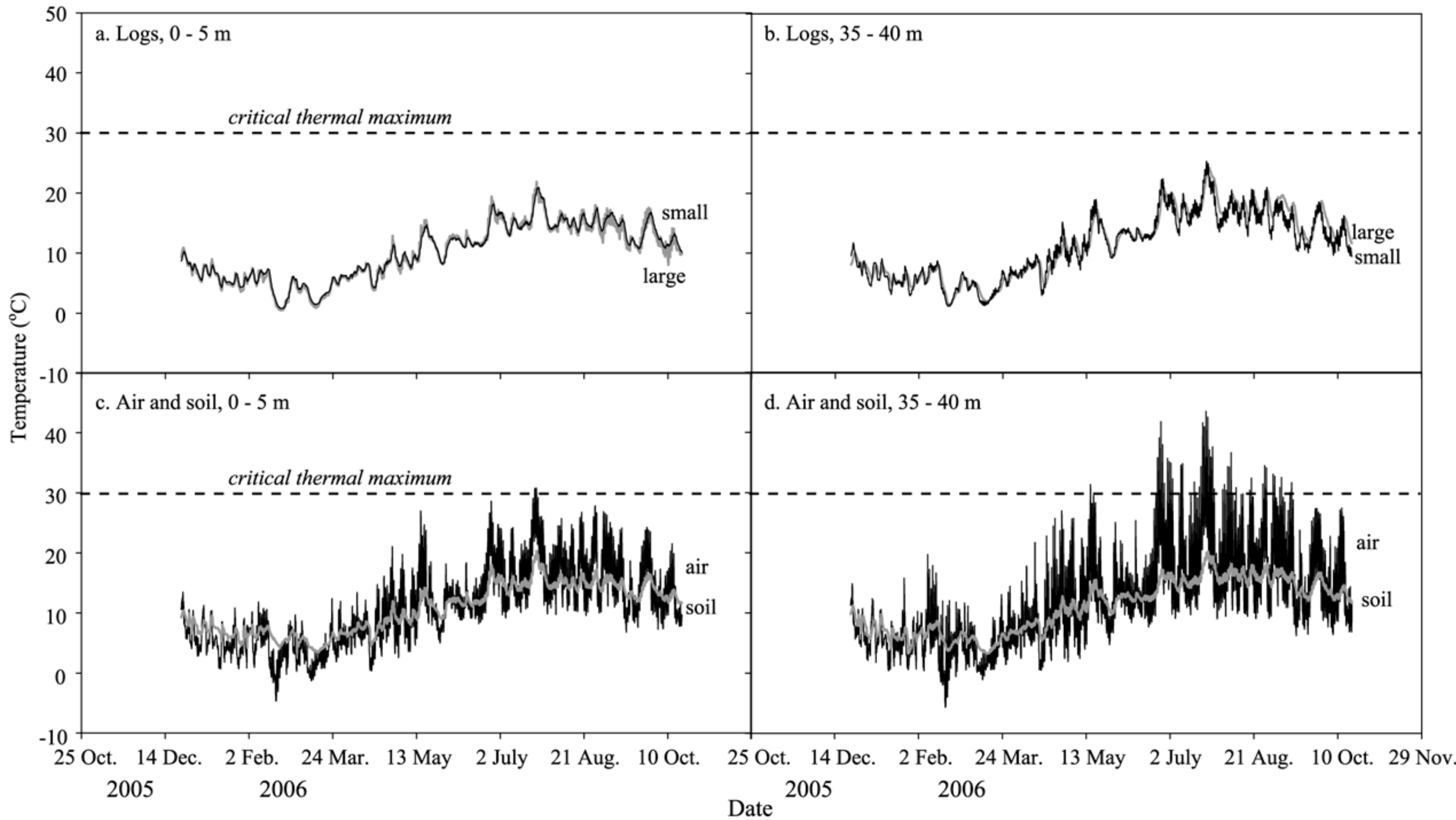


Figure 3.8. Temperature profiles of large and small logs, soil and air during February at two distances from stream (0-5 m and 35-40 m) along a headwater stream with a 15 m buffer and upland thinning in western Oregon.

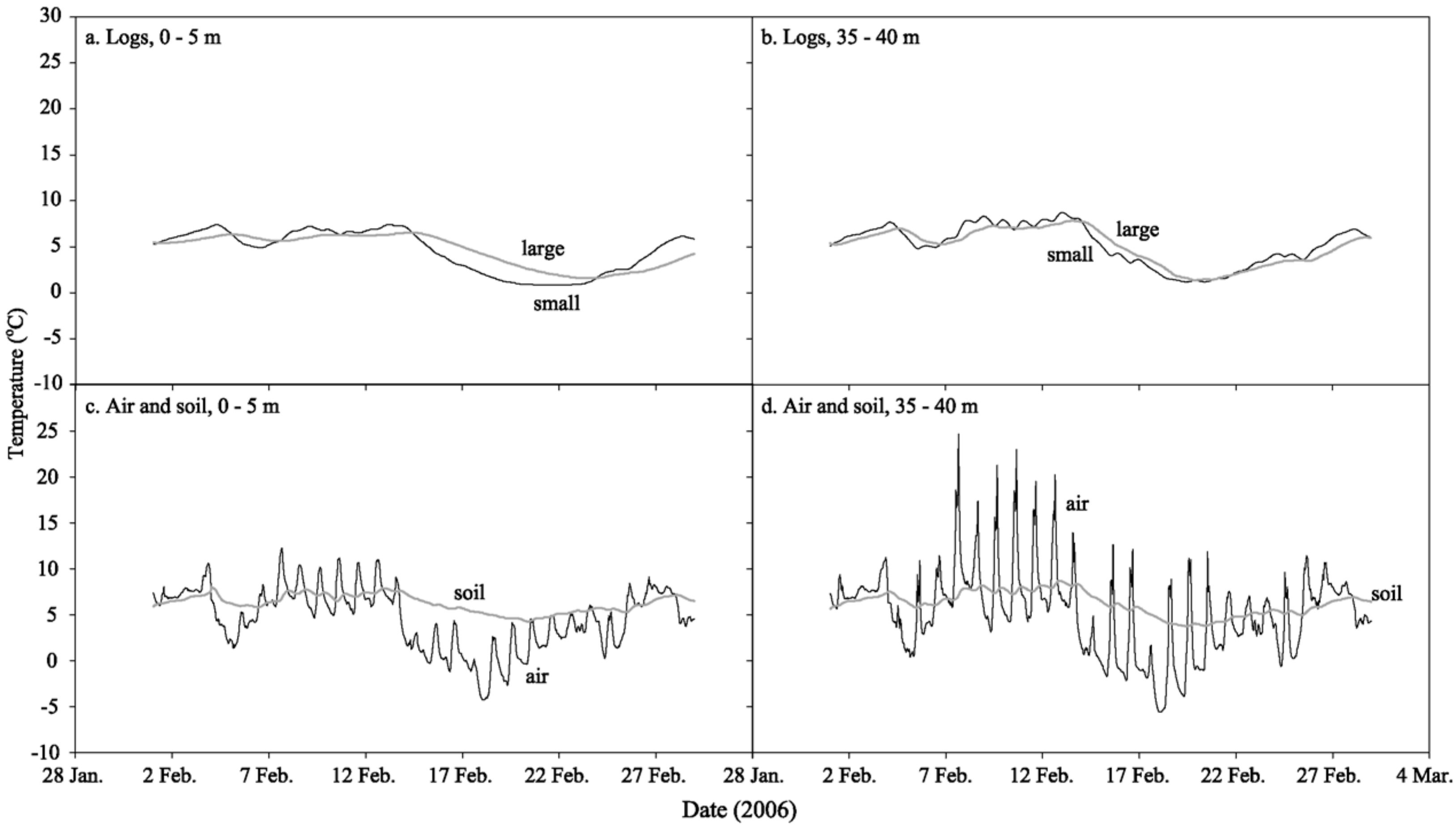
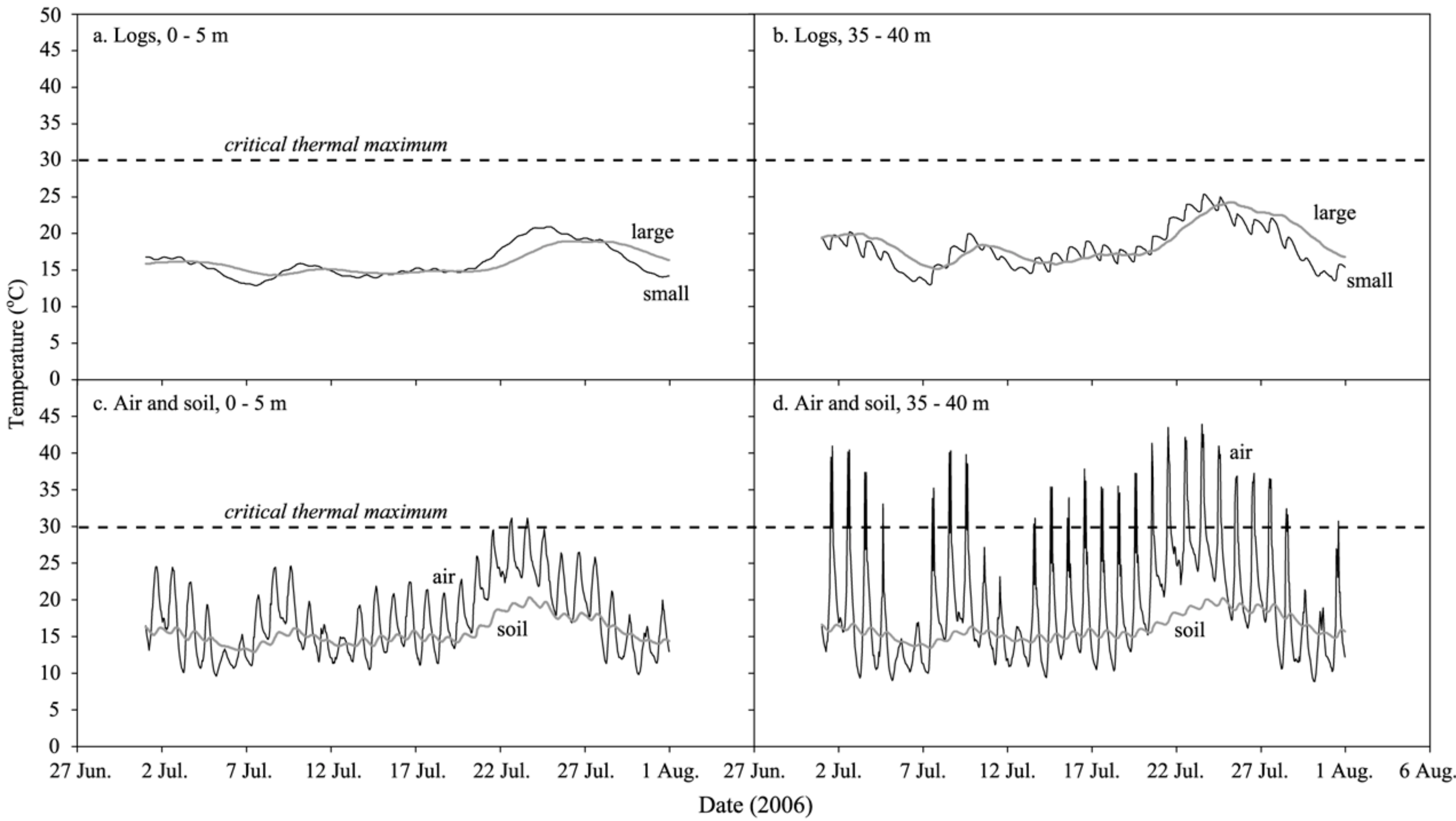


Figure 3.9. Temperature profiles of large and small logs, soil and air during July at two distances from stream (0-5 m and 35-40 m) along a headwater stream with a 15 m buffer and upland thinning in western Oregon. Average critical thermal maximum (~ 30 °C) for western plethodontid salamanders (Brattstrom, 1963) shown by dotted line.



3.3 Unthinned stand

Subtle differences between streamside and upslope mean maximum temperatures were observed along the stream in the unthinned stand (Table 3.5; Figure 3.10). Diel differences in air temperatures between streamside and upslope transects were slight along the stream in the unthinned stand during February (Figure 3.2c) and July (Figure 3.3c). Much less temperature variation was experienced along this stream compared to the 6 m and 15 m buffer streams (Figures 3.2a, b, c and 3.3a, b, c). Similar to the other streams, all substrates (small logs, large logs, soils) protected against the extremes experienced by air temperatures (Table 3.5). Large log temperatures showed less variability in extremes than small logs and showed slight lag times in temperature change relative to small logs (Figures 3.11 and 3.12).

Although all three substrates protected against temperature extremes, soils and large logs were the most effective buffers (Figure 3.10). Soils had slightly cooler mean maximum monthly temperatures compared to large and small logs during July (Tables 3.1 and 3.5). Soils and large logs were generally cooler than small logs during July and warmer during February (Tables 3.1 and 3.5). Large logs experienced very little diurnal fluctuation, whereas small logs tracked diurnal air temperature patterns quite closely and soils did so more subtly. Small logs experienced the greatest diel temperature differences, whereas large logs and soil experienced similar patterns (Table 3.3). In the upslope along this stream, large log temperatures tracked small log temperatures more closely compared to large upslope logs along the stream with the 6 m buffer. Visual comparison between large and small upslope log temperatures along the 15 m buffer stream and along the stream in the unthinned stand show that the large logs track small log temperatures similarly (Figures 3.10b vs. 3.7b; Figures 3.11b vs. 3.8b; Figures 3.12b vs. 3.9b). This is best seen in the upland during July, but also can be seen to a lesser degree streamside (Figure 3.12).

Table 3.5. Mean monthly maximum and minimum temperature comparisons (ANOVA) between air, soil, large logs and small logs along a headwater stream in the Oregon Coast Range in an unthinned stand during February and July. Only findings where $p \leq 0.05$ are shown.

Comparison	Mean Temp Measure	Time	Estimate	95% CI	p-value	Interpretation
Air (streamside vs. upslope)	Maximum	February	0.87	0.78, 0.98	0.02	Air 13% cooler streamside
Air vs. substrates (streamside)			1.30	1.16, 1.46	< 0.0001	Air 30% warmer than substrates
Air vs. substrates (upslope)			1.49	1.33, 1.67	< 0.0001	Air 49% warmer than substrates
Air vs. small logs			1.78	1.52, 2.08	< 0.0001	Air 78% warmer than small logs
Air vs. large logs			1.42	1.21, 1.66	< 0.0001	Air 42% warmer than large logs
Large logs vs. small logs			1.26	1.07, 1.47	0.003	Large logs 26% warmer than small logs
Small logs vs. soil			0.63	0.54, 0.74	< 0.0001	Small logs 37% cooler than soil
Large logs vs. soil			0.80	0.68, 0.93	0.004	Large logs 20% cooler than soil
Air vs. substrates (streamside)	Minimum	February	0.47	0.38, 0.59	< 0.0001	Air 53% cooler than substrates
Air vs. substrates (upslope)			0.63	0.50, 0.78	0.0002	Air 37% cooler than substrates
Air vs. large logs			0.54	0.39, 0.73	0.0002	Air 46% cooler than large logs
Air vs. soil			0.38	0.28, 0.51	< 0.0001	Air 62% cooler than soil
Large logs vs. small logs			1.54	1.14, 2.09	0.004	Large logs 54% warmer than small logs
Small logs vs. soil			0.50	0.37, 0.67	< 0.0001	Small logs 50% cooler than soil
Overall temperatures (streamside vs. upslope)	Maximum	July	0.94	0.92, 0.96	0.0003	Overall temperatures 6% cooler streamside
Air vs. small logs			1.37	1.28, 1.47	< 0.0001	Air 37% warmer than small logs
Air vs. large logs			1.38	1.29, 1.48	< 0.0001	Air 38% warmer than large logs
Air vs. soil			1.59	1.48, 1.70	< 0.0001	Air 59% warmer than soil
Small logs vs. soil			1.13	1.06, 1.21	0.0004	Small logs 13% warmer than soil
Large logs vs. soil			1.14	1.07, 1.22	0.0002	Large logs 14% warmer than soil
Overall temperatures (streamside vs. upslope)	Minimum	July	0.96	0.94, 0.99	0.009	Overall temperatures 4% cooler streamside
Air vs. small logs			0.83	0.79, 0.88	< 0.0001	Air 17% cooler than small logs
Air vs. large logs			0.80	0.76, 0.85	< 0.0001	Air 20% cooler than large logs
Air vs. soil			0.93	0.88, 0.98	0.01	Air 7% cooler than soil
Small logs vs. soil			1.09	1.04, 1.16	0.001	Small logs 9% warmer than soil
Large logs vs. soil			1.16	1.09, 1.22	< 0.0001	Large logs 16% warmer than soil

Figure 3.10. Temperature profiles of large and small logs, soil and air for the 10-month sampling period at two distances from stream (0-5 m and 35-40 m) along a headwater stream in an unthinned stand in western Oregon. Average critical thermal maximum (~ 30 °C) for western plethodontid salamanders (Brattstrom, 1963) shown by dotted line.

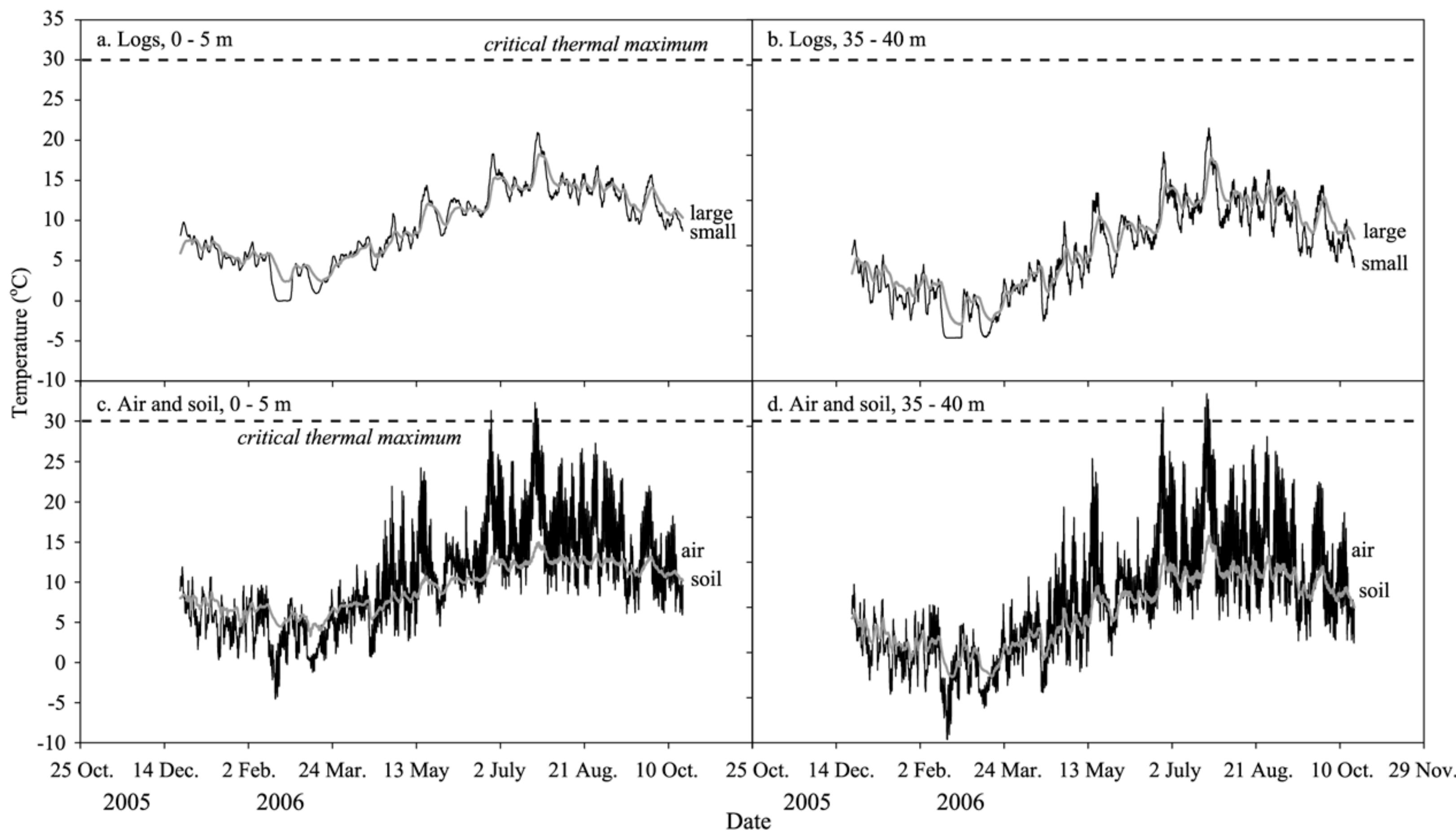


Figure 3.11. Temperature profiles of large and small logs, soil and air during February at two distances from stream (0-5 m and 35-40 m) along a headwater stream in an unthinned stand in western Oregon.

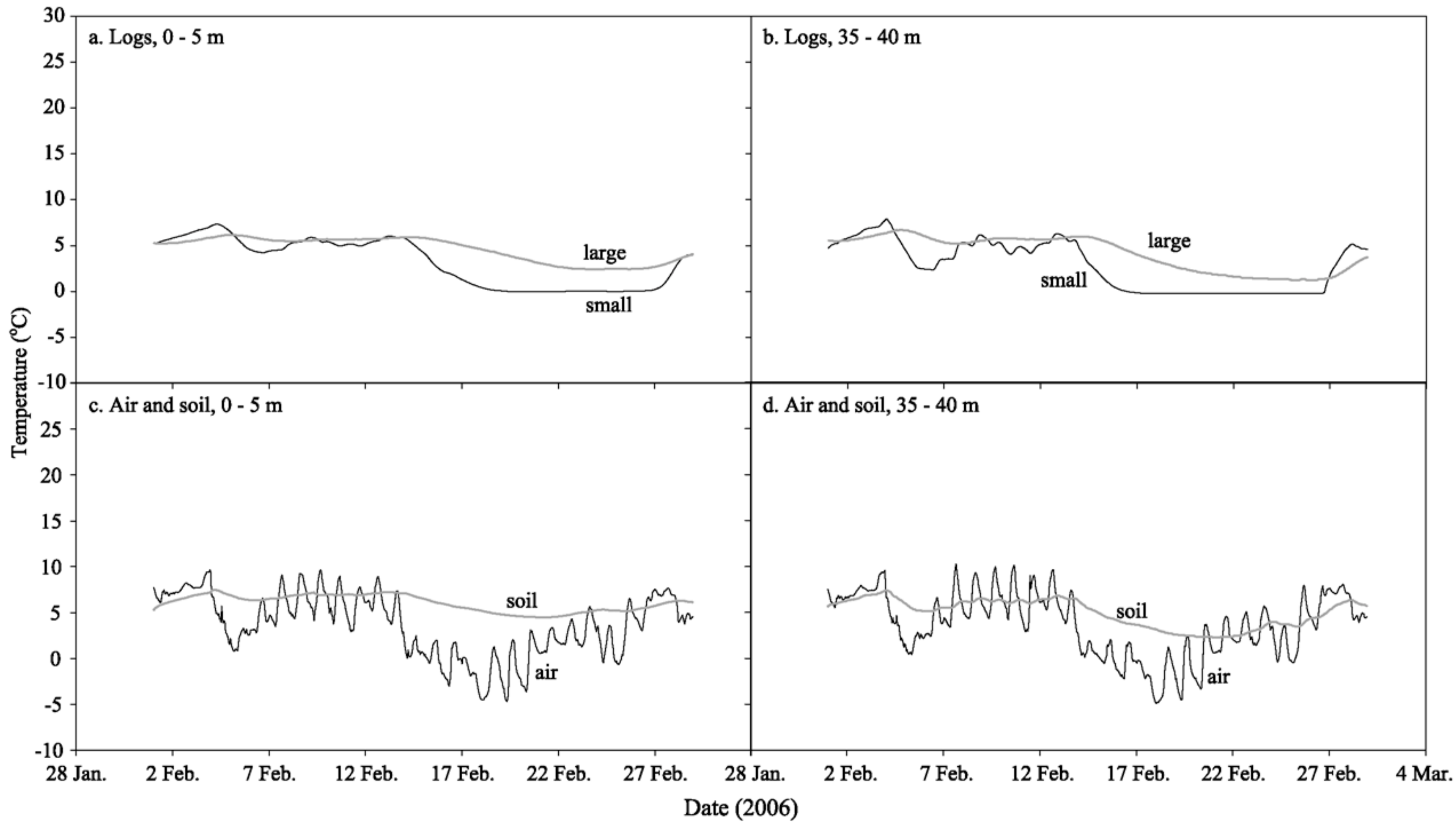
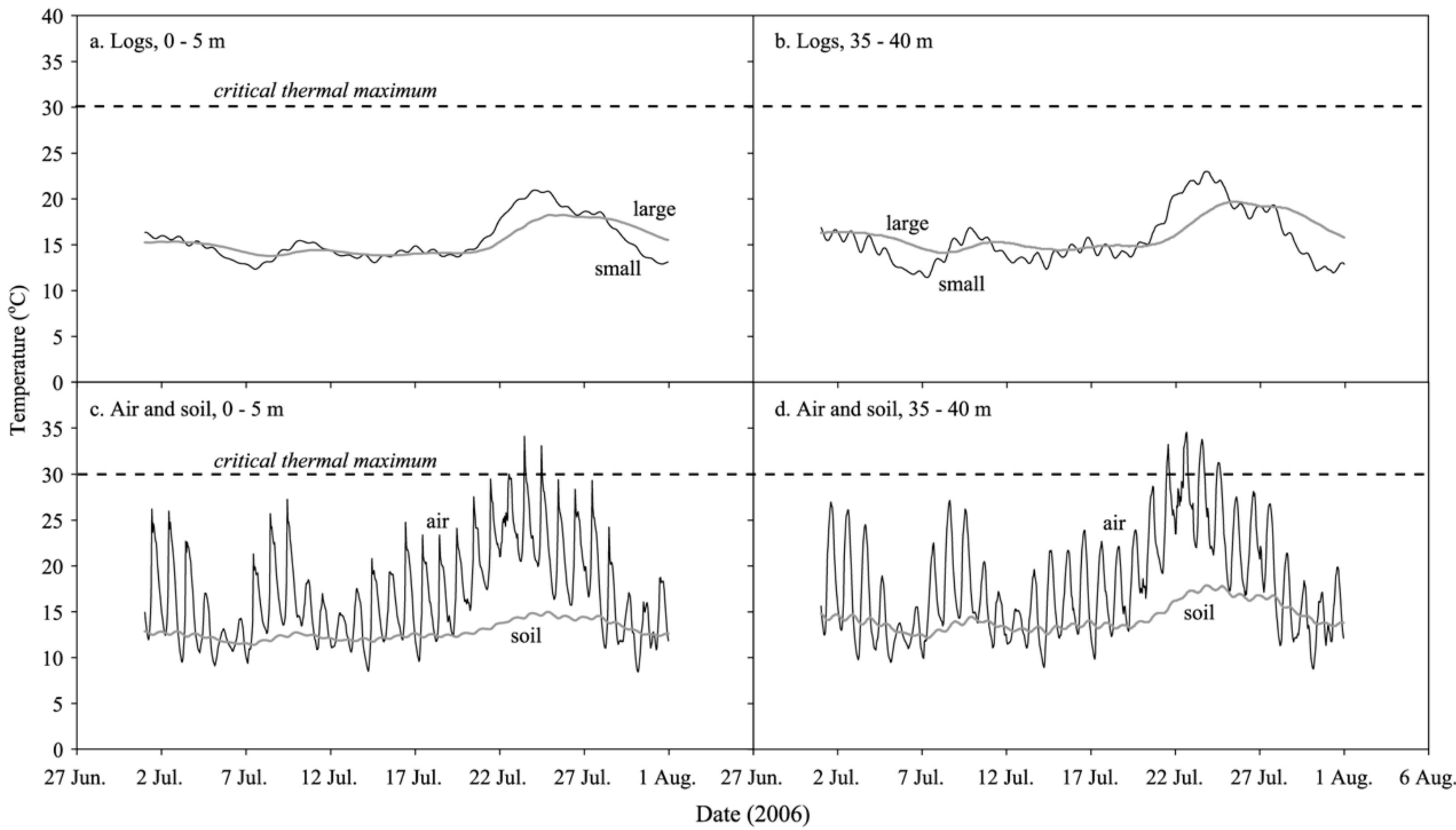


Figure 3.12. Temperature profiles of large and small logs, soil and air during July at two distances from stream (0-5 m and 35-40 m) along a headwater stream in an unthinned stand in western Oregon. Average critical thermal maximum (~ 30 °C) for western plethodontid salamanders (Brattstrom, 1963) shown by dotted line.



4. Discussion

Our results indicated that distance from stream had an influence on temperature patterns, as microclimate gradients extended from the stream to upslope, allowing for cooler conditions within riparian zones (Anderson et al., 2007). Soil, small logs and large logs along all three streams, both streamside and upslope, protected against maximum and minimum thermal extremes reached by air temperatures.

Diurnal temperature fluctuations were evident in small logs and in soil, whereas these fluctuations were either not present or much more subtle in large logs. Additionally, streamside substrates and large upslope logs exhibited more constant temperature profiles and less overall diel variation. These temperature profiles are likely to have biological relevance for the fauna reliant on these forest substrates, such as terrestrial amphibians and their prey.

The patterns seen during our study follow expectations, as direct solar radiation is a major source of air and soil heating (Anderson et al., 2007). The amount of solar radiation reaching the forest floor is primarily dependent on overstory canopy cover and the height and species making up the overstory (Brosofske et al., 1997). Air and substrate temperatures were consistently cooler in areas where canopy cover was retained (e.g., riparian buffers and unthinned upslope).

Maximum air temperatures above logs and soils in thinned upslopes during July were well above critical thermal maxima for western plethodontid salamanders. Log and soil temperatures failed to reach critical thermal maxima under all conditions, demonstrating the ability of soil and small and large logs to protect against thermal extremes that are harmful to plethodontid salamanders. This coincides with the findings of Blessing et al. (1999). They monitored one 0.5-m diameter log that held a clutch of Van Dyke's salamander (*Plethodon vandykei*) eggs and concluded that internal log temperatures were cooler and more constant than ambient air temperatures and that the log protected the eggs against thermal extremes.

Previous research on thermal regimes of downed wood is limited. However, in an old-growth stand in western Washington, logs provided protection from thermal extremes and water loss, where both large and small decay class 3 logs maintained cooler internal temperatures compared to surface temperatures (Marra and Edmonds, 1994). Our study suggest that log size is not a critical factor for buffering temperature extremes in thinned upslope areas, because smaller-diameter downed wood provided similar protections against thermal extremes as larger-diameter downed wood.

Although removal of overstory reduces moisture retention and thermal buffering capabilities of downed wood (Marra and Edmonds, 1996), recruitment of smaller diameter downed wood and partial retention of the canopy (e.g., moderate thinning) may be a viable plethodontid salamander habitat management alternative as the larger logs disappear from managed landscapes over time. Further considerations relative to small logs are: 1) their faster decay rates (Stone et al., 1998); and 2) they may desiccate at a faster rate relative larger logs, depending on their stage of decay (Triska and Cromack, 1980; Rose et al. 2001). Hence, increased downed wood recruitment frequencies of small logs may be needed to provide stable quantities of decaying log habitats on the forest floor for these taxa.

Canopy removal also may result in increased soil temperatures (e.g., clearcut; Chen et al., 1992). However, our study demonstrated that soils in thinned stands at 0.15 m subsurface may provide refugia from temperature extremes for some salamander species. This finding may explain the apparent broad microhabitat associations of some plethodontid salamanders, in which they are found within a variety of substrate types (e.g., western red-back salamander (Dumas, 1956; Ovaska and Gregory, 1989; Blaustein et al., 1995; Dupuis et al., 1995), ensatina (Blaustein et al., 1995; Rundio and Olson, 2007; also see Chapter 2). Other species are more habitat specific, such as the clouded salamander (*Aneides ferreus*), which is largely associated with downed wood (Bury and Corn, 1988; Corn and Bury, 1991; Butts and McComb, 2000; Bull, 2002).

Although cool, moist microhabitats are often available deep in burrows, logs, and rock crevices, the availability of any one of these microhabitats alone may be insufficient to support plethodontid viable salamander populations. As moisture decreases, these salamanders become limited in their foraging activities. Exposed microhabitats (e.g., leaf litter, soil surface, rock faces, vegetation) are where most foraging (and probably courtship) occurs (Feder, 1983). If surface conditions are dry, salamanders remain beneath cover objects (Cunningham, 1960; Jaeger, 1980), choosing progressively larger cover objects (e.g., downed wood) until conditions become too dry, finally retreating underground (Cunningham, 1960; Fraser 1976a, b;

Taub, 1961p; Heatwole, 1962), where little feeding occurs (Feder, 1983). Therefore, it is essential to maintain a variety of habitats within managed forests.

Further research is needed to gain a better understanding of the importance of temperature stability for plethodontid salamanders, and how reduced canopy cover affects temperature regimes of smaller- and larger-diameter downed wood and soils as suitable plethodontid salamander habitat relative to other ecological functions (e.g., prey availability, moisture regimes).

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CHAPTER 4

TERRESTRIAL SALAMANDERS AND THEIR HABITATS IN THINNED HEADWATER FORESTS ALONG STREAMS WITH ALTERNATIVE WIDTHS OF RIPARIAN BUFFERS IN WESTERN OREGON: SYNTHESIS AND MANAGEMENT IMPLICATIONS

As part of the USDI Bureau of Land Management's Density Management Study (Cissel et al., 2006) the primary objective of our research was to examine how upslope thinning (from densities of 600 tph to 200 tph), coupled with two types of riparian buffers (streamside-retention and variable-width), affected terrestrial headwater amphibians and their habitats. In Chapter 2, amphibian captures varied with distance from stream and captures were greatest within 15 m of stream edges. Additionally, percent cover of downed wood was greater streamside compared to upslope areas. Thinning reduced canopy cover, and was associated with percent moss cover in sub-sample units, having the greatest coverage along streams with the variable-width buffer treatment relative to the unthinned reference stand and streamside-retention buffer treatment. Neither thinning nor riparian buffer width had an effect amphibian captures.

Amphibian captures largely consisted of *Plethodon vehiculum*. This species is not restricted to riparian habitats, occurring in upland areas as well (Gomez and Anthony, 1996), and is thought to maintain relatively small home ranges (< 2.5 m; Ovaska, 1988). During our field study and enclosure experiment, *P. vehiculum* were strongly correlated with presence of rocky substrates. However, these salamanders also have an affinity towards downed wood (Aubry et al., 1988; Corn and Bury, 1991) as was evident at the Green Peak study site.

These salamanders seek out cool, moist microsites as a result of physiological requirements (e.g., breathing via gas exchange through their skin; Feder, 1983). The strongest gradient in microclimate from stream to upland occurs within 15 m of the stream center (Anderson et al., 2007), making riparian reserves potential refugia for this and other plethodontid species (Vesely and McComb, 2000).

Although our results from Chapter 2 did not indicate a difference in amphibian distribution among treatments, results from Chapter 3 suggested greater streamside air temperature maxima, as well as greater daily variation along the streamside-retention buffer stream (6 m buffer) relative to conditions experienced along the variable-width buffer stream (minimum of 15 m buffer). This suggests that implementation of variable-width buffers may help ameliorate potential negative impacts to salamanders from temperature extremes that stem from partial removal of overstory via upslope thinning. Increased solar radiation and air flow resulting in increased temperatures and decreased relative humidity as a consequence of loss of canopy cover (Chen et al., 1999; Moore et al., 2005) likely shorten periods of suitable conditions for surface activity by terrestrial salamanders (Welsh and Droege, 2001). During warmer, drier summer months, amphibians may seek out stable, cool, moist conditions found in downed wood (Bury and Corn, 1988; Corn and Bury, 1991; deMaynadier and Hunter, 1995; Maguire, 2002).

In our study, small- and large-diameter downed wood, both streamside and upslope, buffered temperature regimes against air temperature maxima known to be harmful to terrestrial amphibians. Although smaller-diameter downed wood experienced greater daily temperature fluctuations relative to larger-diameter downed wood, the temperature buffering ability of small logs suggest that they may be a viable habitat alternative as input of larger logs decreases on managed forests.

By implementing moderate thinning in upslope areas, coupled with riparian buffers that encompass habitat features relevant to ground-dwelling amphibians (e.g., rocky substrates, downed wood), land managers may be able to minimize negative impacts experienced by terrestrial headwater amphibians as a result of timber harvest in upslope areas.

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