Effectiveness of Riparian Management Zones in Providing Habitat for Wildlife: Resampling at the 10-year Post-treatment Interval

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Report

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Adaptive Management Program
Washington State Department of Natural Resources
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Forest Practices Adaptive Management Program

The Washington Forest Practices Board (FPB) has adopted an adaptive management program in concurrence with the Forests and Fish Report (FFR) and subsequent legislation. The purpose of this program is to:

> Provide science-based recommendations and technical information to assist the board in determining if and when it is necessary or advisable to adjust rules and guidance for aquatic resources to achieve resource goals and objectives. (Forest Practices Rules, WAC 222-12-045)

To provide the science needed to support adaptive management, the FPB made the Cooperative Monitoring, Evaluation and Research Committee (CMER) a participant in the program. The FPB empowered CMER to conduct research, effectiveness monitoring, and validation monitoring in accordance with guidelines recommended in the FFR.

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This technical report contains scientific information from research or monitoring studies that are designed to evaluate the effectiveness of the forest practices rules in achieving one or more of the Forest and Fish performance goals, resource objectives, and/or performance targets. This document was prepared for the Cooperative Monitoring, Evaluation and Research Committee (CMER) and was intended to inform and support the Forest and Fish Adaptive Management program. This project is part of the Wildlife Program within the Wildlife Rule Group, and was conducted under the oversight of the Landscape and Wildlife Advisory Group.

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Chapter 1 EXECUTIVE SUMMARY

Virgil C. Hawkes

This research evaluated the long-term (10-year) post-treatment effects of upland forest harvest and riparian management zone (RMZ) retention on the abundance and diversity of songbirds, small mammals, and terrestrial amphibians in managed forests of eastern and western Washington. The overarching goal of this research was to evaluate the post-treatment interval effects of riparian management on the species richness and diversity of non-game wildlife populations in eastern and western Washington. Related to the overall goal were five objectives:

1. To determine if the RMZ buffers retained along third- and fourth-order streams provided habitat attributes necessary to retain songbird, small mammal, and terrestrial amphibian diversity and abundance.
2. To determine if there was a difference in the abundance and/or diversity of songbirds, small mammals, and terrestrial amphibians in RMZ buffers of two different widths.
3. To determine if either buffer width maintained relative abundance and diversity of songbirds, small mammals, and terrestrial amphibians at levels commensurate to the unlogged control sites.
4. To determine if songbird, small mammal, and terrestrial amphibian relative abundance diversity changed in upland habitats as a result of timber harvest.
5. To identify how habitat structure and complexity varied over time relative to treatment type to determine if habitat structure could explain differences in relative abundance or species diversity.

To achieve these objectives, a replicated Before-After-Control-Impact (BACI) experimental study was conducted in the low-elevation, advanced second-growth, Douglas-fir–dominated forests of inland western Washington and the mid-elevation advanced second-growth, Douglas-fir–dominated forests of northeastern Washington. The experiment utilized three different treatments: (1) no harvest, (2) harvest according to 1988 state regulations, and (3) harvest according to a modified version of state regulations. The harvest was done during 1994. Data were collected during three sampling intervals: pre-harvest (1992 and 1993), first post-harvest period (1995 and 1996) and second post-harvest period (2003 and 2004). The pre-treatment and first post-treatment data (for western Washington) were provided by the Washington Department of Natural Resources; the second post-treatment data were collected by LGL Limited environmental research associates. Pre-treatment or first post-treatment data (i.e., 1995/1996) were collected but are no longer available for the eastern Washington study sites except for those data included in the original report (O'Connell et al. 2000).

Western Washington

Songbirds

A replicated, BACI experimental study was completed in the low-elevation, advanced second-growth, Douglas-fir–dominated forests of inland western Washington to investigate the effects on a selected group of songbirds of three different riparian treatments: (1) no harvest (control; *n* = 6 sites), (2) harvest according to state regulations (state buffer treatment; *n* = 6 sites), and (3) harvest according to a modified version of state regulations (modified buffer treatment; *n* = 5 sites). Sampling occurred along two types of transects: upland and riparian. Three intervals were sampled: (1) pre-harvest, (2) a first post-harvest interval at ~2-years post-harvest, and (3) a second post-harvest interval at ~10-years post-harvest.

Relative abundance of songbirds declined at all three treatment types in both riparian and upland habitats between the pre-harvest and first post-harvest sampling intervals, with the most notable significant declines at upland transects of state and modified buffer sites. Thereafter, relative abundance of breeding birds increased between the first and second post-harvest intervals, with bird abundance becoming significantly greater along the upland transects of state and modified buffer treatments. The abundance of songbirds along riparian transects did not differ significantly with respect to riparian buffer width within or among any sampling intervals.

Pre-harvest songbird alpha diversity (species richness) was similar among all treatments and transects. Significant differences within and between transect and treatment types arose during the second sampling interval and continued through the third sampling interval. The highest alpha diversity occurred along the upland transects of logged sites. The alpha diversity of songbirds along riparian transects did not differ significantly as a function of riparian buffer width within any sampling intervals, although there were a few differences among sampling intervals. The similarity of songbird communities within transect types and among sampling intervals was greatest at control sites throughout this study. Community similarity[1] was lowest in comparisons of upland communities at logged sites between and among sampling intervals irrespective of riparian treatment. Community similarity was greater at the modified buffer treatment than at the state buffer treatment during both post-harvest intervals.

Forestry-induced changes to habitat structure and forest succession altered the abundance, alpha diversity, and makeup of songbird communities by favoring an influx of new species at the expense of those that preferred the forest conditions that typified the treatments during the pre-harvest period. Songbird abundance and alpha diversity appeared to be closely tied to many of the habitat features measured in this study and are shown herein to be affected by forest harvesting and succession. The modified buffer treatment (i.e., comparatively wider riparian buffers) were more effective than the state buffer treatment at maintaining pre-harvest community structure in riparian areas. Patterns of songbird abundance and diversity observed in this study are in keeping with the known ecological requirements of the individual species and with accepted models of the linkages between alpha diversity and seral stage of coniferous forests. However, measures of species abundance and diversity are insufficient to measure the efficacy of riparian buffer width, as changes in some species can be offset by changes in others. Therefore, measures of community similarity are essential in this regard. Finally, a number of study limitations and management implications are presented.

Small Mammals

We assessed the efficacy of two different RMZ prescriptions to provide suitable habitat conditions for small mammals 2- and 10-years post-logging relative to unlogged control sites consisting of maturing second-growth Douglas-fir forests in western Washington. Our findings suggest that there is no global response to RMZ width by small mammals in western Washington. Rather, species showed individual responses that varied over time, treatments, and transects. There is some suggestion of effects of buffer width and logging treatments for some species, but the analysis

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[1] "Community similarity" is a mathematical description of the relationship between the number of species present in a community and the abundance of those species. A mathematical comparison of one community versus another generates a similarity coefficient, which can be used to assess community-level changes.

is confounded by patterns of natural population change over time (some local, some regional) and by likely differences among sites within treatment groups. For species that readily adapt to large-scale habitat alterations, such as *Peromyscus maniculatus* and *P. keeni*, riparian zones may not be necessary; but for others (e.g., *Microtus oregoni*), riparian zones provide important connectivity in the landscape as well as being areas from which such species can recolonize regenerating forest stands. Considering that the RMZ rules have evolved to embrace a site-specific approach, it seems only fitting that forest management in general adopt the same principals when considering biodiversity. Managing for biodiversity on a regional scale does not account for variation at the landscape or site level within the region, which could have dire consequences for species that are limited geographically, that occur at naturally low densities, or that are particularly sensitive to habitat change. As an alternative to the status quo, management that focuses on stand heterogeneity (as opposed to tree-species monocultures with reduced diversity) should produce landscapes with high biocomplexity and biodiversity at multiple scales, and therefore these landscapes should be resilient in the face of disturbance.

Terrestrial Amphibians

The efficacy of two different riparian buffer widths in providing habitat for terrestrial amphibians was assessed using a BACI approach. No global response by terrestrial amphibians to forest harvest or the retention of RMZs was found in western Washington. Rather, species showed individual responses that varied over time and between treatments and transects. The influence of site effects was evident for some species (e.g., Western Red-backed Salamander and Ensatina) while for others, some suggestion of treatment effects existed (e.g., Coastal Giant Salamander and Coastal Tailed Frog), but population changes at both local and regional scales confounded the analysis. A co-inertia analysis (COIA) revealed that amphibian-habitat associations did not change considerably over time and elucidated that the biological response to treatments did not differ from one another. For most species, a buffer width of 14 m was adequate in preserving pre-harvest occupancy and/or abundance patterns. However, for other species (e.g., Coastal Giant Salamander), a buffer width of 30 m was required. In even-aged, second-growth stands of Douglas-fir, a riparian buffer that approximates the conditions provided by the modified prescription would maintain relative abundance and diversity values of all terrestrial amphibians at levels commensurate with those detected prior to logging. The benefits of retaining riparian forest are identified and discussed in the context of maintaining biodiversity and conserving terrestrial amphibians in western Washington.

Eastern Washington

Songbirds

A replicated, experimental study was conducted in the mid-elevation, advanced (mid-seral) second-growth, conifer-dominated forests of northeastern Washington to investigate the efficacy of three different riparian treatments in providing habitat for songbirds: (1) no harvest, (2) upland harvest according to state regulations, and (3) upland harvest according to a modified version of state regulations.

Comparisons of relative abundance among all three sampling intervals for the most common songbird species did not reveal any significant differences over time within treatments and transects. However, differences in the relative abundance of all birds among treatments and transects were apparent during the second post-harvest sampling interval. Overall, forest harvesting led to increased bird abundance along both transect types of modified and state buffer sites. Relative abundance was lowest along riparian transects of the control sites and highest along upland transects of state buffer treatments. Differences in habitat structure resulting from forest harvesting, as elucidated by co-inertia analysis, are believed to be largely responsible for the observed differences in bird abundance, though site effects cannot be discounted entirely.

Both riparian and upland habitats contain important habitats for bird species. For 12 species, riparian habitats (transects) were used by >50% of the individuals within those species. Conversely, 30 species included individuals that used riparian transects for <50% of the records. Upland transects of logged sites supported the greatest alpha diversity during the second post-harvest interval. Increased alpha diversity in logged upland areas likely contributed to increased alpha diversity in the adjacent riparian buffers of modified and state buffer treatment sites. Relatively low values and small ranges in indices of community similarity (Morisita's *C*) for comparisons between the riparian and upland transects suggests that the effects of any differences between treatment types on avian community structure were modest during the second post-harvest sampling interval. That is, community similarity between transect types appeared to be independent of treatment type. Relatively higher similarity values among comparisons of communities along riparian transects during the second post-harvest sampling interval suggested that riparian bird communities were not affected greatly by harvesting.

In conclusion, at upland locations forestry-induced changes to habitat structure and forest succession altered the abundance and alpha diversity of songbird communities, but no notable differences in key parameters of bird ecology between state and modified buffer prescriptions were revealed. This suggests that, despite undergoing greater timber removal, the state buffer treatment is no less capable than the modified buffer treatment of preserving the relative abundance, alpha diversity, and community structure of songbirds 10 years after logging in the riparian areas of coniferous forests of eastern Washington that are comparable to those examined in this study. Finally, a number of study limitations and management implications are presented.

Small Mammals

This study was designed to evaluate the abundance and diversity of small mammals in riparian zones by comparing two different buffer configurations to unlogged control sites. The buffer configurations were based on the RMZ regulations mandated at the time (Washington State Forest Practices Board 1988) and on modifications to the 1988 regulations. Small-mammal response to the retention of RMZs in eastern Washington was highly species-specific, with no global response shown by small mammals as a group. Confounding the interpretation of the results was the fact that the buffer treatments were insufficiently different from each other to allow for any meaningful comparison between the two. As a result, we were unable to draw any conclusions regarding the effectiveness of a narrower, uniform-width buffer compared to a wider, variable-width (modified) buffer in maintaining pre-harvest small-mammal populations on a managed landscape. It was possible, however, to detect some significant changes in populations of several species, specifically the chipmunks (*Tamias* spp.), within the upland habitats that had been selectively logged. A lack of significant changes in the relative abundance of small mammals as a whole within the RMZs suggests that either buffer configuration may be adequate for maintaining small-mammal populations at the levels present in typical second-growth montane coniferous forests of northeastern Washington. At a minimum, a RMZ approximating the state buffer prescription was adequate in maintaining the relative abundance and richness of small mammals in eastern Washington. However, the mid-seral stands that were used as the control condition in this study may represent some of the lowest levels of diversity and abundance of species of any stage during the life of the forest.

Management Implications

Western Washington

Extensive forestry management in the Pacific Northwest has historically included clearcutting, prescribed fire, regeneration by seed trees, protection from forest fires, and salvage of timber killed by windthrow or disease (Carey and Harrington 2001). In the last 20 years, planting of Douglas-fir has increased, with most forest stands >40 years old originating from seed. Current management practices include the retention of seed trees and coarse woody debris and the planting of seedlings. Implicit in the shift to ecosystem management has been the retention of RMZs around perennial fish- and non–fish-bearing streams to protect fish and water quality.

During the past 10–15 years, increased attention has focused on defining and delineating riparian areas. McComb et al. (1993) reported that a buffer width of 50 m adjacent to second- and third-order streams in Oregon would provide marginal habitat and that buffers should be >50 m wide to provide linkages between mature forests. Vesely and McComb (2002) estimated that RMZs 43 m wide would support total salamander abundance and that buffers of 47 m would support species richness at levels similar to unlogged sites. Recently, Crawford and Semlitsch (2007) found that a buffer width of 27 m would encompass 95% of the salamander assemblage and that an additional 50 m would ameliorate edge effects. They recommended that a total buffer of 92.6 m be used in southern Appalachian streams to enable salamander biodiversity to persist. We found that a minimum buffer width of 30 m around third- and fourth-order streams in western Washington would maintain species richness and relative abundance values at levels similar to those recorded prior to logging. This buffer width is suitable for small mammals, most terrestrial amphibians, and songbirds.

Despite these recommendations for buffer width, the minimum width of a RMZ continues to be a geographic issue related to the implementation of region-wide management plans that are too general to account for local variability in species richness and diversity. Yet, the research on the benefits of RMZs is clear (deMaynadier and Hunter 1995; Vesely and McComb 2002; Cushman 2006), and most jurisdictions in Canada and the United Sates require the retention of riparian forest adjacent to upland logged habitats (Lee et al. 2004). The problem faced by land managers is that far too often management plans become specific to one group of organisms and do not use biodiversity preservation as the basis for management.

Although RMZs retain structural heterogeneity and complexity, the production of homogenous, simplified, even-aged stands of second-growth Douglas-fir forests in upland habitats may, over the long term, lead to localized extirpations of species like the Coastal Giant Salamander. Therefore, it is important to consider not only the geographic relationship to riparian management, which will provide an indication of species richness and abundance, but also to consider the frequency of upland rotation and the type of prescription applied to the upland forest. For example, Perkins and Hunter (2006) recommended that riparian management be a two-tiered approach, with no-cut zones in the riparian forest and a partial-cut zone in the adjacent upland. This approach would protect the riparian forest from blowdown and minimize edge effects (deMayandier and Hunter 1998).

For species that decline in abundance because of logging, management strategies to mitigate these declines have been proposed. Carey and Harrington (2001) concluded that management-induced homogeneity and simplification (i.e., reduction of complexity) posed a real danger of small-mammal communities becoming non-supportive of predator populations. They proposed that active management for habitat complexity on long rotations (>90 years) may be necessary to conserve the diversity attributed to maturing forests. Similarly, because terrestrial amphibian communities are inherently linked to many forest ecosystem processes, the development of biodiversity management strategies must be considered at a broader scale and should not be based on single-species conservation. The introduction of heterogeneity into homogenous, even-aged stands has been shown to have positive effects on biodiversity and ecosystem function in the short term (<5 years; Carey 2003), and longer-term studies are required to determine if the positive effects are long lasting.

Since this study's inception in 1991, there have been changes to the Washington Forest Practices Rules and Regulations that have implications for our findings. The Forest Practices Rules of 1988 required a riparian buffer that met the state configuration (i.e., not less than 7.5 m from the ordinary high water mark) for both fish-bearing (Type 3) and non–fish-bearing (Type 4) perennial streams. The development of the modified riparian prescription was an alterative to the state buffer that provided for greater structural complexity, heterogeneity, and ultimately greater diversity for both flora and fauna. Lee et al. (2004) identified the current trend in riparian management prescriptions as tending towards site-specific buffers that vary among broadly similar harvest areas to within a single harvest area. The Forest Practices Rules of 2001 seem to fit this model and, aside from changing how watercourses are defined, are site-specific with many options. Lee et al. (2004) suggest that the primary benefit of this type of prescription is the identification of well-defined criteria to delineate the RMZ. These criteria are specifically defined by the modifying factors selected by jurisdictions, and buffers are applied in a predictable response to these criteria. The caveat to tailor-made buffers is the greater complexity in guidelines. For example, Washington now describes 3 zones of management for Type F streams (Type 3): (1) the core, (2) inner, and (3) outer zones, each with various levels of tree removal, thinning, hardwood conversion, and other activities permitted within these zones depending on the site class of land, the management harvest option, and the bankfull width of the stream. In general, the site class will form the RMZ prescription. It appears that riparian management in western Washington has become (necessarily) site-specific. This implies that applying a one-size-fits-all approach to riparian management has been set aside in favor of a more intensive, site-specific approach, which is consistent with Lee et al. (2004). For Type Np streams, or perennial non–fish-bearing streams (Type 4), the RMZ rules are slightly less complicated; however, there is ample room for site-specific management. Considering that the RMZ rules have evolved to embrace a site-specific approach, it seems only fitting that forest management in general adopt the same principals when considering biodiversity. Managing for biodiversity on a regional scale does not account for variation at the landscape or site level within the region, which could have dire consequences for species that are limited geographically, that occur at naturally low densities, or that are particularly sensitive to habitat change. As an alternative to the status quo, management that focuses on stand heterogeneity (as opposed to tree-species monocultures with reduced diversity) should produce landscapes with high biocomplexity and biodiversity at multiple scales, and therefore these landscapes should be resilient in the face of disturbance (Holling 2001).

Although we did not assess the efficacy of RMZs from a community ecology perspective, management strategies that are implemented at regional scales should be premised on the preservation of biodiversity and not predicated on single-species or single-wildlife group preservation. Further, management strategies should consider habitat connectivity, and all components of the landscape should be managed in concert—not in isolation. Establishing RMZs as a means to preserve non-game wildlife populations may be adequate in even-aged, second-growth forests of western and eastern Washington, where diversity is typically low. However, the efficacy of RMZs as broad-scale, one-size-fits-all management tools needs to be considered for all groups of wildlife, not only those with relatively small home ranges or specific habitat requirements. For example, the habitat needs of deer, elk, bears, marten, and fisher must also be incorporated into landscape management strategies.

Eastern Washington

The type of upland logging applied in eastern Washington produced habitats with high structural complexity relative to the pre-harvest condition, where most stands were even-aged, second-growth forests of Douglas-fir. Establishing partial retention management strategies in upland forests has been promoted as a means to balance the economic and wildlife values of forested habitats in eastern Washington. In this region, it is apparent that the structural complexity and habitat heterogeneity created by partial logging benefited populations of songbirds and small mammals during the second post-harvest sampling interval (i.e., 10 years after logging). In some cases, the relative abundance of certain species increased significantly relative to the control sites, further supporting the benefits of partial retention logging in upland coniferous forests of northeastern Washington. As in western Washington, riparian management has become (necessarily) site-specific. The ability to manage riparian habitats based on the conditions of the site, coupled with the maintenance of partial forest cover in upland forests will create semi-forested habitats that not only maintain richness and relative abundance of non-game wildlife, but that are likely to promote species richness over time, leading to increased biodiversity.

Although we did not assess the efficacy of RMZs from a community ecology perspective, management strategies that are implemented at regional scales should be premised on the preservation of biodiversity and not predicated on single-species or single-wildlife group (e.g., songbirds) preservation. Further, management strategies should consider habitat connectivity, and all components of the landscape should be managed in concert—not in isolation. Establishing RMZs as a means to preserve non-game wildlife populations may be adequate in even-aged, second-growth forests of western and eastern Washington, where diversity is typically low. However, the efficacy of RMZs as broad-scale, one-size-fits-all management tools needs to be considered for all biota, not only those considered in this study. For example, the habitat needs of larger birds such as hawks and owls, and mammals such as deer, elk, bears, and marten, must also be incorporated into landscape management strategies. Equally important are the invertebrate populations that are responsible for decomposition and nutrient recycling, pollination, and a host of other ecological functions as well as being a food source for insectivorous plants and animals.

Riparian Buffer Width

Our ability to determine if the state (narrow) RMZ or the modified (wider) RMZ was better able to provide the habitat conditions necessary to maintain species richness and relative abundance of songbirds, small mammals, and terrestrial amphibians in western Washington was confounded by several factors. In particular, the species-specific response to treatment, transect, and sampling interval affected our ability to identify if one buffer was more effective than the other in providing habitat for wildlife at the 10-year post-treatment interval. For some species groups (e.g., songbirds), species richness increased after logging, regardless of buffer width. This increase was associated with a change in habitat condition caused by logging and not by the retention of a narrow or wide riparian buffer. Similarly, our findings provide no evidence that wider RMZs supported a greater abundance or more diverse community of small mammals than did the narrow RMZs. The same can be said for most species of terrestrial amphibians; however, as indicated above, a riparian buffer width that approximates the modified prescription of 30 m should be implemented.

Because we did not have access to the pre-harvest and first post-harvest data for eastern Washington, we were unable to determine if a narrow buffer was any different from a wider buffer in providing habitat for wildlife 10 years after logging. We did find that both the state and modified buffer configurations had similar species richness and relative abundance values for small mammals and songbirds. At both buffer treatments, the relative abundance and richness values obtained were higher than at the control sites. However, control sites were ~80-year-old second-growth forests that are typically at their lowest ebb in species abundance and diversity. This suggests that logging upland forests using a partial retention strategy increases habitat suitability for small mammals and songbirds—compared to ~80-year-old closed-canopy, single-species, second-growth forests—by creating heterogeneous, uneven aged stands.

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Chapter 2 RIPARIAN HABITATS AND RIPARIAN BUFFERS

James D. Fenneman and Virgil C. Hawkes

Western Washington riparian management zones (RMZs) have been required, at least in some form, since 1976. Between 1976 and 1988, the Washington Forest Practices Rules and Regulations required “streamside management zones” (SMZ) which were intended to provide “stream bank integrity and temperature control” (Washington State Forest Practices Board 1976, 1982). Streamside management zone regulations prescribed riparian buffers 7.5–15 m wide with all nonmerchantable vegetation and “sufficient merchantable timber, if any, necessary to retain 50–75 percent of the summer mid-day shade of the water surface” left within the SMZ (Washington State Forest Practices Board 1976). In 1988, the RMZ regulations were strengthened considerably, requiring riparian buffers up to 30 m wide (Washington State Forest Practices Board 1988). The change in regulations was a result of the Timber, Fish, and Wildlife (TFW) Agreement reached in 1987 among representatives of the Washington State tribes, forest-products industry, environmental community, and natural-resource agencies. The agreement sought to resolve conflicts between these diverse groups by recognizing the common goal of preserving natural resources and at the same time maintaining a viable timber industry.

The term “riparian zone” refers both to the lands adjacent to watercourses (Hall 1998), as well as to the biotic communities living in these habitats, and can be extended vertically into the forest canopy and belowground through the watery, microbe-rich soil (Naiman et al. 2000). Thus, both the biotic and abiotic components of the landscape that are within the influence of a water body can be considered part of the riparian zone. The boundaries of this zone, though sometimes difficult to precisely determine, are typically delineated by changes in soil conditions, vegetation, and hydrology that reflect this aquatic-terrestrial interaction (Naiman and Décamps 1997). Riparian zones are generally characterized by increased primary productivity, higher levels of energy transport, and, often, more frequent natural disturbance (largely in the form of floods) than upland areas (O’Connell et al. 2000).

Riparian zones represent the interface between the purely aquatic environment and the terrestrial environment and, as such, have a set of characteristics uniquely defined by spatial and temporal scales and by the strength of interactions between the adjacent ecological communities (Holland et al. 1991). Riparian zones are influenced by both the aquatic and terrestrial realms. Riparian zones in turn influence these habitats in a variety of ways (O’Connell et al. 2000), including the flow of resources and energy, biological interactions between species, and acting as temporary refuge for species more typically associated with either of the adjoining environments (Naiman and Décamps 1997).

Ecotonal systems, such as riparian habitats, tend to have relatively high biodiversity because they bring together components of adjoining ecosystems as well as contribute components that are specifically adapted for the interface habitats themselves (O’Connell et al. 2000). Although most habitat interfaces are ecologically important, natural riparian zones are considered to represent some of the most diverse, dynamic, and complex biophysical habitats on the terrestrial portion of the planet (Naiman and Décamps 1997). In fact, the riparian forests of the Pacific Coastal Ecoregion of North America are considered the most diverse floristically and structurally (Pollock 1998). This high species diversity, particularly in fluvial systems, is the result of a wide variety of factors such as disturbances caused by floods, spatial heterogeneity created by debris flows, lateral river migration, large woody debris (LWD) input, animal activities, site productivity, landform, and variation in local climate as a function of elevation (Gregory et al. 1991; Naiman et al. 1993; Sagers and Lyon 1997; Naiman et al. 2000; Lee et al. 2004).

Riparian habitats have an ecological importance that far outweighs their abundance on the landscape. Riparian habitats tend to be patchily distributed, even in the wet coastal forests of the Pacific Northwest, occurring in a linear fashion along the edges of streams, lakes, and other wetlands within a matrix of upland habitat (Chappell et al. 2001). The size of the riparian area and the extent of interaction between the land and the water vary with the size of the stream (Bilby 1988, cited in Bolton and Shellberg 2001) and the shape of the terrain. In small, upland streams where the stream flow is typically low, the forest or other adjacent vegetation tends to dominate the stream. Along larger, wider rivers, however, there is a more extensive interaction between the water and forest (Bolton and Shellberg 2001).

Riparian vegetation also has profound impacts on the adjacent aquatic systems. Streamside vegetation filters sediment and pollutants thereby enhancing water quality, stabilizes streambanks by holding soil among interwoven root systems, regulates stream temperatures based on canopy cover, and provides a continual source of woody debris to the stream channel (BC Ministry of Forests 1995). The input of large organic debris, though formerly considered detrimental to stream health (Triska and Cromack 1980), is now recognized as an integral link between terrestrial and aquatic ecosystems (O'Connell et al. 2000). Large woody debris helps define stream structure by retaining gravel and sediment, forming pools, and creating waterfalls (O'Connell et al. 2000). The majority of fish food organisms come from overhanging vegetation and bordering trees, while leaves and twigs that fall into streams are the primary nutrient source that drives aquatic ecosystems (BC Ministry of Forests 1995). The extent and depth of these interactions indicates that maintenance of riparian areas is critical for preserving the health of the aquatic systems along which they occur (Bolton and Shellberg 2001).

Vegetation of the Riparian Zone

Riparian vegetation is determined by the hydrological, topographical, substrate, and microclimatic features of the site. As a result, the vegetation of riparian areas is variable and encompasses a spectrum of aquatic-influenced terrestrial habitats, from grassy or brushy riverbanks to mature bottomland and floodplain forests (Guard 1995). Most riparian areas are characterized by a dynamic, disclimax system with frequent natural disturbances that prevent climax vegetation from becoming permanently established (Naiman et al. 2000) and are often represented by a patchwork of habitats of different ages (Bolton and Shellberg 2001). The mosaic of habitats resulting from an active disturbance regime produces a relatively high diversity of vegetation species compared with adjacent upland areas (Bolton and Shellberg 2001), with broad riparian zones hosting more species than narrow, steep-sided riparian areas (O'Connell et al. 2000).

Disturbance frequency and moisture gradients play key roles in determining the vegetative composition of riparian areas (Naiman et al. 2000), with distinctive communities occupying floodplain, low-terrace, and high-terrace landforms along smaller channels (Rot et al. 2000). Vegetation structure of riparian habitats is typified either by tall deciduous broadleaf shrubland, woodland, or forest (deciduous, coniferous, or mixed), or some mosaic of these habitat types (Chappell et al. 2001). Species such as red alder (*Alnus rubra*), black cottonwood (*Populus balsamifera* ssp. *trichocarpa*), western redcedar (*Thuja plicata*), and western hemlock (*Tsuga heterophylla*) are typical tree species of riparian zones in the Pacific Northwest. The understory is typically composed of a diverse assemblage of shrubs, forbs, grasses, sedges, rushes, and ferns. Species diversity is dependent on the structural age, frequency of disturbance, and substrate type of the habitat (Chappell et al. 2001). Riparian forests tend to have greater structural diversity than upland habitats due to abundant large woody debris in late-seral forests and adjacent stream channels (O'Connell et al. 2000; Chappell et al. 2001).

Disturbance in the Riparian Zone

Most riparian ecosystems, especially those along rivers and streams, are the product of continual disturbance, either in the form of fluvial processes (flooding, debris flows, erosion and subsequent sediment deposition) or nonfluvial processes (fire, wind, plant disease, herbivory) (Gregory et al. 1991). Moreover, most riparian systems are subject to multiple disturbance processes that create a mosaic of different microsites within the zone, allowing a diverse biological community to become established (Bolton and Shellberg 2001). However, the riparian zone itself is not uniformly susceptible to the same disturbance processes; fluvial processes dominate in areas closer to the actual water body, and most non-fluvial processes occur throughout the zone or, in the case of fire, occur more frequently in areas away from open water (Agee 1988). Gregory et al. (1991) suggested that the variety of microsites that result from these complex, high-frequency disturbance regimes along river valleys lead to greater plant-species diversity compared to upland areas. Frequent natural disturbance, and the mix of habitats that results, is considered the key to a healthy riparian system (Bolton and Shellberg 2001).

Most riparian communities are adapted to some degree of natural disturbance, or they occur during a specific time in the successional sequence of forest regrowth after a major disturbance (Chappell et al. 2001). Flooding is the primary form of natural disturbance in most riparian areas, particularly along rivers and streams, although debris flows can also be important, albeit infrequent, in some systems (Chappell et al. 2001). In some riparian communities, such as cottonwood bottomlands, removal of a regular flooding regime can result in the succession of the habitat and loss of the disclimax characteristics that typify it (Lytle and Merritt 2004). Fire, which is often a dominant form of natural disturbance in upland forests, is generally of less importance in riparian systems due to the increased site moisture and more protected position on the landscape of those areas (Chappell et al. 2001). However, the effects of fires on upland areas can be extended into the fluvial system through the resulting increased erosion and sedimentation, increased runoff and flooding, and the input of large woody debris (Chappell et al. 2001). Other forms of natural disturbance, including ungulate herbivory, wind, and flooding due to the activity of beavers (*Castor canadensis*), can also have a major effect on the structure and composition of riparian communities in some areas (Chappell et al. 2001).

Although they are often highly dynamic and adapted to frequent disturbance, riparian systems have the potential to be overwhelmed by the increased number, size, or duration of disturbances that accompany human activities on the landscape (Bolton and Shellberg 2001). Several factors make riparian areas particularly susceptible to human-induced disturbances, including the fact that humans concentrate much of their activity on the landscape into riparian zones; the relatively limited extent of riparian areas compared to upland areas; the extensive riparian-upland interface created by the long, thin shape of most riparian areas; and the unique and often sensitive flora that is often present in riparian areas (Oakley et al. 1985). These factors can have implications for both the terrestrial and aquatic components of the riparian system because altering the ecology of the site can affect the health of the riparian system (Broadmeadow and Nisbet 2004).

Human-induced disturbances on the landscape are varied and include timber harvest, livestock grazing, agriculture, road building, development, impoundment and channelization, and both consumptive and non-consumptive recreation (O'Connell et al. 2000). Timber harvest and its associated activities, however, are among the most prevalent forms of anthropogenic disturbance in forested areas of the Pacific Northwest and can have severe implications for riparian systems. Management of upland areas of a watershed for timber production and harvest can alter the processes that deliver water, wood, and sediment to the riparian area (Bolton and Shellberg 2001). Although opinions vary as to the severity of the effects of these factors on the riparian environment (O'Connell et al. 2000), most studies have shown that effects such as increased water temperatures, increased erosion and sedimentation, overload of large woody debris in the stream, increased water flows, and increased rates of windthrow among remaining trees generally accompany forest-harvest activities within a watershed, particularly when these activities encroach upon the riparian zone itself (Bolton and Shellberg 2001). Additionally, studies have suggested that clearcutting may have greater negative impacts on riparian habitats than single-tree selection (Oakley et al. 1985, cited in O'Connell et al. 2000). The retention of buffer strips of riparian vegetation along watercourses has become an important means of conserving biodiversity and habitat conditions in managed forests. However, many questions still exist regarding the most effective method of applying this mitigation strategy and regarding the use of buffers by terrestrial species (Cockle and Richardson 2002).

Wildlife Use of Riparian Areas

Riparian areas often support the highest diversity and abundance of wildlife in temperate zones because of their diversity of natural features, because they can support species uniquely adapted to riparian and early successional habitats, and because they can support upland species that might depend on riparian habitats for part of their life cycle (Gyug 2000). In the Pacific Northwest, riparian zones may occupy as little as 0.5–2.0% of the landscape yet contain more wildlife species than the surrounding uplands (Kauffman et al. 2001). Wetlands and riparian habitats are considered to represent high value to wildlife due to the presence of water, high structural diversity and cover, abundant forage, high prey densities, and unique habitat features as well as their overall rarity on the landscape (MacKenzie and Moran 2004). Wildlife use of riparian areas differs between the major groups of organisms, and thus riparian habitats serve different functions.

Amphibians and Reptiles

Reptiles and amphibians form an integral component in any riparian environment, including both wetland-associated systems and fluvial-associated systems (Semlitsch and Bodie 2003), yet have received relatively little attention from researchers studying the ecology of these systems (Kauffman et al. 2001). Kauffman et al. (2001) indicate that, of 30 species of reptiles and amphibians inhabiting riparian and wetland habitats in western Washington and Oregon, 26 are closely to moderately associated with such habitats. The herpetological fauna of many regions forms a large component of the terrestrial biomass in riparian habitats and, as such, can contribute significantly to the trophic transfer and energy flow in these ecosystems (Petranka and Murray 2001).

Although some reptiles such as turtles and snakes (especially garter snakes, *Thamnophis* spp.) regularly occur in aquatic and riparian habitats, amphibians, due to their requirement for moist or wet conditions throughout their lives, are especially dependent on these environments in temperate regions. Most amphibian species in the Pacific Northwest, with the exception of the plethodontid (lungless) salamanders, require the presence of aquatic habitats for breeding (MacKenzie and Moran 2004); thus the high quality of the aquatic habitat that is afforded by a healthy riparian zone is critically important for these species. Additionally, most species of amphibians use the moist terrestrial habitats adjacent to watercourses for foraging, at least occasionally, and some are found in this habitat at all times (Semlitsch and Bodie 2003; MacKenzie and Moran 2004). The changes in light, humidity, temperature, structural composition, and other microclimatic features in the riparian zone that accompany forest removal in adjacent areas may affect some amphibian populations because some species are particularly sensitive to environmental alterations within their habitat (Kauffman et al. 2001).

Birds

Riparian habitats provide breeding habitat for more species of birds than any other vegetation type in North America despite being restricted to only 1–2% of the western landscape. The disproportionate abundance and diversity of birds in riparian and wetland ecosystems have been reported in every major biome and in every western state (Kauffman et al. 2001). The high bird diversity in riparian areas is largely due to the juxtaposition of habitat requirements (food, cover, water) as well as the many niches that are provided due to the complexity, age range, number of plant and animal species, and structure of the vegetation (Knight 1988). Birds use riparian and wetland areas not only for breeding but also for foraging, cover, as movement corridors and migrational stopovers. The abundance of migrating neotropical songbirds may be up to 10 times greater in riparian zones than in surrounding upland forests, while the overall species diversity may be as great as 14 times higher (Kauffman et al. 2001). The abundance of resources afforded by riparian vegetation can be of critical importance to migrating birds. For example, migrating Wilson's Warblers (*Wilsonia pusilla*) have been shown to have higher rates of fat deposition and shorter stopover periods, both of which are important for successful migration, when using native riparian willow habitat compared to agricultural or field edge habitats (Yong et al. 1998).

Mammals

As with reptiles, amphibians, and birds, riparian zones are disproportionately important to mammals because of the abundance of food, presence of water, thermal protection, and high structural and landscape diversity (Bolton and Shellberg 2001). As well, the linear shape and the resulting edge of riparian habitat allows mammal species access to two or more habitats within close proximity enabling them to fulfill more than one ecological need within a limited area (Kauffman et al. 2001). This linear shape also permits mammals to use riparian strips as corridors when moving between different geographic areas, particularly where the adjacent upland areas offer little cover (Kauffman et al. 2001). Approximately 65% of the mammal species in Oregon and Washington use riparian areas for some aspect of their life history, with small mammals, bats, mustelids, and most ungulates being particularly tied to riparian areas in these states (Kauffman et al. 2001).

More than any other wildlife group in the Pacific Northwest, mammals have the ability to modify the riparian environment either intentionally, as with beavers, or unintentionally through herbivory, predation, and trampling (Kauffman et al. 2001). These levels of disturbance often help to increase habitat complexity by altering the composition and structure of riparian vegetation, although at extreme levels these forms of disturbance can be detrimental to the ecology of the riparian community (Kauffman et al. 2001). The presence of a healthy ecosystem can help to mitigate the effects of such pressures on the environment, but in habitats that have been previously degraded by human-caused disturbance those negative impacts can become particularly detrimental. Preservation of high-quality habitat is paramount for preserving the integrity of the entire ecological system of the riparian zone.

Riparian Buffers

Retention of riparian habitat in the form of buffer zones along watercourses is widely used to preserve ecosystem integrity in managed forests, especially as it relates to the aquatic environment. Kauffman et al. (2001) suggested that logging riparian forests is likely deleterious to more habitats and wildlife species than logging in any other forest community or landscape in the Pacific

Northwest. Buffers are designed to prevent erosion, filter runoff of contaminants, and protect or enhance fish habitat (Willson and Dorcas 2002; Broadmeadow and Nisbet 2004; Lee et al. 2004). By proxy, these buffers benefit terrestrial ecosystems as well. Erosion control and runoff filtration are of particular importance in smaller, higher-order streams since these streams receive runoff directly from rain and are therefore often the first aquatic habitats to be affected by development and water-borne pollution (Hoffman et al. 1995). These effects can subsequently be passed throughout the entire fluvial system.

The repeated disturbances and abundant edge habitat present in riparian zones historically provided refugia for species of early successional ecosystems in a matrix of old-growth forest, allowing these species to rapidly invade upland areas following natural disturbances. Today, however, as the forested landscape is logged and regenerated, these same zones, in the form of buffers, act as refugia for species more typically associated with mature and old growth forests (O'Connell et al. 2000). Without the retention of buffers along the riparian corridors, some mature forest-dependent species could be lost or depleted across the landscape, reducing the ability of those species to recolonize the upland forests as they mature. Preserving linear buffers through a landscape of logged or regenerating forests allows for the movement of these species among and between patches of remaining mature forests. This motility enables increased levels of gene flow among populations of species with reduced dispersal capabilities or narrow habitat requirements (O'Connell et al. 2000).

Although the retention of riparian management zones was originally intended for preservation of the aquatic environment, riparian buffers are now being designed to incorporate and maintain a wider variety of ecological functions. Several new trends have recently emerged in the field of riparian management, such as placing emphasis on ecological function and natural riparian forest pattern, adoption of a landscape perspective of river networks, development of ecologically sound systems of restoring riparian ecosystem properties, and attention to social needs for riparian resources (Gregory 1997, cited in Naiman et al. 2000). The introduction of these additional considerations has brought about questions regarding the effectiveness of various buffer widths. Narrower buffers (8–30 m) have traditionally been recognized as sufficient for protecting water quality and fish stocks, while wider buffers (60–100 m) may be necessary if the objectives are to maintain pre-logging densities of some wildlife species (Richardson et al. 2002) or to preserve scenic values (O'Connell et al. 2000). It is therefore apparent that care must be taken when designing and implementing riparian buffers to incorporate a variety of ecological factors that will maintain or improve biodiversity, particularly on landscapes managed for timber. The efficacy of riparian management zones (RMZs) can best be determined through well-designed studies based on a Before-After-Control-Impact (BACI) study design. These types of temporal studies, although costly, are the best means for addressing the effectiveness of RMZs in providing habitat for wildlife.

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Chapter 3 WESTERN WASHINGTON SONGBIRDS

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Abstract

A replicated, Before-After-Control-Impact (BACI) experimental study was completed in the low-elevation, advanced second-growth, Douglas-fir–dominated forests of inland western Washington to investigate the effects on a selected group of songbirds of three different riparian treatments: (1) no harvest (control; $n = 6$ sites); (2) harvest according to state regulations (state buffer treatment; $n = 6$ sites); and (3) harvest according to a modified version of state regulations (modified buffer treatment; $n = 5$ sites). Sampling occurred along two types of transects: (1) upland and (2) riparian. Three intervals were sampled: (1) pre-harvest; (2) a first post-harvest interval at ~2-years post-harvest; and (3) a second post-harvest sampling interval at ~10-years post harvest.

Relative abundance of songbirds declined at all three treatment types in both riparian and upland habitats between the pre-harvest and first post-harvest sampling intervals, with the most notable significant declines at upland transects of state and modified buffer sites. Thereafter, relative abundance of breeding birds increased between the first and second post-harvest intervals, with bird abundance becoming significantly greater along the upland transects of state and modified buffer treatments. The abundance of songbirds along riparian transects did not differ significantly with respect to riparian buffer width within or among any sampling intervals.

Pre-harvest songbird alpha diversity (i.e., species richness) was similar among all treatments and transects. Significant differences within and between transect and treatment types arose during the second sampling interval and continued through the third sampling interval. The highest alpha diversity occurred along the upland transects of logged sites. The alpha diversity of songbirds along riparian transects did not differ significantly as a function of riparian buffer width within any sampling intervals, although there were a few differences among sampling intervals. The similarity of songbird communities within transect types and among sampling intervals was greatest at control sites throughout this study. Community similarity was lowest in comparisons of upland communities at logged sites between and among sampling intervals irrespective of riparian treatment. Community similarity was greater at the modified buffer treatment than at the state buffer treatment during both post-harvest intervals.

Forestry-induced changes to habitat structure and forest succession altered the abundance, alpha diversity, and makeup of songbird communities by favoring an influx of new species at the expense of those that preferred the forest conditions that typified the treatments during the pre-harvest period. Songbird abundance and alpha diversity appeared to be closely tied to many of the habitat features measured in this study and shown herein to be affected by forest harvesting and succession. The modified buffer treatment (i.e., comparatively wider riparian buffers) was more effective than the state buffer treatment at maintaining pre-harvest community structure in riparian areas. Patterns of songbird abundance and diversity observed in this study are in keeping with the known ecological requirements of the individual species and with accepted models of the linkages between alpha diversity and seral stage of coniferous forests. However, measures of species abundance and diversity are insufficient to measure the efficacy of riparian buffer width, as changes in some species can be offset by changes in others. Therefore, measures of community similarity are essential in this regard. Finally, a number of study limitations and management implications are presented.

Introduction

Presence or abundance of wildlife in a given area reflects a complex set of interactions between biophysical components of the environment. The cool, wet climate and mountainous terrain of the western slope of the Cascade Mountains have produced a complex drainage network that in turn has strongly influenced the biophysical environment. Within climatic tolerances, soil conditions influence the composition of the vegetation community. Because vegetation directly or indirectly provides a wildlife species with its food and cover[2] requirements, predictable associations between wildlife species and habitat conditions have been identified.

Riparian habitats are essentially those where the soil nutrient and moisture regimes are affected by adjacent water. The soil moisture and nutrient regimes of riparian zones are often greater than at nearby upland areas, with the result that the vegetation communities in riparian zones are considerably different from those at adjacent upland areas. Such differences in vegetation communities often lead to differences in vertebrate (and invertebrate) assemblages. Riparian zones are characterized by higher levels of primary productivity, energy transport, and, often, natural disturbances than upland habitats. Microclimates in riparian zones can be somewhat different than those in the adjacent upland forests.

Recognizing that large-scale habitat perturbations can affect wildlife–habitat associations, managers have sought ways of mitigating the adverse effects of forestry on wildlife habitat. For example, no-harvest reserves, variable retention, and selective harvesting have been used at the stand and landscape levels to ameliorate the effects of forestry on wildlife (Carey and Curtis 1996; Curtis 1997; Franklin et al. 1997; Lindenmayer and Franklin 2002). Contemporary forest managers are bearing an increasing level of responsibility for maintaining biological diversity in forested landscapes (Bunnell et al. 1999). Forest managers are tasked with harvesting a crop of trees and then, in time, returning the site to a condition that provides for most of the same timber and non-timber values as it did pre-harvest—or at least ensuring that comparable values exist somewhere in the landscape (Tuchmann et al. 1996; Moore and Allen 1999).

Riparian zones in the temperate forests of western North America provide for a number of ecological values. Although riparian zones occupy comparatively small portions of forested landscapes, they often support unique species and higher levels of biodiversity and species abundance compared to more abundant, upland forests (Sabo et al. 2005; Peak and Thompson 2006). Riparian zones also play critical roles in watershed hydrology. For these and many other reasons the conservation and management of riparian habitats has garnered much attention in recent years (Naiman et al. 2000).

Riparian buffers are commonly used in an attempt to mitigate the effects of industrial forestry on biodiversity (Knopf et al. 1988). The width of such buffers has been the subject of considerable debate (Knutson and Naef 1997) and study because of trade-offs between ecological and economic values (Berg 1995). If buffers are wider than necessary to provide their stated ecological values, economic revenue from timber harvesting could be unnecessarily foregone. If short-term economic values are maximized by minimizing buffer width, species conservation could be jeopardized. As a means of balancing these often competing objectives, guidelines for riparian buffer width are common throughout the political jurisdictions of North America (Lee et al. 2004).

Western Washington riparian management zones (RMZs) have been required, at least in some form, since 1976. Between 1976 and 1988, the Washington Forest Practices Rules and Regulations

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[2] "Cover" describes habitat that provides such values as security from predators, security from weather extremes, and sites for breeding.

required "streamside management zones" (SMZ) which were intended to provide "stream bank integrity and temperature control" (Washington State Forest Practices Board 1976, 1982). Streamside management zone regulations prescribed riparian buffers 7.5–15 m wide with all non-merchantable vegetation and "sufficient merchantable timber, if any, necessary to retain 50–75 percent of the summer mid-day shade of the water surface" left within the SMZ (Washington State Forest Practices Board 1976). In 1988, the RMZ regulations were strengthened considerably, requiring riparian buffers up to 30 m wide (Washington State Forest Practices Board 1988). The change in regulations was a result of the Timber, Fish, and Wildlife (TFW) Agreement reached in 1987 among representatives of the Washington State tribes, forest-products industry, environmental community, and natural-resource agencies. The agreement sought to resolve conflicts between these diverse groups by recognizing the common goal of preserving natural resources while simultaneously maintaining a viable timber industry.

In 1990, the Wildlife Steering Committee of the Cooperative Monitoring, Evaluation, and Research Committee (CMER) of the TFW Agreement initiated a study to evaluate the efficacy of RMZs in providing habitat for wildlife. The study was designed to evaluate the abundance and diversity of wildlife in RMZs by comparing two different buffer configurations at unlogged control sites. The buffer configurations were based on the RMZ guidelines mandated at the time (Washington State Forest Practices Board 1988) and on modifications to the 1988 guidelines. The wildlife groups selected for study were terrestrial and aquatic amphibians, riparian-associated songbirds, small mammals (rodents and insectivores), and bats. A precursor to this study (O'Connell et al. 2000) compared the diversity and abundance of these wildlife groups before and immediately after logging to identify the short-term temporal effects of logging both within and between treatments. In 2001, CMER initiated a follow-up study to evaluate the long-term effects of RMZs on wildlife. This study is the only one in Washington to evaluate the long-term post-treatment effects of riparian management strategies on wildlife based on data collected at the same sites.

In Washington, many neotropical migrant birds rely heavily on riparian habitat for breeding during spring and summer. Of the 118 species of neotropical migrants in Washington, 67 have an affinity for riparian habitats (Andelman and Stock 1994). The conservation of neotropical migrants in the western United States will depend very much on protection and restoration of riparian woodlands (Bock et al. 1993). When land is converted from natural vegetation to other uses, the retention of adequate riparian buffers can provide remnant habitat sufficient to support a broad variety of bird species. The key question is what is an "adequate" riparian buffer?

Riparian buffer width and breeding-bird (particularly passerine) responses to them have been studied in the forests of western North America and elsewhere (e.g., Kinley and Newhouse 1997; Hagar 1999; O'Connell et al. 2000; Pearson and Manuwal 2001; Shirley and Smith 2005; chapter 3 of the current study). Wider riparian buffers appear to result in greater bird use; Manuwal (1986) found a 58% increase in bird use with a 50% increase in the size of the riparian zone. Many researchers note high bird diversity and abundance in riparian areas (Bottorff 1974; Stevens et al. 1977; Hehnke and Stone 1978; Knopf 1985; Knight 1988), while others found no relationship between riparian stand and bird diversity (Carey 1988).

Pearson and Manuwal (2001) found that within ~2-years post-harvest, riparian buffers retained after the adjacent upland was logged did not differ in terms of overall abundance relative to unlogged control sites; but the number of species and average species turnover increased in narrow buffers (~7.5 m wide), while both species turnover and alpha diversity were similar to the control sites at wider buffers (~30 m wide). The 10-year post-treatment efficacy of RMZs to provide habitat for songbirds has not been studied in western Washington.

The objectives of this study were to evaluate the post-treatment efficacy of RMZs in providing habitat for songbirds. Specifically, we compared the relative abundance, alpha diversity (species richness), and community similarity of breeding songbirds in riparian and upland habitats across time and treatments.

Methods

This study was a component of a larger study evaluating the efficacy of RMZs in providing habitat for selected groups of wildlife. The larger study was implemented in 1991, with pre-treatment sampling in 1992 and 1993. The first post-treatment sampling occurred in 1995 and 1996, with a second post-treatment sample (this study) in 2003 and 2004. The temporal nature of this study required that sampling techniques used in the pre-harvest and first post-harvest sampling intervals be repeated in the second post-harvest sampling interval to ensure consistency among the data sets from each sampling interval.

The study design was based on a split-plot repeated-measures design, with 6 sites assigned to each of three treatments (*n* = 18). Each site was split into two distinct habitat types: (1) terrestrial riparian and (2) upland. Terrestrial riparian habitats occurred within 5 m of the ordinary high-water mark adjacent to a perennial stream and extended to the point where the vegetation visibly shifted to an upland type. Upland habitats extended upland from the zone of transition from riparian to upland vegetation. Upland sampling occurred 100 m upslope from terrestrial riparian habitat.

The 18 sites were assigned to one of three treatments:

1. **Control**: This treatment consisted of continuous stands of maturing second growth (65–75 years old) dominated by coniferous trees, with no upland harvesting for the duration of the study.
2. **State**: This treatment consisted of a logged upland forest with a narrow forested buffer retained in the riparian zone. The RMZ was based on the Washington Forest Practices Rules and Regulations in place in 1988 concerning buffer width and number of leave trees for Type 3 (fish-bearing, perennial) streams (Table 3-1). Streams of Type 2, 4, and 5 that were structurally similar to the Type 3 streams were also used in the study but were buffered according to Type 3 regulations to keep the experimental treatments uniform (Appendix 3-A). The RMZ extended from the ordinary high-water mark to the line where vegetation changes from riparian to upland but was not less than 15 m wide. The number and arrangement of leave trees was modified to accommodate the logging operation, and the RMZ width was expanded as necessary to include swamps, bogs, marshes, or ponds adjacent to the stream.
3. **Modified**: This treatment consisted of a logged upland forest with a wider, more variable forested buffer retained in the riparian zone. The RMZ was based on harvest prescriptions developed by O'Connell et al. (2000). The prescription for the modified buffer departed from the 1988 Forest Practices Rules and Regulations in three ways:

    i) The 1988 guidelines specified a minimum canopy cover based on stream temperature classification and the elevation at the midpoint of the stream. If the cover requirement is met, selective cutting is allowed in the no-entry zone (i.e., within 7.5 m of the ordinary high-water mark). The modified prescription differs from the 1988 state prescription in that harvest is prohibited within the no-entry zone. Cover requirements are the same as those indicated in the1988 rules (WAC-222-30-040 [2] Temperature Control, Washington State Forest Practices Board [1988]). Specifically, WAC-222-30-040 (2) states that:

> "All nonmerchantable vegetation that provides mid-summer and mid-day shade of the water surface should be retained; and
>
> Sufficient merchantable timber, if any, necessary to retain 50% of the summer mid-day shade of the water surface, except when ambient water temperatures exceed 15.5° C for a 7-day period. In this case, 75% of the shade should be retained."

ii) The modified prescription applies 1988 guidelines for selective harvest in state RMZs to a variable width located 7.5–22.7 m from the ordinary high-water mark. This prescription will produce a buffer that is at least 7.5 m from the ordinary high-water mark along the riparian/upland boundary.

iii) The modified prescription increases the number of wildlife reserve trees, recruitment trees, and downed logs left for each acre logged. State Forest Practices Rules and Regulations required only three wildlife reserve trees, two green recruitment trees, and two downed logs left for each logged acre. Unless the wildlife tree violated Washington Department of Labor and Industry requirements, all wildlife reserve trees were retained at modified sites. Wildlife reserve trees are defined as follows:

- **Type 1**: Live tree with defective or deformed sound tops, trunks, and roots.
- **Type 2**: Dead tree with a sound top, trunk, and roots.
- **Type 3**: Live or dead tree with unstable tops or upper portions.
- **Type 4**: Live or dead tree with unstable trunk or roots, with or without bark. This includes "soft" snags as well as live trees with unstable roots caused by root rot or fire. Type 4 reserve trees are the most dangerous.

Table 3-1. Leave-tree requirements per water type and average RMZ width for western Washington (see WAC 222-30-020 [4c]) using January 1988 or November 1988 rules.

Water type / Avg. width (m)	RMZ Max. width (m)	Ratio of conifers: Deciduous / Size of leave trees	# Trees / 300 m (each side) by bed material	
			Gravel/Cobble	Boulder/Bedrock
1 & 2 / ≥22	30	Representative of stand	50	25
1 & 2 / ≤22	22	Representative of stand	100	50
3 / ≥1.5	15	2:1 / 30 cm or next largest available	75	25
3 / ≤1.5	7.5	1:1 / 15 cm diameter or next largest available	25	25

Study Area

The 18 study sites were distributed throughout western Washington in Cowlitz, Grays Harbor, King, Lewis, Pierce, and Thurston counties (west of the Cascade Mountains on the Olympic Peninsula, in southwestern Washington, the Puget Trough, and on the western slopes and crest, or the Washington Cascades physiographic provinces) (Figure 3-1).

Study sites were established in 1992 and 1993 and were selected to minimize variation in forest age and composition, elevation, moisture condition, and water type. Accordingly, the results of the study are applicable to sites meeting the same criteria that were used to select the 18 study sites:

1. Low elevation (<620 m).
2. Second-growth forest (55–66 years old) dominated by Douglas-fir (*Psuedotsuga menziesii*).
3. Type 3 water according to state forest regulations. Type 4 was selected if streams differed in only the presence of salmonids. Water types are defined in Table 3-2.
4. Predominantly coniferous riparian canopy with deciduous tree component.
5. At least 500 m in stream length; and
6. Road access within 500 m.

Figure 3-1. Distribution of western Washington study sites. West Fork Falls Creek replaced Vail in 2003.

Table 3-2. Water-typing criteria (WAC 222-16-030) used in Washington in 1988. Source: Washington State Forest Practices Board (1988).

	Water Type				
Parameter	**1**	**2**	**3**	**4**	**5**
Channel Width	N/A	≥ 6 m between OHWM	**Anadromous:** > 1.5m between OHWM Resident Game: > 3 m between OHWM	> 0.6 m between OHWM	> 0.6 m between OHWM
Gradient	N/A	< 4%	**Anadromous:** < 12% not upstream of a falls > 3m in height Resident Game: < 12%	N/A	N/A
Flow	N/A	N/A	**Anadromous:** N/A Resident Game: > 0.3 CFS at summer low flow	N/A	N/A
Impoundment	N/A	Water surface area of < 0.4 ha at seasonal low flow	**Anadromous:** Water surface area of < 0.4 ha at seasonal low flow Resident Game: Water surface area of < 0.2 ha at seasonal low flow	N/A	N/A
Fisheries	N/A	Used by substantial numbers of anadromous or resident game fish for spawning and rearing and migration	Used by significant numbers of anadromous or resident game fish for spawning and rearing and migration	Not used by significant numbers of fish	Not used by significant numbers of fish
Diversion	N/A	Domestic use for > 100 residences or campsites. Includes upstream reach of 450 m or until the drainage area is < 50%, whichever is less	Domestic use for > 10 residences or campsites. Includes upstream reach of 450 m or until the drainage area is < 50%, whichever is less	N/A	N/A
Other	All water within their OHWM inventoried as “Shoreline of the State” excluding related wetlands	Streams flowing through campgrounds available to public having ≥ 30 campsites	Contributes > 20% of flow to Type 1 or 2 water. Anadromous fish impoundments have outlet to stream with anadromous fish	N/A	All natural waters not classified as Type 1, 2, 3 ,or 4 or seepage areas, ponds and drainways having short run-off periods

Field Sampling

Breeding-Songbird Surveys

Birds were surveyed using 15 m fixed-radius point counts (Verner 1985). Small-radius point counts reduce the problem associated with differences in the ability to detect birds along riparian and upland habitats caused by stream noise. Small-radius point counts also allowed us to examine differences in bird abundance along narrow buffers of potential habitat at logged sites. In each stand, 10 riparian stations were established along the edge of the stream, with 5 stations spaced evenly on each side of the stream (Figure 3-2). Each riparian station was located approximately 15 m from the usual high-water line, 100 m from other stations and (at state and modified sites) approximately 50 m from the edge of the stand. Ten point-count stations were located parallel and 100 m upslope from the riparian stations in the adjacent uplands. Reference flags were placed 15 m to each side of each station to facilitate distance estimation. Censuses usually started within 30 minutes of sunrise and were completed within 5 hours. On several occasions, start times were delayed because of weather (e.g., rain or wind). Two observers worked one site simultaneously, with one observer in the riparian transect on one side of the creek and one observer in the opposite upland transect. Upon arriving at a survey point, observers remained stationary and quiet for 1–2 minutes to allow birds to settle and then recorded all birds heard or seen during a 6-minute period. Birds detected outside the 15 m radius during the observation period were tallied separately. For riparian point counts and transects, the location of each bird was noted (i.e., within riparian buffer/outside riparian buffer). For transects (riparian and upland), birds outside the buffer were identified as streamside or upland. Flyovers (i.e., birds seen or heard flying over the tree canopy at a point count/transect but not directly associated with the terrestrial environment at that site) were identified as such. To minimize observer bias, observers were rotated among the study sites and transects. To avoid bias associated with visiting riparian or upland habitats first, travel routes along transects and transect survey starting order were alternated. Logistical constraints meant that some individual sites were surveyed simultaneously by the two observers (e.g., one surveying the upland while the other surveyed the riparian). For pairs of sites that were close together, observers usually worked alone on a site. Each transect was visited between mid-April and the first few days of July (30 April–25 June 1993; 3 May–28 June 1995; 29 April–1 July 1996; 24 May–27 June 1997 [2 sites only]; 11 May–3 July 1998 [2 sites only]; 19 April–1 July 2003; 24 April–19 June 2004).

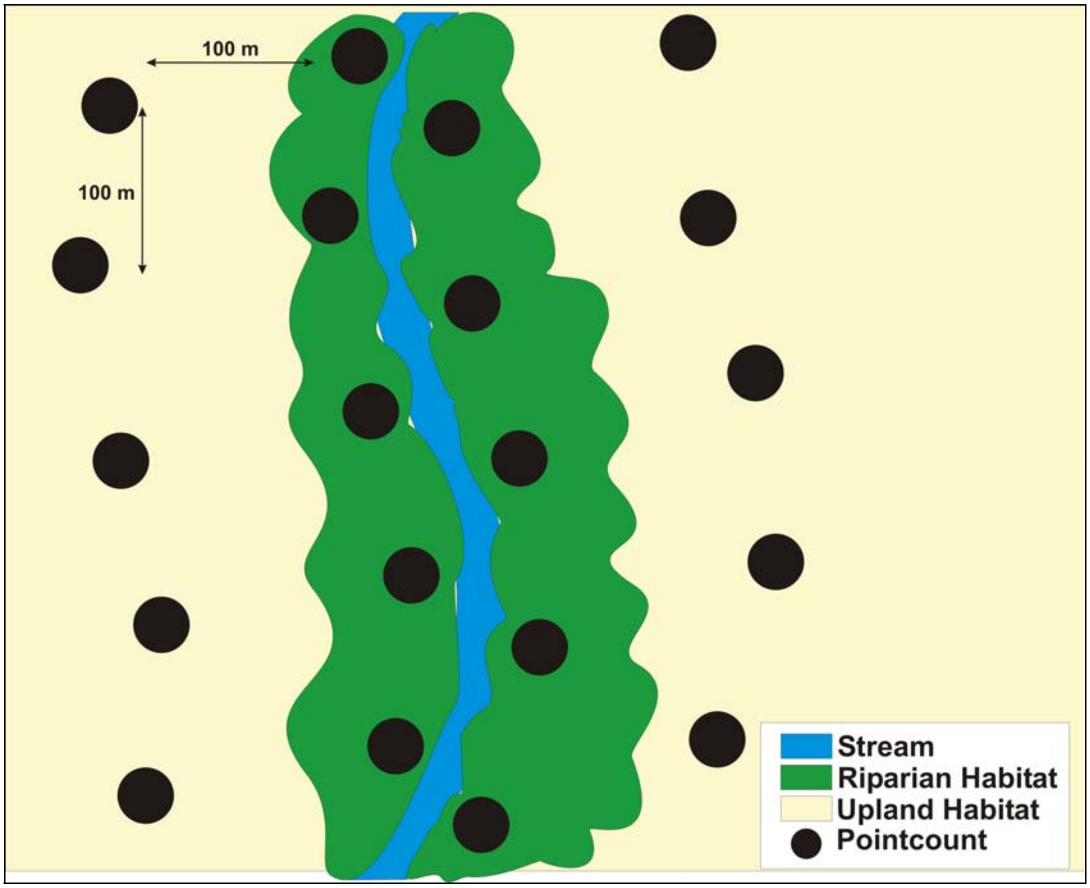

Figure 3-2. Schematic of a typical site set up to sample for songbirds using small-diameter radius point counts during western Washington songbird surveys, 1992–2004.

All sites, regardless of treatment type, were surveyed 4–5 times during one year (1993) of pre-treatment sampling before any forest harvesting[3] occurred at the modified and state buffer sites. Following forest harvesting, most sites were sampled 5–6 times during each of two years (1995/1996). Two exceptions were a control site (Abernathy) that was monitored in each of four years, and a modified buffer site (Ryderwood 1557) that was surveyed in 1997 and 1998 owing to delays in harvesting that site (i.e., logged in 1996). With one exception, all sites were monitored 6 times in 2003 and 2004. One control site (Vail) had been logged after 1998, so a replacement site (West Fork Falls Creek) was established in 2003. Following the 2003 field season, the Griffen Creek modified buffer site was subjected to silvicultural activities (primarily thinning). Because identical activities did not occur at the other modified buffer sites, this site was removed from the study.

Surveys were spaced throughout the breeding season to account for differences in breeding chronology among species. No survey was conducted during heavy precipitation or wind speeds that exceeded ~20 km/h (i.e., modified Beaufort wind scale code >5; Table 3-3). In 2003/2004, ambient temperature (°C), wind strength (Beaufort wind scale), and precipitation class were recorded at each site. Air temperature (°C) was recorded using a mercury thermometer. Every attempt was made to avoid counting individual birds more than once. If the riparian buffer was narrower than the diameter of a point-count circle on logged sites, then we recorded whether the bird was detected in the forested buffer or in the clearcut portion of the circle. Appendix 3-B lists the data that were collected at each site during each survey.

Table 3-3. Modified Beaufort wind scale used in breeding-bird surveys.

Code	Definition
1	calm (<2 km/h)
2	light air (2–5 km/h)
3	leaves rustle (6–12 km/h)
4	leaves and twigs constantly move (13–19 km/h)
5	small branches move, dust rises (20–29 km/h)
6	small trees sway (30–39 km/h)
7	large branches moving, wind whistling (40–50 km/h)

Vegetation and Habitat Structure Sampling

Vegetation sampling occurred once during each sampling interval and coincided with the primary period of productivity and when most plants would be readily identifiable, which in this case was mid- to late summer. Vegetation data from the pre-treatment and first post-treatment sampling intervals were provided by the Washington Department of Natural Resources. Vegetation data for the second post-treatment sampling interval were collected by LGL Limited environmental research associates.

Riparian and upland vegetation and habitat attributes were measured at each site within a series of plots: 12 plots were established in the riparian habitat and 10 plots were established in the upland habitat. Each plot consisted of four 10 x 8 m large quadrats (Figure 3-3). Riparian plots extended 8 and 16 m from the ordinary high-water mark, which was usually within 5 m of the stream edge. Upland habitats were sampled in plots approximately 100 m upslope from the riparian transects. Within each large quadrat we established 2 x 2 m and 1 x 1 m small quadrats for the collection of

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[3] With some exceptions, harvesting at most sites occurred during 1994. One state buffer site (Kapowsin) was logged in March 1995 (prior to the 1995 bird survey); a modified buffer site (Ms Black) was logged in January 1995, and another (Ryderwood 1557) was logged in 1996.

ground-cover measurements (Figure 3-4). These small quadrats were located 1, 4, 7, and 10 m from the streamside edge of each plot.

At each small quadrat, we estimated the percentage cover of herbaceous and woody vegetation, rock, litter, and bare soil at 1, 4, 7, and 10 m from the streamside edge. The percentage cover of shrubs was estimated in large quadrats 2 and 3 at each plot in the riparian and upland sampling areas. Shrubs were grouped into three categories: (1) berry-producing, (2) evergreen, or (3) other deciduous.

Percentage of downed wood was estimated from large quadrats 2 and 3 at each plot. Wood was considered down if its angle of incidence with the ground was <45°. Each piece of downed wood was categorized by diameter (cm; 2 classes) and decay class (3 classes; Table 3-4).

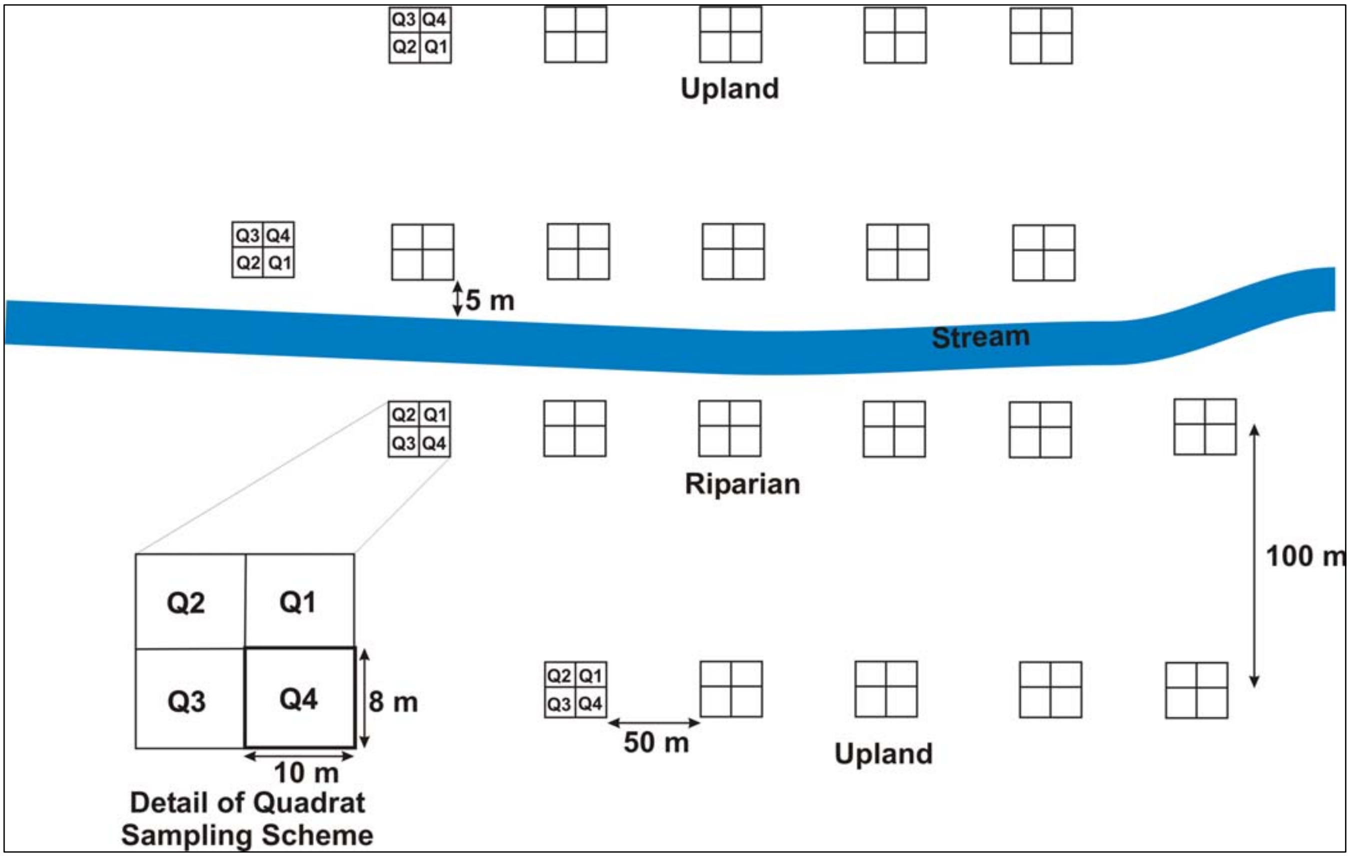


Figure 3-3. Schematic of the vegetation sampling scheme using plots consisting of four 10 x 8 m large quadrats situated on each side of the stream at each study site.

Table 3-4. Diameter- and decay-class categories to be used for downed-wood measurements in large quadrats 2 and 3.

Diameter Class (cm)		Decay Class	
1	10–30	1	Structurally sound wood with intact limbs
2	>30	2	Reduced structural integrity and some limb loss
		3	Minimal structural integrity and presence of epiphytes

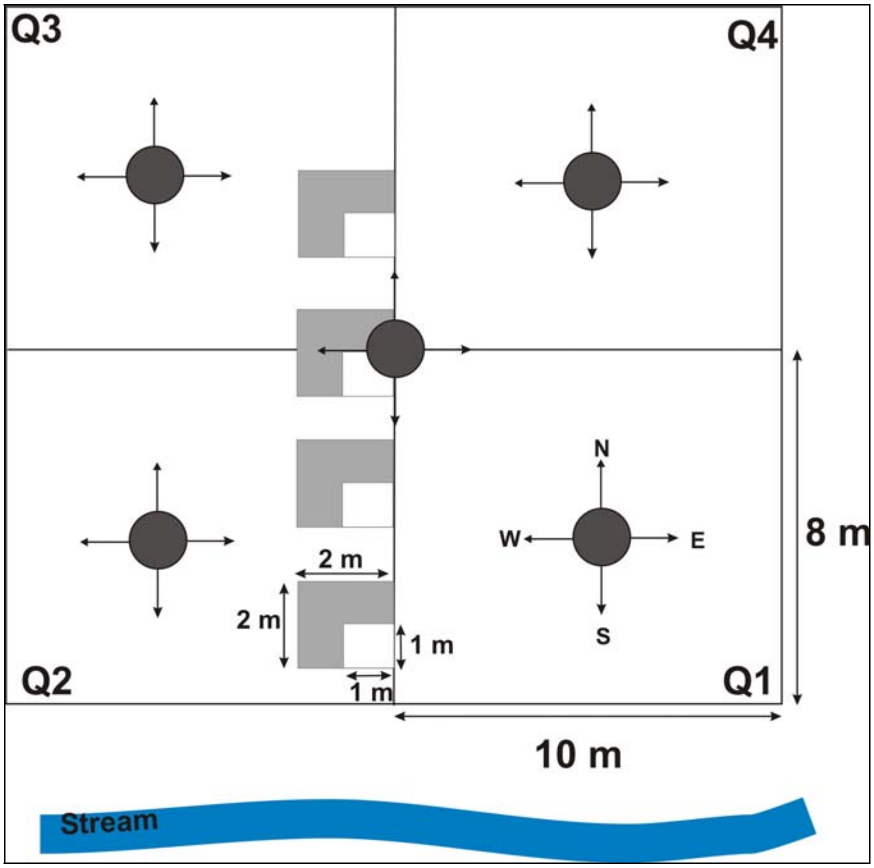


Figure 3-4. Schematic of vegetation sampling design showing the location of the 2 x 2 m and 1 x 1 m small quadrats at 1, 4, 7, and 10 m from the edge of the plots and the locations from which canopy-cover data were collected (black circles).

Snags were counted in all 4 large quadrats and grouped according to quadrat location. Snags were classified as short (<1.5 m), medium (1.5–15 m) and tall (>15 m) in three diameter classes: Class 1 = all limbs attached and structurally sound; Class 2 = losing limbs and showing reduces structural integrity; and Class 3 = about to fall down due to minimal structural integrity.

Percentage cover of sapling trees was estimated from large quadrats 2 and 3. Saplings between 1 and 3 m in height were included regardless of whether they originated from the ground, a stump, or downed wood. Each sapling was identified to species, and we estimated percentage cover for each species encountered in each quadrat. We did not include tree species in our analyses that were <1 m tall.

Trees >3 m in height were counted in all 4 large quadrats and grouped according to quadrat (1, 2, 3, or 4) and diameter size (10 cm, 10–50 cm, 50–100 cm, >100 cm diameter at breast height [DBH]). All trees with split boles were counted as more than one tree (with the exception of vine maple) if the split occurred below breast height. Trees with more than half of the bole outside the quadrat were not counted.

Riparian buffer width was measured from five plots on either side of the stream and was measured from the ordinary high-water mark to the outermost edge of forest. Slope distance rather than horizontal distance from the outermost tree to the normal high-water mark was measured.

To estimate canopy cover we used a spherical, convex densiometer at the outer corners of each quadrat and at the center point where the 4 large quadrats met. At each of the five points, we took four readings: (1) facing the stream; (2) away from the stream; (3) downstream; and (4) upstream (Figure 3-4). We estimated canopy cover from each of the five locations in each plot. Other data recorded at each site included air and soil temperature, relative humidity, slope, and aspect.

Statistical Analyses

Statistical analyses were performed using SAS V9.1 (© 2002–2003), R V2.2.1 (© 2005), and Microsoft Excel 2002 (© 1985–2003). Prior to performing all analyses, data were assessed to see if they fit a normal distribution using the Analyst Application in SAS, which produces four tests of normality (Shapiro–Wilk, Kolomogerov–Smirnov, Cramer–von Mises, and Anderson–Darling). Comparisons of relative abundance were done using log-transformed means ($\log[x + 1]$), which approximated a normal distribution. The critical value of alpha was set at 0.1 and beta at 0.8 (*a priori*) for all analyses, which is appropriate for landscape-based studies and when analyzing data obtained from populations that are highly variable.

We used detection rate as a measure of relative abundance. Detection rate was calculated as the number of detections of each species divided by the number of visits to a given habitat. For example, if riparian habitats were visited 6 times in one year and Golden-crowned Kinglet was documented 27 times, the detection rate would be $27 \div 6 = 4.5$. For each species, an average detection rate was derived for each sampling interval; each sampling interval consisted of two years. Yearly data from each sampling interval were combined because they were found not to be statistically different. Statistical tests were done only for the most common (i.e., abundant) species.

The following references were evaluated when developing avian species selection criteria: O'Connell et al. (2000); Pearson and Manuwal (2001); Sallabanks et al. (2002); and Shirley (2004). The following criteria were used to select data considered suitable for analysis except where noted otherwise:

- Detections within 15 m of the point count were included.
- Detections in the point-count sampling period of 0–6 minutes were included.
- All species detected >15 times at ≥50% of sites pre-harvest were included (as per O'Connell et al. 2000).
- Species that did not meet the previous criterion, but whose occurrence increased notably over the course of the study (i.e., if detection rate and/or the number of sites the species occurred at increased), were included.
- Flyovers and non-breeding migrants were excluded.

These criteria were used to maintain congruency with the analyses performed by O'Connell et al. (2000). Where appropriate, post-hoc multiple range tests were used to account for experiment-wise error to avoid Type I errors (i.e., errors that are made when a difference is observed and there really is not one). When making multiple pairwise comparisons, the probability by chance alone of having at least one comparison that is significant is very high; this could lead to a failure to reject the null hypothesis when it is actually true (Shaffer 1995). When pairwise comparisons of the means were made, the Bonferroni method was applied to control the family-wise error rate.

Alpha Diversity and Community Similarity

We used the number of species found in each habitat type (i.e., riparian or upland) of each site during each sampling interval as a measure of alpha diversity (species richness). Alpha diversity data from riparian transects were used to test for differences relative to buffer width. To test whether logging influenced alpha diversity, we combined data from the buffer treatment groups (i.e., state and modified riparian buffers) because both groups received similar upland treatments (clearcut logging).

We used Morisita's coefficient of similarity (C) as a measure of community similarity because of its superior utility, relative to other measures, in pairwise comparisons of communities (Krebs 1999). Morisita's index (Morisita 1959; Horn 1966; Brower et al. 1990) measures community similarity and is based on Simpson's index of dominance (Simpson 1949). It calculates the probability that specimens randomly drawn from two sites will be of the same species, relative to the probability that specimens randomly drawn from the same site will be of the same species. This index is desirable because sample size and diversities of the samples have little influence on its calculation (Morisita 1959; Wolda 1981). Morisita's index returns a value from 0.0 (no similarity) to 1.0 (identical) and was chosen because it is affected little by large variations in sample size.

Relative Abundance

We used the number of times each species was detected at a site as a measure of relative abundance for each species. Comparisons were made between mean relative abundance measures, transformed as $\log(x + 1)$ to approximate a normal distribution. We tested variation in relative abundance only for the most common species; sample size was too small for analysis of other species.

Using PROC MIXED we conducted repeated-measures ANOVAs to test hypotheses about variation in relative abundance, followed by orthogonal and non-orthogonal contrasts (Table 3-5) of main effects (treatment, transect, sampling interval) and their interactions. The main experimental units for which measurements were repeatedly collected were the individual transects in riparian and upland habitats. The data from each transect were pooled to derive a treatment mean for each of the three sampling intervals.

Contrasts were grouped into two sets of questions: differences *between* and differences *within* sampling intervals. In each question set, non-orthogonal contrasts were used to answer questions about the mean relative abundance of the more common species detected. Based on the objective of the overall study, the sampling intervals, treatments, and transects were grouped to best answer the questions of differences between the state and modified buffer treatments relative to the control sites over time. The comparisons derived were based on the following assumptions:

1. All sites selected for this study were similar enough that results derived for one site could be extrapolated across all sites of a particular treatment and, as such, pooling of sites into treatments was acceptable.
2. The upland habitats at the state and modified treatments represented the same treatment and thus could be pooled into a post-cut condition defined by clearcut logging followed by replanting.
3. A comparison of the control site uplands with the pooled state and modified uplands (i.e., pre-cut vs. post-cut) would address upland logging effects (i.e., does logging uplands affect the relative abundance of songbirds?).
4. A comparison of riparian transects of the control sites to the pooled state and modified condition would determine if logging affects relative abundance of songbirds irrespective of buffer width.
5. A comparison between the modified and state riparian transects would provide a measure of the effect of a wide buffer vs. a narrow buffer and the effect that buffer width has on the relative abundance of songbirds.

The contrasts in Table 3-5 consist of two question sets. Question set 1 consists of 12 contrasts that compare the relative abundance of songbirds across time to determine if there are temporal relationships due to treatment or habitat type (transect). The 12 contrasts in question set 2 evaluate relative abundance within each sampling interval relative to treatment and habitat, which is particularly important for the pre-treatment sampling interval in order to establish a baseline of similarity or difference prior to treatment application.

Non-orthogonal contrasts ask specific questions and are a compromise between a "fishing expedition" (all possible pairwise *a posteriori* comparisons) and completely orthogonal (and therefore independent) contrasts. The maximum number of pairwise comparisons that could have been made for each species was 153. We made 24 contrasts, of which 18 are orthogonal. The 6 non-orthogonal contrasts are designed to ask pertinent questions of the data to determine if there are measurable treatment effects on the relative abundance of songbirds across time. Specifically, we wanted to know if the relative abundance of songbirds had a relationship with one or more treatments and/or habitat types. Contrasts are preferred to multiple ANOVAs with post-hoc multiple range tests to control FWE, because contrasts can reduce the number of questions asked of the data (and therefore avoid data mining) and because, in many cases, many of the comparisons of the means are not of interest.

There is little clarity in the literature regarding the "right" way to approach the use of non-orthogonal contrasts. Some authors have indicated that non-orthogonal contrasts are acceptable provided the redundancy is noted (Mead 1990). Others have commented that orthogonality is desirable but not essential, because a reasonable approximation of the probabilities exists for non-orthogonal contrasts. Miller (1981) indicated that because multiple comparisons are being made (regardless of orthogonality) the results should be corrected for experiment-wise error using a correction factor such as Scheffé, Bonferroni, or Sidak. Miller (1981) discusses methods for pairwise comparison that can be adapted for general contrasts, suggesting that Bonferroni be used for additive inequality and Sidak for multiplicative inequality and that these results need to be corrected when the number of comparisons is large. The term "large" is vague and conveys nothing about when to account for experiment-wise error when using contrasts analyses in repeated-measures ANOVAs. We did not consider the number of comparisons to be large relative to the number of comparisons that were possible for each species. In addition, because we were analyzing a temporal data set that included three sampling intervals with 17 contrasts per sampling interval, we could have generated up to 51 orthogonal contrasts. However, although there is ambiguity in the literature, we elected to adjust the critical value of alpha but expanded our definition of a statistically significant result to include *p*-values up to and including 0.1.

Table 3-5. Orthogonal and non-orthogonal contrasts used to identify differences in relative abundance (RA) among and within sampling intervals (SI) relative to treatment (control, modified, state) and transect (riparian, upland). SI 1 = 1992/1993; SI 2 = 1995/1996; SI 3 = 2003/2004.

	Sampling Interval, Transect, and Treatment Type	Question
Among	CONTROL RIPARIAN - SI 2 vs. SI 3	Is RA different in SI 2 compared to SI 3 for control riparian transects?
	CONTROL RIPARIAN - pre-cut vs. post-cut	Is RA pre-harvest different from post-harvest for control riparian transects?
	CONTROL UPLAND - SI 2 vs. SI 3	Is RA different in SI 2 compared to SI 3 for control upland transects?
	CONTROL UPLAND - pre-cut vs. post-cut	Is RA pre-harvest different from post-harvest for control upland transects?
	MODIFIED RIPARIAN - SI 2 vs. SI 3	Is RA different in SI 1 compared to SI 3 for modified riparian transects?
	MODIFIED RIPARIAN - pre-cut vs. post-cut	Is RA pre-harvest different from post-harvest for modified riparian transects?
	MODIFIED UPLAND - SI 2 vs. SI 3	Is RA different in SI 2 compared to SI 3 for modified upland transects?
	MODIFIED UPLAND - pre-cut vs. post-cut	Is RA pre-harvest different from post-harvest for modified upland transects?
	STATE RIPARIAN - SI 2 vs. SI 3	Is RA different in SI 2 compared to SI 3 for state riparian transects?
	STATE RIPARIAN - pre-cut vs. post-cut	Is RA pre-harvest different from post-harvest for state riparian transects?
	STATE UPLAND - SI 2 vs. SI 3	Is RA different in SI 2 compared to SI 3 for state upland transects?
	STATE UPLAND - pre-cut vs. post-cut	Is RA pre-harvest different from post-harvest for state upland transects?
Within	SI 1 RIPARIAN - Control vs. cut	In SI 1, was riparian RA at the control sites different from the treatments?
	SI 1 RIPARIAN - Modified vs. State	In SI 1, was riparian RA different between the modified and state?
	SI 1 UPLAND - Control vs. cut	In SI 1, was upland RA at the control sites different from the treatments?
	SI 1 UPLAND - Modified vs. State	In SI 1, was upland RA different between the modified and state?
	SI 2 RIPARIAN - Control vs. cut	In SI 2, was riparian RA at the control sites different from the treatments?
	SI 2 RIPARIAN - Modified vs. State	In SI 2, was riparian RA different between the modified and state?
	SI 2 UPLAND - Control vs. cut	In SI 2, was upland RA at the control sites different from the treatments?
	SI 2 UPLAND - Modified vs. State	In SI 2, was upland RA different between the modified and state?
	SI 3 RIPARIAN - Control vs. cut	In SI 3, was riparian RA at the control sites different from the treatments?
	SI 3 RIPARIAN - Modified vs. State	In SI 3, was riparian RA different between the modified and state?
	SI 3 UPLAND - Control vs. cut	In SI 3, was upland RA at the control sites different from the treatments?
	SI 3 UPLAND - Modified vs. State	In SI 3, was upland RA different between the modified and state?

Co-inertia Analysis

The program R V2.2.1 (© 2005) was used to investigate habitat relationships between abundant or common species and the habitat attributes measured at each site and treatment. For this study, a co-inertia analysis (COIA) was used and was based on the ade4 package (Chessel et al. 2005). COIA is a multivariate statistical technique that provides for the visualization of the structure of ecological data via ordination. COIA is very general and some existing methods appear as special cases of it (Dray et al. 2003). For example, interbattery analysis (Tucker 1958) is mathematically equivalent to a simultaneous principle components analysis (PCA) on two ecological tables followed by a COIA (i.e., PCA–PCA COIA). When table X contains qualitative variables and Y contains species numbers, it is usual to cross tables X and Y to obtain a matrix containing the distribution of species among the categories of environmental variables. A simple canonical analysis (CA) of this new table allows one to ordinate the species and the environmental classes (analysis of ecological profiles; Montana and Greig-Smith 1990). Binary discriminant analysis (Strahler 1978), which has been used in ecology (Del Moral 1982; Huang and Del Moral 1988), is mathematically equivalent to Romane's CA (Dray et al. 2003). Although this approach allows plotting of species and environmental classes (Ben-Shahar 1987; Ben-Shahar and Skinner 1988), no information about ordination of sites is available.

COIA is similar to other ecological ordination techniques, such as a canonical correspondence analysis (CCA; ter Braak 1986) and redundancy analysis (RDA; Wollenberg 1977); however, COIA is a general coupling method that maximizes the covariance between the variables of two tables. Separate tables X and Y can be analyzed by various methods, leading to different coupling methods. This approach aims to find a site score that is a linear combination of environmental variables maximizing the variance of species centroids (i.e., separation of species niches). Unlike CCA, COIA does not have a regression step (Thiolouse et al. 2004). This difference has important implications for ecological data analyses: the regression step implies that in CCA and RDA the number of samples must be high compared to the number of environmental variables; COIA does not have this constraint. CCA also has an additional constraint (the total variance must be equal to 1) and must be avoided in the case of numerous environmental variables, which is common in biological studies and is the case in this study.

COIA is a paired ecological table analysis that relies on the covariance matrix produced from (in this case) a simultaneous PCA of the environmental and species data (Figure 3-5). Each table is related through a common number of rows, which in this case represents the number of sites. The output of the PCA can be visualized as two different representations of the sites in space: one relating to the relative abundance of species; the other to the habitat attributes. The covariance matrices produced by the PCA are then joined via COIA so that the relationship between species relative abundance and habitat data can be viewed. The resulting biplot allows for visualization of how the two initially unrelated data sets covary at the same sites. From the biplot, a description of the habitat variables can be derived for species of interest. The ability to quickly assign habitat relationships based on the relative position of species to habitat variables provides an effective way to describe the habitat relationships for each species in each sampling interval and for each treatment.

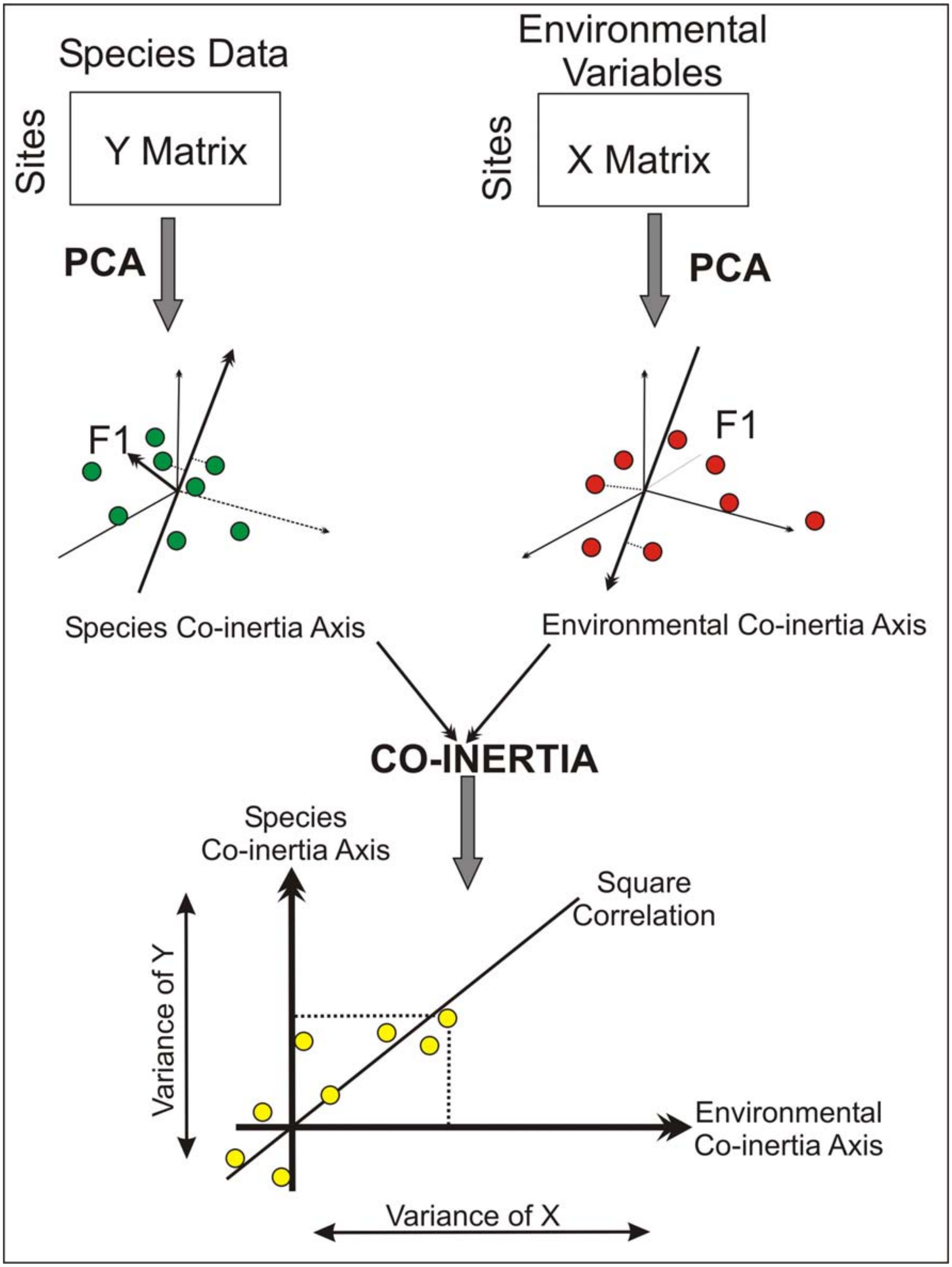


Figure 3-5. Schematic showing the principles of co-inertia analysis. The two ecological data tables X and Y produce two representations of the sites in two hyperspaces. Separate analyses find axes maximizing inertia (=variation) in space (F1 [first factorial axis]). Co-inertia analysis maximizes the square covariance between the projections of the sites on the co-inertia axes. Source: modified from Dray et al. (2003).

The relationship between species relative abundance and environmental variables can be viewed on a biplot. One approach is to view the species relative abundance and environmental variable biplots side by side to look for relationships. Alternatively, and as we have done, the species relative abundance biplot can be superimposed onto the environmental variable biplot for easier interpretation. When this is done, an environmental gradient can be described for each axis, with the horizontal axis (axis 1) representing the primary environmental gradient and the vertical axis (axis 2) describing a secondary gradient. Species that occur nearer the origin are indicative of (a) species with too few detections to infer habitat associations, (b) habitat generalists, or (c) species that respond to intermediate conditions along the gradients described by the axes. Species that are plotted farther away from the origin can be regarded as correlated with a particular habitat variable or suite of variables. The length of the vector from the origin to the location of the habitat variable in space is indicative of the strength of the correlation between the habitat variable and the gradient described for each axis. Unlike the more commonly used multivariate analyses, such as canonical correspondence analysis (ter Braak 1986), for which the number of environmental variables is limited to the number of sites minus one, COIA has no limitations on the number of environmental variables that can be used regardless of the number of sites sampled (Thioulouse et al. 2004).

A typical biplot resulting from a COIA is shown in Figure 3-6. This biplot for all riparian transects combined reveals an environmental gradient strongly driven by forest openness (i.e., negative crown closure) along axis 1. In addition, axis 2 shows a gradient from stands with tall evergreen and berry-producing shrubs to stands with an understory dominated by small Douglas-fir. Under these conditions, the detection rate of Wilson's Warbler and Swainson's Thrush is highly correlated with large deciduous trees, while the detection rate of Brown Creeper is correlated with axis 1, or increasing canopy cover.

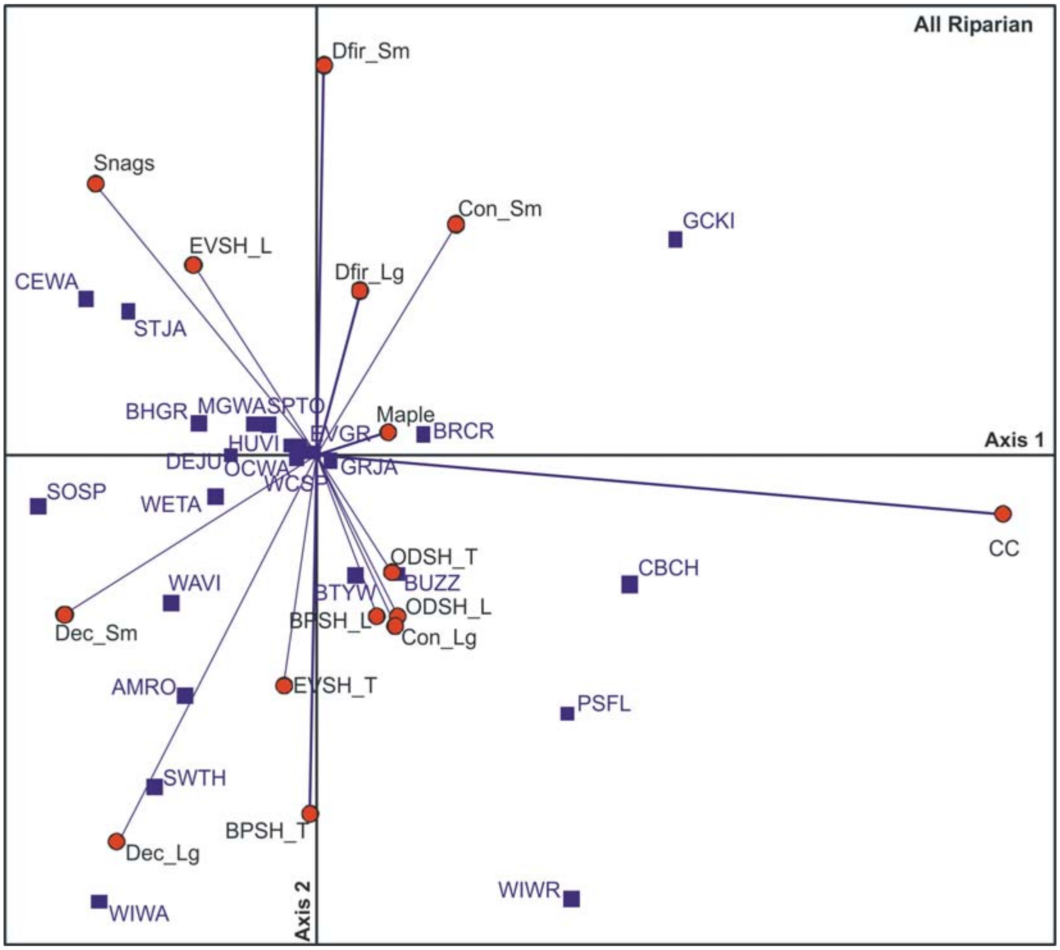


Figure 3-6. Sample biplot depicting songbird–habitat relationships in riparian habitats of all study sites averaged across time. CC = Canopy cover; BPSH_T = Tall berry-producing shrubs; EVSH_T = Tall evergreen shrubs; ODSH_T = Tall other shrubs; BPSH_L = Low berry-producing shrubs; EVSH_L = Low evergreen shrubs; ODSH_L = Low other shrubs; Snags = Snags; Maple = Bigleaf maple; Dec_Sm = Small deciduous trees; Dec_Lg = Large deciduous trees; Con_Sm = Small conifers; Con_Lg = Large conifers; Dfir_Sm = Small Douglas-fir; Dfir_Lg = Large Douglas-fir. AMRO = American Robin; BHGR = Black-headed Grosbeak; BRCR = Brown Creeper; BTYW = Black-throated Gray Warbler; BUZZ = Hermit/Townsend's Warbler; CBCH = Chestnut-backed Chickadee; CEWA = Cedar Waxwing; DEJU = Dark-eyed Junco; EVGR = Evening Grosbeak; GCKI = Golden-crowned Kinglet; GRJA = Gray Jay; HUVI = Hutton's Vireo; MGWA = MacGillivray's Warbler; OCWA = Orange-crowned Warbler; PSFL = Pacific-slope Flycatcher; SOSP = Song Sparrow; SPTO = Spotted Towhee; STJA = Steller's Jay; SWTH = Swainson's Thrush; WAVI = Warbling Vireo; WCSP = White-crowned Sparrow; WETA = Western Tanager; WIWA = Wilson's Warbler; WIWR = Winter Wren.

Results

General

Survey effort during the period of 1993 through 2004 is summarized in Table 3-6. Effort was consistent within sampling intervals but varied modestly among intervals.

Table 3-6. Number of site visits for breeding-bird surveys in western Washington, 1993–2004. 1993 = pre-harvest sampling interval; 1995–1998 = first post-harvest sampling interval; 2003/2004 = second post-harvest sampling interval.

		Year						
Treatment	**Site Name**	**1993**	**1995[a]**	**1996**	**1997**	**1998**	**2003**	**2004**
Control	Abernathy	5	5	6	6[b]	6[b]	6	6
	Elbe Hills	5	5	6			6	6
	Hotel Creek	5	5	6			6	6
	Porter Creek	5	5	6			6	6
	Taylor Creek	5	5	6			6	6
	Vail Control	5	5	6				
	West Fork Falls Creek	-	-	-			6	6
Modified Buffer	Blue Tick	5	5	6			6	6
	Eleven Creek 31	5	5	6			6	6
	Griffen Creek	5	5	6			6	-
	Ms. Black	5	5	6			6	6
	Ryderwood 1557[c]	5	5[b]	-	6[a]	6	5	6
	Side Rod	5	5	6			6	6
State Buffer	Eleven Creek 32	4	5	6			6	6
	Kapowsin	5	5	6			6	6
	Night Dancer	5	5	6			6	6
	Potpourri	4	5	6			6	6
	Ryderwood 860	5	5	6			6	6
	Simmons Creek	5	5	6			6	6

[a] First year of post-harvest monitoring of modified buffer and state buffer sites
[b] Data excluded from analyses
[c] Logged in 1996

The vast majority of bird detections were of passerines (i.e., songbirds; Figure 3-7). Proportions were arcsine transformed to ensure that all species groups were represented in the figure. Songbirds comprised 97.8%, 94.3%, and 91.8% of all bird detections during the three sequential sampling intervals.

Figure 3-8 shows attrition in the number of bird species and individuals detected (excluding flyovers) along the riparian and upland transects of the study sites according to three sets of increasingly restrictive filtering criteria:

- A total of 91 species of birds comprising 52,847 individuals were detected within the 0–6 min sampling interval.
- A total of 69 species of birds comprising 10,060 individuals were detected within 15 m of the point count within the 0–6 min sampling interval (Appendix 3-D).
- A total of 51 species of songbirds comprising 9358 individuals were detected within 15 m of the point count within the 0–6 min sampling interval.

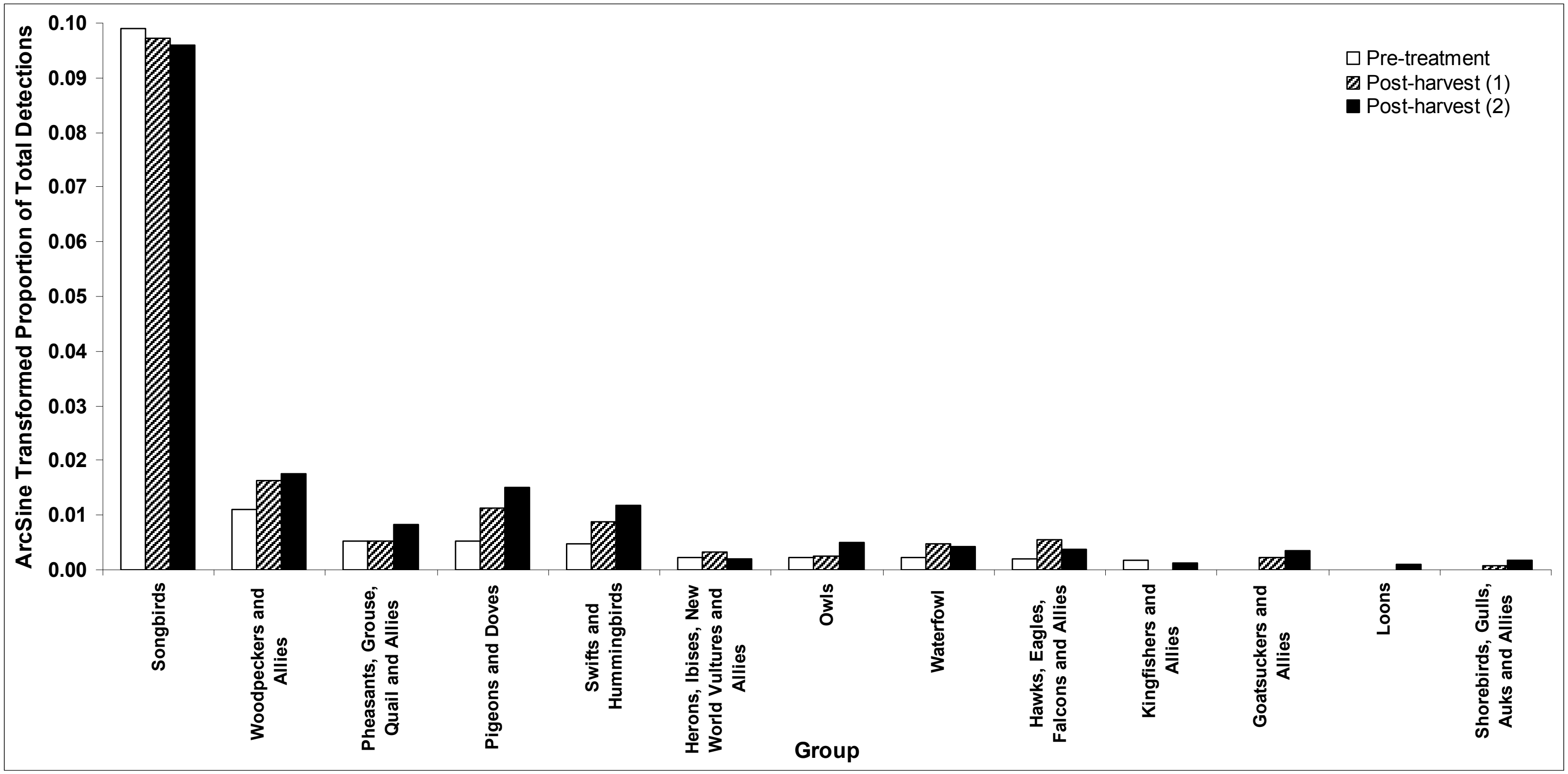


Figure 3-7. Proportion of birds (arcsine transformed) by taxonomic grouping, detected at point counts along the riparian and upland transects of study sites during breeding-bird surveys in western Washington for each sampling interval (includes flyovers, all distances, and point-count sampling duration of 0–6 min).

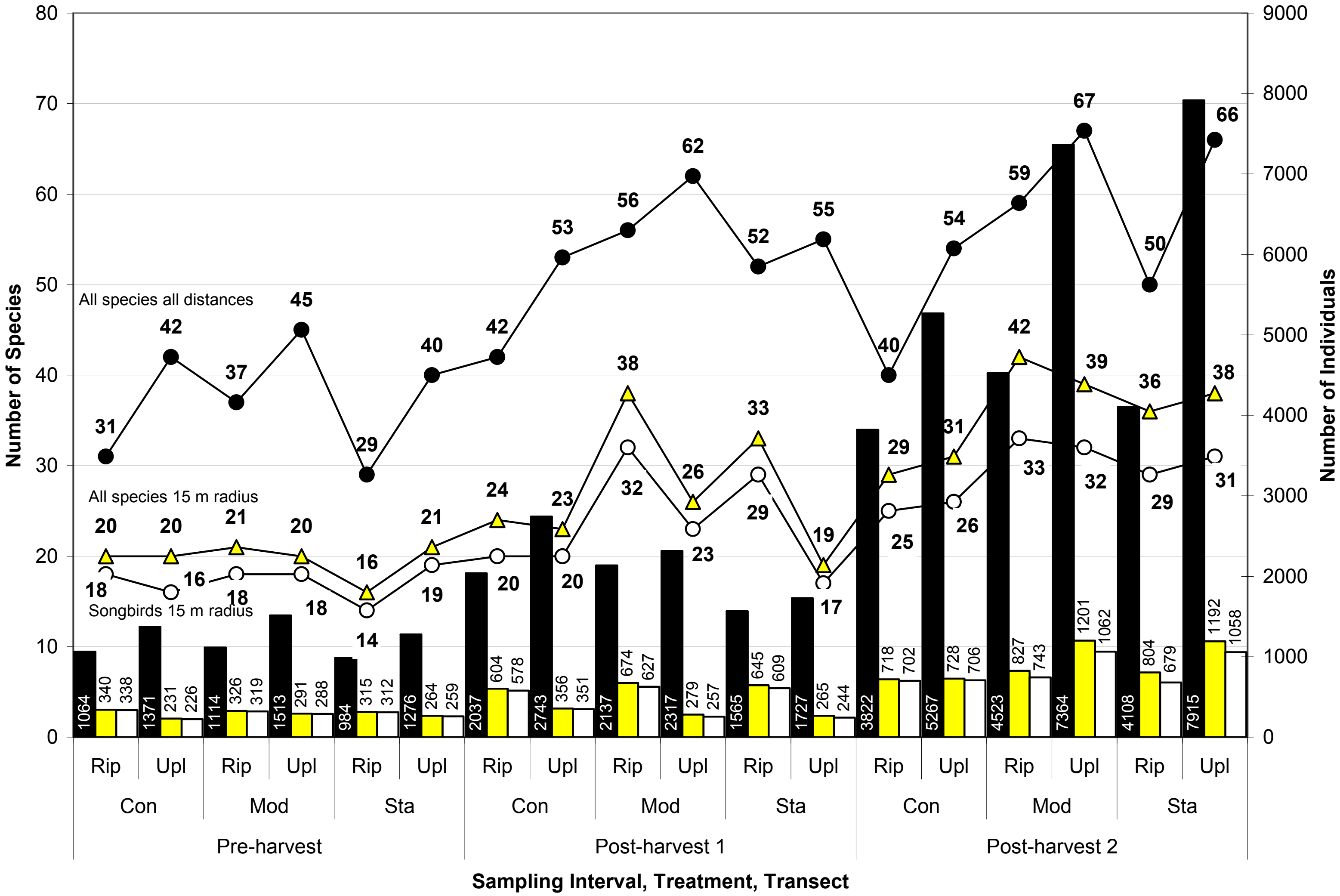


Figure 3-8. Number of species (points) and individuals (bars) detected at point counts along the riparian (Rip) and upland (Upl) transects of control (Con), modified (Mod), and state (Sta) sites during breeding-bird surveys in western Washington during three sampling intervals according to three sets of criteria, all of which exclude flyovers: (1) all bird species detected during the 0–6 min interval (black circles and bars), (2) as for 1 but only those within 15 m of the point count (yellow triangles and bars), (3) as for 2 except only songbirds (white circles and bars). Note that lines connecting numbers of species are for display purposes only (i.e., they do not reflect any trends or other relationships among numbers). Readers are cautioned that comparisons using these raw data are biased by unequal sampling effort among sites and sampling intervals (see Table 3-6).

Applying the additional species selection conditions (i.e., detected >15 times at ≥50% of sites pre-harvest, detection rate and/or the number of sites the species occurred at increased, flyovers were excluded; as described in the Methods section for those species analyzed in depth) to the 51 species of songbirds resulting from the third set of criteria above resulted in 24 species comprising 8700 individuals. This total is 16.5% of all birds detected, excluding flyovers.

Relative Abundance Comparisons

Relative abundance (mean detection rates) of songbirds (i.e., all species combined) along riparian and upland transects for each treatment type during the three sampling intervals differed significantly ($F_{17,974} = 15.3$; $p < 0.001$), with several notable patterns (Figure 3-9). Mean detection rates along the three types of riparian transects changed least over time, with none of the differences being significant (Figure 3-10). All mean detection rates declined between the pre-harvest sampling interval and the first post-harvest sampling interval; mean detection rates then increased between the first post-harvest sampling interval and the second post-harvest sampling interval, though not necessarily back to or above rates in the pre-harvest sampling interval. Note that not all changes were significant (Figure 3-10). The lowest relative abundance occurred along the upland transects of the logged sites during the first post-harvest sampling interval. The highest relative abundance occurred along the upland transects of the logged sites during the third sampling interval.

A key function of the control treatment was to allow for an interpretation of naturally occurring changes in the relative abundance of songbirds over time. The mean numbers of birds detected along riparian and upland transects at those sites are plotted for each of the three sampling intervals (Figure 3-9). Mean detection rates at unlogged sites changed modestly over time, but there were no statistically significant differences between the upland and riparian transects of control sites within any sampling period (Figure 3-10).

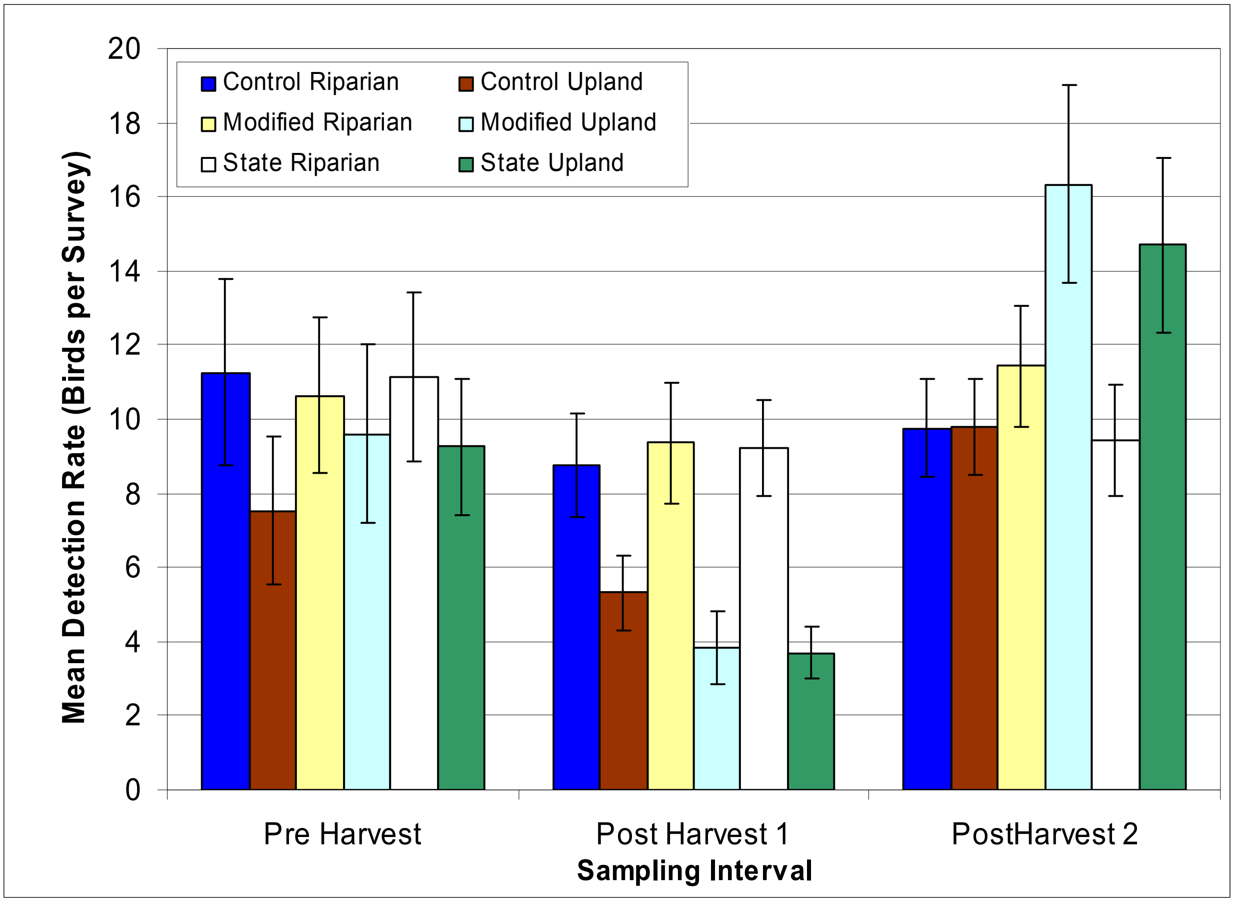
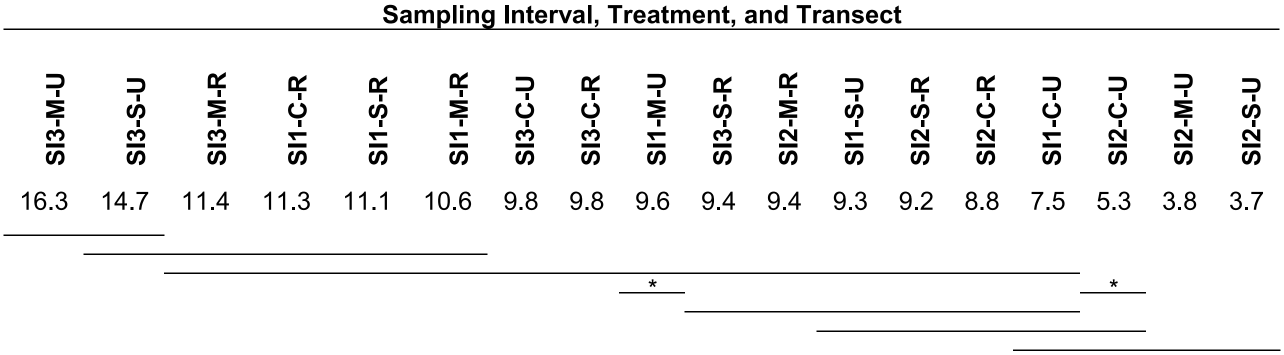


Figure 3-9. Mean (and 95% CI) number of songbirds detected within 15 m of the point count during bird surveys in western Washington for each treatment and sampling interval (excludes flyovers). Forest harvesting at modified buffer and state buffer sites occurred between the pre-harvest, two-year post-harvest, and ten-year post-harvest sampling intervals.



Sampling Interval, Treatment, and Transect

SI3-M-U	**SI3-S-U**	**SI3-M-R**	**SI1-C-R**	**SI1-S-R**	**SI1-M-R**	**SI3-C-U**	**SI3-C-R**	**SI1-M-U**	**SI3-S-R**	**SI2-M-R**	**SI1-S-U**	**SI2-S-R**	**SI2-C-R**	**SI1-C-U**	**SI2-C-U**	**SI2-M-U**	**SI2-S-U**
16.3	14.7	11.4	11.3	11.1	10.6	9.8	9.8	9.6	9.4	9.4	9.3	9.2	8.8	7.5	5.3	3.8	3.7
								*							*		

Figure 3-10. Results of comparisons of mean bird detection rates (i.e., mean number of birds per survey) according to sampling interval (SI 1–3), treatment (C = control; M = modified; S = state) and transect (R = riparian, U = upland). Means that are connected by an underline or that have an asterisk (*) on the same row are not significantly different (at α = 0.05, Tukey–Kramer HSD; unequal sample sizes). See also Figure 3-9.

The relative abundance of individual species responded in a number of different ways, not necessarily consistent with the patterns of overall relative abundance of songbirds (Appendix 3-E). Non-orthogonal contrasts (Table 3-7) of the mean relative abundance values of 24 selected[4] species of songbirds show that individual species and the group of 24 as a whole responded very differently to the different combinations of treatment and transect type. Though a change in the abundance of one species could be offset by a change in another, many patterns were generally consistent with those that might be inferred from differences evident in Figure 3-9, with the greatest number of significant differences as follows (Table 3-7):

1. **Between upland transects of the control group and the modified and state buffer treatments (logged sites) during the second post-harvest sampling interval.** Six species had higher detection rates at the control sites than at logged sites (Chestnut-backed Chickadee, Golden-crowned Kinglet, Brown Creeper, Winter Wren, Pacific-slope Flycatcher, and Hermit/Townsend's Warbler), and 10 had higher detection rates at logged sites than at control sites (Swainson's Thrush, Dark-eyed Junco, Song Sparrow, Black-headed Grosbeak, American Robin, Cedar Waxwing, Wilson's Warbler, MacGillivray's Warbler, Orange-crowned Warbler, and Spotted Towhee) (Appendix 3-E and Table 3-7) . Some species that were detected at upland transects of control sites during the pre-harvest and/or first post-harvest sampling interval were not detected there during the second post-harvest sampling interval, including MacGillivray's Warbler, Orange-crowned Warbler, and Song Sparrow. Hermit/Townsend's Warbler was detected on upland transects at modified and state buffer sites only during the first two sampling intervals.
2. **Between the upland transects of modified and state buffer sites between the pre- and combined post-harvest intervals and between the two post-harvest intervals.** At state buffer sites the mean detection rates of 16 species were significantly different pre- versus post-logging, while at modified buffer sites 15 species showed significant changes. Comparisons between the two post-harvest sampling intervals showed that at modified buffer sites, 14 species had different detection rates, while at the state buffer sites 13 species had significantly different detection rates.

   At the state buffer treatment, mean detection rates of 10 species increased in the post-logging period (Swainson's Thrush, Dark-eyed Junco, Song Sparrow, Black-headed Grosbeak, American Robin, Cedar Waxwing, MacGillivray's Warbler, Orange-crowned Warbler, Spotted Towhee, and White-crowned Sparrow), while the mean detection rates of 6 species decreased after logging (Chestnut-backed Chickadee, Golden-crowned Kinglet, Brown Creeper, Winter Wren, Pacific-slope Flycatcher, and Black-throated Gray Warbler) (Appendix 3-E and Table 3-7). When species detection rates increased on state uplands after logging, the species was not detected prior to timber harvest. The only exception was the relative abundance of Swainson's Thrush, which increased after logging. Of the 6 species showing a decrease in mean detection rate after logging, only 1 was not documented from upland transects of the state buffer treatment (Brown Creeper). All others were present, but their detection rates declined.

   At the modified buffer treatment, mean detection rates of 8 species increased in the post-logging period (Swainson's Thrush, Dark-eyed Junco, Song Sparrow, Black-headed Grosbeak, MacGillivray's Warbler, Orange-crowned Warbler, Spotted Towhee, and White-crowned Sparrow), while the mean detection rates of 7 species decreased after logging (Chestnut-backed Chickadee, Golden-crowned Kinglet, Brown Creeper, Winter

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[4] Comprising a subset of the 51 species that met the criteria for analysis (see Methods).

Wren, Pacific-slope Flycatcher, Hermit/Townsend's Warbler, and Black-throated Gray Warbler) (Appendix 3-E and Table 3-7). When species detection rates increased on modified uplands after logging, the species was not detected prior to timber harvest. Exceptions were Dark-eyed Junco and Swainson's Thrush. Mean detection rates of Dark-eyed Junco and Swainson's Thrush increased after logging. For species showing a decrease in mean detection rate after logging, none were entirely absent, although the relative abundance of some was very low.

Of the 14 species showing a significant change in mean detection rate between the first and second post-harvest sampling intervals, 7 were not documented immediately after logging on upland transects at the modified or state buffer sites (Black-headed Grosbeak, Evening Grosbeak, MacGillivray's Warbler, Orange-crowned Warbler, Swainson's Thrush, Warbling Vireo, and Wilson's Warbler). Of these species, Evening Grosbeak was not detected at the modified buffer treatment in the first post-harvest sampling interval, and both Black-headed Grosbeak and MacGillivray's Warbler were not detected on upland transects at the state buffer treatment immediately after logging. Thirteen of the 14 species showed an increase in mean detection rate 10 years after logging compared to the immediate post-treatment numbers. Only the mean detection rate of Dark-eyed Junco decreased 10 years after logging.

3. **Between the riparian transects of state buffer sites between the pre- and combined post-harvest intervals.** In this contrast, the mean detection rate increased for 8 species (Dark-eyed Junco, Song Sparrow, American Robin, Cedar Waxwing, Steller's Jay, Warbling Vireo, Wilson's Warbler, and Western Tanager) and decreased for 5 (Chestnut-backed Chickadee, Golden-crowned Kinglet, Brown Creeper, Winter Wren, and Hermit/Townsend's Warbler) (Appendix 3-E and Table 3-7). Where increases were observed, 6 species were not detected prior to logging. Where decreases were observed, only the Hermit/Townsend's hybrid was not observed after logging.
4. **Between the riparian transects of unlogged sites and logged sites during the second post-harvest sampling interval.** The mean detection rates of 11 species changed significantly along riparian transects ~10 years after logging. Along riparian transects of logged sites, lower mean detection rates were observed for 4 species (Golden-crowned Kinglet, Brown Creeper, Winter Wren, and Pacific-slope Flycatcher), while 7 species showed an increase (Song Sparrow, Black-headed Grosbeak, American Robin, Cedar Waxwing, Steller's Jay, Wilson's Warbler, and Western Tanager) (Appendix 3-E and Table 3-7). Three species (Black-headed Grosbeak, Cedar Waxwing, and Song Sparrow) were not detected at all on riparian transects of unlogged sites.

Conversely, no more than 3 of 24 species demonstrated a significant change in detection rates for eight of the contrasts (Table 3-7). Finally, very little change was evident for the riparian and upland transects of unlogged sites when the pre- and post-harvest periods were compared. In both of those comparisons, 3 primarily insectivorous species were involved (Table 3-7; Brown Creeper, Chestnut-backed Chickadee, and Golden-crowned Kinglet).

By the second post-harvest sampling interval (i.e., 10-years post-harvest), species detection rates did not differ between the upland transects of state and modified buffer sites (Table 3-7), except for Evening Grosbeak, which was least abundant on state buffer sites. Additionally, only 5 species showed a significant difference in detection rates between the riparian transects of state and modified buffer sites during the second post-harvest sampling interval.

Using data from Table 3-7, non-orthogonal contrasts that resulted in five or more significant differences within treatment types across sampling intervals are shown in Figure 3-11. Similarly, non-orthogonal contrasts that resulted in five or more significant differences among treatment types within sampling intervals are shown in Figure 3-12.

Figure 3-11 shows that most changes in bird detection rates occurred at the riparian and upland transects of modified and state buffer sites between the pre-harvest and combined post-harvest intervals (i.e., control vs. logged) and between the two post-harvest sampling intervals. Some changes between the riparian and upland transects of control sites also occurred between the two post-harvest sampling intervals.

Figure 3-12 shows that during the first sampling interval (i.e., pre-harvest) comparisons between the transect types of the three treatments revealed no differences (0 as per Table 3-7) in detection rates. During the second sampling interval, 6 of 24 species exhibited different detection rates along upland transects of the control and logged sites. During the 10-year post-harvest sampling interval, differences in detection rates along upland and riparian transects of the control and logged sites were evident. Additionally, the contrast between the riparian transects of modified and state buffer sites revealed 5 species with significantly different detection rates within the third sampling interval.

Figure 3-13 depicts the sampling intervals at each treatment type for each transect type in which the relative abundance of the 24 analyzed species was greatest. Although a number of significant differences were recorded at the logged sites over time, many of those species also exhibited significant differences at the control sites.

Table 3-7. *F*-value probabilities of non-orthogonal contrasts of bird detection rates for combinations of transect types (riparian, upland) among the three sampling intervals (SI 1–3) at each of the treatment types (control, modified, state) during breeding-bird surveys in western Washington, 1992–2004. Significant values (α = 0.05) are indicated with bold italics and shading. The number of significant values per analysis combination is shown. Not all 0.05 values are significant due to rounding for display purposes. Species codes are explained in Appendix 3-C.

Sampling Interval (SI), Transect, and Treatment Type	CBCH	GCKI	SWTH	DEJU	BRCR	SOSP	WIWR	BHGR	AMRO	CEWA	PSFL	STJA	WAVI	WIWA	MGWA	WETA	BUZZ	OCWA	SPTO	BTYW	EVGR	WCSP	GRJA	HUVI	Count Sig.
SI 3 Upland - Control vs. Mod. & State	***0.04***	***0.00***	***0.00***	***0.00***	***0.00***	***0.00***	***0.00***	***0.00***	***0.00***	***0.00***	***0.00***	0.44	0.07	***0.00***	***0.00***	0.22	***0.00***	***0.00***	***0.00***	0.21	0.08	0.09	0.26	0.84	16
State Upland - SI 1 vs SI 2&3	***0.00***	***0.00***	***0.01***	***0.00***	***0.00***	***0.00***	***0.00***	***0.00***	***0.03***	***0.02***	***0.00***	0.76	0.90	0.53	***0.00***	0.96	0.12	***0.00***	***0.00***	***0.00***	1.00	***0.04***	0.31	0.50	16
Modified Upland - SI 1 vs SI 2&3	***0.00***	***0.00***	***0.00***	***0.00***	***0.00***	***0.00***	***0.00***	***0.01***	0.06	0.09	***0.00***	0.51	0.34	0.06	***0.00***	0.96	***0.01***	***0.00***	***0.00***	***0.00***	0.33	***0.04***	0.07	0.99	15
Modified Upland - SI 2 vs SI 3	***0.00***	0.07	***0.00***	***0.00***	0.99	***0.00***	0.05	***0.00***	***0.00***	***0.00***	0.84	***0.01***	***0.04***	***0.00***	***0.00***	0.22	0.99	***0.00***	***0.00***	0.90	***0.02***	0.13	0.70	0.97	14
State Riparian - SI 1 vs SI 2&3	***0.00***	***0.00***	0.62	***0.03***	***0.01***	***0.00***	***0.03***	0.05	***0.00***	***0.02***	0.07	***0.02***	***0.00***	***0.00***	0.17	***0.04***	***0.00***	0.55	0.34	0.47	0.25	1.00	0.44	0.06	13
State Upland - SI 2 vs SI 3	***0.00***	0.85	***0.00***	***0.00***	1.00	***0.00***	0.06	***0.00***	***0.01***	***0.00***	0.75	0.22	***0.04***	***0.00***	***0.00***	***0.04***	1.00	***0.00***	***0.00***	0.48	1.00	0.11	0.12	0.13	13
SI 3 Riparian - Control vs. Modified & State	0.17	***0.00***	0.51	0.90	***0.00***	***0.00***	***0.03***	***0.02***	***0.05***	***0.01***	***0.01***	***0.00***	0.20	***0.01***	0.14	***0.03***	0.44	0.52	0.10	0.90	0.81	0.62	0.34	0.50	11
Modified Riparian - SI 1 vs SI 2&3	***0.00***	***0.01***	***0.01***	0.11	0.14	0.19	0.10	***0.03***	0.20	0.13	0.20	***0.00***	***0.01***	***0.00***	0.97	0.17	***0.00***	1.00	0.85	***0.00***	0.41	0.60	0.29	0.60	9
Modified Riparian - SI 2 vs SI 3	***0.00***	0.32	***0.00***	***0.00***	0.43	0.49	***0.02***	***0.03***	***0.05***	0.25	0.90	***0.00***	***0.00***	0.95	0.40	***0.00***	0.98	0.99	0.68	0.72	0.06	0.56	0.95	0.96	9
Control Upland - SI 2 vs SI 3	***0.00***	***0.00***	***0.02***	0.17	***0.00***	0.78	***0.00***	0.68	0.90	0.62	***0.00***	0.29	1.00	0.55	1.00	***0.00***	0.53	1.00	1.00	0.11	***0.02***	1.00	0.93	0.58	8
State Riparian - SI 2 vs SI 3	0.05	0.52	***0.05***	***0.00***	0.77	***0.04***	***0.00***	0.15	0.49	***0.04***	0.26	***0.00***	0.34	0.24	***0.01***	***0.03***	1.00	0.19	0.06	0.09	0.74	1.00	1.00	0.74	8
Control Riparian - SI 2 vs SI 3	***0.00***	***0.01***	***0.01***	***0.00***	***0.00***	0.26	0.83	0.68	0.47	0.88	***0.05***	0.55	0.65	***0.04***	0.83	0.19	0.78	1.00	1.00	0.76	0.72	1.00	0.69	0.56	7
SI 2 Upland - Control vs. Mod. & State	***0.03***	***0.00***	0.78	***0.00***	0.12	0.32	0.50	0.90	0.48	0.67	***0.01***	0.83	1.00	0.77	0.72	0.89	***0.00***	1.00	0.66	1.00	0.67	***0.00***	0.10	1.00	6
SI 3 Riparian - Modified vs. State	0.05	0.72	***0.01***	0.99	0.82	***0.02***	***0.03***	0.56	0.40	0.20	0.07	***0.03***	***0.05***	0.46	0.11	0.13	0.98	0.25	0.06	0.26	0.73	0.37	0.76	0.18	5
Control Riparian - SI 1 vs SI 2&3	***0.00***	***0.01***	0.28	0.13	***0.00***	0.59	0.31	0.69	0.53	0.64	0.22	0.87	0.90	0.27	0.75	0.56	0.08	1.00	1.00	0.17	0.60	1.00	0.37	0.41	3
Control Upland - SI 1 vs SI 2&3	***0.00***	***0.04***	0.89	0.44	***0.00***	0.91	0.92	0.85	0.97	0.80	0.69	0.39	1.00	0.11	1.00	0.36	0.78	1.00	1.00	0.40	0.15	1.00	0.09	0.80	3
SI 1 Riparian - Modified vs. State	0.11	***0.04***	0.97	1.00	0.58	0.84	0.50	1.00	0.17	1.00	0.66	0.69	1.00	0.37	0.77	0.18	0.94	1.00	1.00	0.09	0.32	1.00	0.64	1.00	1
SI 2 Riparian - Control vs. Modified & State	0.93	0.32	0.81	0.50	0.17	0.34	0.19	0.24	0.14	0.38	0.58	0.71	***0.00***	0.98	0.91	0.29	0.24	1.00	0.94	0.36	0.30	0.86	0.57	0.81	1
SI 3 Upland - Modified vs. State	0.35	0.11	0.36	0.29	0.99	0.09	0.71	0.40	0.93	0.54	0.91	0.12	0.96	0.14	0.09	0.37	0.99	0.81	1.00	0.60	***0.03***	0.89	0.19	0.16	1
SI 1 Riparian - Control vs. Modified & State	0.92	0.71	1.00	1.00	0.70	0.91	0.66	1.00	0.31	1.00	0.65	0.38	0.55	0.58	0.86	0.72	0.61	1.00	1.00	0.85	0.96	1.00	0.36	1.00	0
SI 1 Upland - Control vs. Mod. & State	0.19	0.64	0.98	0.49	0.61	1.00	0.65	1.00	0.89	0.83	0.09	0.40	0.74	0.75	1.00	0.80	0.36	1.00	1.00	0.14	0.37	1.00	0.12	1.00	0
SI 1 Upland - Modified vs. State	0.45	0.89	0.81	0.75	0.95	1.00	0.73	1.00	0.81	0.70	0.61	0.85	0.57	0.72	1.00	0.65	0.34	1.00	1.00	0.89	0.12	1.00	0.84	1.00	0
SI 2 Riparian - Modified vs. State	0.74	1.00	0.37	0.24	0.49	0.21	0.47	0.91	0.73	0.68	0.43	0.62	0.72	0.08	0.85	0.54	1.00	1.00	0.89	0.45	0.08	0.76	0.72	0.10	0
SI 2 Upland - Modified vs. State	0.87	1.00	0.77	0.43	1.00	0.39	0.65	0.84	0.99	0.88	1.00	1.00	1.00	1.00	0.53	0.81	1.00	1.00	0.87	1.00	1.00	0.95	0.76	1.00	0
Count Significant	13	12	11	10	9	9	9	8	7	7	7	7	7	7	6	6	5	5	5	3	3	3	0	0	

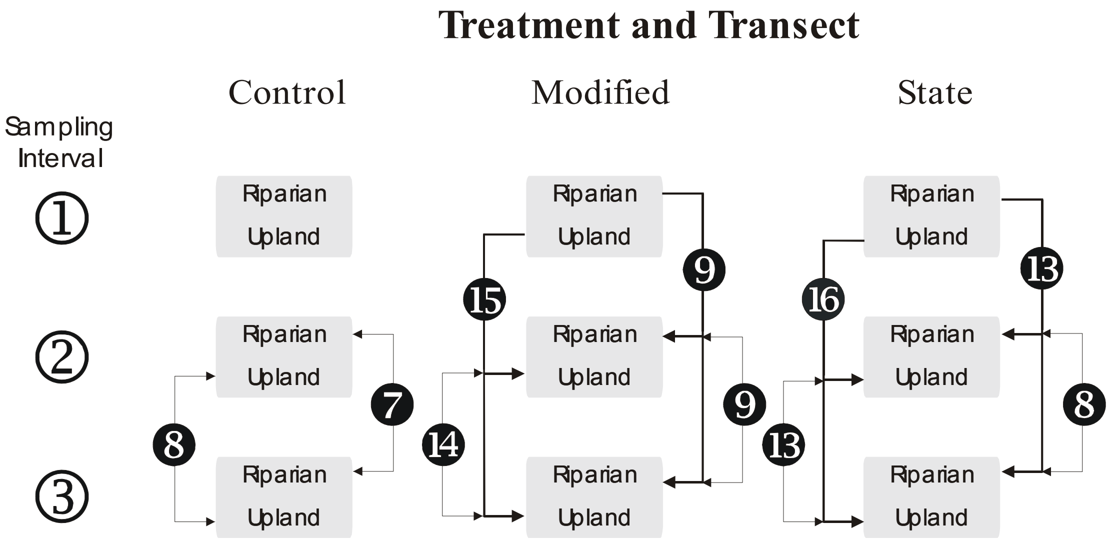


Figure 3-11. Number of species exhibiting a significant difference in mean detection rates according to contrasts for combinations of habitat types *among* the three sampling intervals for each treatment type during breeding-bird surveys in western Washington. Thicker lines denote significant contrasts between pre-harvest and a grouping of the post harvest periods. Only those contrasts with ≥5 species exhibiting significant differences are shown.

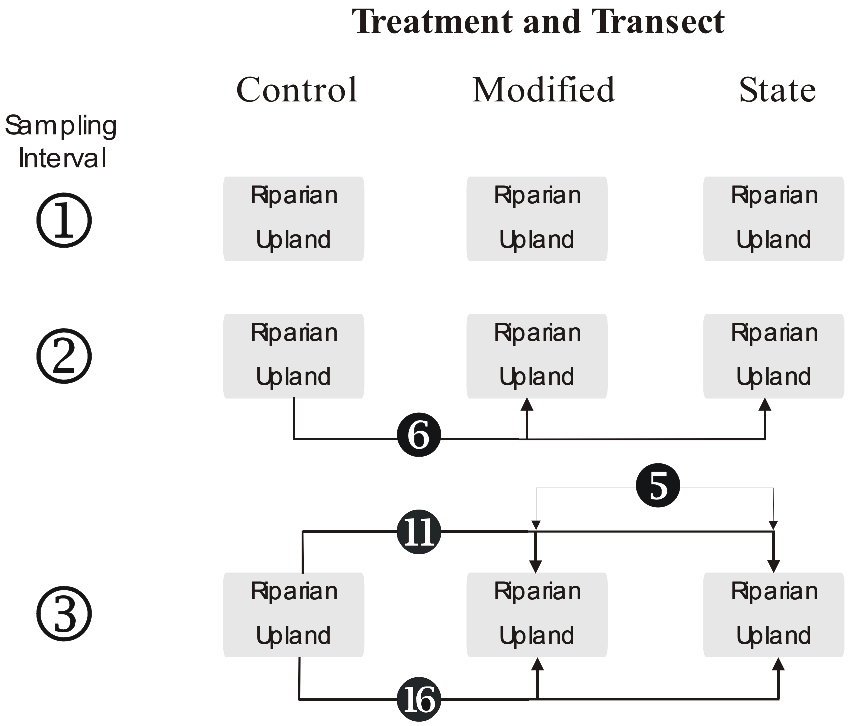
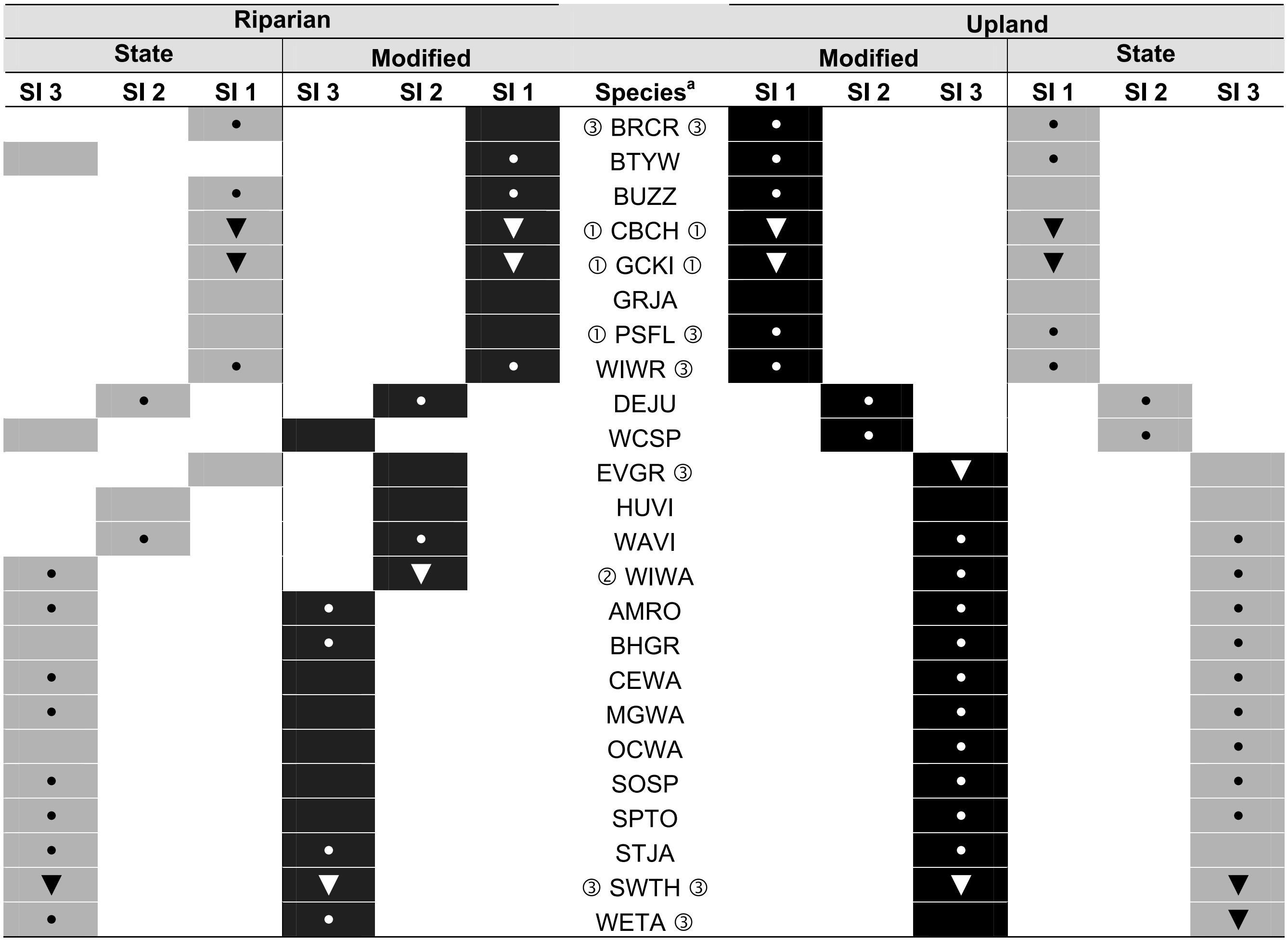


Figure 3-12. Number of species exhibiting a significant difference in mean detection rates according to contrasts for combinations of habitat types *within* each of the three sampling intervals at each of the treatment types during breeding-bird surveys in western Washington. Thicker lines with arrows denote significant contrasts between unlogged sites and a grouping of modified and state buffer sites. Only those contrasts with ≥5 species exhibiting significant differences are shown.



Riparian							Upland					
State			Modified				Modified			State		
SI 3	SI 2	SI 1	SI 3	SI 2	SI 1	Species^a	SI 1	SI 2	SI 3	SI 1	SI 2	SI 3
		▨ •			■	③ BRCR ③	■ •			▨ •		
▨					■ •	BTYW	■ •			▨ •		
		▨ •			■ •	BUZZ	■ •			▨		
		▨ ▼			■ ▼	① CBCH ①	■ ▼			▨ ▼		
		▨ ▼			■ ▼	① GCKI ①	■ ▼			▨ ▼		
		▨			■	GRJA	■			▨		
		▨			■	① PSFL ③	■ •			▨ •		
		▨ •			■ •	WIWR ③	■ •			▨ •		
	▨ •			■ •		DEJU		■ •			▨ •	
▨			■			WCSP		■ •			▨ •	
		▨		■		EVGR ③			■ ▼			▨
	▨			■		HUVI			■			▨
	▨ •			■ •		WAVI			■ •			▨ •
▨ •				■ ▼		② WIWA			■ •			▨ •
▨ •			■ •			AMRO			■ •			▨ •
▨			■ •			BHGR			■ •			▨ •
▨ •			■			CEWA			■ •			▨ •
▨ •			■			MGWA			■ •			▨ •
▨			■			OCWA			■ •			▨ •
▨ •			■			SOSP			■ •			▨ •
▨ •			■			SPTO			■ •			▨ •
▨ •			■ •			STJA			■ •			▨
▨ ▼			■ ▼			③ SWTH ③			■ ▼			▨ ▼
▨ •			■ •			WETA ③			■			▨ ▼

[a] For those species exhibiting a significant difference at control sites over time (Table 3-7), the SI with the highest mean relative abundance value for control sites (per Appendix 3-E) is indicated: riparian on left, upland on right. For example, the (significantly) highest mean relative abundance of PSFL was during SI 3 (③) at the upland and SI 1 (①) at riparian transects of control sites.

Figure 3-13. Sampling intervals (SI) with highest relative abundance (shaded cells) of 24 species by habitat type at the two logged treatments. SI 2 was the first post-harvest sampling interval. A "•" or "▼" (black or white) denotes a significant difference in comparisons between pre- and post-harvest or between the two post-harvest intervals. The significance of values marked with "▼"(black or white) as attributable to a treatment effect is suspect, as they correspond to a significant difference for that species (in the same direction—higher or lower) at the upland or riparian transects of control sites during the same sampling period. For example, the statistically significant highest relative abundance of WIWA at riparian transects of modified buffer sites during SI 2 is suspect because relative abundance of WIWA was significantly higher at riparian transects of control sites during SI 2. This concurrent peak at control sites raises doubt that the peak at modified buffer sites necessarily reflected a treatment effect.

Diversity and Community Comparisons

Plots of total numbers of songbird species detected by treatment and transect show that there was a consistent trend toward increased alpha diversity (i.e., species richness; Figure 3-14) over the three sampling intervals. Only upland transects at state buffer sites showed a decline in alpha diversity (i.e., between the pre- and first post-harvest sampling intervals).

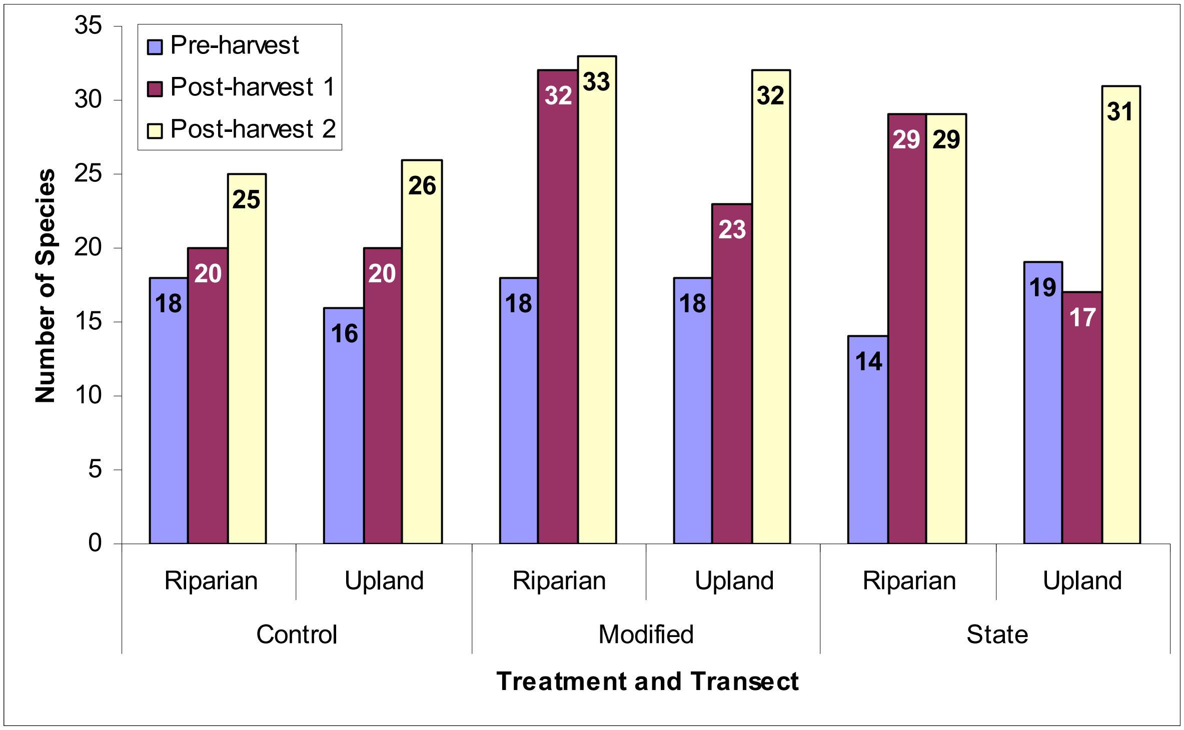


Figure 3-14. Total number of songbird species detected within 15 m of the point count during bird surveys in western Washington for each treatment type, transect, and sampling interval. Forest harvesting at modified and state buffer sites occurred between the "pre-harvest" and "post-harvest 1" sampling intervals.

The mean numbers of songbird species detected along each transect type for each treatment type during the three sampling intervals are shown in Figure 3-15. Means were significantly different ($F_{17,90} = 8.9$; $p < 0.001$). Means testing (Figure 3-16) revealed a number of significant differences, but none were evident in any comparisons during the pre-harvest sampling interval. That is, during the first sampling interval, alpha diversity was similar at all transects, showing no significant differences at any particular transect of a given treatment type. Significant differences within and between transect and treatment types first became evident during the second sampling interval and continued through the third sampling interval. Six of the eight highest mean numbers of songbird species occurred during the 10-year post-harvest sampling interval, with the two highest values occurring along the upland transects of logged sites during the 10-year post-harvest sampling interval.

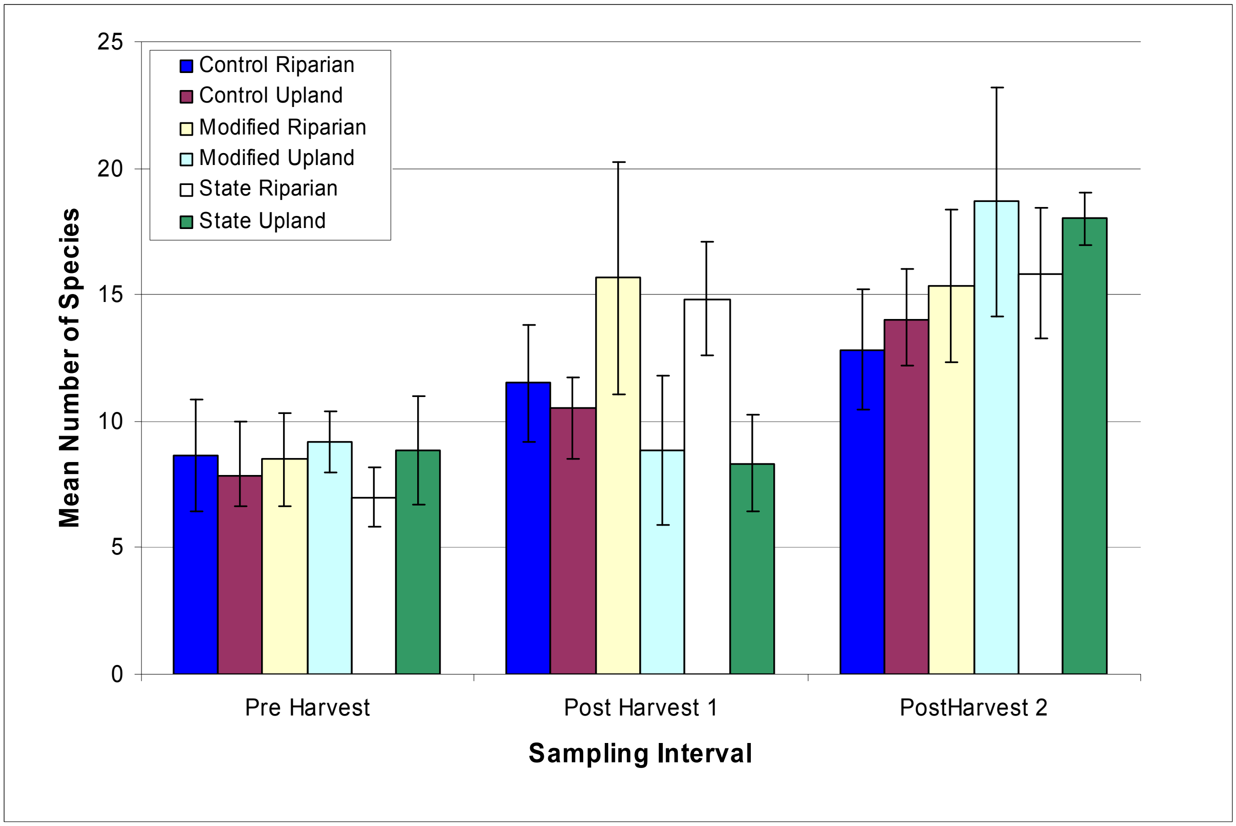
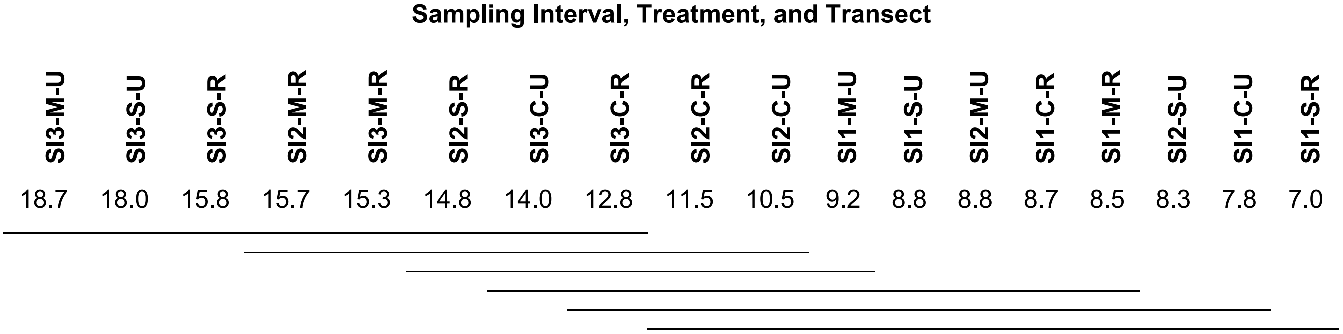


Figure 3-15. Mean number (95% CI) of songbird species detected within 15 m of the point counts along each transect type at each site (grouped by treatment type) during each of three sampling intervals in western Washington, 1992–2004 (excluding flyovers). Each mean was computed from the mean value for each transect, treatment and sampling interval (i.e., *n* = 6 for all plotted values). Forest harvesting at modified and state buffer sites occurred between the pre-harvest and first post-harvest sampling intervals.



Sampling Interval, Treatment, and Transect

SI3-M-U	SI3-S-U	SI3-S-R	SI2-M-R	SI3-M-R	SI2-S-R	SI3-C-U	SI3-C-R	SI2-C-R	SI2-C-U	SI1-M-U	SI1-S-U	SI2-M-U	SI1-C-R	SI1-M-R	SI2-S-U	SI1-C-U	SI1-S-R
18.7	18.0	15.8	15.7	15.3	14.8	14.0	12.8	11.5	10.5	9.2	8.8	8.8	8.7	8.5	8.3	7.8	7.0

Figure 3-16. Results of comparisons of mean numbers of songbird species according to sampling interval (SI 1–3), treatment (C = control; M = modified buffer; S = state buffer), and transect (R = riparian, U = upland). Means that are connected by an underline are not significantly different (at α = 0.05, Tukey–Kramer HSD; unequal sample sizes). See also

Figure 3-15.

Morisita's index of similarity (*C*) was calculated for each transect, treatment, and sampling interval using data for the 51 species of songbirds detected within 15 m of the point-count center (excluding flyovers) (Figure 3-17). The avian communities along riparian and upland transects at control sites and along the riparian transects of modified buffer sites were the most similar (stable) over time. Conversely, the upland transects of modified and state buffer sites were the most dissimilar. The community similarity of modified and state buffer treatment riparian transects was less than that of the control group in all comparisons. Along the riparian transects of modified and state buffer sites, the pre- and post-harvesting contrasts showed no appreciable changes in community similarity. Although the avian communities along upland transects of modified and state buffer sites remained relatively dissimilar (i.e., low values of Morisita's *C*), the rising trends in Morisita's *C* for those transects indicated that over time the similarity of logged sites to their pre-harvest conditions increased. Community similarity values on upland transects at modified and state buffer treatments were equally low when the first and second post-harvest sampling intervals were compared. This suggests that clearcut logging of upland habitats of the modified and state buffer sites affected the similarity of songbird communities to a similar degree, regardless of the width of the riparian buffer. Further, the low values of Morisita's *C* in those contrasts indicate that each community underwent considerable changes between the second and third sampling intervals (i.e., if the changes had been modest, higher values of Morisita's *C* would have resulted).

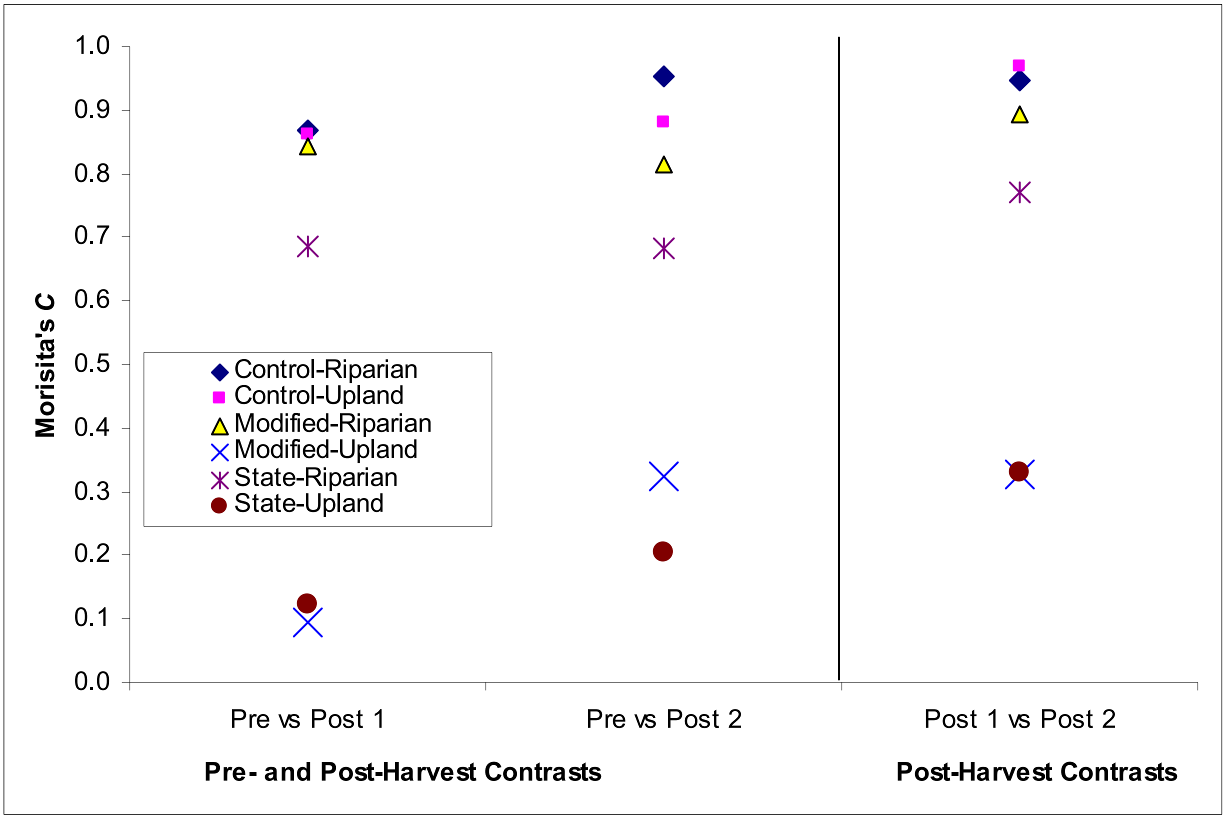


Figure 3-17. Morisita's index of similarity for songbird communities along transects (riparian, upland) at each treatment type (control, modified buffer, state buffer) for three sampling interval (SI) contrasts (Pre = pre-harvest; Post 1 = first post-harvest sampling interval; Post 2 = second post-harvest sampling interval) in western Washington, 1992–2004. As *C* approaches 1, communities are more similar.

Bird–Habitat Relationships

The co-inertia analysis (COIA) combined information on songbird communities and habitat variables measured at the sites. The biplots generated from the COIA depict the bird-habitat relationships during each sampling interval or along each transect (Figure 3-18; Figure 3-19). On each biplot, axis 1 represents the primary environmental gradient describing most of the variance; axis 2 corresponds to a secondary environmental gradient. The correlation of each environmental variable with axes 1 and 2 is indicated by its position relative to each axis with variables closer to axis 1 or 2 more strongly correlated to that particular axis; variables occupying space between are correlated with both axes. Vector length is directly proportional to the degree of correlation: positive correlation to the right of the origin for axis 1 and above the origin for axis 2; negative correlation to the left of the origin along axis 1 and below the origin along axis 2. The relative position of a given species is indicative of its response to the environmental gradients described by the axes. Species nearer the origin are indicative of (a) species with too few detections to infer habitat associations, (b) habitat generalists, or (c) species that respond to intermediate conditions along the gradients described by the axes. Species farther from the origin are more abundant in environments that are dominated by particular variables. Beyond revealing the habitat features associated with higher levels of a species' relative abundance, biplots do not provide quantitative information on the relative abundance of a species, nor do they allow for comparisons of relative abundance among species.

The biplot for all riparian habitats (Figure 3-18) revealed an environmental gradient that was strongly driven by forest openness (i.e., crown closure) along axis 1. In addition, axis 2 showed a gradient from stands with tall evergreen and berry-producing shrubs to stands with an understory dominated by small Douglas-fir. Combined, axes 1 and 2 explained 77.6% of the variation in the two data sets (axis 1: 47.8%; axis 2: 29.8%). Notable habitat feature associations of breeding birds are summarized in Table 3-8.

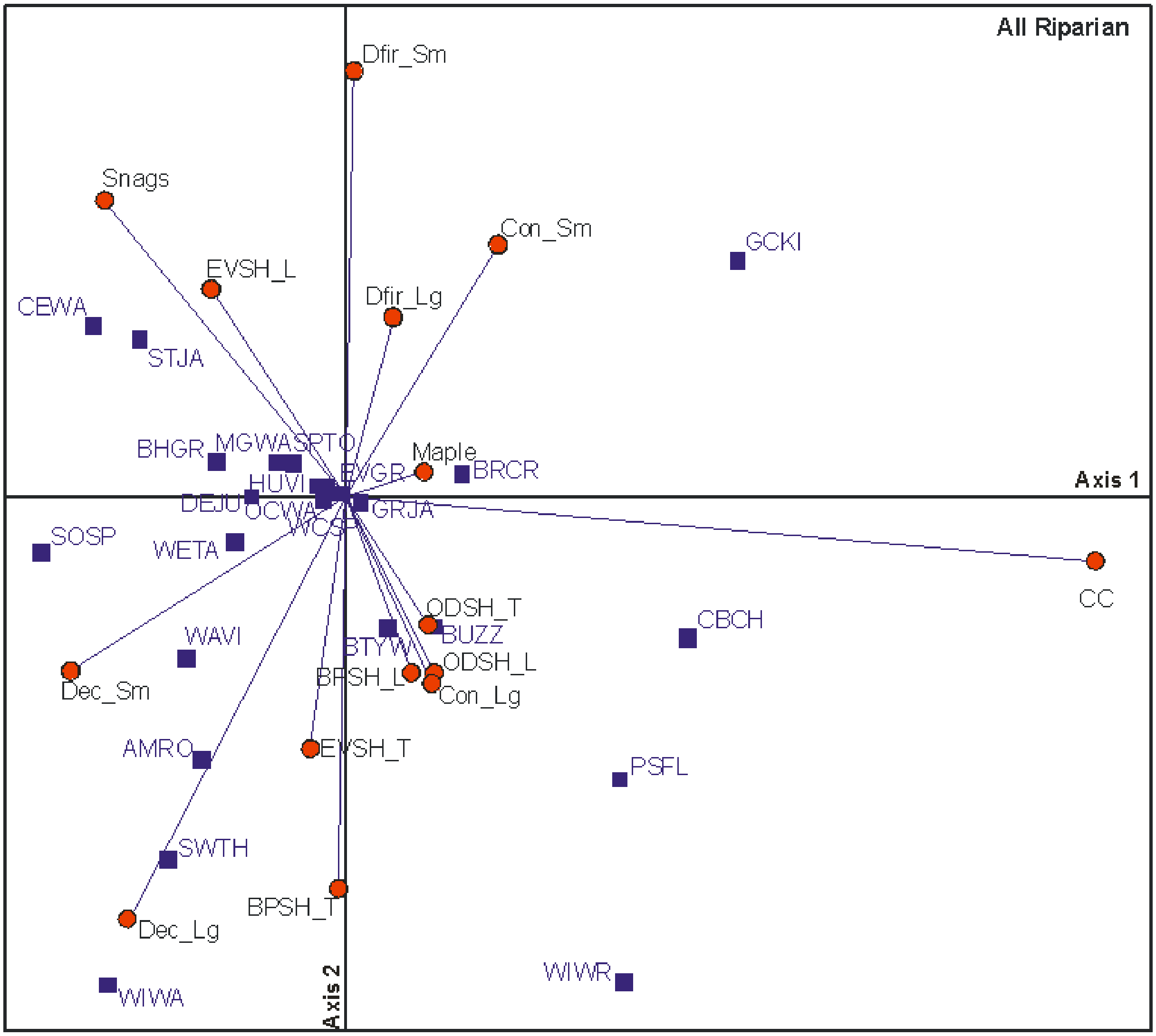


Figure 3-18. Biplot of songbird–habitat relationships for all riparian transects during the three sampling intervals, combined. Blue squares indicate species position in space; red circles indicate position of each habitat attribute. See Appendix 3-F for more information on habitat codes. CC = Canopy cover; BPSH_T = Tall berry-producing shrubs; EVSH_T = Tall evergreen shrubs; ODSH_T = Tall other shrubs; BPSH_L = Low berry-producing shrubs; EVSH_L = Low evergreen shrubs; ODSH_L = Low other shrubs; Snags = Snags; Maple = Bigleaf maple; Dec_Sm = Small deciduous trees; Dec_Lg = Large deciduous trees; Con_Sm = Small conifers; Con_Lg = Large conifers; Dfir_Sm = Small Douglas-fir; Dfir_Lg = Large Douglas-fir. AMRO = American Robin; BHGR = Black-headed Grosbeak; BRCR = Brown Creeper; BTYW = Black-throated Gray Warbler; BUZZ = Hermit/Townsend's Warbler; CBCH = Chestnut-backed Chickadee; CEWA = Cedar Waxwing; DEJU = Dark-eyed Junco; EVGR = Evening Grosbeak; GCKI = Golden-crowned Kinglet; GRJA = Gray Jay; HUVI = Hutton's Vireo; MGWA = MacGillivray's Warbler; OCWA = Orange-crowned Warbler; PSFL = Pacific-slope Flycatcher; SOSP = Song Sparrow; SPTO = Spotted Towhee; STJA = Steller's Jay; SWTH = Swainson's Thrush; WAVI = Warbling Vireo; WCSP = White-crowned Sparrow; WETA = Western Tanager; WIWA = Wilson's Warbler; WIWR = Winter Wren.

Table 3-8. Notable habitat associations of songbird species detected within 15 m of the point counts along riparian transects during all sampling intervals combined (see Figure 3-18). Species presented in Figure 3-18 but not listed here are considered generalists with regard to the gradients depicted by the two axes of that figure.

Species[a]	Notable Habitat Feature Associations
SOSP	Open stands with a deciduous component and with the least canopy cover
WIWA; SWTH; AMRO; WAVI	Stands dominated by large deciduous trees and an understory of deciduous trees and tall evergreen and berry-producing shrubs
WIWR; PSFL	Mixed-conifer (including Douglas-fir) and deciduous stands with high canopy cover
GCKI	Coniferous (including Douglas-fir) stands with high canopy cover
CBCH	Stands with large conifers, "other" deciduous shrubs, and high canopy cover
STJA; CEWA	Open stands with low evergreen shrubs, snags, and small Douglas-fir

[a] SOSP = Song Sparrow; WIWA = Wilson's Warbler; SWTH = Swainson's Thrush; AMRO = American Robin; WAVI = Warbling Vireo; WIWR = Winter Wren; PSFL = Pacific-slope Flycatcher; GCKI = Golden-crowned Kinglet; CBCH = Chestnut-backed Chickadee; STJA = Steller's Jay; CEWA = Cedar Waxwing

The biplot for all upland habitats (Figure 3-19) revealed an environmental gradient that was driven by the openness of conifer-dominated forests. Axis 2 showed a gradient from stands comprising small Douglas-fir, small deciduous trees, and berry-producing and evergreen shrubs. The two axes explained nearly all of the variation in the data (axis 1: 73.2%; axis 2: 24.4%). Notable habitat feature associations of breeding birds are summarized in Table 3-9.

Although sample sizes (total number of individuals) varied greatly, bird species occurred along a continuum from those that were abundant mostly in riparian areas to those that used upland areas almost exclusively (Figure 3-20). At least 75% of Red-breasted Sapsucker (RBSA), Olive-sided Flycatcher (OSFL), Ruffed Grouse (RUGR), Northern Flicker (NOFL), Warbling Vireo (WAVI), Hutton's Vireo (HUVI), Winter Wren (WIWR), Hairy Woodpecker (HAWO), and Pacific-slope Flycatcher (PSFL) were recorded along the riparian transects, whereas ≤25% of House Wren (HOWR), Black-capped Chickadee (BCCH), Dark-eyed Junco (DEJU), MacGillivray's Warbler (MGWA), Bewick's Wren (BEWR), Red Crossbill (RECR), Spotted Towhee (SPTO), White-crowned Sparrow (WCSP), Orange-crowned Sparrow (OCSP), Willow Flycatcher (WIFL), Common Yellowthroat (COYE), and American Goldfinch (AMGO) occurred there.

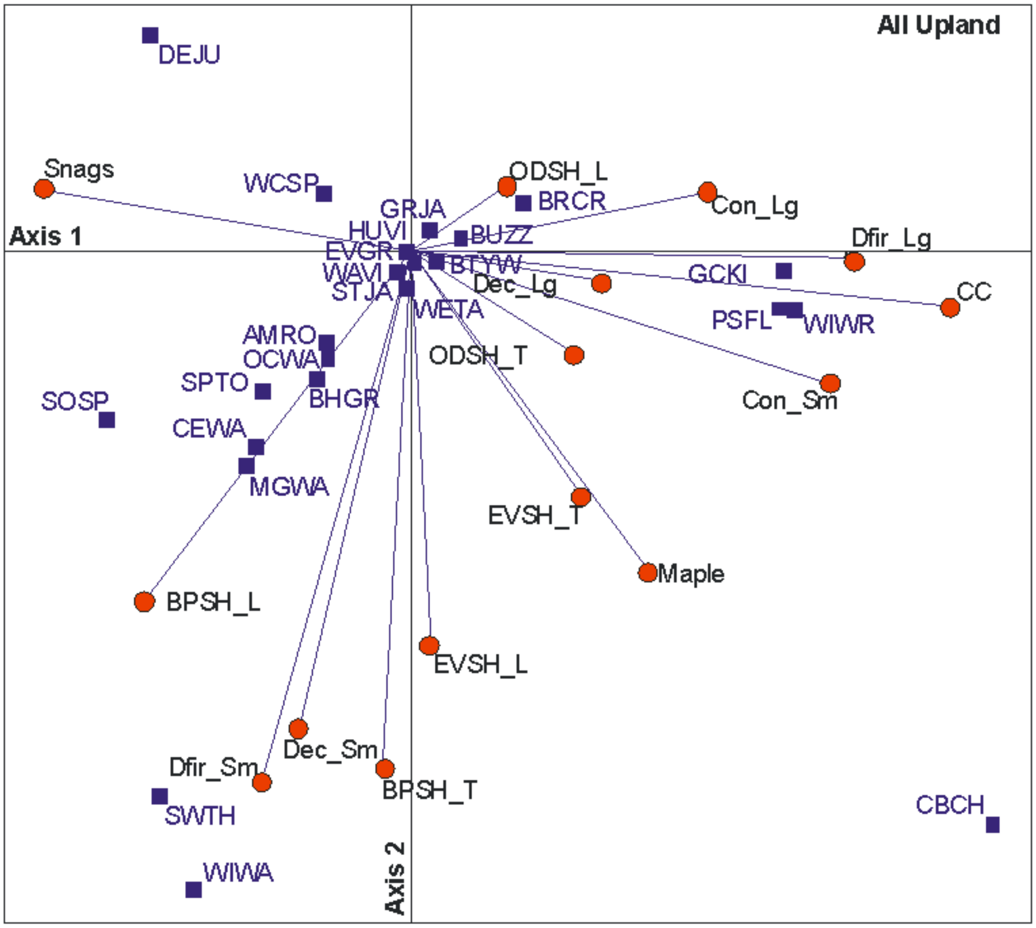


Figure 3-19. Biplot of songbird–habitat relationships for all upland transects during the three sampling intervals, combined. Blue squares indicate species position in space; red circles indicate position of each habitat attribute. See Appendix 3-F for more information on habitat codes. CC = Canopy cover; BPSH_T = Tall berry-producing shrubs; EVSH_T = Tall evergreen shrubs; ODSH_T = Tall other shrubs; BPSH_L = Low berry-producing shrubs; EVSH_L = Low evergreen shrubs; ODSH_L = Low other shrubs; Snags = Snags; Maple = Bigleaf maple; Dec_Sm = Small deciduous trees; Dec_Lg = Large deciduous trees; Con_Sm = Small conifers; Con_Lg = Large conifers; Dfir_Sm = Small Douglas-fir; Dfir_Lg = Large Douglas-fir. AMRO = American Robin; BHGR = Black-headed Grosbeak; BRCR = Brown Creeper; BTYW = Black-throated Gray Warbler; BUZZ = Hermit/Townsend's Warbler; CBCH = Chestnut-backed Chickadee; CEWA = Cedar Waxwing; DEJU = Dark-eyed Junco; EVGR = Evening Grosbeak; GCKI = Golden-crowned Kinglet; GRJA = Gray Jay; HUVI = Hutton's Vireo; MGWA = MacGillivray's Warbler; OCWA = Orange-crowned Warbler; PSFL = Pacific-slope Flycatcher; SOSP = Song Sparrow; SPTO = Spotted Towhee; STJA = Steller's Jay; SWTH = Swainson's Thrush; WAVI = Warbling Vireo; WCSP = White-crowned Sparrow; WETA = Western Tanager; WIWA = Wilson's Warbler; WIWR = Winter Wren.

Table 3-9. Notable habitat associations of bird species detected within 15 m of the point counts along upland transects during all sampling intervals combined (see Figure 3-19). Species presented in Figure 3-19 but not listed here are considered generalists with regard to the gradients depicted by the two axes of that figure.

Species[a]	**Notable Habitat Feature Associations**
GCKI; PSFL; WIWR	Conifer- and Douglas-fir–dominated stands with high crown closure
CBCH	Conifer- and Douglas-fir–dominated stands having high crown closure and with maple, evergreen shrubs, and berry-producing shrubs
WIWA; SWTH; SOSP; MGWA; CEWA; SPTO; BHGR; OCWA; AMRO	Open stands of small Douglas-fir, small deciduous trees, and with evergreen and berry-producing shrubs
DEJU; WCSP	Open areas with snags, fewer trees, and low levels of shrubby ground cover

[a] GCKI = Golden-crowned Kinglet; PSFL = Pacific-slope Flycatcher; WIWR = Winter Wren; CBCH = Chestnut-backed Chickadee; WIWA = Wilson’s Warbler; SWTH = Swainson’s Thrush; SOSP = Song Sparrow; MGWA = MacGillivray’s Warbler; CEWA = Cedar Waxwing; SPTO = Spotted Towhee; BHGR = Black-headed Grosbeak; OCWA = Orange-crowned Warbler; AMRO = American Robin; DEJU = Dark-eyed Junco; WCSP = White-crowned Sparrow.

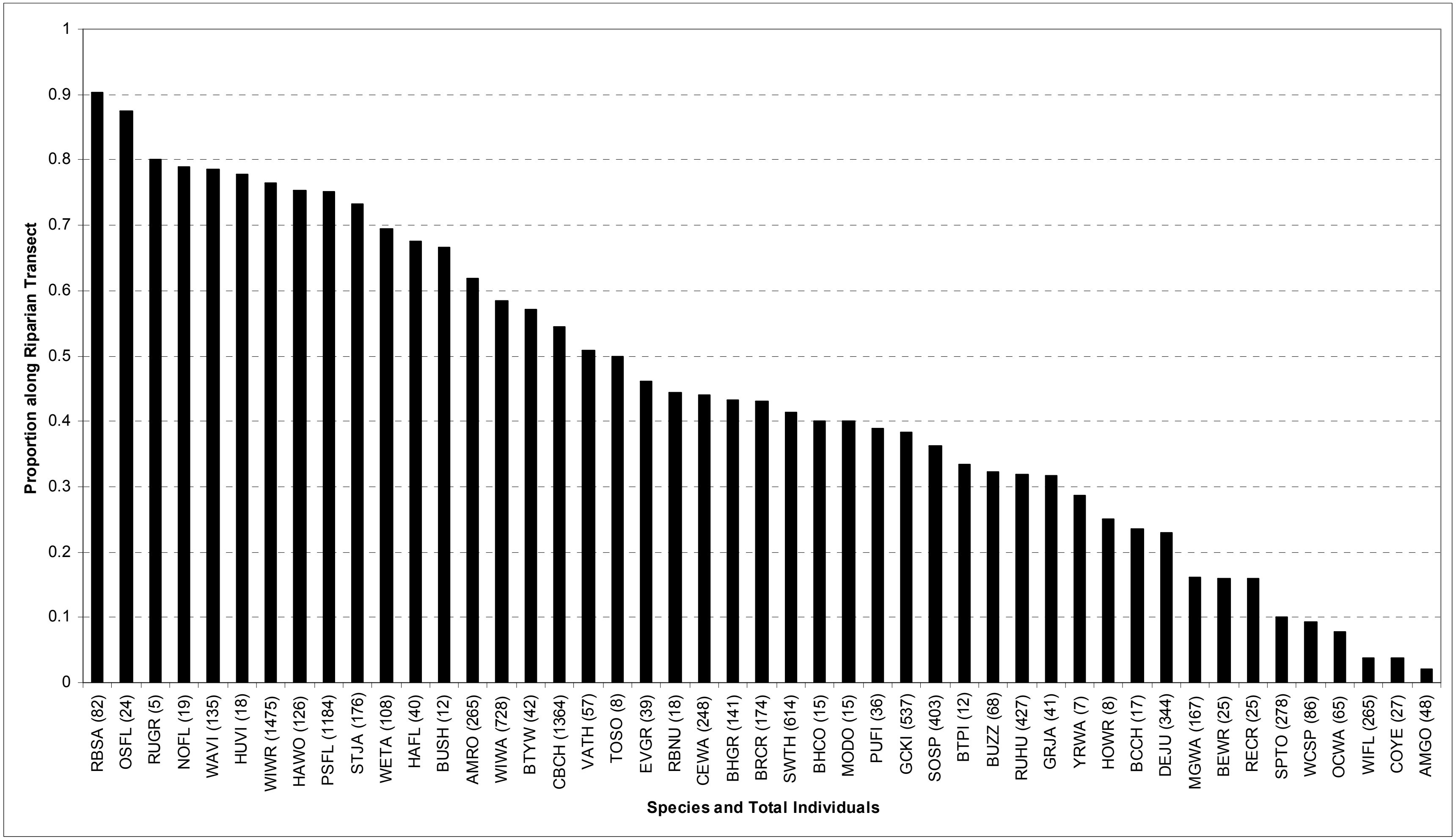


Figure 3-20. Proportion of bird species detected along riparian transects during the three (pooled) sampling intervals (1992–2004) at the three (pooled) treatment types during breeding-bird surveys in western Washington (from Appendix 3-D). Data include detections within the 0–6 min interval and within 15 m of the point count, but exclude flyovers. Total number of individuals detected follows species code (see Appendix 3-C for a list of codes). Only species detected ≥5 times are shown.

Discussion

Natural Variation in Relative Abundance at Control Sites

Birds surveyed during this study are subjected to a wide range of factors that can influence their demographics. Many such factors have little or nothing to do with the structural habitat conditions on their breeding grounds. Weather, predation, disease, and habitat conditions of non-breeding habitats can directly or indirectly affect bird populations. Jones et al. (2005) found that tree nesters and foliage gleaners exhibited spatial synchrony in abundance over distances of tens of kilometers and that neotropical migrants exhibited higher spatial synchrony in abundance than did short-distance migrants or year-round residents. They found that such synchrony within and among six species of long-distance, migratory, insectivorous birds was correlated with the abundance of lepidopteron larvae, a key prey item. In turn, the abundance of larvae has been linked to variations in climatic conditions. One purpose of the control sites in this study is to elucidate this variability over time.

During the course of this study, it is likely that conditions within the study area had the greatest effect on numbers of birds. Although food was not directly measured during this study, it is notable that of the three species (Golden-crowned Kinglet, Chestnut-backed Chickadee, and Brown Creeper) that exhibited a significant difference in relative abundance along the riparian and upland transects of control sites after logging (Table 3-7), all are highly insectivorous (especially during the breeding season) year-round residents of western Washington's forests (Ingold and Galati 1997; Dahlsten et al. 2002; Hejl et al. 2002). For these three species, at control sites there was a general trend of decline following the pre-harvest sampling interval, followed by an increase between the first and second post-harvesting sampling intervals (Appendix 3-E). This general pattern was also reflected by the mean detection rate of all birds at control sites during the course of the study (Figure 3-9); this anecdotally supports the possibility that a non-structural habitat factor (e.g., food availability) influenced some breeding-bird populations over the course of the study more so than any successional changes in habitat structure or composition. In comparing control sites between the two post-harvest sampling intervals, the relative abundance values of a number of other species were also significantly different: Pacific-slope Flycatcher differed along both transect types; Evening Grosbeak, Western Tanager, and Winter Wren differed along upland transects; and Dark-eyed Junco and Wilson's Warbler differed along riparian transects. The relative abundance of these species all increased 10 years after logging except for the latter two.

Relative Abundance and Habitat Relationships

The absence of significant differences in relative abundance of species among the transects of the three site types before logging is important for the experimental layout of the sites because it suggests a lack of any location (site) effects for the treatment types. Examination of habitat conditions for the upland transects of each treatment type shows that the pre-harvest conditions of the modified and state buffer sites were very similar to the control sites and that the control sites remained relatively constant across time (Figure 3-21). Clustering of the post-harvest sampling interval data for the modified and state buffer sites provides evidence that the logging prescriptions produced ecologically similar results with regard to habitat structure over time. This is exactly what would be expected considering that forest harvest prescriptions along upland transects were expected to be the same at modified and state buffer sites. Immediately after harvesting, the upland transects were largely devoid of vegetation. By 10 years later, modified and state buffer sites showed clear signs of succession, supporting an increased abundance of Douglas-fir, deciduous trees, and berry-producing shrubs.

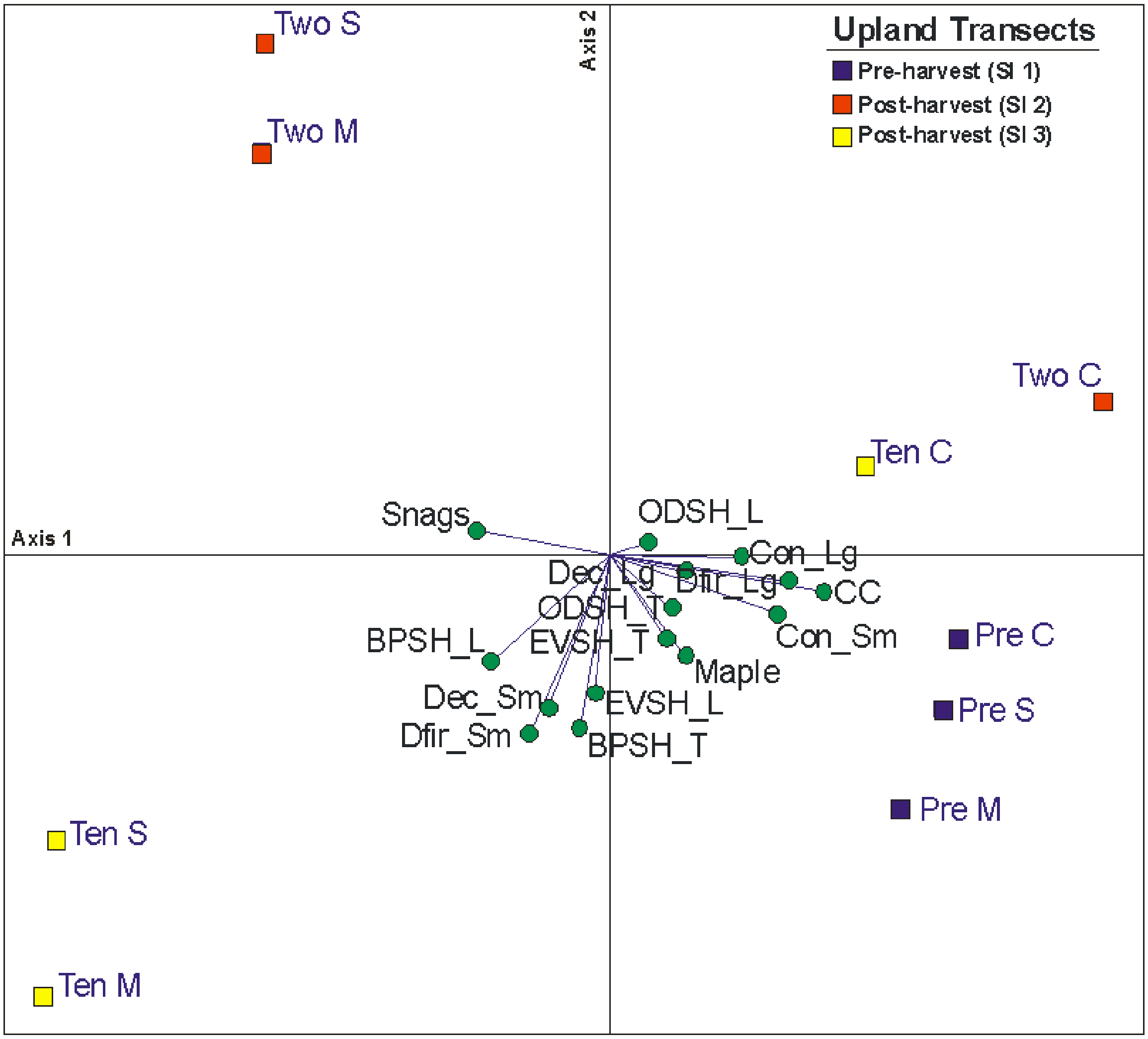


Figure 3-21. Biplot of upland transect habitat condition by treatment and time interval. The data in this figure represent the mean condition as derived from the normalized score of the PCA on the habitat data matrix. See Appendix 3-F for more information on habitat codes. Pre = Sampling Interval (SI) 1; Two (years post-harvest) = SI 2; Ten (years post-harvest) = SI 3. CC = Canopy cover; BPSH_T = Tall berry-producing shrubs; EVSH_T = Tall evergreen shrubs; ODSH_T = Tall other shrubs; BPSH_L = Low berry-producing shrubs; EVSH_L = Low evergreen shrubs; ODSH_L = Low other shrubs; Snags = Snags; Maple = Bigleaf maple; Dec_Sm = Small deciduous trees; Dec_Lg = Large deciduous trees; Con_Sm = Small conifers; Con_Lg = Large conifers; Dfir_Sm = Small Douglas-fir; Dfir_Lg = Large Douglas-fir.

The situation along riparian transects was not as straightforward, probably due to factors such as comparatively greater heterogeneity in vegetation structure along riparian transects both within and among treatment types. Control sites remained fairly static, being characterized primarily by high canopy closure (Figure 3-22). Pre-harvest conditions at modified and state buffer sites were similar with regard to canopy cover, but state buffer sites on average supported greater amounts of Douglas-fir and had lesser amounts of deciduous tree cover and shrub cover than control and modified buffer sites at all time intervals. This latter difference is indicative of a site effect rather than a treatment effect. The greatest axis 1 difference for state buffer sites was that crown closure declined 10 years after logging relative to the immediate post-harvest sampling interval, possibly as a result of blowdown (which was evident at some sites within the state buffer treatment). At modified buffer sites, a temporal response along axis 2 was observed most likely as a result of increases in berry-producing and other deciduous shrubs, deciduous trees, and tall evergreen shrubs. It is possible that harvesting in the adjacent upland areas created an "edge effect" in the riparian buffers, whereby factors such as increased light penetration (see Harper et al. 2005) promoted the observed increases in understory growth. If site effects were indeed prevalent, then the lack of a similar increase in understory growth at state buffer sites could have reflected reduced rates of seed dispersal due to a reduced abundance of such plants to begin with. It is also possible that edaphic (i.e., soil nutrient and moisture) conditions at state buffer sites were sufficiently different to produce a different form or rate of forest succession compared to modified buffer sites.

For riparian transects, relative abundance of songbirds over time was fairly consistent among treatment types and, as mentioned above, showed a consistent trend: decreasing immediately after logging relative to pre-harvest numbers and then increasing 10 years after logging relative to the immediate post-harvest period. Although relative abundance of songbirds was lowest at the upland transects of modified and state buffer sites immediately after logging, it was not different from that at the control treatment during the same interval (Figure 3-9). As shown for the control group, the pattern of declining relative abundance of songbirds immediately after logging with an increase in the 10-year post-harvest sampling interval occurred at all treatments and transect types. This also supports the hypothesis that regional numbers of some bird species declined and then increased synchronously during the study.

Evidence of a treatment effect was greatest for the upland transects of modified and state buffer sites. In both cases, a decline in overall mean relative abundance immediately after logging was followed by a substantial increase in the 10-year post-harvest sampling interval (Figure 3-9). Despite this increase, not all species had recovered to pre-harvest levels by that point in time (e.g., Brown Creeper, Pacific-slope Flycatcher, and others near the top of Figure 3-13).

The following three subsections discuss changes in the relative abundance of selected species that likely resulted from experimental effects. As such, discussion in these subsections centers on how forest harvesting and subsequent succession along the upland and riparian transects of modified and state buffer sites could have influenced relative abundance. Except where noted, it is assumed that both modified and state buffer sites are under discussion. Further, the discussion focuses on species shown in Figure 3-13 with unconfounded, significant peaks in relative abundance. Graphic depictions of how habitat conditions for each treatment and transect changed over time are shown in Figure 3-21 and Figure 3-22.

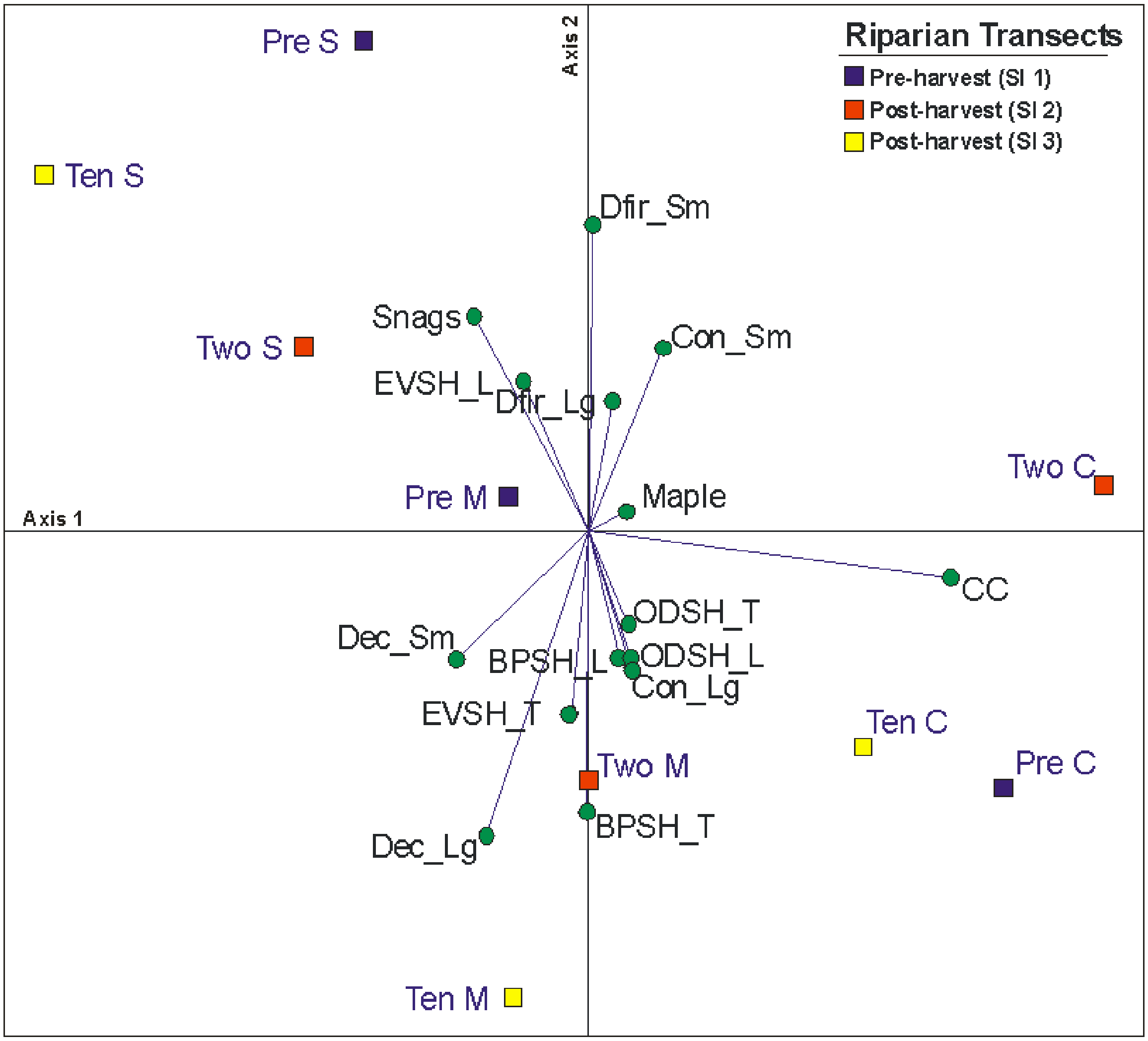


Figure 3-22. Biplot of riparian transect habitat condition by treatment and time interval. The data in this figure represent the mean condition as derived from the normalized score of the PCA on the habitat data matrix. See Appendix 3-F for more information on habitat codes. Pre = Sampling Interval (SI) 1; Two (years post-harvest) = SI 2; Ten (years post-harvest) = SI 3. CC = Canopy cover; BPSH_T = Tall berry-producing shrubs; EVSH_T = Tall evergreen shrubs; ODSH_T = Tall other shrubs; BPSH_L = Low berry-producing shrubs; EVSH_L = Low evergreen shrubs; ODSH_L = Low other shrubs; Snags = Snags; Maple = Bigleaf maple; Dec_Sm = Small deciduous trees; Dec_Lg = Large deciduous trees; Con_Sm = Small conifers; Con_Lg = Large conifers; Dfir_Sm = Small Douglas-fir; Dfir_Lg = Large Douglas-fir.

Species Most Abundant during the Pre-harvest Sampling Interval at Modified and State Buffer Sites

The relative abundance of several species was highest before logging along the riparian and/or upland transects: Brown Creeper (BRCR), Black-throated Gray Warbler (BTYW), Hermit/Townsend's Warbler (BUZZ), Pacific-slope Flycatcher (PSFL), and Winter Wren (WIWR). Relative abundance of Chestnut-backed Chickadee (CBCH) and Golden-crowned Kinglet (GCKI) also peaked along upland and riparian transects, but those peaks were confounded by similar significant peaks in relative abundance at control sites, suggesting that one or more unknown regional effects had a greater effect on relative abundance than any treatment effects—particularly regarding Chestnut-backed Chickadee (see Appendix 3-E).

Brown Creeper is a resident species whose breeding habitat is characterized by late successional stages of coniferous forests and mixed coniferous–deciduous forests. In the Pacific Northwest, it prefers mesic conifer forests with high canopy cover. Large trees and snags provide foraging (primarily invertebrates) and nesting (primarily in bark crevices) habitat (Hejl et al. 2002). Pre-harvest, high canopy cover, and the availability of large trees along the upland transects provided Brown Creeper with the habitat conditions that allowed the species to reach peak relative abundance during this study. The decline of those habitat attributes post-harvest, in light of the abundance of this species at control sites during the 10-year post-harvest sampling interval is consistent with the observed reduction in Brown Creeper after logging (i.e., in both post-harvest sampling intervals). Considering the information in Figure 3-22, the decline in relative abundance of Brown Creeper along riparian transects after logging likely reflected forestry-related changes in the adjacent upland areas more so than any changes to the riparian transect per se.

Black-throated Gray Warbler is a neotropical migrant songbird that nests in trees, gleans insects from foliage, and has been shown to prefer second-growth coniferous forests ~50–60 years old (Bryant et al. 1993; Guzy and Lowther 1997). In the Coast Range of Oregon, abundance of Black-throated Gray Warbler declined following the thinning of ~35- to 45-year-old Douglas-fir forests (Hayes et al. 2003). Because little is known about many aspects of the Black-throated Gray Warbler's natural history, including its breeding biology (Guzy and Lowther 1997), it is difficult to say what factor(s) caused the post-harvest reduction in abundance of this species. Considering the general similarity in trends of relative abundance over time along the two transects of the control and treatment sites, it is possible that this species was responding to habitat conditions or regional effects that were independent of the experiment.

Hermit Warbler and Townsend's Warbler (combined due to local hybridization [Rohwer and Wood 1998]) are migrants that prefer coniferous forests with high canopy volumes where they glean invertebrates from foliage and nest high above the ground (Pearson 1997; Wright et al. 1998). In the Cascade Mountains of Washington, Hermit Warbler is most abundant in young (35–79 years) and relatively dry old-growth (200 years) stands dominated by Douglas-fir, and is less abundant in mature forest stands (Manuwal 1991). It is largely absent from stands with a large component of western hemlock, western redcedar, and mountain hemlock (Chappell and Ringer 1983). The post-treatment declines in these bird species along the upland transects of modified and state buffer sites most likely reflected reductions in crown closure and availability of large coniferous (including Douglas-fir) trees in the upland areas.

Pacific-slope Flycatcher is a migratory forest songbird that feeds almost exclusively on insects caught in the air or gleaned from foliage of trees and shrubs (Lowther 2000). In coastal Washington, it occurs in dense hemlock-fir conifer forests and in deciduous growth of riparian areas (Lowther 2000). In this study, the high crown closure and an abundance of large conifers (including Douglas-fir) were the most likely reasons for the peak in relative abundance of Pacific-slope Flycatcher along upland transects during the pre-harvest sampling interval. The substantial reductions in crown closure and availability of large conifers (including Douglas-fir) most likely caused the number of Pacific-slope Flycatcher to decline to negligible levels along the upland transects post-harvesting.

Winter Wren is a migratory, partially migratory, and resident songbird. Its breeding habitat is characterized by large conifers (including Douglas-fir), coarse woody debris, and proximity to freshwater (Hejl et al. 2002). Invertebrates, and especially insects, are important foods that are captured during searches on the ground, on woody debris, on low vegetation, and along streams (especially when snow cover is present) (Van Horne and Bader 1990; Hejl et al. 2002). In this study, the high crown closure and an abundance of large conifers (including Douglas-fir) were the most likely reasons for the higher relative abundance of Winter Wren observed prior to logging. The substantial reductions in crown closure and availability of large conifers (including Douglas-fir) at upland areas most likely caused the number of Winter Wren to decline to negligible levels along the riparian and upland transects post-harvest. Considering the information on important habitat parameters for Winter Wren as shown in Figure 3-22, the post-harvest decline in relative abundance of Winter Wren along riparian transects likely reflected habitat changes in the adjacent upland areas more so than any changes to the riparian transect per se.

The change in relative abundance observed for Golden-crowned Kinglets between logged and unlogged sites in the 10-year post-harvest sampling interval is likely attributable to a treatment effect. Golden-crowned Kinglet is resident in the study area and feeds primarily on soft-bodied arthropods and their eggs (Ingold and Galati 1997). This species nests high in the trees of coniferous, mixed coniferous–deciduous, and deciduous forests with closed or open canopies (Beedy 1981; Peck and James 1987). Understory appears to be an unimportant habitat attribute (Franzreb and Ohmart 1978; Titterington et al. 1979; Wetmore et al. 1985). In this study, the high crown closure and an abundance of large conifers (including Douglas-fir) were the most likely reasons for the higher relative abundance of Golden-crowned Kinglet observed prior to logging. The substantial reductions in crown closure and availability of large conifers (including Douglas-fir) at upland locations most likely caused the number of Golden-crowned Kinglet to decline along riparian and upland transects post-harves.

Species Most Abundant 2 Years after Logging at Modified and State Buffer Sites

Only 2 species were most abundant along upland transects of modified and state buffer sites during the 2-year post-harvest sampling interval (Figure 3-13): Dark-eyed Junco and White-crowned Sparrow. Along riparian transects, only the Warbling Vireo was most abundant during the 2-year post-harvest sampling interval.

Habitat conditions during the immediate post-harvest period were ideal for White-crowned Sparrow, which is a ground- and low-shrub-nesting, resident or short-distance migrant songbird. This species prefers edges and open, patchy areas of grass and bare ground in order to forage on seeds and insects; dense shrubs or small, thick conifers for roosting and nest concealment; standing or running water on or near its territory; and tall coniferous trees, generally on the periphery of its territory (DeWolfe and DeWolfe 1962). By creating patches of bare ground and grass at previously forested sites, forestry operations can provide new habitat for White-crowned Sparrows (DeWolfe and DeWolfe 1962; Chilton et al. 1990). Forest harvesting along the upland transects of modified and state buffer sites produced suitable habitat conditions for White-crowned Sparrows during the first post-harvest sampling interval. Ten years after logging, habitat conditions declined in suitability—most likely due to increases in the height and percentage cover of trees and shrubs.

Dark-eyed Junco is a habitat generalist and ground-nesting songbird that is resident throughout much of northern and western North America (Nolan et al. 2002). In general, it prefers edges and open areas similar to those described above for White-crowned Sparrow. Steventon et al. (1998) found that Dark-eyed Junco abundance 1–2 years after conifer harvesting was highest on those sites where the greatest amount of timber (percent cover) was removed. This same pattern of peak Dark-eyed Junco abundance 1–2 years post-harvest was also documented by Hayes et al. (2003). At modified and state buffer sites, Dark-eyed Junco was also most abundant immediately post-harvest. Ten years after logging, habitat conditions declined in suitability—most likely due to increases in the height and percentage cover of trees and shrubs.

Warbling Vireo is a neotropical migrant songbird that prefers nesting in deciduous trees (Gardali and Ballard 2000), is strongly associated with mature mixed deciduous woodlands in riparian areas and occurs near forest edges and interiors (Gardali and Ballard 2000). Along riparian transects, Warbling Vireo was most abundant immediately after logging. The best explanation for the observed increase is the creation of edge habitat, as the immediate post-logging sampling interval occurred too soon after harvesting for there to be any meaningful succession-based changes in habitat structure. This is further substantiated by the stable numbers at control transects over time (Appendix 3-E).

Species Most Abundant 10 Years after Logging at Modified and State Buffer Sites

Along upland transects the relative abundance of 7 species was highest during the second post-harvest sampling interval: American Robin (AMRO); Black-headed Grosbeak (BHGR); Cedar Waxwing (CEWA); MacGillivray's Warbler (MGWA); Orange-crowned Warbler (OCWA); Song Sparrow (SOSP); and Spotted Towhee (SPTO). During this interval, Steller's Jay (STJA) was most abundant at state and modified buffer sites (Figure 3-13).

The relative abundance of 4 species (American Robin, Black-headed Grosbeak, Steller's Jay, and Western Tanager) was highest 10 years after logging on riparian transects of the modified buffer treatment. On riparian transects of the state buffer treatment, Wilson's Warbler, American Robin, Cedar Waxwing, MacGillivray's Warbler, Song Sparrow, Spotted Towhee, Steller's Jay, and Western Tanager were most abundant in the 10-year post-harvest sampling interval (Figure 3-13).

American Robin is a resident songbird that occupies a very broad ecological niche. It feeds primarily on invertebrates and fruit (berries) and nests in shrubs and trees. In Douglas-fir and western hemlock forests of southwestern Washington it prefers partially logged stands over clearcuts and old growth (Sallabanks and James 1999). Black-headed Grosbeak is a neotropical migrant songbird that breeds in a wide range of open or deciduous habitats characterized by trees, shrubs, edge, and proximity to riparian areas (Grinnell and Miller 1944; Hill 1995; Campbell et al. 2001). The shrubs and small trees that characterized the upland transects of modified and state buffer sites 10 years after logging provided suitable habitat for both American Robins and Black-headed Grosbeaks. It is possible that habitat conditions along riparian transects 10 years after logging were improved for both of these species, but it is more likely that these species were responding to increased habitat suitability along the adjacent upland. For example, use of the riparian areas might have been a result of territorial males using the trees for singing perches to advertise nesting territories centered in the upland.

Cedar Waxwing is resident in the study area. Its diet consists primarily of sugary fruits, but insects are gleaned from foliage and hawked from aerial perches during the summer (Witmer et al. 1997). Cedar Waxwing prefers to nest in trees (primarily deciduous) and shrubs in open and riparian areas with numerous shrubs and small trees (Campbell et al. 1997). The open areas, together with shrubs and small trees that characterized the upland and riparian transects of modified and state buffer sites 10 years after logging, provided suitable habitat for this songbird.

MacGillivray's Warbler is a neotropical migrant songbird that breeds in riparian habitat and regenerating coniferous forests where it nests close to the ground in low shrubs (Campbell et al. 2001). Insects are the main prey of MacGillivray's Warbler and are taken on or just above the ground (Pitocchelli 1995). Although MacGillivray's Warbler is common throughout much of its breeding range, its biology is not well understood (Pitocchelli 1995). The shrubs and/or small trees that characterized the upland transects of state buffer sites and the upland transects of modified buffer sites 10 years after logging provided suitable habitat for MacGillivray's Warbler. It is possible that habitat conditions along riparian transects of state buffer sites during the 10-year post-harvest sampling period were improved, but it is more likely that MacGillivray's Warbler was responding to increased habitat suitability along the adjacent upland. For example, use of the riparian areas might have been a result of territorial males using the trees for singing perches to advertise nesting territories centered in the upland.

Orange-crowned Warbler is a migratory species that breeds in the study area and winters in the southern United States and Central America. Orange-crowned Warbler prefers habitats with shrubs and low vegetation where it nests on or near the ground (Sogge et al. 1994). It feeds primarily on insects gleaned from vegetation but also consumes berries, fruit, and sap from trees tapped by sapsuckers (Sogge et al. 1994). The shrubs and small trees that characterized the upland transects 10 years after logging provided suitable habitat for this species.

Song Sparrow is resident in the study area and prefers to breed on the ground or in low vegetation in areas near freshwater that support dense patches of low vegetation, such as shrubs or herbaceous plants that often characterize young, regenerating forest stands (Campbell et al. 2001). During the breeding season, Song Sparrow feeds primarily on insects captured on the ground, on vegetation, or in-air (Arcese et al. 2002). The shrubs and small trees that characterized the upland transect 10 years after logging provided suitable habitat for this species. It is possible that habitat conditions along riparian transects of state buffer sites during the 10-year post-harvest sampling interval were improved, but it is more likely that Song Sparrow was responding to increased habitat suitability along the adjacent upland. For example, use of the riparian areas might have been a result of territorial males using the trees for singing perches to advertise nesting territories centered in the upland.

Spotted Towhee is resident in the study area and is a ground-nesting bird that prefers to breed near the edges between coniferous or deciduous forest stands and in open areas supporting dense patches of low vegetation, such as shrubs or herbaceous plants that often characterize young, regenerating forest stands (Campbell et al. 2001). Spotted Towhee forages primarily on invertebrates, seeds, and fruits on or near the ground (Greenlaw 1996). The forest edge habitat as well as the shrubs and small trees that characterized the upland transects 10 years after logging provided suitable habitat for this species. The modest increase in relative abundance observed along riparian transects of state buffer sites might have been in response to increased edge habitat and low-evergreen-shrub cover.

Steller’s Jay is resident in the study area and prefers nesting in deciduous, mixed, or coniferous forests, in trees with a wide range of ages (Campbell et al. 1997). It forages on the ground, in shrubs, and in trees on a wide variety of animal and plant food, including arthropods, nuts, seeds, berries, fruits, small vertebrates, and eggs and young of smaller birds (Greene et al. 1998). Sieving and Willson (1998) found Steller’s Jay to be a principal nest predator, with a preference for coniferous–deciduous habitat interfaces. Marzluff et al. (2004) concluded that Steller’s Jay preferred fragmented habitats and forest edges and that concentration of use along forest–human land-use interfaces might explain increased risk of nest predation to other birds in such settings. In a study in western Washington, point counts of corvid (including Steller’s Jay) abundance had the strongest correlation with predation on artificial nests containing eggs (Luginbuhl et al. 2001). Considering the overall increase in relative abundance of songbirds 10 years after logging (Figure 3-9), it is possible that the increased relative abundance of Steller’s Jays along riparian transects reflected increased abundance of prey (i.e., eggs and nestlings) or increased accessibility to those prey (e.g., loss of cover and edge effects) more so than any specific changes in habitat structure lacking such implications for predation.

Wilson’s Warbler is a neotropical migrant whose breeding territories are usually located in deciduous forest habitats with extensive shrub thickets, where it usually nests at the base of shrubs or small trees or under dense bunches of grass or sedge (Ammon and Gilbert 1999; Campbell et al. 2001). Key prey items include larval insects and spiders, which are taken by gleaning, hovering or sallying (Ammon and Gilbert 1999). Large and small deciduous trees, together with the ground cover provided by shrubs along the upland transects of modified and state buffer sites, provided suitable habitat for Wilson's Warbler. Along the riparian transects of state buffer sites, increased relative abundance of Wilson's Warbler likely reflected habitat conditions in the adjacent upland more than conditions in the riparian buffer.

Western Tanager is a neotropical migrant that prefers edge habitat and open coniferous and mixed coniferous–deciduous forests, where it nests mostly in conifers (Campbell et al. 2001) and forages in the forest canopy primarily on insects, with lesser amounts of fruits and berries taken (Hudon 1999). Increased relative abundance of Western Tanager 10 years after logging likely reflected suitable openings and edge habitats created by the riparian buffers at modified buffer sites and, to a lesser degree, at state buffer sites. The similar patterns in abundance of Western Tanager across all three treatment types suggests that Western Tanager was also responding to habitat conditions or regional effects that were independent of the experiment.

Diversity and Similarity

Although alpha diversity increased at all sites (including control sites) and in both habitat types (i.e., riparian and upland) during the course of the study, the magnitude of increase observed at the modified and state buffer treatments 10 years after logging is indicative of a treatment effect. For riparian habitats of the modified and state buffer treamtents, the magnitude of increase in alpha diversity (for songbirds identified within 15 m of the point-count station) was 45% and 69% greater than the change observed at the control treatment, which also increased relative to pre-harvest values. Alpha diversity in upland habitats also changed 10 years after logging (relative to the pre-harvest values) for all treatments; however, the magnitude of that change was smaller when compared with the changes observed in riparian habitats. Alpha diversity increased by 78% in upland habitats at the modified treatment and by 63% on both the state buffer and control treatments, with the difference in change only 15% for the modified buffer treatment. Forest harvesting led to increased alpha diversity—possibly via two pathways. First, diversity along riparian transects at the modified and state buffer sites increased, with most of the increase observed immediately after logging. This indicates that habitat changes resulting from upland harvesting likely had a greater effect on diversity than did successional changes in vegetation structure that occurred between the two post-harvest sampling intervals. Second, forest harvesting had mixed effects on alpha diversity immediately after logging along upland transects of modified and state buffer sites, being higher at modified and lower at state buffer sites. Ten years after logging, alpha diversity along upland transects of both treatment types was similar to that of the diversity along riparian transects (all of which exceeded the diversity at riparian or upland transects of the control sites). This delay suggests that phenological changes that occurred over the ~10-year post-treatment period served to increase habitat suitability for the greatest number of species occurring in these habitat types.

Riparian and upland transects of the control sites supported the most stable communities of songbirds when data from all pairs of sampling intervals were compared, indicating the lowest levels of species turnover (Figure 3-17). Conversely, the avian communities at upland transects of modified and state buffer sites showed the lowest degrees of community similarity—especially when the pre- and immediate post-harvest communities were compared. Ten years after logging, communities along the upland transects at logged sites (modified and state) were equally dissimilar from their pre-harvest conditions, providing an indication that the width of the nearby riparian buffer did not influence the avian community in upland habitats. The changes in avian communities of upland logged habitats are not surprising given that in this study these are the habitats that have undergone the most dramatic changes. The consistent pattern of community similarity observed for the riparian transects of modified and state buffer sites, with modified riparian sites being more stable, indicated that the difference in the width of the riparian buffers was enough to elicit a measurable response in avian communities over time. In this regard, and as concluded by Pearson and Manuwal (2001), modified buffers were more effective than state buffers at maintaining pre-harvest community structure over the course of the study. For many species, the highest relative abundance calculated for upland transects of modified and state buffer sites occurred 10 years after logging, a trend that was mirrored (for the same species) on riparian transects in the same sampling interval. The fact that American Robin, Black-headed Grosbeak, Cedar Waxwing, MacGillivray's Warbler, Orange-crowned Warbler, Song Sparrow, Spotted Towhee, and Steller's Jay (Figure 3-13)—which were associated with habitat attributes typified by forest harvesting (Figure 3-18 and Figure 3-19)—were abundant in riparian areas adjacent to logged areas means that the avian community of a given riparian buffer is influenced by the nearby logged area. This is particularly true for the comparatively narrower riparian buffers of the state buffer treatment, which had considerably more significant peak relative abundance values 10 years after logging than did the riparian transects of the modified buffer treatment during the same interval.

Forest communities develop through a continuum that, for conceptual purposes, has been grouped into sequential stages (e.g., Thomas 1979; Bunnell et al. 1999). Figure 3-23 depicts 6 seral (structural) stages of forest succession[5] as they are conceptualized for a forest stand culminating in an old-growth climax. Stage 1 begins with a vegetation community dominated by non-woody plants such as rushes, grasses, and forbs. By stage 2, woody shrubs and shade-intolerant conifer seedlings are present. At stage 3, conifers begin to dominate the site as they compete with other plants for light, water, and nutrients. During stage 4, the conifer canopy closes, effectively shading out vegetation on a forest floor that is becoming increasingly barren. At stage 5, conifers are mature, and weaker trees begin to die. Those trees will stand as snags that will eventually fall to the forest floor, where they become coarse woody debris. Shade-tolerant species of conifers become established in the understory. During stage 6, shade-intolerant species of conifers are gradually outcompeted by the shade-tolerant ones. Those shade-tolerant species continue to germinate and grow up in the understory. The number of standing dead trees and downed logs increases. Dead and fallen trees create gaps in the canopy that, together with other stand-mediated changes to microclimatic conditions, create suitable conditions for shade-intolerant species. The increasing diversity of plant species—and their increasing density, size, age, and state of decay—promotes the structural heterogeneity of the forest during stage 6.

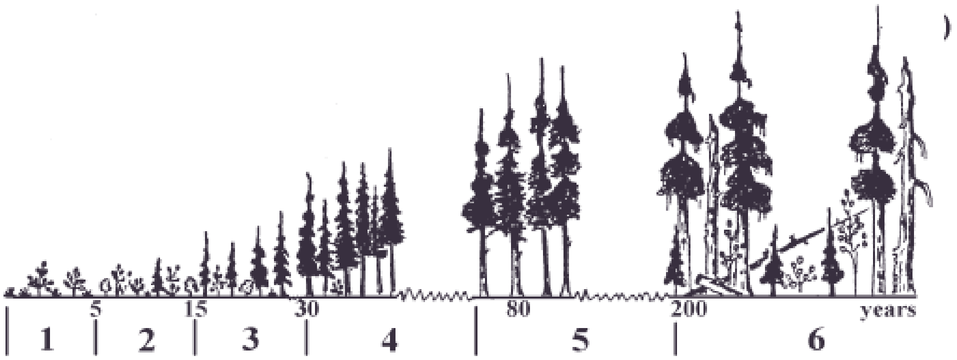


Figure 3-23. Conceptual model of the seral (structural) stages (1–6) and timeline (years) of natural coniferous-forest succession. The seral stages are: 1 = grass–forb; 2 = shrub–seedling; 3 = pole–sapling; 4 = young; 5 = mature; 6 = old growth. Source: Bunnell et al. (1999).

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[5] Other authors have used 4 stages (e.g., Franklin 1990; Klenner et al. 2000).

The increases in alpha diversity recorded at modified and state buffer sites during the 10-year post-harvest sampling interval fits well with standard conceptual models. Coniferous forests typically display the pattern of alpha diversity depicted in Figure 3-24. In that pattern, alpha diversity is relatively highest during the earliest and latest seral stages. As the tree canopy closes (around stage 4), the understory is shaded out, resulting in a decline in habitat suitability for many species. As succession progresses, the structural diversity and number of plant species in the forest increases. This brings about an improvement in habitat suitability which in turn facilitates an increase in wildlife alpha diversity (richness). Although wildlife-species richness in early and late successional stages can be similar and can have some species in common, the makeup of the wildlife communities is notably different. Thus, increases in relative abundance (at upland sites) and species diversity following forest harvesting of mid-seral second-growth stands in the study area is expected because pre-harvest stands in the present study were at a seral stage of ~4–5 (i.e., second-growth stands dominated by Douglas-fir ~65–75 years old), which is when natural levels of diversity and abundance are typically lowest (Figure 3-24). By harvesting sites and then allowing succession to proceed to stage ~2 (i.e., shrub-seedling stage), an increase in species diversity is expected. Similarly, over time the seral stage of the buffers will advance, potentially resulting in further increases in species diversity.

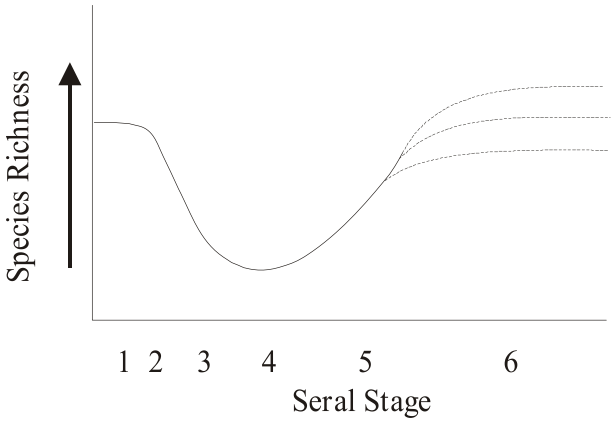

Figure 3-24. Conceptual model of the relative change in alpha diversity (species richness; number of species) for wildlife communities as a function of forest seral stage, showing three possible outcomes (dashed lines) of lesser, similar, and greater alpha diversity during the earliest and latest stages. Refer to Figure 3-23 for a description of seral stages. Source: adapted from Franklin (1990).

Though not a novel finding by any means, this study clearly shows that the habitat suitability for a given species of bird is tied to measures of the physical environment that can be altered both directly and indirectly by forest harvesting. The existence of this link means that forest managers have a degree of control over, and thus responsibility for, their effects on avian communities.

Conclusions and Management Implications

Forest harvesting alters the abundance, alpha diversity, and community structure of songbirds in the Douglas-fir–dominated forests of inland western Washington. Because many of those changes are predictable, it is possible to plan industrial activities to meet scientifically and socially derived conservation objectives. The geographic scale of this study (western Washington) also allows inference of study results across a large land base, particularly where similar habitats occur (i.e., those conforming to the study site criteria). Given that the sites selected for inclusion in this study represent the largest managed forest type in western Washington, the scale at which the study conclusions can be applied is substantial.

The purpose of this study was not to comment on specific objectives of songbird conservation but rather to assess songbird responses to two different stand-level harvesting prescriptions that differ primarily in the extent to which riparian buffers were retained after adjacent second-growth coniferous forests were clearcut. It is hoped that information gained by this study will assist managers and biologists in meeting the conservation objectives adopted by government and/or industry.

This study presents a short-term (~10-year) view of forestry–wildlife interactions at the stand level,[6] using mid-seral coniferous forests as a baseline at a landscape level instead of old-growth or climax seral forests. This presents a few challenges. First, mid-seral forests are typically lowest in alpha diversity. As a result, any forest harvesting is likely to cause an increase in alpha diversity within a few years of harvest. Second, it is unclear how unlogged buffers will progress through time on an industrial forest land base. For example, if the buffers are to be retained in perpetuity, they would conceivably reach old-growth status while the surrounding upland passed through multiple rotations. Yet, according to this study, they would continue to be greatly influenced by the adjacent harvesting. As such, they would need to be wider to be immune from such influences. Third, and most importantly, the conservation of wildlife such as songbirds requires that management occur at landscape scales or larger. While site prescriptions are necessary to deal with stand-level conservation objectives, they are only part of a management hierarchy. Further, conserving ecological communities comprising other taxonomic groups (i.e., plants, invertebrates, amphibians, reptiles, and mammals) and birds other than songbirds will often require management actions very different from those that would be prescribed for songbirds alone.

A number of factors potentially influence the veracity of this study's results or limit the extent to which the conclusions can be used to aid in managing and conserving songbirds in coniferous forests of the Pacific Northwest. The purpose of presenting these limitations is to make the reader aware of (1) factors that could have influenced the results of the study and (2) factors that should be considered if this study is to be used to guide formal policy development.

Data Accuracy and Study Design—Much has been written on the subject of observer bias and the effects of overcounting, undercounting, and misidentifying species (e.g., Bart 1985). We acknowledge that the database assimilated over the course of this study contains all three types of errors to some unknown extent. Although steps were taken to reduce this bias (see Methods section), its potential influence on the results must be considered nonetheless.

The study only examined abundant, conspicuous songbirds that allow observers to be within 15 m. As such, rare, reclusive, nocturnal, quiet, and widely ranging species are likely to go underrepresented in the analyses. Notable taxa are the Accipitridae (diurnal raptors) and Strigidae (typical owls), both of which require survey techniques much different from those employed here. Even some of the normally conspicuous songbirds will reduce their vocalizations during some parts of their reproductive cycle.

In describing the results of the first two sampling intervals of this study, Pearson and Manuwal (2001) concluded that strip transects or territorial mapping would provide better sample sizes and statistical power than the 15 m radius point-count method. Indeed, as depicted in Figure 3-8, a

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[6] "Stand level" describes the level of forest management at which a relatively homogeneous land unit can be managed under a single prescription or set of treatments to meet well-defined objectives. Stand-level management occurs on a land base ranging in size from a few to several thousand hectares. "Landscape level" describes planning areas delineated on the basis of topographic or geographic features. Typically they cover a watershed or series of watersheds and range in size from ~5000 to 100,000+ ha.

substantial amount of data collected during the study was excluded from most analyses. Although methods other than those used here might improve sample sizes and statistical power, changing survey methods partway through a long-term study would introduce potentially insurmountable problems to the process of comparing data sets among sampling intervals.

Nesting Success—An underlying assumption untested by this study is that relative abundance of songbirds is indicative of habitat quality. To test this assumption, nesting success rates (a measure of fitness) would need to be monitored over time. Measuring nesting success is costly and logistically difficult. Until there are compelling reasons to believe that monitoring trends in relative abundance are incapable of revealing wildlife responses to habitat, it seems reasonable to continue monitoring using the methods in this study.

Habitat Assessment—Food is a critical habitat feature for breeding songbirds; however, this study did not measure food directly. While measures of the relative abundance of berry-producing shrubs likely provide a reasonable approximation of some food types, the influence of the abundance of insects, arthropods, and other invertebrates on the relative abundance of birds during the study is unknown. This study examined species–habitat relationships for only a portion (<20%) of the year. Spring is the most important time of year for neotropical and other migrants in the Pacific Northwest, but many species occur in the area year-round. Factors that influence habitat conditions during winter (e.g., roosting habitat, presence of and access to food) must also be considered.

Extensive forestry management in the Pacific Northwest has historically included clearcutting, prescribed fire, regeneration by seed trees, protection from forest fires, and salvage of timber killed by windthrow or disease (Carey and Harrington 2001). In the last 20 years, planting of Douglas-fir has increased, with most forest stands >40 years old originating from seed. Current management practices include the retention of seed trees and coarse woody debris and the planting of seedlings. Implicit in the shift to ecosystem management has been the retention of riparian management zones (RMZs) around perennial fish- and non–fish-bearing streams, which have been required in some form since 1978.

Since this study's inception in 1991, there have been changes to the Washington Forest Practices Rules and Regulations that have implications on our findings. The Forest Practices Rules of 1988 required a riparian buffer that met the state configuration (i.e., not less than 7.5 m from the ordinary high-water mark) for both fish-bearing (Type 3) and non–fish-bearing (Type 4) perennial streams. The development of the modified riparian prescription was an alterative to the state buffer that provided for greater structural complexity, heterogeneity, and ultimately greater diversity for both flora and fauna. Lee et al. (2004) identified the current trend in riparian management prescriptions as tending toward site-specific buffers that vary among broadly similar harvest areas to within a single harvest area. The Forest Practices Rules of 2001 seem to fit this model and, aside from changing how watercourses are defined, are site-specific with many options. Lee et al. (2004) suggest that the primary benefit of this type of prescription is the identification of well-defined criteria to delineate the riparian zone. These criteria are specifically defined by the modifying factors selected by jurisdictions, and buffers are applied in a predictable response to these criteria. The caveat to tailor-made buffers is the greater complexity in guidelines. For example, Washington now describes 3 zones of management for Type F streams (Type 3): (1) the core, (2) inner, and (3) outer zones, with various levels of tree removal, thinning, hardwood conversion, and other activities permitted within those zones depending on the site class of land, the management harvest option, and the bankfull width of the stream. In general, the site class will form the RMZ prescription. It appears that riparian management in western Washington has become (necessarily) site-specific. This implies that applying a one-size-fits-all approach to riparian management has been set aside in favor of a more intensive approach that considers the specifics of the site, which is consistent with Lee et al. (2004).

For Type Np streams, or perennial non–fish-bearing streams (Type 4), the RMZ rules are slightly less complicated; however, there is ample room for site-specific management. Considering that the RMZ rules have evolved to embrace a site-specific approach, it seems only fitting that forest management in general adopt the same principals when considering biodiversity. Managing for biodiversity on a regional scale does not account for variation at the landscape or site level within the region, which could have dire consequences for species that are limited geographically, that occur at naturally low densities, or that are particularly sensitive to habitat change. As an alternative to the status quo, management that focuses on stand heterogeneity (as opposed to tree-species monocultures with reduced diversity) should produce landscapes with high biocomplexity and biodiversity at multiple scales, and therefore these landscapes should be resilient in the face of disturbance (Holling 2001).

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Appendices

Appendix 3-A. Ecological description of each site sampled in western Washington, 1992–2004. Refer to Figure 3-1 for distribution of study sites. – = data not available; OP/SW = Olympic Peninsula and SW Washington; PT = Puget Trough; WSC = western slopes and crest, or Washington Cascades physiographic provinces.

			Physiographic	Elevation (m ASL)			Stand Age			Stream Type		Bankfull		Riparian	
Site Name	Treatment	County	Province	Min	Max	Slope (%)	Year 1	Year 14	Dominant Tree	1988	2000	Width (m)	Shape	CC (%)	Lithology
Abernathy	Control	Cowlitz	OP/SW	362	491	16.0	55-65	68-78	Douglas-fir	3	F	5.04	Sloping	71.0	Basalt flows
Elbe Hills	Control	Pierce	WSC	554	609	10.0	55-67	68-80	Douglas-fir	4	F	3.20	V-shaped	91.0	Volcanclastic deposits or rocks
Hotel Creek	Control	King	PT	277	303	4.0	55-71	68-84	Douglas-fir	4	F	2.38	V-shaped	99.0	Continetal glacial outwash - Fraser Age
Porter Creek	Control	Thurston	PT	285	290	3.0	55-75	68-88	Douglas-fir	3	F	5.98	V-shaped	95.0	Basalt flows and flow breccias
Taylor Creek	Control	King	PT	415	424	2.0	55-81	68-94	Douglas-fir	2	F	6.47	V-shaped	93.8	Continetal glacial till - Fraser Age
Vail	Control	Lewis	PT				55-82	68-95	Douglas-fir	5	N				Andesite Flows
West Fork Falls Creek	Control	Thurston	PT	379	463	20.0	55-83	68-96	Douglas-fir	3	F	3.26	V-shaped	92.0	Basalt flows and flow breccias
Bluetick	Modified	Grays Harbor	PT	110	151	7.0	55-66	68-79	Douglas-fir	4	F	2.02	Undercut	94.0	Marine sedimentary rocks
Eleven Creek 31	Modified	Lewis	PT	493	561	6.0	55-68	68-81	Douglas-fir	4	N	3.30	V-shaped	92.5	Andesite flows
Griffen Creek	Modified	King	WSC	157	188	5.0	55-70	68-83	Douglas-fir	2	F	4.06	V-shaped	93.0	Continetal glacial till - Fraser Age
Ms. Black	Modified	Thurston	PT	67	92	3.0	55-73	68-86	Douglas-fir	3	F	1.82	Undercut	83.0	Basalt flows and flow breccias
Ryderwood 860	Modified	Cowlitz	OP/SW	303	337	8.0	55-78	68-91	Douglas-fir	3	F		V-shaped	75.0	Nearshore sedimentary rocks
Side Rod	Modified	Pierce	WSC	520	548	6.0	55-79	68-92	Douglas-fir	4	N	2.70	V-shaped	90.0	Volcanclastic deposits or rocks
Eleven Creek 32	State	Lewis	PT	415	501	10.0	55-69	68-82	Douglas-fir	4	N	1.32	V-shaped	98.0	Andesite flows
Kapowsin	State	Pierce	WSC	443	450	2.0	55-72	68-85	Douglas-fir	3	N	3.08	V-shaped	51.0	Basaltic andesite flows
Night Dancer	State	Grays Harbor	PT	125	201	11.0	55-74	68-87	Douglas-fir	5	F	1.44	V-shaped	96.0	Marine sedimentary rocks
PotPourri	State	Thurston	PT	239	249	7.0	55-76	68-89	Douglas-fir	3	F	5.18	V-shaped	74.0	Basalt flows and flow breccias
Ryderwood 1557	State	Cowlitz	OP/SW	333	376	7.0	55-77	68-90	Douglas-fir	4	N	3.30	V-shaped	77.0	Nearshore sedimentary rocks
Simmons Creek	State	Lewis	WSC	422	471	8.0	55-80	68-93	Douglas-fir	3	F	3.22	V-shaped	80.0	Basaltic andesite flows

Appendix 3-B. Data collected at each point-count station during bird surveys.

Field	Definition	Value
Observer	Observer	Initials
Date	Date	Day, month, year
Site Number	Number of site (numbered alphabetically)	1–18
Site Name	Name of site	Name (e.g., “Abernathy”)
Countpoint Number	Song bird point-count station	1–10
Countpoint Location	Riparian or upland	R or U
Valley Side	Indicates side of creek relative to parking location	Parking or across
Species	Bird species	4-letter species code; "NONE" = no birds
Flock Count	Number of birds in a group/flock	Number
Sex	Male, female, unknown	M/F/U
Age	Adult, juvenile, unknown	A/J/U
Flyover	Observation involving a bird flying overhead	Yes/no
Distance	Estimated distance of bird from point count (m)	0–15; 15–30; 30–50; 50–100; >100
Time Interval	Period of time when detection was made (min)	Pre-start; 0–3; 3–5; 5–6
Detection	Type of detection	Sight/call/song/drum/wing
Habitat	Habitat type for specific detection	Buffer/logged/forest/unknown
Comments	Comments	Any other information
Cloud Cover	Cloud cover	1 = clear; 2 = cloudy <50%; 3 = cloudy >50% but <100%; 4 = 100% cloud cover
Air Temperature	Air temperature	Degrees C
Precipitation	Precipitation	Nil; fog; misty; drizzle; light rain; hard rain; hail; snow
Wind	Wind speed (estimate)	Km/h
Time Start	Survey start time (per site)	24-hour time
Time End	Survey end time (per site)	24-hour time

Appendix 3-C. Species codes and common names of all birds encountered during breeding-bird surveys in western Washington, 1992–2004.

Family/CODE	Name
Gaviidae	**Loons**
COLO	Common Loon
Ardeidae	**Bitterns, Egrets, and Herons**
GBHE	Great Blue Heron
Cathartidae	**New World Vultures**
TUVU	Turkey Vulture
Anatidae	**Swans, Geese, and Ducks**
CAGO	Canada Goose
Accipitridae	**Kites, Eagles, Hawks, and Allies**
NOHA	Northern Harrier
SSHA	Sharp-shinned Hawk
COHA	Cooper's Hawk
SWHA	Swainson's Hawk
RTHA	Red-tailed Hawk
Falconidae	**Falcons**
AMKE	American Kestrel
Phasianidae	**Partridges, Pheasant, Grouse, Ptarmigan, and Turkey**
RNPH	Ring-necked Pheasant
BLGR	Blue Grouse
RUGR	Ruffed Grouse
Odontophoridae	**American Quail**
CAQU	California Quail
Scolopacidae	**Sandpipers, Phalaropes, and Allies**
COSN	Common Snipe
Columbidae	**Pigeons and Doves**
BTPI	Band-tailed Pigeon
MODO	Mourning Dove
Strigidae	**Typical Owls**
WSOW	Western Screech Owl
GHOW	Great Horned Owl
NPOW	Northern Pygmy Owl
BAOW	Barred Owl
NSWO	Northern Saw-whet Owl
Caprimulgidae	**Goatsuckers**
CONI	Common Nighthawk
Apodidae	**Swifts**
VASW	Vaux's Swift
Trochilidae	**Hummingbirds**
ANHU	Anna's Hummingbird
RUHU	Rufous Hummingbird
Alcedinidae	**Kingfishers**
BEKI	Belted Kingfisher
Picidae	**Woodpeckers**
RBSA	Red-breasted Sapsucker
DOWO	Downy Woodpecker
HAWO	Hairy Woodpecker
NOFL	Northern Flicker
PIWO	Pileated Woodpecker
Tyrannidae	**Tyrant Flycatchers**
OSFL	Olive-sided Flycatcher
WWPE	Western Wood-peewee
WIFL	Willow Flycatcher
HAFL	Hammond's Flycatcher
PSFL	Pacific-slope Flycatcher
Vireonidae	**Vireos**
CAVI	Cassin's Vireo
HUVI	Hutton's Vireo
WAVI	Warbling Vireo
SOVI	Solitary Vireo
Corvidae	**Jays, Magpies, Crows, and Ravens**
GRJA	Gray Jay
STJA	Steller's Jay

Family/CODE	Name
AMCR	American Crow
NOCR	Northwestern Crow
CORA	Common Raven
Hirundinidae	**Swallows**
TRES	Twree Swallow
VGSW	Violet-green Swallow
BARS	Barn Swallow
Paridae	**Chickadees**
BCCH	Black-capped Chickadee
CBCH	Chestnut-backed Chickadee
BUSH	Bushtit
Sittidae	**Nuthatches**
RBNU	Red-breasted Nuthatch
WBNU	White-breasted Nuthatch
Certhiidae	**Creeper**
BRCR	Brown Creeper
Troglodytidae	**Wrens**
BEWR	Bewick's Wren
HOWR	House Wren
WIWR	Winter Wren
Regulidae	**Kinglets**
GCKI	Golden-crowned Kinglet
RCKI	Ruby-crowned Kinglet
Muscicapidae	**Bluebirds, Thrushes, and Allies**
WEBL	Western Bluebird
TOSO	Townsend's Solitaire
SWTH	Swainson's Thrush
HETH	Hermit Thrush
AMRO	American Robin
VATH	Varied Thrush
Bombycillidae	**Waxwings**
CEWA	Cedar Waxwing
Parulidae	**Wood Warblers**
OCWA	Orange-crowned Warbler
YEWA	Yellow Warbler
YRWA	Yellow-rumped Warbler
BTYW	Black-throated Gray Warbler
BUZZ	Hermit/Townsend's Warbler
MGWA	MacGillivray's Warbler
COYE	Common Yellowthroat
WIWA	Wilson's Warbler
YBCH	Yellow-breasted Chat
Thraupidae	**Tanagers**
WETA	Western Tanager
Cardinalidae	**Cardinals, Grosbeaks, and Allies**
BHGR	Black-headed Grosbeak
Emberizidae	**Towhees, Sparrows, Longspurs, and Allies**
SPTO	Spotted Towhee
SOSP	Song Sparrow
LISP	Lincoln's Sparrow
WCSP	White-crowned Sparrow
GCSP	Golden-crowned Sparrow
DEJU	Dark-eyed Junco
Icteridae	**Blackbirds, Orioles, and Allies**
RWBL	Red-winged Blackbird
COGR	Common Grackle
BHCO	Brown-headed Cowbird
Fringillidae	**Cardueline Finches and Allies**
PUFI	Purple Finch
RECR	Red Crossbill
PISI	Pine Siskin
AMGO	American Goldfinch
EVGR	Evening Grosbeak

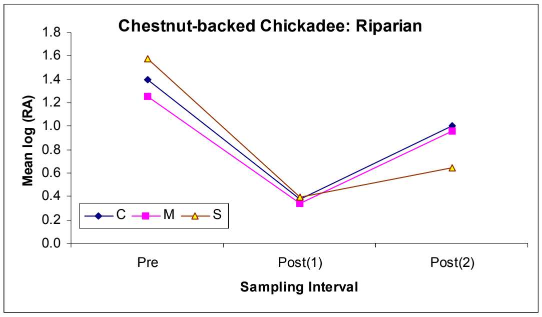
Appendix 3-D. All birds (excluding flyovers and data excluded from two sites; see Table 3-6) detected during the 0–6 min sampling interval and within 15 m of the point count along the riparian (R) and upland (U) transects of the three different sites during three sampling intervals. Readers are cautioned regarding the use of these data for comparative purposes because of unequal sampling effort among sites and sampling intervals (see Table 3-6). Refer to Appendix 3-B for data collected and Appendix 3-C for species names.

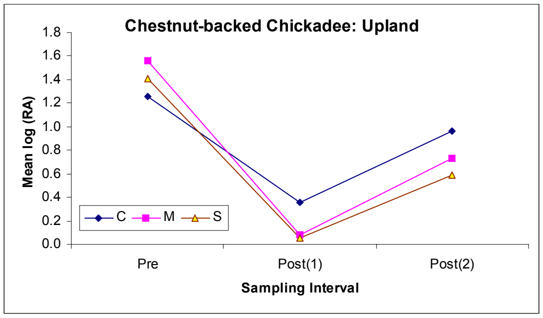
	Site, Sampling Interval, Transect																	
	Control						Modified						State					
	1		2		3		1		2		3		1		2		3	
Species	R	U	R	U	R	U	R	U	R	U	R	U	R	U	R	U	R	U
BRCR	4	6	16	16	37	62	4	8	6		1		6	7	1			
BTYW	3	2	4	2	2	4	5	4	2		1		2	4			5	2
BUZZ	4	5	6	21	2	14	5	4	1				4	2				
CBCH	96	79	45	53	120	128	77	116	46	7	123	85	112	90	37	3	86	61
GCKI	34	46	37	76	49	104	20	42	3	1	12	24	37	36	3		11	2
GRJA			2	12	3	5	2	3	3	1	1		1	2	1			5
PSFL	65	23	161	59	150	122	62	42	112		98	2	51	42	117		74	4
WIWR	88	34	181	63	167	134	101	37	159	12	120	8	78	40	152	16	81	4
DEJU			7	9		1		4	20	91	1	40		2	50	91	1	27
EVGR	1			1	2	10		3	8			7	2		4		1	
HUVI			2		1	1			1		1				5		4	3
WAVI	2		8		4				27		5	13		2	38	1	22	13
WIWA	20	16	49	4	43	21	20	9	69		99	119	9	13	24	1	92	120
AMRO	7	1	15	6	12	9	7	1	28	11	32	31			34	9	29	33
BHGR			3		3	2			7	3	19	32			12		17	43
CEWA				5	6	1		2	14	2	25	52			17	7	47	70
MGWA	1				1		1		2	8	4	75			1	4	17	53
OCWA												29					5	31
SOSP				2			1		23	33	18	71			54	39	50	112
SPTO			1							21	3	100			2	21	22	108
STJA	3		7	3	16	7		2	13	6	47	17	1	3	8	1	34	8
SWTH	6	7	22	7	48	35	6	6	35	2	82	155	6	8	14	4	35	136
WCSP									1	30	3	6			4	36		6
WETA	1	1	2		7	12	4	1	15		24	5		2	5	1	17	11
AMGO												10				6	1	31
ANHU									1									
BAOW					1													

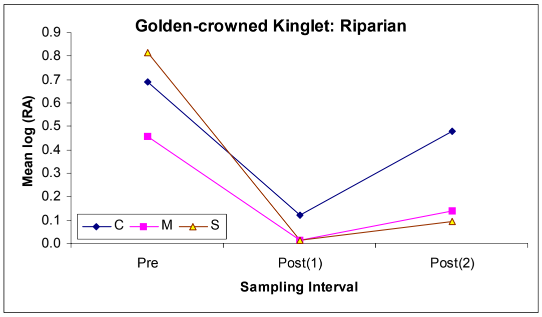
	Site, Sampling Interval, Transect																	
	Control						Modified						State					
	1		2		3		1		2		3		1		2		3	
Species	R	U	R	U	R	U	R	U	R	U	R	U	R	U	R	U	R	U
BCCH	1	1			2			3			1	8						1
BEWR										8	1	7			1		2	6
BHCO									2		1	5					3	4
BLGR	1																	
BTPI		1	2	1		4								1			2	1
BUSH									6			2					2	2
CAQU		1							2									
CAVI						1					2							
CONI											1							
CORA						1	1				2							
COYE										1		14					1	11
DOWO									2		2							
GCSP											2					1		
HAFL		1	4	2	2	7	1	1	9	1	1		1	1	9			
HAWO		1	16	2	7	12	1	1	20	4	16	2	2		18	7	15	2
HETH	1	1												1				
HOWR										6					2			
MODO											5	7					1	2
NOFL											5	2			1		9	2
OSFL									5		3	3			7		6	
PISI									3	1								
PIWO												1						
PUFI	1				2	1	1			1	4	17					6	3
RBNU				1	1	6			2	1	5	2						
RBSA						1	1		8	2	28	3			3		34	2
RCKI												1						1
RECR					4	2						19						
RUGR			2		1						1	1						
RUHU	1	2	6	2	7	4	5	2	14	16	25	123	1	4	14	14	63	124
SOVI									2					1				
SSHA																		1
SWHA						1												

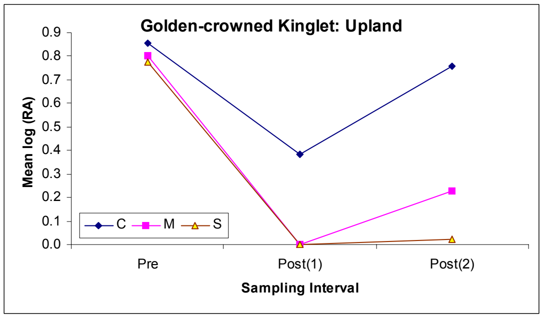
	Site, Sampling Interval, Transect																	
	Control						Modified						State					
	1		2		3		1		2		3		1		2		3	
Species	**R**	**U**	**R**	**U**	**R**	**U**	**R**	**U**	**R**	**U**	**R**	**U**	**R**	**U**	**R**	**U**	**R**	**U**
TOSO										1					4	3		
TRES															1			
TUVU											1							
VATH		2	6	8	18	15			1		1	1	2	2			1	
WBNU				1														
WIFL									1	9	1	102			1		7	144
WSOW																	1	
WWPE									1									
YEWA						1												
YRWA		1					1							1	1			3
Grand Total	340	231	604	356	718	728	326	291	674	279	827	1201	315	264	645	265	804	1192
Species (n)	20	20	24	23	29	31	21	20	38	26	42	39	16	21	33	19	36	38

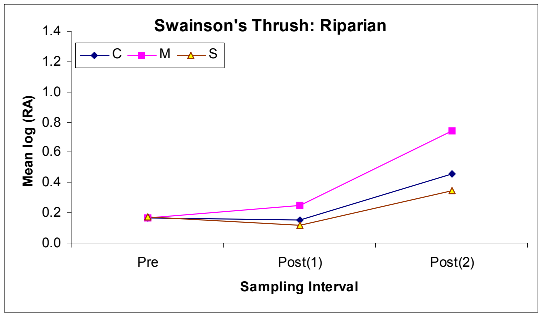
Appendix 3-E. Changes in relative abundance (RA) for 24 selected bird species across treatments (C = control, M = modified, S = state), transects (riparian, upland), and sampling interval during breeding-bird surveys in western Washington, 1993–2004. Data show mean detection rates (log transformed to address normality). Plots are presented in descending order of the number of significant differences in F-probabilities on non-orthogonal contrasts (see Table 3-7).

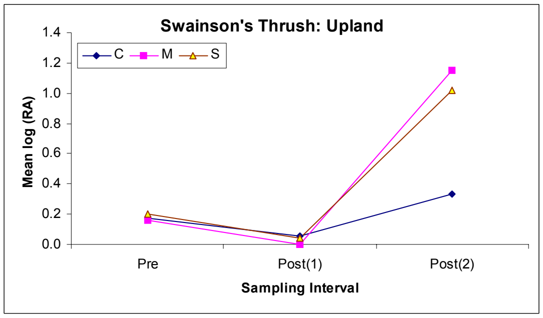


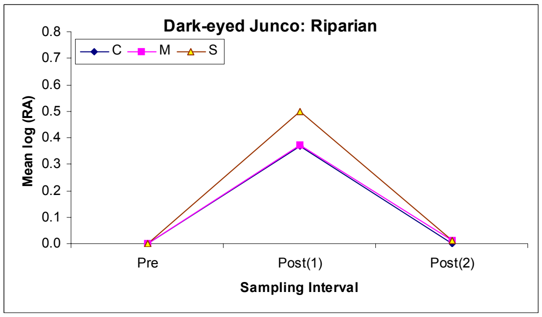


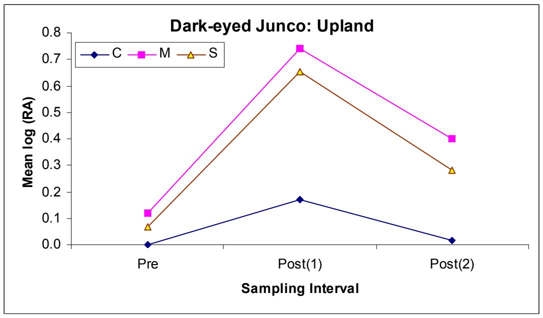












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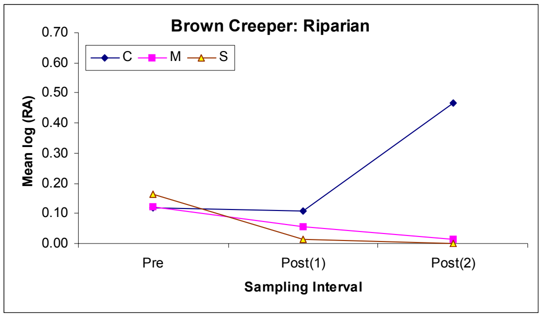

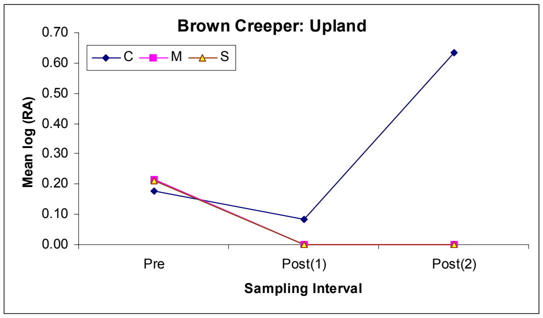

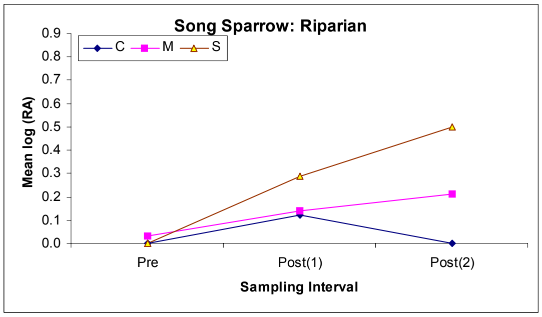

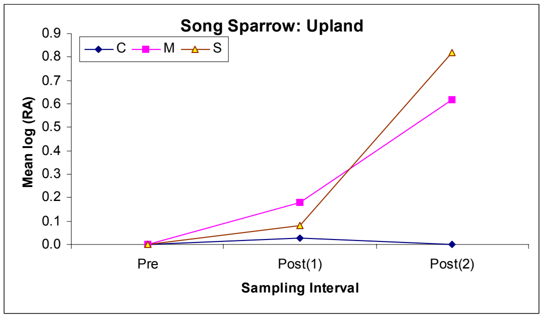

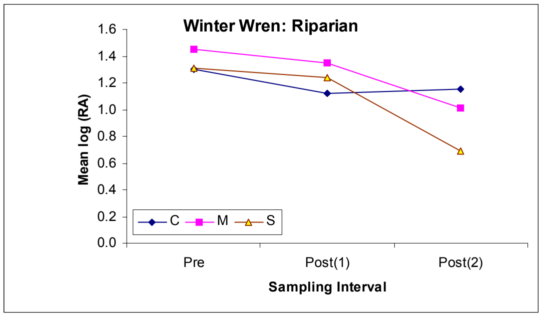

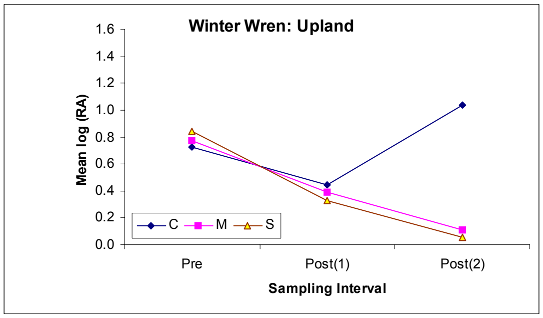

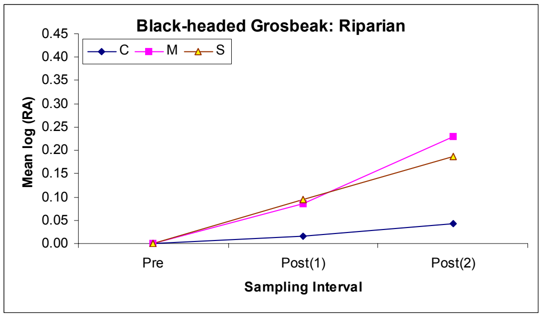

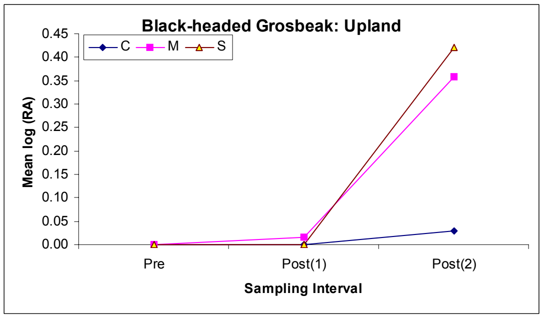

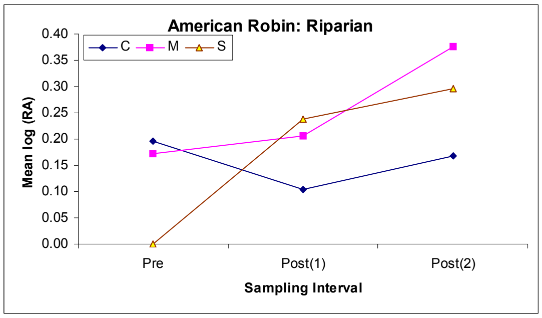

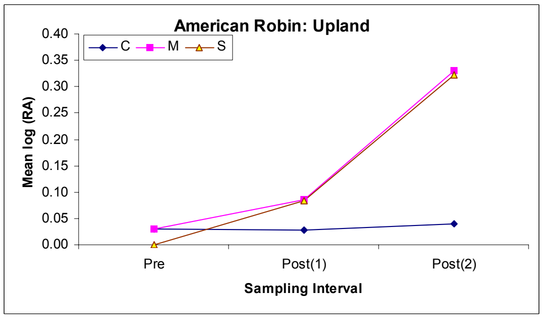

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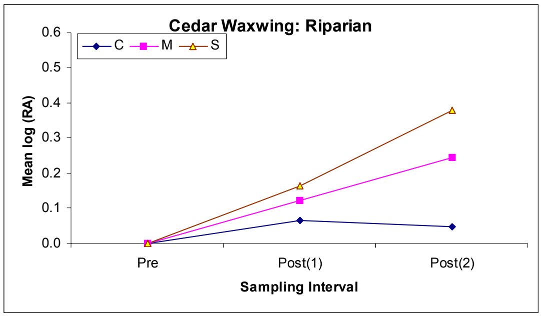
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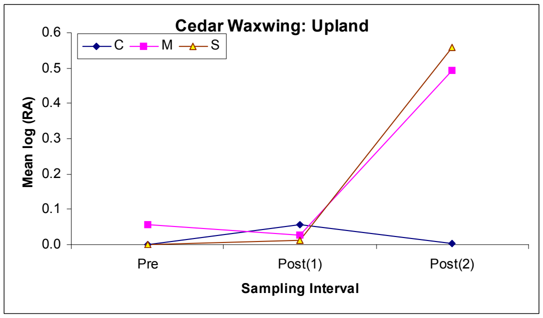
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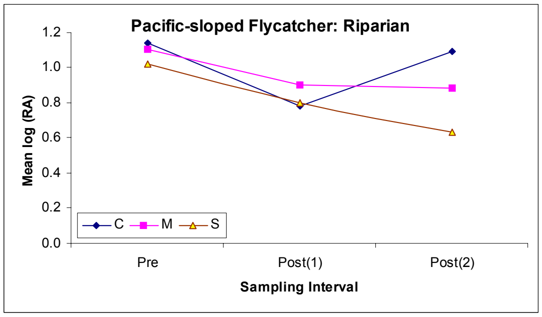
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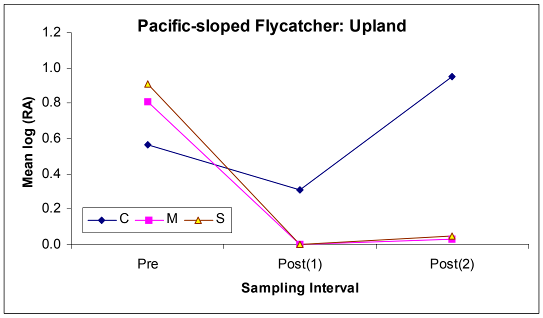
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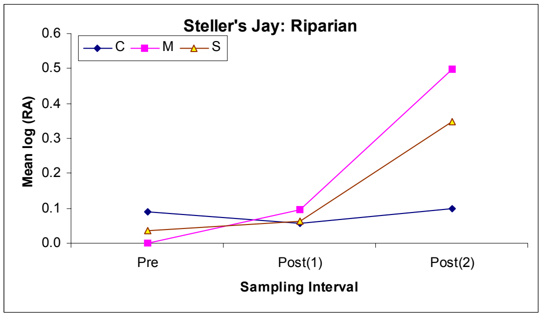
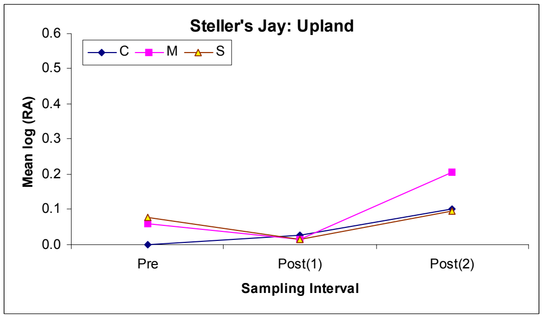

Steller's Jay: Riparian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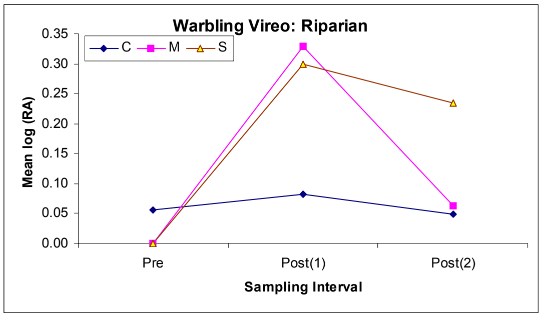
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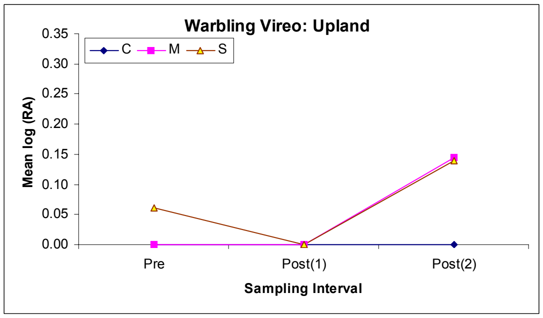
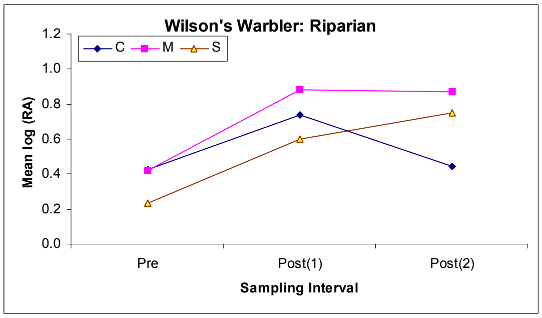
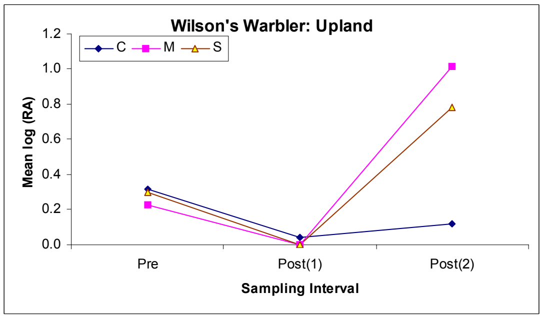

Warbling Vireo: Upland


Wilson's Warbler: Riparian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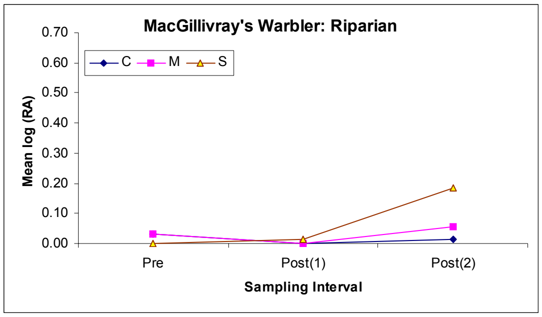

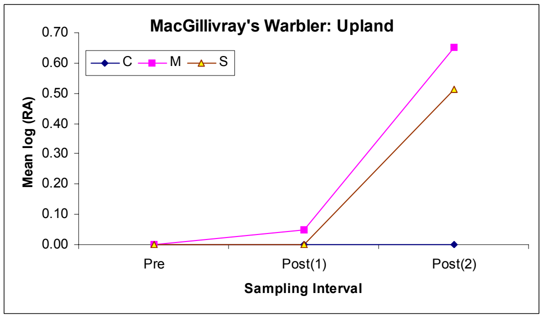

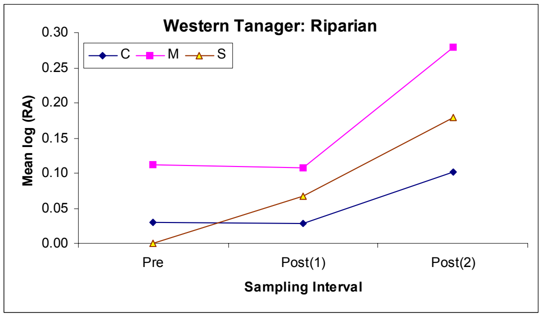

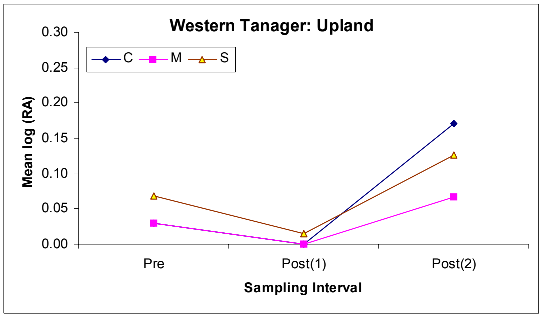

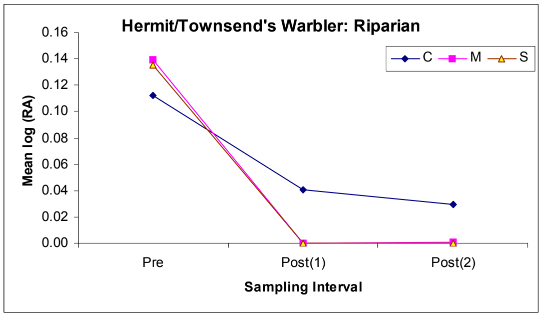

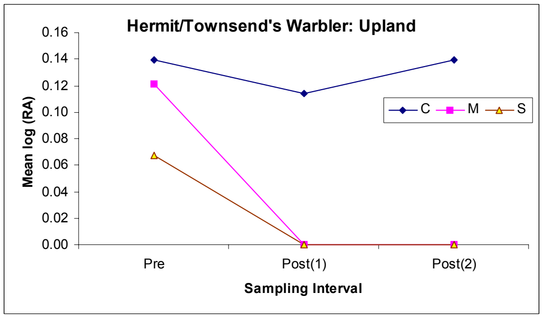

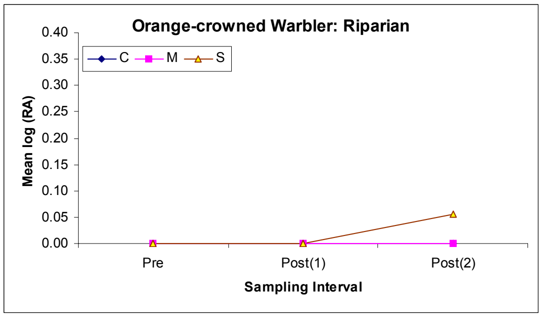

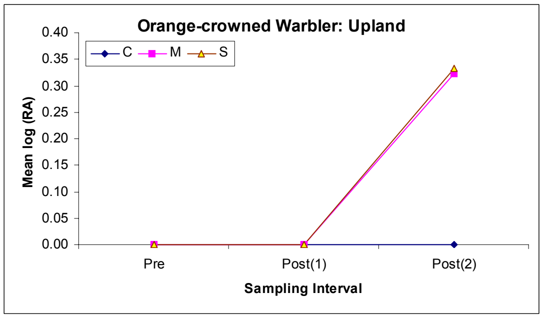

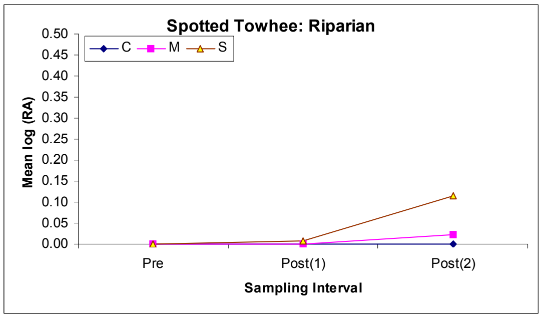

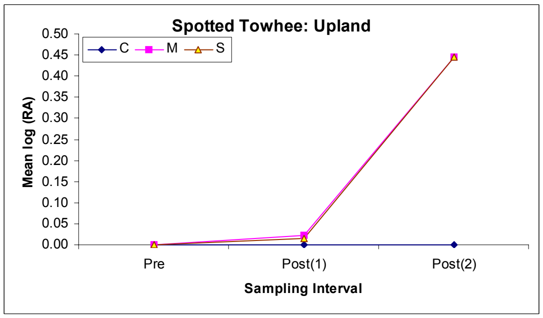

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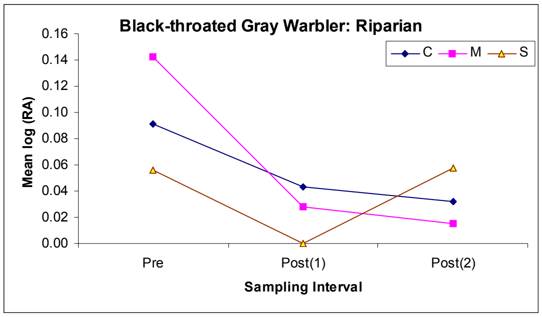

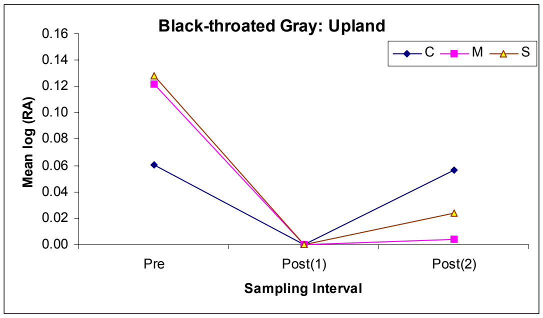

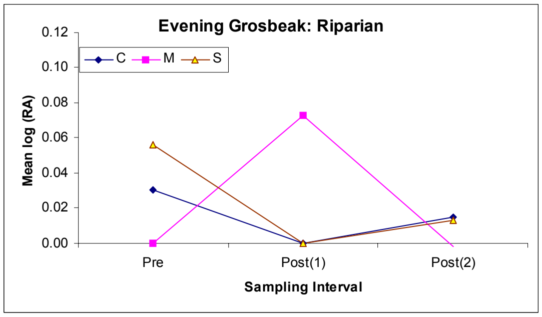

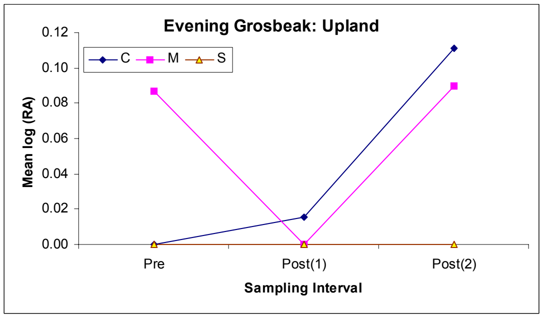

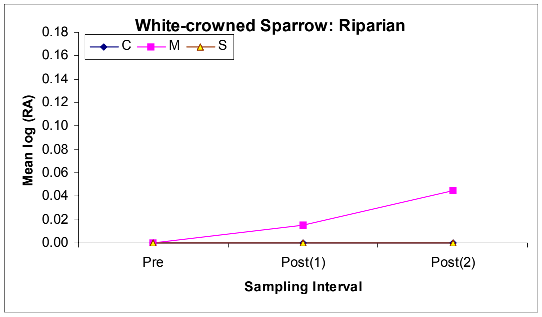

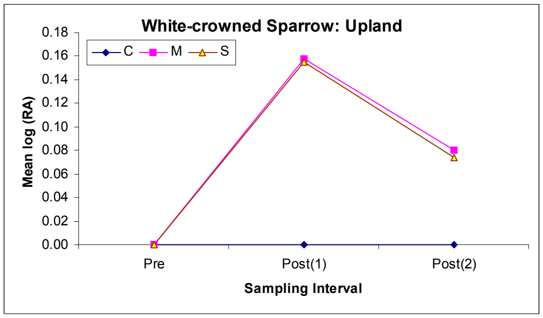

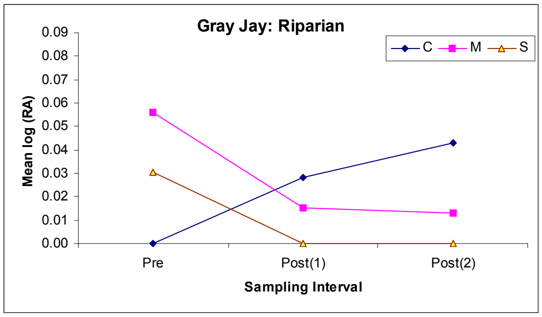

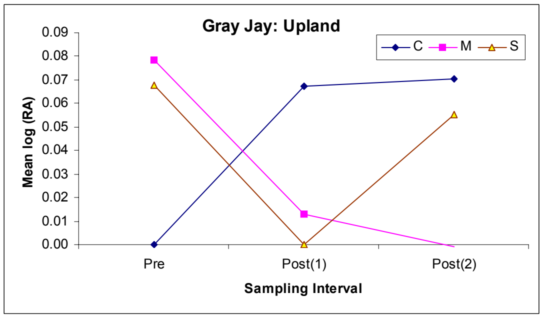

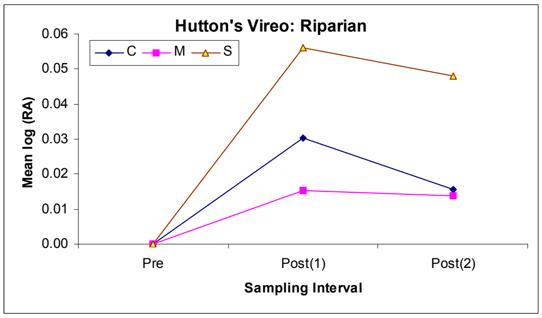

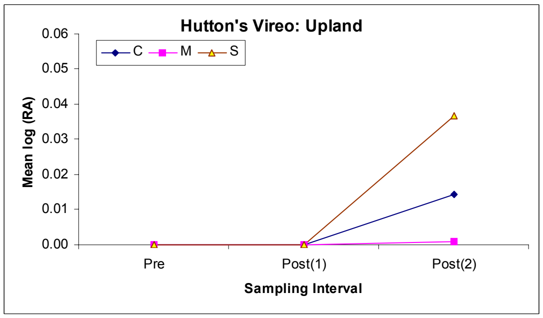

Appendix 3-F. Explanation of habitat codes used in the co-inertia analysis.

Code	Definitions
Shrub_T	Number of shrubs >1.5 m (deciduous, coniferous, berry-producing, etc)
Shrub_L	Number of shrubs <1.5 m (deciduous, coniferous, berry-producing, etc)
CC	Overstory canopy cover
CWD	Percent cover of coarse woody debris
Fern	Percent cover of ferns
Grass	Percent cover of grass
Herb	Percent cover of herbaceous plants
Litt	Percent cover of deciduous leaf litter
LTD	Deciduous leaf litter depth
Moss	Percent cover of moss
Rock	Percent cover of rock
Soil	Percent cover of exposed soil
Maple	Number of vine maple
Dec_Sm	Number of small deciduous (bigleaf maple, alder, cottonwood) trees
Dec_Lg	Number of large deciduous (bigleaf maple, alder, cottonwood) trees
Con_Sm	Number of small coniferous (hemlock and cedar) trees
Con_Lg	Number of large coniferous (hemlock and cedar)trees
Dfir_Sm	Number of small Douglas-fir trees
Dfir_Lg	Number of large Douglas-fir trees

Chapter 4 WESTERN WASHINGTON SMALL MAMMALS

Virgil C. Hawkes and Gordon Glova

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Abstract

We assessed how effective riparian management zones (RMZ) of varying widths were in providing suitable habitat conditions for small mammals 2- and 10-years post-logging relative to the control condition of unlogged, maturing second-growth Douglas-fir forests in western Washington. We used a Before-After-Control-Impact (BACI) approach to determine if a narrow RMZ was as effective as a wider, more variable RMZ in providing habitat attributes necessary to maintain species richness and relative abundance across time. Our findings suggest that there is no global response to RMZ width by small mammals in western Washington. Rather, species showed individual responses that varied over time, treatments, and transects. There is some suggestion that buffer width and logging treatments affect some species, but the analysis is confounded by patterns of natural population change over time (some local, some regional) and likely by differences among sites within treatment groups. For species that readily adapt to large-scale habitat alterations, such as *Peromyscus maniculatus* and *P. keeni*, riparian zones may not be necessary; but for other species (e.g., *Microtus oregoni*), riparian zones provide important connectivity in the landscape as well as being areas from which such species can recolonize regenerating forest stands. Considering that the RMZ rules have evolved to embrace a site-specific approach, it seems only fitting that forest management in general adopt the same principals when considering biodiversity. Managing for biodiversity on a regional scale does not account for variation at the landscape or site level within the region, which could have dire consequences for species that are limited geographically, that occur at naturally low densities, or that are particularly sensitive to habitat change. As an alternative to the status quo, management that focuses on stand heterogeneity (as opposed to tree-species monocultures with reduced diversity) should produce landscapes with high biocomplexity and biodiversity at multiple scales, and therefore these landscapes should be resilient in the face of disturbance.

Introduction

In the Pacific Northwest, one of the leading causes of habitat alteration of both forested and aquatic ecosystems is logging and its associated activities (Harris 1984). The response of vertebrate animals to logging in both riparian and upland habitats has garnered significant study and the resulting impacts have been negative, positive, or neutral (Cockle and Richardson 2003). Studies of the effects of forest practices on wildlife groups (small mammals, amphibians, and birds) have been extensive, ranging from site-specific studies to general reviews (Pough et al. 1987; Raymond and Hardy 1991; Andrén 1994; deMaynadier and Hunter 1995). To ameliorate the various habitat-related impacts on wildlife, mitigative measures have been applied in an attempt to reduce these effects and maintain biodiversity. The retention of riparian management zones (RMZs) immediately adjacent to watercourses has received considerable attention in the last few decades; however, relatively few studies have evaluated the efficacy of RMZs in providing habitat for wildlife and, to our knowledge, no study has evaluated the temporal trends of small mammals relative to RMZs over an extended (i.e., >2-year) time frame in western Washington.

Small mammals perform several important roles in the forest ecosystem, including serving as prey, dispersing mycorrhizal fungi, and contributing to the structural complexity of a site through selective herbivory and seed dispersal (Kelsey and West 1998; Sirotnak 2000; Cockle and Richardson 2003). Certain species of small mammals (e.g., Pacific Water Shrew, *Sorex bendirii*; Kepkay and Cathro 1998) depend on the habitat and structural heterogeneity provided by riparian zones, which are dynamic ecosystems characterized by high structural complexity, high species diversity relative to the adjacent upland, and a high frequency of natural disturbance (Naiman et al. 2000). The retention of riparian habitats in managed forests may be important for the conservation of small mammals, especially for those species dependent on habitat attributes that either only occur in riparian zones or are more abundant.

As Cockle and Richardson (2003) point out, many studies have looked at the effects of forest harvesting on wildlife; however, the majority have been conducted in upland forests rather than riparian areas. Despite the widespread acceptance of RMZs as a measure to protect streamside biodiversity, few studies have tested whether these riparian buffers are effective in providing either habitat or corridors for small mammals. Previous studies in upland habitats have shown that logging often leads to changes in the abundance of some species of small mammals but does not affect overall species richness (Martell and Radvanyi 1977; Kirkland 1990; McComb et al. 1993).

Western Washington RMZs have been required, at least in some form, since 1976. Between 1976 and 1988, the Washington Forest Practices Rules and Regulations required “streamside management zones” (SMZ) which were intended to provide “stream bank integrity and temperature control” (Washington State Forest Practices Board 1976, 1982). SMZ regulations prescribed riparian buffers 7.5–15 m wide with all nonmerchantable vegetation and “sufficient merchantable timber, if any, necessary to retain 50–75 percent of the summer mid-day shade of the water surface” left within the SMZ (Washington State Forest Practices Board 1976). In 1988, the RMZ regulations were strengthened considerably, requiring riparian buffers up to 30 m wide (Washington State Forest Practices Board 1988). The change in regulations was a result of the Timber, Fish, and Wildlife (TFW) Agreement reached in 1987 among representatives of the Washington State tribes, forest-products industry, environmental community, and natural-resource agencies. The agreement sought to resolve conflicts between these diverse groups by recognizing the common goal of preserving natural resources while simultaneously maintaining a viable timber industry.

In 1990, the Wildlife Steering Committee of the Cooperative Monitoring, Evaluation, and Research Committee (CMER) of the TFW Agreement initiated a study to evaluate the efficacy of

RMZs in providing habitat for wildlife. The study was designed to evaluate the abundance and diversity of wildlife in RMZs by comparing two different buffer configurations at unlogged control sites. The buffer configurations were based on the RMZ guidelines mandated at the time (Washington State Forest Practices Board 1988) and on modifications to the 1988 guidelines. The wildlife groups selected for study were terrestrial and aquatic amphibians, riparian-associated songbirds, small mammals (rodents and insectivores), and bats. A precursor to this study (O'Connell et al. 2000) compared the diversity and abundance of these wildlife groups before and immediately after logging to identify the short-term temporal effects of logging both within and between treatments. In 2001, CMER initiated a follow-up study to evaluate the long-term effects of RMZs on wildlife. This study is the only one in Washington to evaluate the long-term post-treatment effects of riparian management strategies on wildlife based on data collected at the same sites.

The objectives of this study were to evaluate the effectiveness of two RMZ prescriptions to provide habitat for small mammal populations at the 10-year post-treatment interval. The data collected 10-years post-harvest were compared to data collected both pre-harvest and 2-years post-harvest to identify temporal changes in small-mammal relative abundance and species richness relative to an unlogged control site (~65–75-year-old Douglas-fir stands) and two treatments: (1) clearcuts with streams buffered as per the 1988 state regulations (buffer width not <7.5 m) and (2) clearcuts with streams buffered to include features such as snags and unharvestable live trees (modified RMZ width from 7.5 to 22.7 m, depending on local conditions). This study is a continuation of a study implemented in 1991 that evaluated the immediate post-logging efficacy of RMZs in providing habitat for selected groups of wildlife (O'Connell et al. 2000).

Methods

This study was a component of a larger study evaluating the efficacy of RMZs in providing habitat for selected groups of wildlife. The larger study was implemented in 1991, with pre-treatment sampling in 1992 and 1993. The first post-treatment sampling occurred in 1995 and 1996, with a 10-year post-treatment sample (this study) in 2003 and 2004. The temporal nature of this study required that sampling techniques used in the pre-harvest and 2-year post-harvest sampling intervals be repeated in the 10-year post-harvest sampling interval to ensure consistency among the data sets from each sampling interval.

The study design was based on a split-plot repeated-measures design, with 18 sites assigned to one of three treatments. Each site was split into two distinct habitat types: (1) terrestrial riparian and (2) upland. Terrestrial riparian habitats occurred within 5 m of the ordinary high-water mark adjacent to a perennial stream and extended to the point where the vegetation visibly shifted to an upland type. Upland habitats extended upland from the zone of transition from riparian to upland vegetation. Upland sampling occurred 100 m upslope from terrestrial riparian habitat.

The 18 sites were assigned to one of three treatments:

1. **Control**: This treatment consisted of continuous stands of maturing second growth (65–75 years old) dominated by coniferous trees, with no upland harvesting for the duration of the study.
2. **State**: This treatment consisted of a logged upland forest with a narrow forested buffer retained in the riparian zone. The RMZ was based on the Washington Forest Practices Rules and Regulations in place in 1988 concerning buffer width and number of leave trees for Type 3 (fish-bearing, perennial) streams (Table 4-1). Streams of Type 2, 4, and 5 that were structurally similar to the Type 3 streams were also used in the study but were buffered according to Type 3 regulations to keep the experimental treatments uniform (Appendix

4-A). The RMZ extended from the ordinary high-water mark to the line where vegetation changes from riparian to upland but was not less than 15 m wide. The number and arrangement of leave trees was modified to accommodate the logging operation, and the RMZ width was expanded as necessary to include swamps, bogs, marshes, or ponds adjacent to the stream.

3. **Modified**: This treatment consisted of a logged upland forest with a wider, more variable forested buffer retained in the riparian zone. The RMZ was based on harvest prescriptions developed by O'Connell et al. (2000). The prescription for the modified buffer departed from the 1988 Forest Practices Rules and Regulations in three ways:

   i) The 1988 guidelines specified a minimum canopy cover based on stream temperature classification and the elevation at the midpoint of the stream. If the cover requirement was met, selective cutting was allowed in the no-entry zone (i.e., within 7.5 m of the ordinary high-water mark). The modified buffer prescription differed from the state buffer prescription in that harvest was prohibited within the no-entry zone. Cover requirements were the same as those indicated in the 1988 rules (WAC-222-30-040 [2] Temperature Control, WSFPB 1988). Specifically, WAC-222-30-040 (2) states that:

   > "All nonmerchantable vegetation that provides mid-summer and mid-day shade of the water surface should be retained; and
   >
   > Sufficient merchantable timber, if any, necessary to retain 50% of the summer mid-day shade of the water surface, except when ambient water temperatures exceed 15.5° C for a 7-day period. In this case, 75% of the shade should be retained."

   ii) The modified prescription applied 1988 guidelines for selective harvest in state buffer RMZs to a variable width located 7.5–22.7 m from the ordinary high-water mark. This prescription produced a buffer of at least 7.5 m from the ordinary high-water mark along the riparian/upland boundary.

   iii) The modified prescription increased the number of wildlife reserve trees, recruitment trees, and downed logs left for each acre logged. State Forest Practices Rules and Regulations required only three wildlife reserve trees, two green recruitment trees, and two downed logs left for each logged acre. Unless the wildlife tree violated Washington Department of Labor and Industry requirements, all wildlife reserve trees were retained on modified buffer sites. Wildlife reserve trees were defined as follows:

- **Type 1**: Live tree with defective or deformed sound tops, trunks, and roots.
- **Type 2**: Dead tree with a sound top, trunk, and roots.
- **Type 3**: Live or dead tree with unstable top or upper portion.
- **Type 4**: Live or dead tree with unstable trunk or roots, with or without bark. This includes "soft" snags as well as live trees with unstable roots caused by root rot or fire. Type 4 reserve trees are the most dangerous.

Table 4-1. Leave-tree requirements per water type and average riparian management zone (RMZ) width for western Washington (WAC 222-30-020 [4c]) using January 1988 or November 1988 rules.

Water type / Avg. width (m)	RMZ Max. width (m)	Ratio of conifers: Deciduous / Size of leave trees	# Trees / 300 m (each side) by bed material	
			Gravel/Cobble	Boulder/Bedrock
1 & 2 / ≥22	30	Representative of stand	50	25
1 & 2 / ≤22	22	Representative of stand	100	50
3 / ≥1.5	15	2:1 / 30 cm or next largest available	75	25
3 / ≤1.5	7.5	1:1 / 15 cm diameter or next largest available	25	25

Study Area

The 18 study sites were distributed throughout western Washington in Cowlitz, Grays Harbor, King, Lewis, Pierce, and Thurston counties (west of the Cascade Mountains on the Olympic Peninsula, in southwestern Washington, the Puget Trough, and on the western slopes and crest, or the Washington Cascades physiographic provinces) (Figure 4-1).

Study sites were established in 1992 and 1993 and were selected to minimize variation in forest age and composition, elevation, moisture condition, and water type. Initially, 18 sites were selected for this study based on the following criteria:

1. Low elevation (<620 m).
2. Second-growth forest (55–66 years old) dominated by Douglas-fir (*Psuedotsuga menziesii*).
3. Type 3 water according to state forest regulations. Type 4 was selected if streams differed in only the presence of salmonids. Water types are defined in Table 4-2.
4. Predominantly coniferous riparian canopy with deciduous tree component.
5. At least 500 m in stream length.
6. Road access within 500 m.

Figure 4-1. Distribution of western Washington study sites. West Fork Falls Creek replaced Vail in the ten-year post-harvest sampling interval.

Table 4-2. Water-typing criteria (WAC 222-16-030) used in Washington in 1988. Source: Washington State Forest Practices Board (1988).

	Water Type				
Parameter	**1**	**2**	**3**	**4**	**5**
Channel Width	N/A	≥ 6 m between OHWM	**Anadromous:** > 1.5m between OHWM Resident Game: > 3 m between OHWM	> 0.6 m between OHWM	> 0.6 m between OHWM
Gradient	N/A	< 4%	**Anadromous:** < 12% not upstream of a falls > 3m in height Resident Game: < 12%	N/A	N/A
Flow	N/A	N/A	**Anadromous:** N/A Resident Game: > 0.3 CFS at summer low flow	N/A	N/A
Impoundment	N/A	Water surface area of < 0.4 ha at seasonal low flow	**Anadromous:** Water surface area of < 0.4 ha at seasonal low flow Resident Game: Water surface area of < 0.2 ha at seasonal low flow	N/A	N/A
Fisheries	N/A	Used by substantial numbers of anadromous or resident game fish for spawning and rearing and migration	Used by significant numbers of anadromous or resident game fish for spawning and rearing and migration	Not used by significant numbers of fish	Not used by significant numbers of fish
Diversion	N/A	Domestic use for > 100 residences or campsites. Includes upstream reach of 450 m or until the drainage area is < 50%, whichever is less	Domestic use for > 10 residences or campsites. Includes upstream reach of 450 m or until the drainage area is < 50%, whichever is less	N/A	N/A
Other	All water within their OHWM inventoried as “Shoreline of the State” excluding related wetlands	Streams flowing through campgrounds available to public having ≥ 30 campsites	Contributes > 20% of flow to Type 1 or 2 water. Anadromous fish impoundments have outlet to stream with anadromous fish	N/A	All natural waters not classified as Type 1, 2, 3 ,or 4 or seepage areas, ponds and drainways having short run-off periods

Field Sampling

Small-mammal trapping occurred at 18 sites in the western Washington study area during three sampling intervals (Table 4-3). Site-specific monitoring varied somewhat over the years, owing in part to difficulties related to site selection and setup at the outset, logging of a control site, and delayed logging at one of the modified buffer sites (Appendix 4-B). At each site, four transects (each 400 m long) were established—two in the riparian and two in the adjacent upland area (one of each on either side of the stream)—with the upland transects located approximately 100 m upslope of the riparian transects (Figure 4-2). Centered on each transect, 36 stations 10 m apart were established, with 2 snap traps baited with a mixture of whole oats and peanut butter set at each station (36 × 2 × 4 = 288 traps per site). Additionally, 18 pitfall traps (two #10 cans taped together, with the ends removed except the end of the bottom tin, which was left intact) were buried in the soil at 15 m intervals along both the riparian and upland transects on one side of the stream (36 per site); these pitfall traps were centered on the snap-trap locations along the transects (Figure 4-2). The snap traps were checked daily for four consecutive days; the pitfall traps were checked every three to five days for four weeks. Captured animals were weighed and measured, identified to species whenever possible in the field, labelled, bagged, and frozen.

Table 4-3. Sampling dates for each sampling interval during which small mammals were trapped in western Washington.

Sampling Interval	Year	Sites	Start	End	Sampling Effort (days)
1	1992	13	13-Oct-92	14-Dec-92	62
	1993	18	11-Oct-93	3-Nov-93	23
2	1995[a]	18	3-Oct-95	18-Nov-95	46
	1996	17	8-Oct-96	11-Nov-96	34
	1997[b]	2	8-Oct-97	15-Nov-97	38
	1998	2	17-Oct-98	7-Nov-98	21
3	2003	17	16-Oct-03	1-Dec-03	46
	2004	17	11-Oct-04	22-Nov-04	42

[a] First year of post-harvest monitoring at the modified and state treatments
[b] Data excluded from analysis

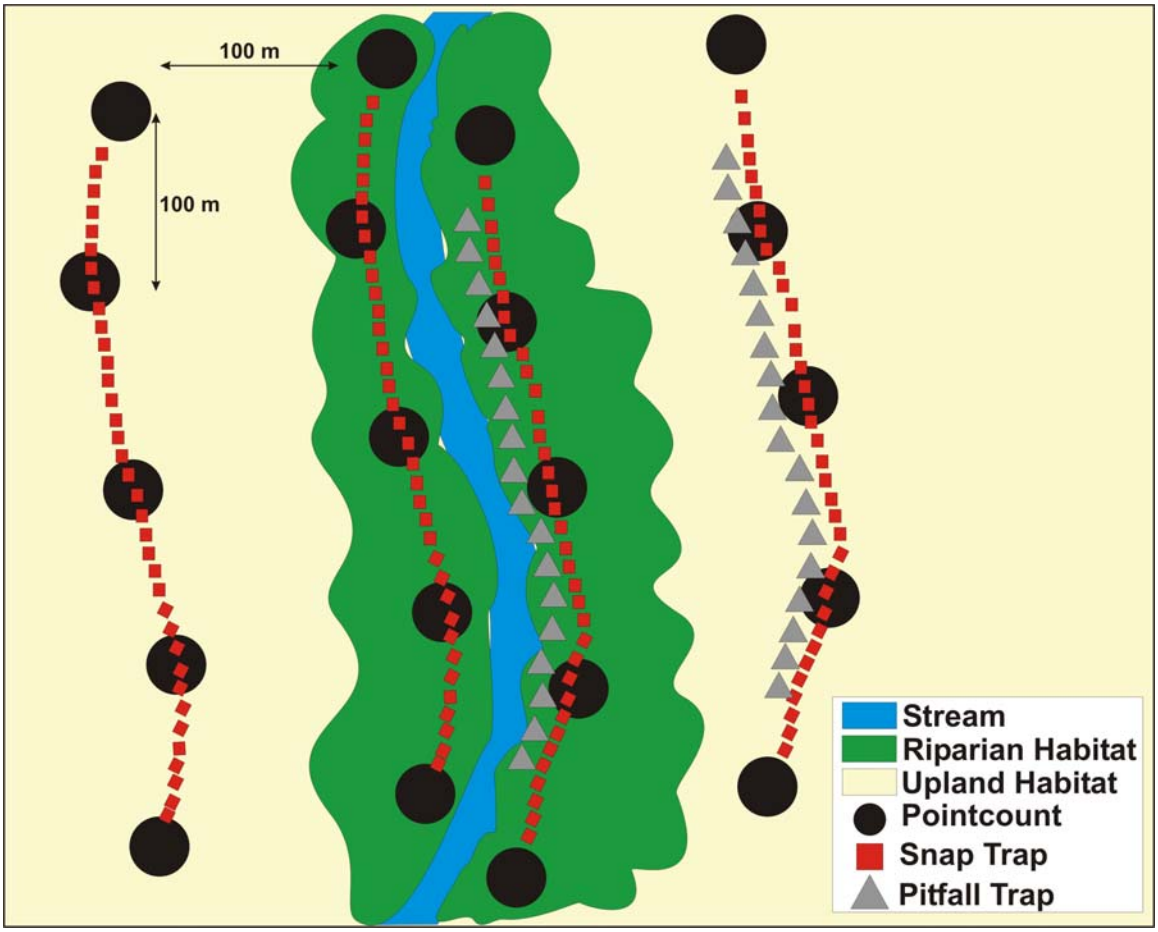

Figure 4-2. Schematic of sampling design at each site in western Washington showing the distribution of snap trap (red squares) and pitfall traps (gray triangles) relative to the riparian and upland transects. Point counts demarcate the beginning and end of each transect.

Statistical Analyses

Statistical analyses were performed using SAS V9.1 (© 2002–2003), R V2.2.1 (© 2005), and Microsoft Excel 2002 (© 1985–2003). Prior to performing all analyses, data were assessed to see if they fit a normal distribution using the Analyst Application in SAS, which produces four tests of normality (Shapiro–Wilk, Kolomogerov–Smirnov, Cramer–von Mises, and Anderson–Darling). Comparisons of relative abundance were done using log-transformed means ($\log[x + 1]$), which approximated a normal distribution. Where appropriate, post hoc multiple range tests were used to account for experiment-wise error and thereby to reduce Type I errors (Shaffer 1995). When pairwise comparisons of means were made, the Bonferroni method (Miller 1981) was applied to control the family-wise error (FWE) rate. The critical value of alpha was set at 0.1 and beta at 0.8 (*a priori*) for all analyses, which is appropriate for landscape-based studies and when analyzing data obtained from populations that are highly variable.

Species Richness and Community Similarity

We used the number of species found in each habitat type (i.e., riparian or upland) of each site during each sampling interval as a measure of species richness. Species richness data from riparian transects were used to test for differences relative to buffer width. To test whether logging influenced species richness or diversity, we combined data from the buffer treatment groups (i.e., state and modified riparian buffers). Because both groups received similar upland treatments (clearcut logging), pooling the treatments would provide a sense of whether or not the retention of riparian habitat, regardless of width, provides for the persistence of terrestrial species richness.

We used Morisita's coefficient of similarity (C) as a measure of species diversity because of its superior utility, relative to other measures, in pairwise comparisons of communities (Krebs 1999). Morisita's index (Morisita 1959; Horn 1966; Brower et al. 1990) measures community similarity and is based on Simpson's index of dominance (Simpson 1949). It calculates the probability that specimens randomly drawn from two sites will be of the same species, relative to the probability that specimens randomly drawn from the same site will be of the same species. This index is desirable because sample size and diversities of the samples have little influence on its calculation (Morisita 1959; Wolda 1981). Morisita's index returns a value from 0.0 (no similarity) to 1.0 (identical) and was chosen because it is affected little by large variations in sample size.

Relative Abundance

We used the catch per 100 trap nights as a measure of relative abundance for each species, adjusted for missed nights or non-functional traps (Nelson and Clark 1973). Comparisons were made between mean relative abundance measures, transformed as $\log(x + 1)$ to approximate a normal distribution. We tested variation in relative abundance only for the most common species; sample size was too small for analysis of other species.

Using PROC MIXED we ran a repeated-measures ANOVAs to test hypotheses about variation in relative abundance, followed by orthogonal and non-orthogonal contrasts (Table 4-4) of main effects (treatment, transect, sampling interval) and their interactions. The main experimental units for which measurements were repeatedly collected were the individual transects in riparian and upland habitats, and the data from each transect were pooled to derive a treatment mean for each of the three sampling intervals.

Contrasts were grouped into two sets of questions: differences *among* and differences *within* sampling intervals. In each question set, non-orthogonal contrasts were used to answer questions about the mean relative abundance of the more common species detected. Based on the objective of the overall study, the sampling intervals, treatments, and transects were grouped to best answer the questions of differences between the state and modified buffer sites relative to the control sites over time. The comparisons derived were based on the following assumptions:

1. All sites selected for this study were similar enough that results derived for one site could be extrapolated across all sites of a particular treatment; as such, pooling of sites into treatments was acceptable.
2. The upland habitats at the state and modified treatments represented the same treatment and thus could be pooled into a post-cut condition, defined by clearcut logging followed by replanting.
3. A comparison of the control uplands with the pooled state and modified uplands (i.e., pre-cut vs. post-cut) would address logging effects (i.e., does logging uplands affect the relative abundance of small mammals?).
4. A comparison of the control riparian transects to the pooled state and modified condition would determine if logging affects relative abundance of small mammals irrespective of buffer width.
5. A comparison between the modified and state riparian transects would provide a measure of the effect of a wide buffer versus a narrow buffer, and the effect that buffer width has on the relative abundance of small mammals.

The contrasts used are summarized in Table 4-4 . Question set 1 consists of 12 contrasts that compare the relative abundance of small mammals across time to determine if there are temporal relationships due to treatment or habitat type (transect). The contrasts in question set 2 evaluate relative abundance within each sampling interval relative to treatment and habitat type. This is particularly important for the pre-treatment sampling interval to establish a baseline of similarity or difference prior to treatment application.

Non-orthogonal contrasts ask specific questions and are a compromise between a “fishing expedition” (all possible pairwise *a posteriori* comparisons) and completely orthogonal (and therefore independent) contrasts. The maximum number of pairwise comparisons that could have been made for each species was 153, of which we made 42; 22 of which are orthogonal. The 20 non-orthogonal contrasts are designed to ask pertinent questions of the data to determine if there are measurable treatment effects on the relative abundance of small mammals across time. Specifically, we wanted to know if the relative abundance of small mammals had a relationship with one or more treatments and/or habitat types. Contrasts are preferred to multiple ANOVAs with post hoc multiple range tests to control FWE, because contrasts can reduce the number of questions asked of the data (and therefore avoid data mining) and because, in many cases, many of the comparisons of the means are not of interest.

There is little clarity in the literature regarding the “right” way to approach the use of non-orthogonal contrasts. Some authors have indicated that non-orthogonal contrasts are acceptable provided the redundancy is noted (Mead 1990). Others have commented that orthogonality is desirable but not essential, because a reasonable approximation of the probabilities exist for non-orthogonal contrasts. Miller (1981) indicated that because multiple comparisons are being made (regardless of orthogonality) the results should be corrected for experiment-wise error using a correction factor such as Scheffé, Bonferroni, or Sidak. Miller (1981) discusses methods for pairwise comparison that can be adapted for general contrasts, suggesting that Bonferroni be used for additive inequality and Sidak for multiplicative inequality and that these results need to be corrected when the number of comparisons is large. The term “large” is vague and conveys nothing about when to account for experiment-wise error when using contrasts analyses in repeated-measures ANOVAs. We did not consider the number of comparisons to be large relative to the number of comparisons that were possible for each species. In addition, because we were analyzing a temporal data set that included three sampling intervals with 17 contrasts per sampling interval, we could have generated up to 51 orthogonal contrasts. However, although there is ambiguity in the literature, we elected to adjust the critical value of alpha but expanded our definition of a statistically significant result to include *p*-values up to and including 0.1.

Table 4-4. Orthogonal and non-orthogonal contrasts used to identify differences in relative abundance (RA) among and within sampling intervals (SI) relative to treatment (control, modified, state) and transect (riparian, upland). SI 1 = 1992/1993; SI 2 = 1995/1996**; SI 3 = 2003/2004.**

	Sampling Interval, Transect, and Treatment Type	Question
Among	CONTROL RIPARIAN - SI 2 vs. SI 3	Is RA different in SI 2 compared to SI 3 for control riparian transects?
	CONTROL RIPARIAN - pre-cut vs. post-cut	Is RA pre-harvest different from post-harvest for control riparian transects?
	CONTROL UPLAND - SI 2 vs. SI 3	Is RA different in SI 2 compared to SI 3 for control upland transects?
	CONTROL UPLAND - pre-cut vs. post-cut	Is RA pre-harvest different from post-harvest for control upland transects?
	MODIFIED RIPARIAN - SI 2 vs. SI 3	Is RA different in SI 1 compared to SI 3 for modified riparian transects?
	MODIFIED RIPARIAN - pre-cut vs. post-cut	Is RA pre-harvest different from post-harvest for modified riparian transects?
	MODIFIED UPLAND - SI 2 vs. SI 3	Is RA different in SI 2 compared to SI 3 for modified upland transects?
	MODIFIED UPLAND - pre-cut vs. post-cut	Is RA pre-harvest different from post-harvest for modified upland transects?
	STATE RIPARIAN - SI 2 vs. SI 3	Is RA different in SI 2 compared to SI 3 for state riparian transects?
	STATE RIPARIAN - pre-cut vs. post-cut	Is RA pre-harvest different from post-harvest for state riparian transects?
	STATE UPLAND - SI 2 vs. SI 3	Is RA different in SI 2 compared to SI 3 for state upland transects?
	STATE UPLAND - pre-cut vs. post-cut	Is RA pre-harvest different from post-harvest for state upland transects?
Within	SI 1 RIPARIAN - Control vs. cut	In SI 1, was riparian RA at the controls different from the treatments?
	SI 1 RIPARIAN - Modified vs. State	In SI 1, was riparian RA different between the modified and state?
	SI 1 UPLAND - Control vs. cut	In SI 1, was upland RA at the controls different from the treatments?
	SI 1 UPLAND - Modified vs. State	In SI 1, was upland RA different between the modified and state?
	SI 2 RIPARIAN - Control vs. cut	In SI 2, was riparian RA at the controls different from the treatments?
	SI 2 RIPARIAN - Modified vs. State	In SI 2, was riparian RA different between the modified and state?
	SI 2 UPLAND - Control vs. cut	In SI 2, was upland RA at the controls different from the treatments?
	SI 2 UPLAND - Modified vs. State	In SI 2, was upland RA different between the modified and state?
	SI 3 RIPARIAN - Control vs. cut	In SI 3, was riparian RA at the controls different from the treatments?
	SI 3 RIPARIAN - Modified vs. State	In SI 3, was riparian RA different between the modified and state?
	SI 3 UPLAND - Control vs. cut	In SI 3, was upland RA at the controls different from the treatments?
	SI 3 UPLAND - Modified vs. State	In SI 3, was upland RA different between the modified and state?

Co-inertia Analysis

We used the program R V2.2.1 (© 2005) to investigate habitat relationships between abundant or common species and the habitat attributes measured at each site and treatment. For this study, a co-inertia analyses (COIA) was used and was based on the ade4 package (Chessel et al. 2005). COIA is a multivariate statistical technique that provides for the visualization of the structure of ecological data via ordination. COIA is very general, and some existing methods appear as special cases of it (Dray et al. 2003). For example, interbattery analysis (Tucker 1958) is mathematically equivalent to a simultaneous principle components analysis (PCA) on two ecological tables followed by a COIA (i.e., PCA–PCA COIA). When table X contains qualitative variables and Y contains species numbers, it is usual to cross tables X and Y to obtain a matrix containing the distribution of species among the categories of environmental variables. A simple canonical analysis (CA) of this new table allows one to ordinate the species and the environmental classes (analysis of ecological profiles; Montana and Greig-Smith 1990). Binary discriminant analysis (Strahler 1978), which has been used in ecology (Del Moral 1982; Huang and Del Moral 1988), is mathematically equivalent to Romane's CA (Dray et al. 2003). Although this approach allows plotting of species and environmental classes (Ben-Shahar 1987; Ben-Shahar and Skinner 1988), no information about ordination of sites is available.

COIA is similar to other ecological ordination techniques, such as a canonical correspondence analysis (CCA; ter Braak 1986) and redundancy analysis (RDA; Wollenberg 1977); however, COIA is a general coupling method that maximizes the covariance between the variables of two tables. Separate tables X and Y can be analyzed by various methods, leading to different coupling methods. This approach aims to find a site score that is a linear combination of environmental variables, maximizing the variance of species centroids (i.e., separation of species niches). Unlike CCA, COIA does not have a regression step (Thiolouse et al. 2004). This difference has important implications for ecological data analyses: the regression step implies that in CCA and RDA the number of samples must be high compared to the number of environmental variables; COIA does not have this constraint. CCA also has an additional constraint (the total variance must be equal to 1) and must be avoided in the case of numerous environmental variables, which is common in biological studies and is the case in this study.

COIA is a paired ecological table analysis that relies on the covariance matrix produced from (in this case) a simultaneous PCA of the environmental and species data (Figure 4-3). Each table is related through a common number of rows, which in this case represents the number of sites. The output of the PCA can be visualized as two different representations of the sites in space: one relating to the relative abundance of species, the other to the habitat attributes. The covariance matrices produced by the PCA are then joined via COIA so that the relationship between species relative abundance and habitat data can be viewed. The resulting biplot allows for visualization of how the two initially unrelated data sets covary at the same sites. From the biplot, a description of the habitat variables can be derived for species of interest. The ability to quickly assign habitat relationships based on the relative position of species to habitat variables provides an effective way to describe the habitat relationships for each species in each sampling interval and for each treatment.

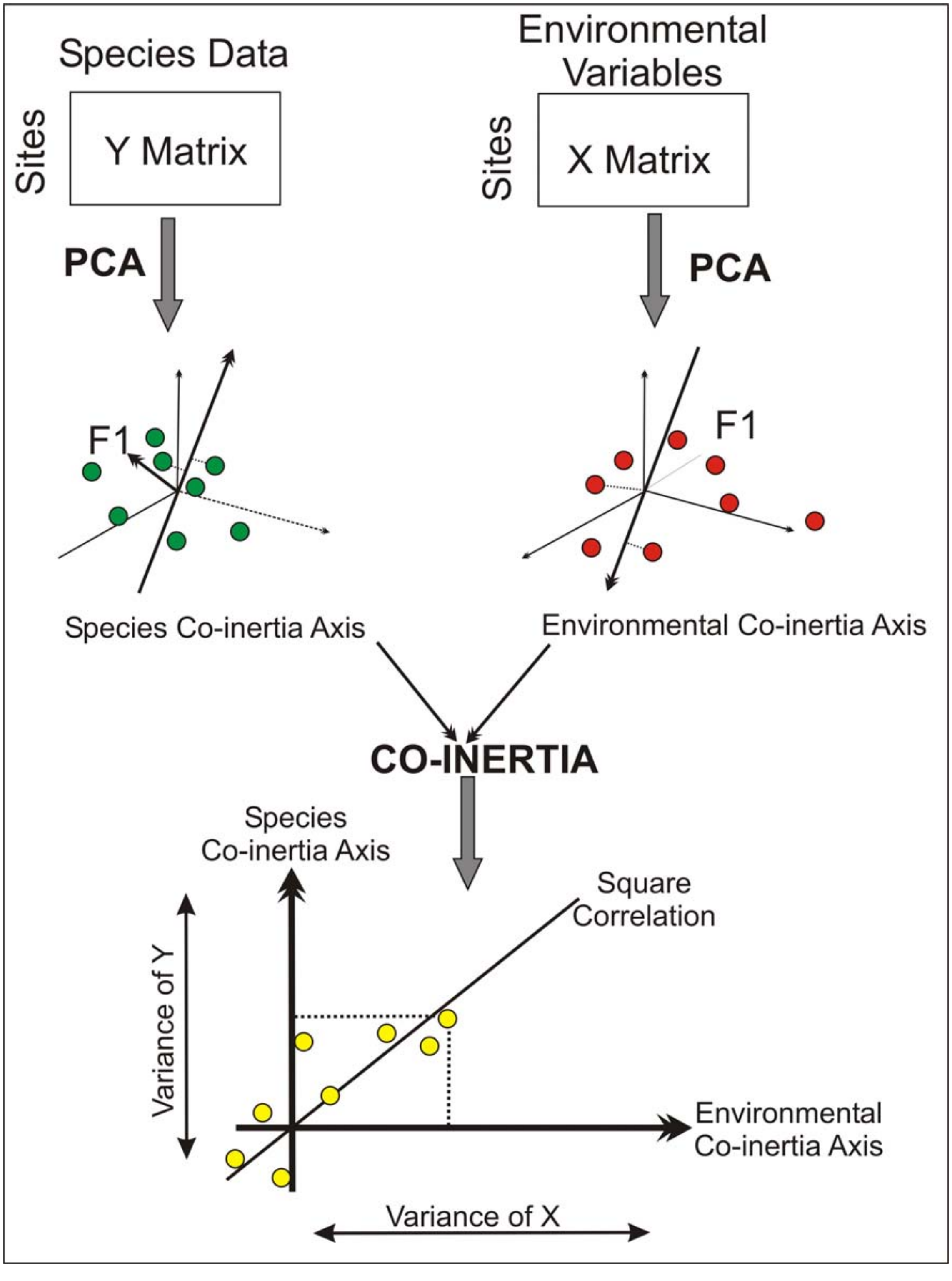

Figure 4-3. Schematic showing the principles of co-inertia analysis. The two ecological data tables X and Y produce two representations of the sites in two hyperspaces. Separate analyses find axes maximizing inertia (=variation) in space (F1 [first factorial axis]). Co-inertia analysis maximizes the square covariance between the projections of the sites on the co-inertia axes. Source: modified from Dray et al. (2003).

The relationship between species relative abundance and environmental variables can be viewed on a biplot. One approach is to view the species relative abundance and environmental variable biplots side by side to look for relationships. Alternatively, the species relative abundance biplot can be superimposed onto the environmental variable biplot for easier interpretation. When this is done, an environmental gradient can be described for each axis, with the horizontal axis (axis 1) representing the primary environmental gradient and the vertical axis (axis 2) describing a secondary gradient. Species that occur nearer the origin are indicative of (a) species with too few detections to infer habitat associations, (b) habitat generalists, or (c) species that respond to intermediate conditions along the gradients described by the axes. Species that are plotted farther away from the origin can be regarded as correlated with a particular habitat variable or suite of variables. The length of the vector from the origin to the location of the habitat variable in space is indicative of the strength of the correlation between the habitat variable and the gradient described for each axis. Unlike the more commonly used multivariate analyses, such as canonical correspondence analysis (ter Braak 1986), for which the number of environmental variables is limited to the number of sites minus one, COIA has no limitations on the number of environmental variables that can be used regardless of the number of sites sampled (Thioulouse et al. 2004).

A typical biplot resulting from a COIA is shown in Figure 4-4. In this example, the COIA combined information on small-mammal relative abundance and habitat variables measured along upland habitats of the control and logged treatments prior to logging. The biplots provide an opportunity to compare how the relative abundance of select species is associated with habitat attributes and how the relative abundance of those species varies relative to treatment. The correlation of each environmental variable with axes 1and 2 is indicated by its position relative to each axis, with variables closer to axis 1 or 2 more strongly correlated to the particular axis; variables occupying space between are correlated with both axes. Vector length is directly proportional to the degree of correlation: positive correlation to the right of the origin for axis 1 and above the origin for axis 2; negative correlation to the left of the origin along 1 one and below the origin along axis 2. Species farther from the origin are more abundant in environments that are dominated by particular variables. For example, the pre-harvest riparian biplot revealed an environmental gradient tending toward mixed coniferous–deciduous forests with high structural complexity and increasing forest openness along axis 1, while the gradient associated with axis 2 describes increasing forest openness and increasing herbaceous growth (Figure 4-4). The riparian biplot reveals that the abundance of the *Peromyscus* species (all are considered generalists) is not strongly correlated with either axis 1 or axis 2. The species' position on the biplot indicates that the abundance of *Peromyscus* is more likely to be higher in habitat characterized by both increasing habitat complexity and increasing canopy cover.

Results

Prior to completing all statistical analyses, the data were partitioned in two ways: (1) only those sites sampled in all three sampling intervals; and (2) all sites sampled, including sites used as replacement sites, in all three sampling intervals. The results of all analyses were consistent between both data sets, and the results reported below are based on the analyses that included all sites sampled. We assumed that the inclusion of West Fork Falls Creek would not introduce bias into the data and the two parallel analyses verfied that assumption. The regional scale of this study lends itself to this type of site-level replacement, because all sites within the region are considered part of the available sampling population. Given that the replacement site was selected using the same criteria as the original sites, it is representative of the region in which sampling occurred.

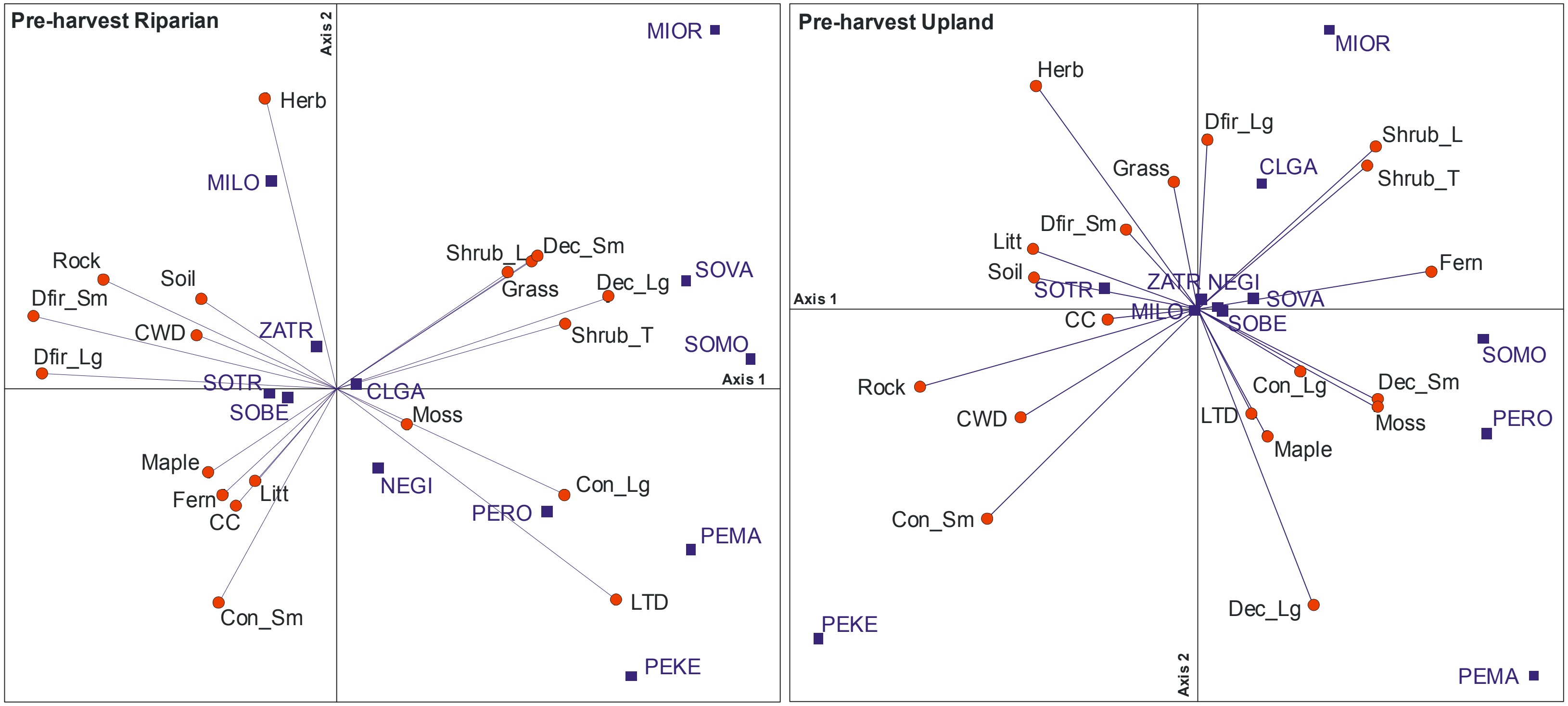

Figure 4-4. Typical biplot showing small-mammal–habitat relationships in riparian and upland habitats prior to logging. Shrub_T = tall shrubs; Shrub_L = low shrubs; CC = overstory canopy cover; CWD = coarse woody debris; Fern = fern cover; Grass = grass cover; Herb = herbaceous cover; Litt = deciduous leaf-litter cover; LTD = deciduous leaf-litter depth; Moss = moss cover; Rock = rock cover; Soil = exposed soil; Maple = vine maple; Dec_Sm = bigleaf maple and red alder <1.3 m; Dec_Lg = bigleaf maple and red alder >1.3 m; Con_Sm = western redcedar and western hemlock < 1.3 m; Con_Lg = western redcedar and western hemlock >1.3 m; Dfir_Sm = Douglas-fir <1.3 m; Dfir_Lg = Douglas-fir >1.3 m. CLGA = *Clethrionomys gapperi*; MILO = *Microtus longicaudus*; MIOR = *Microtus oregoni*; NEGI = *Neurotrichus gibbsii*; PEKE = *Peromyscus keeni*; PEMA = *Peromyscus maniculatus*; PERO = *Peromyscus* species; SOBE = *Sorex bendirii*; SOMO = *Sorex monticolus*; SOTR = *Sorex trowbridgii*; SOVA = *Sorex vagrans*; ZATR = *Zapus trinotatus*.

The numbers of snap- and pitfall-trap nights sampled along riparian and upland transects at each site during each sampling interval are tabulated for comparison of distribution of sampling effort (Table 4-5, Table 4-6, Table 4-7). Site- and trap -specific data were not available for the first two sampling intervals (only trap-night totals were provided). For these intervals, we assumed[7] that the number of set snap traps was the same and that, on average, the number of traps set off without captures was similar to what we encountered during the second post-harvest period (i.e., 10 years after logging). We also evaluated how many of our pitfall trap nights were affected by extraneous circumstances, such as heavy rains, snow cover, and blowdown. Based on the proportion of each transect at each site that was replicated from the previous sampling intervals, we applied a correction factor to the number of pitfall-trap nights; in most cases, the adjustment to the number of pitfall-trap nights was minimal. For the pre-harvest and 2-year post-harvest controls, the average trap nights calculated for the 10-year post-harvest controls were used. For modified and state treatments in the 2-year post-harvest interval, the average number of trap nights obtained for each treatment in the 10-year post-harvest sampling interval was applied. Values in Table 4-7 were used in all subsequent analyses.

Table 4-5. Pitfall-trap nights obtained for each site during each sampling interval (SI). Values for the pre-harvest (SI 1) and first post-harvest (SI 2) intervals are based on averages of values obtained during the ten-year post-harvest period (SI 3).

		SI 1				SI 2				SI 3			
Pitfall Traps		1992		1993		1995		1996		2003		2004	
Site Name	Treatment	Rip	Upl	Rip	Upl	Rip	Upl	Rip	Upl	Rip	Upl	Rip	Upl
Abernathy	Control			504	504	504	504	504	504	486	486	486	486
Elbe Hills	Control			504	504	504	504	504	504	486	486	462	465
Hotel Creek	Control	504	504	504	504	504	504	504	504	486	486	468	468
Porter Creek	Control	504	504	504	504	504	504	504	504	468	468	486	486
Taylor Creek	Control	504	504	504	504	504	504	504	504	486	482	468	468
Vail	Control	504	504	504	504	504	504	504	504				
West Fork Falls Creek	Control									558	558	462	486
Blue Tick	Modified			504	504	504	504	504	504	486	461	486	423
Eleven 31	Modified	504	504	504	504	504	504	504	504	432	432	468	468
Griffen Creek	Modified	504	504	504	504	504	504	504	504				
Ms Black	Modified	504	504	504	504	504	504	504	504	524	570	486	513
Ryderwood 860	Modified	504	504	504	504	504	504	504	504	432	432	486	486
Side Rod	Modified			504	504	504	504	504	504	486	486	468	468
Eleven 32	State	504	504	504	504	504	504	504	504	432	432	468	468
Kapowsin	State	504	504	504	504	504	504	504	504	504	504	468	468
Night Dancer	State			504	504	504	504	504	504	486	486	486	486
Potpourri	State	504	504	504	504	504	504	504	504	468	468	486	486
Ryderwood 1557	State	504	504	504	504	504	504	504	504	486	486	486	486
Simmons Creek	State	504	504	504	504	504	504	504	504	504	504	468	468

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[7] Site- and transect-specific trapping effort data were not avaialbe for the first two sampling intervals. To generate relative abundance values for use in the temporal analyses, we needed to generate a site- and transect-specific estimate of trapping effort.

Table 4-6. Snap-trap nights obtained for each site during each sampling interval (SI). Values for the pre-harvest (SI 1) and first post-harvest (SI 2) periods are based on averages of values obtained during the ten-year post-harvest period (SI 3).

		SI 1				SI 2				SI 3			
Snap Traps		1992		1993		1995		1996		2003		2004	
Site Name	Treatment	Rip	Upl	Rip	Upl	Rip	Upl	Rip	Upl	Rip	Upl	Rip	Upl
Abernathy	Control			533	517	533	517	533	517	508	490	512	507
Elbe Hills	Control			533	517	533	517	533	517	520	512	515	526
Hotel Creek	Control	533	517	533	517	533	517	533	517	537	543	546	557
Porter Creek	Control	533	517	533	517	533	517	533	517	456	487	534	542
Taylor Creek	Control	533	517	533	517	533	517	533	517	543	553	546	547
Vail	Control	533	517	533	517	533	517	533	517				
West Fork Falls Creek	Control									648	423	531	520
Blue Tick	Modified			533	517	513	525	513	525	512	503	464	522
Eleven 31	Modified	533	517	533	517	513	525	513	525	515	562	529	538
Griffen Creek	Modified	533	517	533	517	513	525	513	525				
Ms Black	Modified	533	517	533	517	513	525	513	525	527	533	542	540
Ryderwood 860	Modified	533	517	533	517	513	525	513	525	493	470	532	545
Side Rod	Modified			533	517	513	525	513	525	500	512	522	525
Eleven 32	State	533	517	533	517	490	474	490	474	486	612	552	544
Kapowsin	State	533	517	533	517	490	474	490	474	389	352	546	551
Night Dancer	State			533	517	490	474	490	474	467	412	469	511
Potpourri	State	533	517	533	517	490	474	490	474	480	443	536	414
Ryderwood 1557	State	533	517	533	517	490	474	490	474	515	489	532	549
Simmons Creek	State	533	517	533	517	490	474	490	474	389	352	517	467

Table 4-7. Total trap nights calculated by summing snap-trap nights and pitfall-trap nights for each site and sampling interval (SI). Rip = riparian; Upl = upland.

		SI 1				SI 2				SI 3			
Snap and Pitfall		1992		1993		1995		1996		2003		2004	
Site Name	Treatment	Rip	Upl	Rip	Upl	Rip	Upl	Rip	Upl	Rip	Upl	Rip	Upl
Abernathy	Control			1037	1021	1017	1029	1017	1029	998	989	950	1008
Elbe Hills	Control			1037	1021	1037	1021	1037	1021	1006	998	977	991
Hotel Creek	Control	1037	1021	1037	1021	1037	1021	1037	1021	1023	1029	1014	1025
Porter Creek	Control	1037	1021	1037	1021	1037	1021	1037	1021	924	955	1020	1028
Taylor Creek	Control	1037	1021	1037	1021	1037	1021	1037	1021	1029	1035	1014	1015
Vail	Control	1037	1021	1037	1021	1037	1021	1037	1021				
West Fork Falls Creek	Control									1206	981	993	1006
Blue Tick	Modified			1037	1021	1017	1029	1017	1029	998	964	950	945
Eleven-31	Modified	1037	1021	1037	1021	1017	1029	1017	1029	947	994	997	1006
Griffen Creek	Modified	1037	1021	1037	1021	1017	1029	1017	1029				
Ms Black	Modified	1037	1021	1037	1021	1017	1029	1017	1029	1051	1103	1028	1053
Ryderwood 860	Modified	1037	1021	1037	1021	1017	1029	1017	1029	925	902	1018	1031
Side Rod	Modified			1037	1021	1017	1029	1017	1029	986	998	990	993
Eleven-32	State	1037	1021	1037	1021	994	978	994	978	918	1044	1020	1012
Kapowsin	State	1037	1021	1037	1021	994	978	994	978	893	856	1014	1019
Night Dancer	State			1037	1021	994	978	994	978	953	898	955	997
Potpourri	State	1037	1021	1037	1021	994	978	994	978	948	911	1022	900
Ryderwood 1557	State	1037	1021	1037	1021	994	978	994	978	1001	975	1018	1035
Simmons Creek	State	1037	1021	1037	1021	994	978	994	978	893	856	985	935

Relative Abundance Comparisons

General

The total catch—comprising 20 species for riparian and upland transects combined in the control, modified, and state buffer sites for the pre-, 2- and 10-year post-harvest periods—amounted to 3808, 5435, and 4286 animals, respectively, for each treatment (Table 4-8). In each of the three sampling periods, the catch was dominated by deer mice (*Peromyscus* spp.), followed by shrews (*Sorex* spp., particularly *trowbridgii*) and then voles (mainly *Microtus* spp.) Overall, these three groups constituted 95.6% of the total catch, with voles making up 16.6%, shrews 30.6%, and deer mice 48.4%. Some of the less common small mammals included the Shrew Mole (*Neurotrichus gibbsii*) and the Pacific Jumping Mouse (*Zapus trinotatus*), which comprised 3.5% and 0.9% of the catch, respectively. Among the rarer specimens caught were the Northern Flying Squirrel (*Glaucomys sabrinus*), Ermine (*Mustela erminea*), Coast Mole (*Scapanus oranius*), and Townsend's Mole (*S. townsendii*), and chipmunks (*Tamias* spp.). We could not identify the species of 201 individuals of the genus *Sorex* due to poor specimen condition. Juvenile deer mice that had not reached the tail-length criterion allowing classification as *P. maniculatus or P. keeni* were classified as *Peromyscus* spp. In addition, 89 individuals in the genus *Microtus* could not be assigned to species due to poor specimen condition. Where possible, these data were included in statistical calculations, but they were not included in analyses for individual species.

Natural Variation

To convey an appreciation of naturally occurring changes in abundance of small-mammal populations over time, the mean numbers of animals captured in riparian and upland control sites were plotted by species for each of the three sampling periods (Figure 4-5). From this plot, it is evident that quite large in relative abundance occurred for some species in riparian habitats. For example, the relative abundance of the Western Redbacked Vole (*Clethrionomys gapperi*) exhibited a 17-fold increase in the 10-year post-harvest period relative to pre-harvest conditions. *Peromyscus maniculatus* relative abundance remained stable between the pre-harvest and immediate post-harvest intervals, with a 5.5-fold decrease in relative abundance 10 years after logging. Over time, the relative abundance of individuals assigned to the *Peromyscus* spp. category increased first by 4 times and then by an additional 8 times, for a 12-fold increase 10 years after logging relative to pre-harvest values. *Microtus oregoni* relative abundance decreased immediately after logging, with a 12-fold decrease in the 10-year post-harvest period.

Similar trends were noted for the same four species on upland transects: the relative abundance of Western Redbacked Vole increased 15-fold in the 10-year post-harvest interval relative to pre-harvest numbers; *P. maniculatus* decreased by a factor of 5.7 in the 10-year post-harvest period; *Peromyscus* spp. increased 7-fold 2 years after logging, with an additional 3-fold increase in the 10-year post-harvest sampling interval; and the relative abundance of *M. oregoni* decreased by a factor of 5 on upland transects of the control in the 10-year post-harvest period. On both riparian and upland transects, the relative abundance of *Neurotrichus gibbsii* increased between the pre- and 2-year post-harvest periods, followed by a decline to pre-harvest levels at the end of the study period. In general, the changes in relative abundance noted for a given species were consistent between habitat types (Figure 4-5).

Table 4-8. Relative abundance expressed as a proportion of total captures per transect and treatment for each species caught during each sampling interval, 1992–2004. R = riparian; U = Upland.

	Pre-harvest						Two-years Post-harvest						Ten-years Post-harvest					
	Control		Modified		State		Control		Modified		State		Control		Modified		State	
Species Code	R	U	R	U	R	U	R	U	R	U	R	U	R	U	R	U	R	U
Clethrionomys gapperi	0.4	1.0	0.3	5.0	0.6	2.8	0.5	1.6	0.1	0.3	0.0	0.0	4.3	8.9	10.6	13.0	3.2	9.1
Glaucomys sabrinus	0.0	0.0	0.0	0.0	0.1	0.1	0.0	0.1	0.0	0.0	0.1	0.0	0.0	0.1	0.3	0.1	0.0	0.0
Microtus longicaudus	3.2	0.2	3.8	0.1	4.9	0.1	6.5	0.0	5.5	1.4	4.8	1.6	2.4	0.3	6.0	1.8	5.2	1.4
Microtus oregoni	8.5	8.7	3.5	4.1	8.6	4.6	6.6	4.6	10.4	24.1	18.8	19.6	0.6	1.5	4.8	10.2	4.6	8.9
Microtus spp.	1.6	0.0	0.7	0.1	0.6	0.0	1.5	0.0	2.8	0.6	1.7	0.4	0.0	0.0	0.5	0.6	0.0	0.3
Microtus townsendii	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.2	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Mustela erminea	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.1	0.2	0.4	0.1	0.1	0.2	0.0	0.2	0.1	0.0	0.1
Mus musculus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0
Neurotrichus gibbsii	5.9	3.8	6.3	4.1	4.9	4.4	5.1	6.0	6.9	1.8	2.9	0.4	3.3	2.6	2.6	4.0	1.1	0.7
Peromyscus keeni	16.4	29.1	26.1	25.4	27.8	33.2	16.0	25.4	9.3	2.1	8.3	2.5	14.7	17.5	15.8	3.3	11.8	3.1
Peromyscus maniculatus	7.1	8.4	10.0	13.8	4.6	13.0	6.2	8.5	8.6	12.9	6.4	13.0	1.1	1.8	1.5	0.3	1.7	1.0
Peromyscus spp.	2.8	2.9	6.8	11.2	5.5	8.0	10.4	14.5	29.2	35.4	23.0	45.0	30.4	38.7	34.2	42.8	47.7	55.1
Scapanus orarius	0.0	0.0	0.1	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.1	0.1	0.5	0.1	0.2	0.0	0.2	0.2
Scapanus townsendii	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.1
Sorex bendirii	2.2	0.4	1.6	1.6	1.9	0.4	3.2	0.6	0.4	0.0	1.0	0.1	1.3	0.3	2.2	1.2	1.8	0.5
Sorex cinerus	0.4	0.0	0.0	0.0	0.0	0.1	0.0	0.1	0.0	0.0	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0
Sorex monticolus	13.2	8.6	9.2	6.8	6.7	4.3	7.1	3.9	3.2	3.3	2.6	2.0	8.1	5.2	6.3	4.5	4.3	3.6
Sorex palustris	0.0	0.0	0.1	0.1	0.1	0.1	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Sorex spp.	1.0	0.2	0.7	0.7	0.1	0.4	0.2	0.2	0.3	0.2	0.3	0.3	5.7	3.9	2.7	4.0	3.5	3.6
Sorex trowbridgii	26.9	35.4	27.0	25.1	29.9	26.9	31.4	32.7	19.1	12.1	20.5	12.7	24.1	17.2	9.9	11.2	11.2	9.8
Sorex vagrans	9.3	1.5	3.3	1.2	2.7	1.5	3.6	1.1	1.1	4.3	3.0	1.7	3.3	1.8	1.4	2.4	2.6	1.5
Tamias spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.1
Tamias townsendii	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.4	0.2	0.3	0.1	0.0	0.0	0.0	0.1	0.0	0.1
Zapus trinotatus	1.0	0.0	0.3	0.3	0.6	0.0	1.7	0.2	1.9	0.9	5.8	0.4	0.2	0.0	0.9	0.0	0.9	0.8
Total Captures	**494**	**526**	**693**	**676**	**672**	**747**	**663**	**823**	**937**	**1028**	**860**	**1124**	**632**	**727**	**587**	**669**	**652**	**1019**

Relative Abundance in Riparian and Upland Habitats

For most species, mean catch was greater in upland habitats compared to the riparian at the control sites in all three sampling periods (Figure 4-5). With few exceptions, these differences were more pronounced in the captures of *Peromyscus*, *Sorex*, and *Clethrionomys*. A plot of the total catch of small mammals for riparian and upland transects, separately and combined (to convey a sense of total small-mammal abundance relative to treatment) and within treatments for each of three sampling intervals (Figure 4-6), reveals that total catch in riparian and upland transects combined was highest for modified buffer sites in all three sampling periods. Also, the total catch was generally higher in upland than riparian transects, and the difference was more pronounced in modified and state buffer sites than in the controls during the 10-year post-harvest period. In addition, among the different treatment levels, total catch for both riparian and upland transects was lowest in the control sites in the pre-harvest period.

Differences in the numbers of animals caught by species between riparian and upland transects within treatment levels and sampling periods were tested statistically using a t-test of the pooled snap and pitfall catch data. Most species did not show a significant difference (at $\alpha = 0.05$) between the total captures on riparian and upland transects. Those that did are presented in Table 4-9. Among the modified buffer sites in all three sampling periods, and control sites in the 10-year post-harvest period, differences in the numbers caught between riparian and upland transects did not differ significantly ($p > 0.05$) for any of the species. For the whole of the sampling matrix, in only nine situations (5.6% of the total comparisons) did the catch by species differ significantly between riparian and upland transects (Table 4-9). Prior to logging, the relative abundance of *S. trowbridgii* was 1.4 times greater on upland transects of the control sites (only marginally significant; $p = 0.05$), and the relative abundance of *P. maniculatus* was 3.3 times higher ($p = 0.02$) on upland transects compared to riparian transects of the state buffer treatment. Immediately after logging, two species were more abundant on riparian transects of the control sites (*S. monticolus*; $p = 0.03$; *S. bendirii*; $p = 0.04$), two species were more abundant on upland transects of the state buffer treatment (*P. maniculatus*; $p = 0.01$; *Peromyscus* spp.; $p = 0.008$), while only one species was more abundant on riparian transects of the state buffer treatment (*M. longicaudus*; $p = 0.002$). During the 10-year post-harvest sampling interval, the relative abundance of *Peromyscus* spp. continued to be more abundant ($p = 0.01$) on upland transects compared to riparian transects of the state buffer treatment. Similarly, the relative abundance of *M. oregoni* was 3.2 times greater in upland habitats relative to riparian habitats at the state buffer treatment ($p = 0.03$).

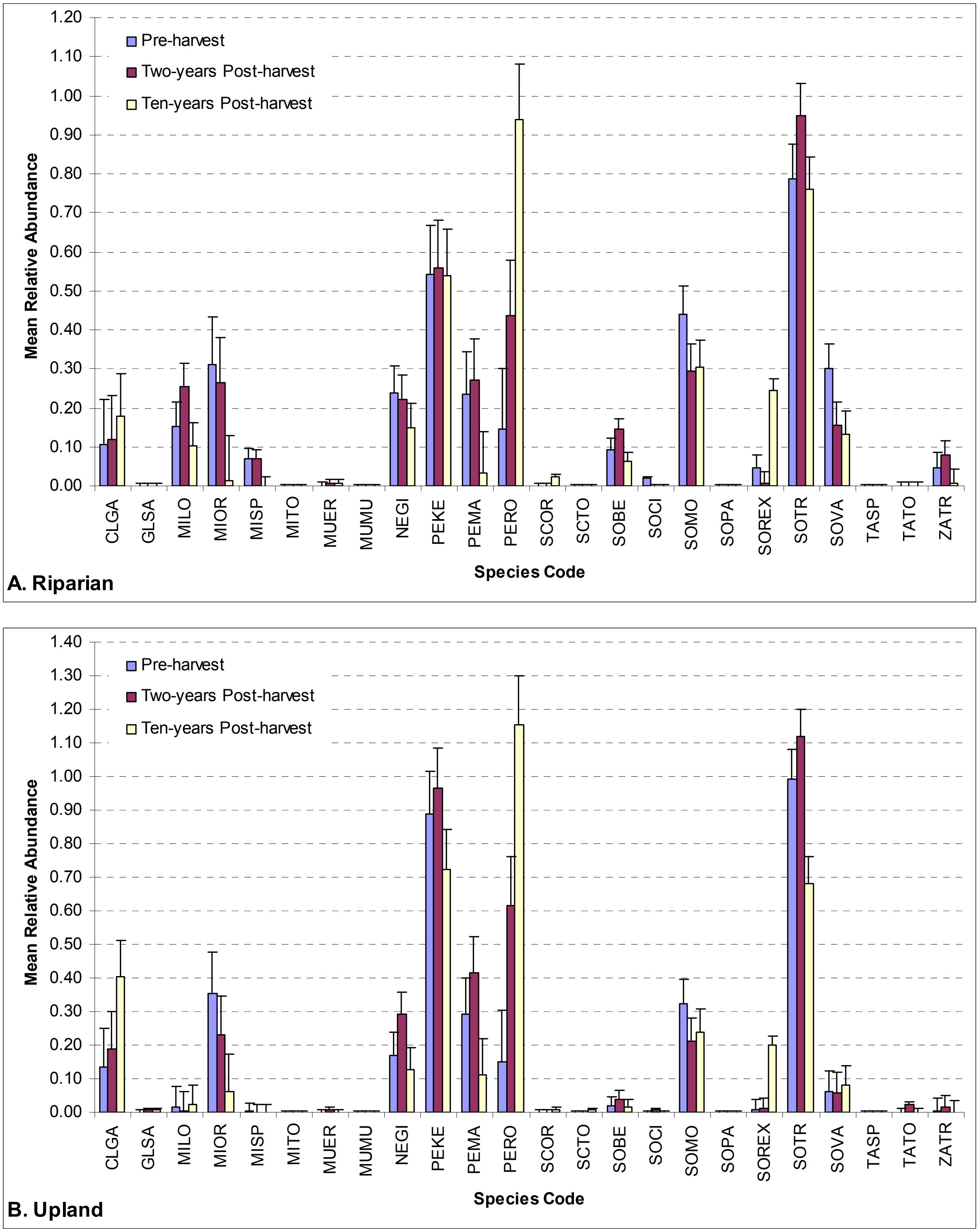

Figure 4-5. Mean number of captures (+SE) per habitat (A = Riparian, B = Upland) for each species of small mammal caught at the western Washington study sites, 1993–2004. Pre-harvest = 1993; Two-years post-harvest = 1995/1996; Ten-years post-harvest = 2003/2004. Species codes are explained in Appendix 4-C.

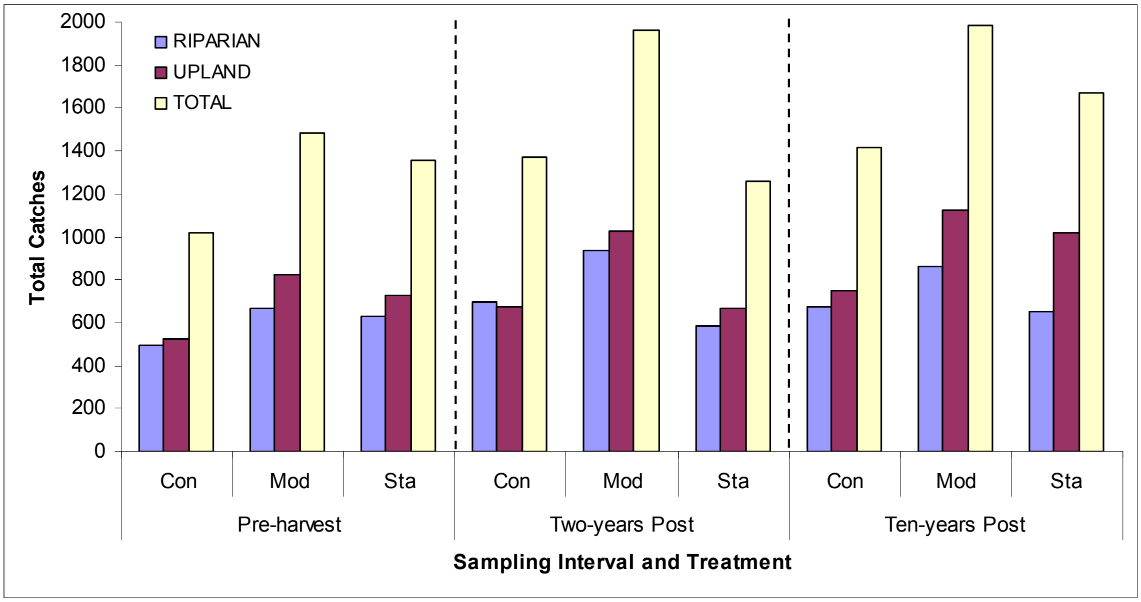

Figure 4-6. Total small-mammal captures per sampling interval, treatment, and transect at the western Washington study sites. Pre-harvest = 1993; Two-years post = 1995/1996; Ten-years post = 2003/2004; Con = control; Mod = modified; Sta = state.

Table 4-9. Species with a significant difference (α = 0.05) in total captures between riparian (Rip) and upland (Upl) transects by treatment for each sampling interval. Data from snap and pitfall trapping combined. Species codes are explained in Appendix 4-C. 1 = 1992/1993; 2 = 1995/1996; 3 = 2003/2004.

			Captures			Sites/Transect			
Sampling Interval	Treatment	Species	Rip	Upl	Total	Rip	Upl	Total	*P*
1	Control	SOTR	133	186	319	6	6	6	0.046
1	State	PEMA	31	97	128	6	6	6	0.015
2	Control	SOMO	47	32	79	6	6	6	0.032
2	Control	SOBE	21	5	26	5	4	5	0.037
2	State	PERO	198	506	704	6	6	6	0.008
2	State	PEMA	55	146	201	6	6	6	0.013
2	State	MILO	41	18	59	5	6	6	0.002
3	State	PERO	311	561	872	6	6	6	0.010
3	State	MIOR	30	91	121	6	6	6	0.029

For comparative purposes, the total catch by species for riparian and upland transects within treatment levels was standardized (captures per 100 trap nights) for each of the three sampling periods (Table 4-10). In this study, catch rates on riparian and upland transects for some species differ between treatments. For example, the mean relative abundance of *Peromyscus* spp. is higher on both riparian and upland transects of the modified and state buffer treatments relative to the control; and relative abundance was always higher on upland transects relative to riparian habitats, a trend that is noted across time. *M. longicaudus* was more abundant in riparian habitats at the modified and state buffer treatments compared to the control sites, with upland relative abundance equal among treatments in the pre-harvest sampling interval.

Table 4-10. Small-mammal relative abundance (captures per 100 trap nights) for all species of small mammals caught at western Washington study sites for each transect and treatment during each sampling interval. R = riparian; U = upland.

	Pre-harvest						Two-years Post-harvest						Ten-years Post-harvest					
	Control		Modified		State		Control		Modified		State		Control		Modified		Sstate	
Species	R	U	R	U	R	U	R	U	R	U	R	U	R	U	R	U	R	U
Clethrionomys gapperi	0.02	0.05	0.02	0.33	0.04	0.19	0.02	0.11	0.01	0.02	0.00	0.00	0.33	0.75	0.67	0.88	0.18	0.81
Glaucomys sabrinus	0.00	0.00	0.00	0.00	0.01	0.01	0.00	0.01	0.00	0.00	0.01	0.00	0.00	0.01	0.02	0.01	0.00	0.00
Microtus longicaudus	0.15	0.01	0.25	0.01	0.30	0.01	0.35	0.00	0.43	0.11	0.34	0.15	0.13	0.03	0.35	0.13	0.29	0.15
Microtus oregoni	0.41	0.45	0.24	0.27	0.52	0.31	0.35	0.31	0.79	2.01	1.36	1.87	0.03	0.09	0.29	0.69	0.26	0.83
Microtus spp.	0.08	0.00	0.05	0.01	0.04	0.00	0.08	0.00	0.21	0.05	0.13	0.03	0.00	0.00	0.03	0.04	0.00	0.02
Microtus townsendii	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.02	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Mustela erminea	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.01	0.02	0.03	0.01	0.01	0.01	0.00	0.01	0.01	0.00	0.01
Mus musculus	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00
Neurotrichus gibbsii	0.28	0.20	0.43	0.27	0.30	0.31	0.27	0.40	0.53	0.15	0.21	0.03	0.17	0.16	0.15	0.26	0.06	0.06
Peromyscus keeni	0.78	1.50	1.76	1.68	1.68	2.25	0.85	1.71	0.71	0.18	0.60	0.24	0.80	1.24	0.95	0.24	0.64	0.28
Peromyscus maniculatus	0.34	0.43	0.67	0.91	0.28	0.89	0.33	0.57	0.66	1.08	0.46	1.24	0.06	0.17	0.09	0.02	0.09	0.09
Peromyscus spp.	0.14	0.15	0.46	0.74	0.34	0.55	0.55	0.97	2.24	2.95	1.66	4.31	1.66	2.64	2.03	2.90	2.60	4.92
Scapanus orarius	0.00	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.01	0.01	0.02	0.01	0.01	0.00	0.01	0.02
Scapanus townsendii	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.01
Sorex bendirii	0.11	0.02	0.11	0.11	0.12	0.03	0.17	0.04	0.03	0.00	0.08	0.01	0.07	0.02	0.13	0.10	0.11	0.04
Sorex cinerus	0.02	0.00	0.00	0.00	0.00	0.01	0.00	0.01	0.00	0.00	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00
Sorex monticolus	0.63	0.44	0.63	0.45	0.40	0.29	0.38	0.26	0.25	0.28	0.18	0.20	0.42	0.32	0.37	0.30	0.25	0.35
Sorex palustris	0.00	0.00	0.01	0.01	0.01	0.01	0.00	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Sorex spp.	0.05	0.01	0.05	0.05	0.01	0.03	0.01	0.02	0.02	0.02	0.03	0.03	0.30	0.23	0.16	0.27	0.20	0.32
Sorex trowbridgii	1.28	1.82	1.82	1.66	1.80	1.82	1.67	2.20	1.47	1.00	1.48	1.22	1.28	1.03	0.62	0.80	0.61	0.89
Sorex vagrans	0.44	0.08	0.22	0.08	0.16	0.10	0.19	0.07	0.08	0.36	0.22	0.16	0.17	0.11	0.10	0.18	0.17	0.16
Tamias spp.	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.00	0.01
Tamias townsendii	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.03	0.02	0.03	0.01	0.00	0.00	0.00	0.01	0.00	0.01
Zapus trinotatus	0.05	0.00	0.02	0.02	0.04	0.00	0.09	0.02	0.15	0.07	0.42	0.04	0.01	0.00	0.05	0.00	0.05	0.07

During both post-harvest periods, differences in relative abundance are evident for several species in upland transects (Table 4-10). Immediately following logging, the relative abundance of *M. oregoni* was more than 6 times greater on upland transects of the modified and state buffer treatments relative to the control group. Relative abundance of *M. longicaudus* also increased on upland transects of logged sites following logging compared to control sites, where this species was not detected. A similar trend was noted for *P. maniculatus*, with an approximate 2-fold increase in relative abundance on upland transects of logged sites immediately following logging. The *Peromyscus* group (i.e., the *Peromyscus* that could not be assigned to species) also increased substantially on upland transects of logged sites compared to the control, with a 3- (modified buffer) to 4.4-fold (state buffer) increase observed. The only shrew to exhibit this response was *S. vagrans*, with a 5-fold increase on upland transects of the modified buffer treatment and a 2.3-fold increase on state buffer uplands.

The relative abundance of five species was greater in riparian habitats at the modified buffer treatment relative to the state treatment (Table 4-11). For example, the relative abundance of *Neurotrichus gibbsii* was 2.5 times greater on riparian transects at the modified buffer treatment. Four species were more abundant on riparian transects at the state buffer treatment compared to modified buffer riparian habitats (e.g., *S. vagrans*, *S. bendirii*) (Table 4-11), while the relative abundance of one species (*S. trowbridgii*) was equivalent on riparian transects at the modified and state buffer treatments immediately after logging.

Ten years after logging, trends evident in the 2-year post-harvest interval continued. Four of the five species that had higher relative abundance on logged uplands relative to the control sites immediately after logging continued to be more abundant in upland habitats at logged sites 10 years after logging. The one exception was *P. maniculatus*, which had lower relative abundance on logged uplands relative to control sites 10 years after timber harvest.

On riparian transects, seven species were more abundant at the modified buffer treatment compared to the state buffer treatment, and one (*S. vagrans*) was more abundant at the state buffer treatment compared to the modified buffer treatment. Four species had equivalent relative abundance on riparian transects at both treatments (Table 4-10). Four of the five species that were more abundant on riparian transects at the modified buffer treatment immediately following logging continued to be more abundant at the modified buffer treatment 10 years following logging. However, the relative abundance of *P. maniculatus* no longer differed between riparian transects of the logged treatments. At both logged treatments, the relative abundance of *C. gapperi* increased relative to pre-harvest and immediate post-harvest levels, with higher relative abundance observed at the modified buffer treatment (Table 4-10). Two years after logging, the relative abundance of *M. oregoni* was higher on riparian transects at the state buffer treatment. Ten years later, this species was slightly more abundant on riparian transects at the modified buffer sites.

Table 4-11. Species of small mammals with higher relative abundance on riparian transects at the modified or state treatment two-years post-logging.

Species	Treatment	Increase
Microtus longicaudus	Modified	1.26
Neurotrichus gibbsii	Modified	2.52
Peromyscus keeni	Modified	1.18
Peromyscus maniculatus	Modified	1.43
Sorex monticolus	Modified	1.39
Microtus oregoni	State	1.72
Sorex bendirii	State	2.67
Sorex trowbridgii	State	1.01
Sorex vagrans	State	2.75
Zapus trinotatus	State	2.80

The relative abundance data in Table 4-10 were logarithmically (log) transformed to normalize the distribution of the data and were statistically analyzed by orthogonal and non-orthogonal contrasts (see Methods). The results of this analysis for the various contrasting combinations both within and between treatments and sampling periods are presented for 12 of the more common species of small mammals (Table 4-12). The relative abundance of several species of small mammals differed significantly between the control and logged sites (i.e., data from modified and state buffer uplands pooled) over the 10-year period of study (Table 4-12). Prior to logging, the relative abundance of *P. keeni* was 2.2 times greater ($p = 0.002$) on riparian transects at modified and state buffer treatments relative to the control sites. On upland transects, the relative abundance of *P. maniculatus* was significantly greater (2-fold; $p = 0.003$). The relative abundance of both species was equivalent at state and modified buffer sites (Figure 4-7). None of the sites had been logged in the pre-treatment interval, making identification of these pre-treatment differences important for subsequent comparisons to avoid mistakenly assigning treatment effects to significant differences in relative abundance.

During the immediate post-harvest sampling interval (1995 and 1996), the relative abundance of *Zapus trinotatus* was significantly higher ($p = 0.002$) on riparian transects at logged sites compared to riparian transects at the control sites. Similarly, the relative abundance of the *Peromyscus* group increased on riparian transects at the logged sites immediately following logging ($p = 0.003$; Figure 4-8). Both *Z. trinotatus* and *Peromyscus* were 3 times more abundant on riparian transects at the logged sites compared to the control. A third species, *M. oregoni* was also more than 3 times more abundant on riparian transects at logged sites relative to the control sites ($p = 0.046$). For this species, the real difference in relative abundance was between the riparian transects of the state buffer and control treatments, with both of these treatments not statistically different from the modified buffer sites (Figure 4-8). Pooling data from the modified and state buffer treatments and comparing the pooled value to the control sites diluted this difference, which is apparent in Figure 4-8. The relative abundance of one species (*S. bendirii*) decreased by a factor of 3 on riparian transects at the logged sites, a significant difference relative to the relative abundance at control sites ($p = 0.002$; Figure 4-8).

M. oregoni and *Peromyscus maniculatus* were both more abundant on upland transects of the logged treatments immediately after logging compared to the upland habitats of the control group. The relative abundance of *M. oregoni* was 6 times greater ($F_{1,55.3} = 20.9$; $p < 0.001$; Figure 4-9), an increase over the pre-harvest condition. The relative abundance of *P. maniculatus* was 2 times greater on logged uplands ($F_{1,28.9} = 5.7$; $p = 0.02$; Figure 4-9), which was consistent with pre-logging data. The *Peromyscus* group was also more abundant on logged uplands relative to control sites, with nearly 4 times as many individuals captured ($F_{1,67.9}$ = 20.4; $p < 0.001$; Figure 4-9). The relative abundance of *P. keeni* was more than 8 times lower on logged uplands compared to the control group ($F_{1,44.4}$ = 25.9; $p < 0.001$; Figure 4-9), which differed from pre-harvest conditions (Table 4-10). The relative abundance of both *N. gibbsi* and *S. trowbridgii* was lower on logged uplands (*N. gibbsi*: $F_{1,75.9}$ = 7.8; $p < 0.001$; *S. trowbridgii*: $F_{1,118.7}$ = 19.7; $p < 0.001$), representing a change from pre-harvest conditions when both species were equally abundant on upland and riparian transects (Figure 4-9; Table 4-10).

During the 10-year post-harvest sampling interval, the relative abundance of *S. trowbridgii* was higher on riparian transects at the control sites relative to logged treatments ($F_{1,124.1}$ = 13.7; $p < 0.001$), with a 2-fold decrease on logged sites (Figure 4-10). *P. keeni* was over 4 times more abundant on upland transects of the control sites compared to the logged treatments, a trend that continued from the immediate post-harvest sampling interval ($F_{1,51.9}$ = 6.9; $p = 0.01$; Figure 4-10). Finally, *M. oregoni* continued to be more abundant on upland transects of logged sites compared to unlogged treatments, with more than 8 times as many individuals captured at logged sites ($F_{1,64.7}$ = 8.3; $p = 0.005$; Figure 4-10).

One of the more important outcomes of the contrasts analysis is that, with the exception of *Z. trinotatus*, the differences in mean relative abundance between modified and state buffer sites for each of the other 11 species were non-significant (at $\alpha = 0.05$) in both riparian and upland habitats and all sampling periods (Table 4-12). The relative abundance of *Z. trinotatus* in riparian habitats was significantly greater (2.8-fold) at the state buffer than at the modified buffer sites during the 10-year post-harvest period ($F_{1,58.0}$ = 10.3; $p = 0.002$), a result driven largely by an increase in relative abundance observed on riparian transects at the state buffer treatment (Figure 4-11). For most species it appears that both logging and the width of the riparian buffer were not determinates of small-mammal relative abundance.

Also, catch rates of several species differed significantly between sampling periods for each of the three treatment levels (Table 4-12), although the differences were significant for fewer situations at the control sites (11) than in the state buffer (17) and modified buffer (19) sites. The species for which the captures differed significantly between sampling periods included *M. oregoni*, *P. keeni*, *P. maniculatus*, *Peromyscus* spp., and *S. trowbridgii*. The catch of six species in the upland state buffer sites differed significantly between pre- and post-harvest periods. In riparian modified buffer sites, the catch of six species differed significantly between the two post-harvest periods (Table 4-12).

Plots of mean relative abundance for both riparian and upland transects for 12 of the more common species within treatments by sampling interval (Figure 4-12) indicate how relative abundance changed over time relative to treatment, sampling interval, and transect. The main findings from these plots are as follows:

- Two years after logging, the abundance of *C. gapperi* decreased at the upland logged sites to levels below that of the controls. Ten years after logging, abundance of *C. gapperi* increased over pre-harvest levels and is consistent with the increase at controls for that period.
- After logging, the abundance of *M. oregoni* increased markedly at upland logged sites but declined to intermediate levels 10 years after logging.
- The abundance of *N. gibbsii* declined at upland logged sites following logging and returned to pre-harvest levels 10 years after logging.
- The abundance of *P. keeni* declined markedly at upland logged sites following logging and remained at a low level in the 10-year post-harvest sampling interval.

- *P. maniculatus* showed no response to logging: the species decline in abundance at the logged sites between the two post-harvest periods is consistent with the decline at the controls over that period.
- The increasing abundance of the *Peromyscus* group at the logged sites is consistent with the rise at the controls, although the increase was greater at the logged sites than at the controls.
- Among the four *Sorex* species, the response to logging appears to be moderate and variable, with all except *S. vagrans* showing a decline in abundance.
- Immediately after logging, the relative abundance of *Z. trinotatus* increased significantly on riparian transects at state buffer sites but returned to pre-harvest levels in the 10-year post-harvest sampling interval.
- The abundance of *Microtus longicaudus* remained relatively stable on both riparian and upland transects at all treatments (i.e., control, modified, and state buffer sites) and sampling intervals.

Table 4-12. *F*-value probabilities of non-orthogonal contrasts of small-mammal relative abundance for combinations of transect types (riparian, upland) among the three sampling intervals (SI 1–3) at each of the treatment types (control, modified, state) in western Washington, 1992–2004. Significant values[a] (α = 0.05) are indicated with bold italics and shading. The number of significant values per species and contrast are shown. Species codes are explained in Appendix 4-C.

Contrast	CLGA	MILO	MIOR	NEGI	PEKE	PEMA	PERO	SOBE	SOMO	SOTR	SOVA	ZATR	Count Sig.
Control Riparian: SI 2 vs. SI 3	0.797	0.047	0.073	0.466	0.969	0.027	0.006	0.039	0.862	0.263	0.812	0.104	4
Control Riparian: SI 1 vs. SI 2 + SI 3 cut[b]	0.791	0.589	0.173	0.539	0.893	0.409	0.001	0.627	0.063	0.351	0.012	0.940	2
Control Upland: SI 2 vs. SI 3	0.126	0.898	0.246	0.067	0.027	0.000	0.015	0.536	0.621	0.001	0.689	0.705	4
Control Upland: SI 1 vs. SI 2 + SI 3 cut[b]	0.153	0.916	0.101	0.538	0.563	0.586	0.000	0.781	0.216	0.513	0.860	0.955	1
Modified Riparian: SI 2 vs. SI 3	0.004	0.768	0.051	0.006	0.205	0.015	0.925	0.015	0.090	0.001	0.786	0.053	6
Modified Riparian: SI 1 vs. SI 2 + SI 3 cut[b]	0.155	0.205	0.169	0.178	0.005	0.874	0.000	0.436	0.016	0.001	0.106	0.159	4
Modified Upland: SI 2 vs. SI 3	0.000	0.829	0.000	0.334	0.458	0.000	0.710	0.011	0.393	0.341	0.059	0.107	4
Modified Upland: SI 1 vs. SI 2 + SI 3 cut[b]	0.962	0.132	0.000	0.396	0.000	0.067	0.000	0.089	0.260	0.001	0.024	0.912	5
State Riparian: SI 2 vs. SI 3	0.246	0.563	0.002	0.119	0.708	0.003	0.051	0.369	0.489	0.000	0.623	0.000	5
State Riparian: SI 1 vs SI 2 + SI 3 cut[b]	0.967	0.806	0.374	0.090	0.000	0.831	0.000	0.447	0.061	0.001	0.629	0.001	4
State Upland: SI 2 vs. SI 3	0.002	0.868	0.078	0.754	0.807	0.000	0.586	0.357	0.142	0.152	0.974	0.612	2
State Upland: SI 1 vs. SI 2 + SI 3 cut[b]	0.830	0.052	0.000	0.023	0.000	0.034	0.000	0.914	0.836	0.001	0.413	0.308	6
SI 1 Riparian: Control vs. cut	0.789	0.469	0.690	0.598	0.022	0.619	0.432	0.830	0.492	0.059	0.055	0.805	1
SI 1 Riparian: Modified vs. State	0.844	0.555	0.443	0.290	0.883	0.568	0.985	0.822	0.198	0.992	0.579	0.698	0
SI 1 Upland: Control vs. cut	0.127	0.971	0.390	0.615	0.388	0.034	0.107	0.202	0.625	0.770	0.887	0.783	1
SI 1 Upland: Modified vs. State	0.597	0.980	0.845	0.852	0.320	0.912	0.767	0.063	0.371	0.819	0.892	0.823	0
SI 2 Riparian: Control vs. cut	0.861	0.847	0.047	0.627	0.505	0.289	0.003	0.002	0.168	0.330	0.591	0.007	4
SI 2 Riparian Modified vs. State	0.949	0.964	0.396	0.057	0.501	0.620	0.397	0.286	0.645	0.881	0.250	0.002	1
SI 2 Upland: Control vs. cut	0.450	0.111	0.000	0.007	0.000	0.024	0.000	0.267	0.744	0.000	0.069	0.422	6
SI 2 Upland: Modified vs. State	0.853	0.668	0.245	0.335	0.838	0.760	0.237	0.823	0.631	0.492	0.106	0.635	0
SI 3 Riparian: Control vs. cut	0.143	0.111	0.202	0.352	0.956	0.457	0.309	0.315	0.669	0.000	0.697	0.489	1
SI 3 Riparian Modified vs. State	0.117	0.808	0.973	0.476	0.192	0.480	0.384	0.639	0.208	0.834	0.573	0.875	0
SI 3 Upland: Control vs. cut	0.203	0.157	0.005	0.883	0.011	0.686	0.059	0.163	0.803	0.098	0.556	0.561	2
SI 3 Upland: Modified vs. State	0.210	0.907	0.567	0.132	0.866	0.791	0.059	0.174	0.966	0.809	0.862	0.214	0
Count Sig.	3	1	8	3	8	9	11	4	1	9	2	4	

[a] Values close to 0.05 (e.g., 0.051) were considered significant and are highlighted in this table

[b] Data for SI 2 and SI 3 combined for each treatment

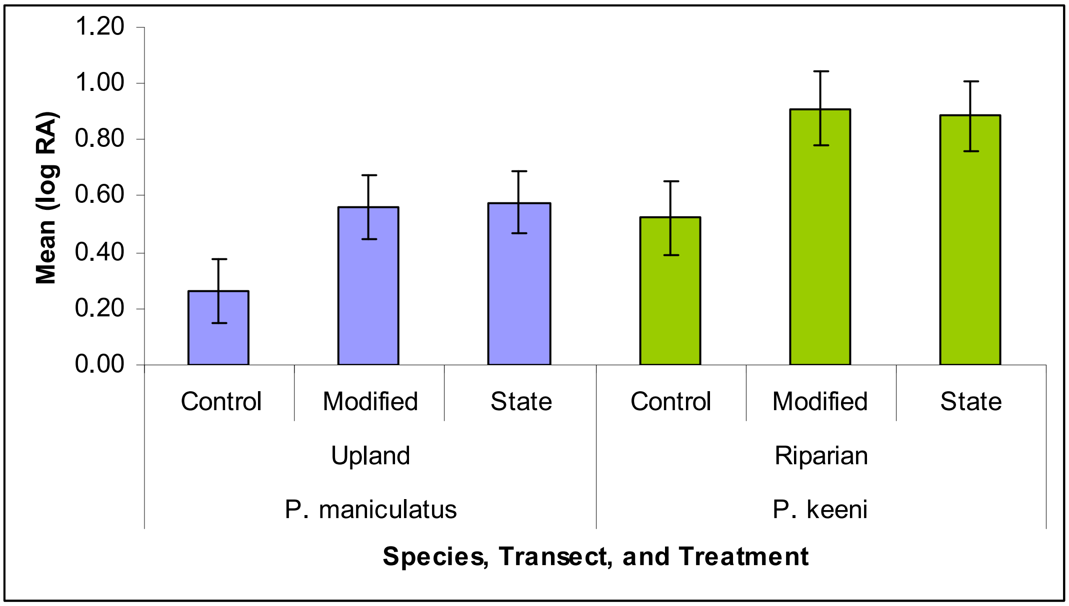

Figure 4-7. Mean relative abundance (log RA; ±1SE) of *Peromyscus maniculatus* on upland transects and *P. keeni* on riparian transects at all 18 western Washington study sites prior to logging. The RA of both species is significantly lower (at α = 0.05) on the controls for the transect shown. Comparisons between species are not being made.

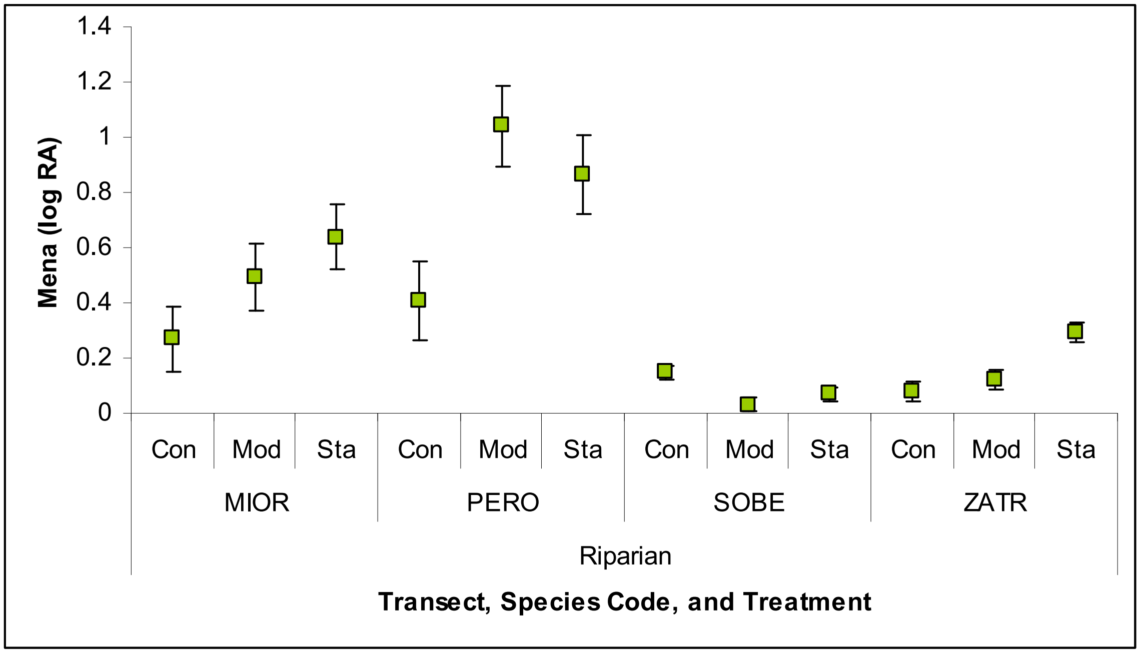

Figure 4-8. Mean relative abundance (log RA; ±SE) of *Microtus oregoni* (MIOR), *Peromyscus* spp. (PERO), *Sorex bendirii* (SOBE), and *Zapus trinotatus* (ZAPR) on riparian transects at the control (Con), modified (Mod) and state (Sta) treatments in the two-year post-harvest sampling interval (SI 2 = 1995/1996). Comparisons among species are not made.

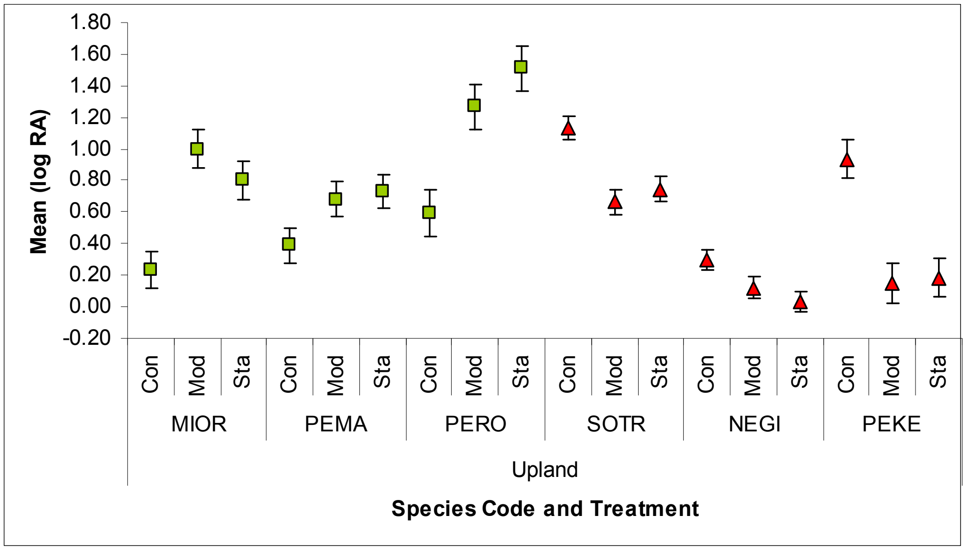

Figure 4-9. Mean relative abundance (log RA; ±SE) of *Microtus oregoni* (MIOR), *Peromyscus maniculatus* (PEMA), *Peromyscus* spp. (PERO), *Sorex trowbridgii* (SOTR), *Neurotrichus gibbsii* (NEGI), and *Peromyscus keeni* (PEKE) on upland transects of the control (Con), modified (Mod) and state (Sta) treatments in the two-year post-harvest sampling interval (SI 2 = 1995/1996). Comparisons among species are not made. Squares indicate increase in RA at logged sites relative to the control; triangles indicate decrease.

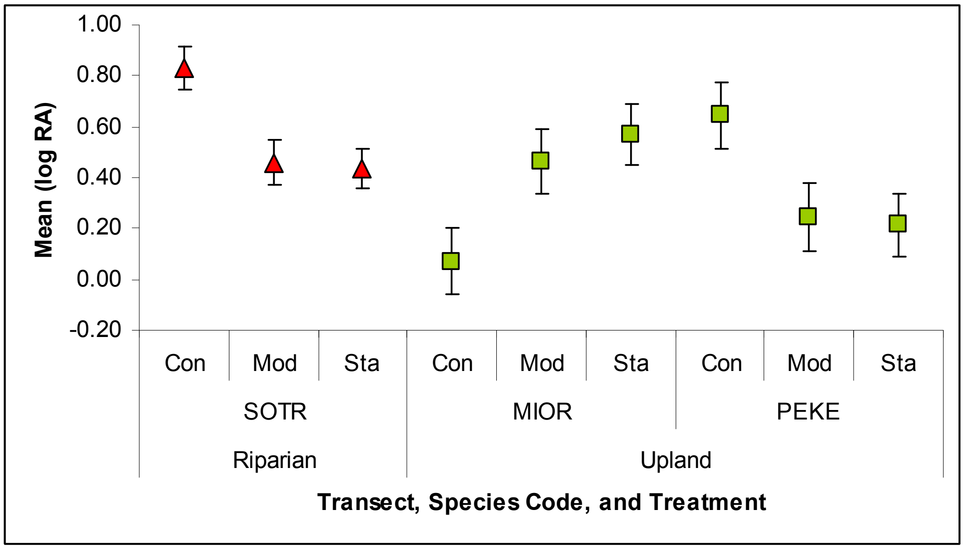

Figure 4-10. Mean relative abundance (log RA; ±SE) of *Sorex trowbridgii* (SOTR), *Peromyscus keeni* (PEKE), and *Microtus oregoni* (MIOR) on riparian (SOTR) and upland (MIOR, PEKE) transects at the control (Con), modified (Mod), and state (Sta) treatments ten-years post-logging (SI 3 = 2003/2004). Comparisons among species are not made. Squares indicate increase in RA at logged sites relative to the control; triangles indicate decrease.

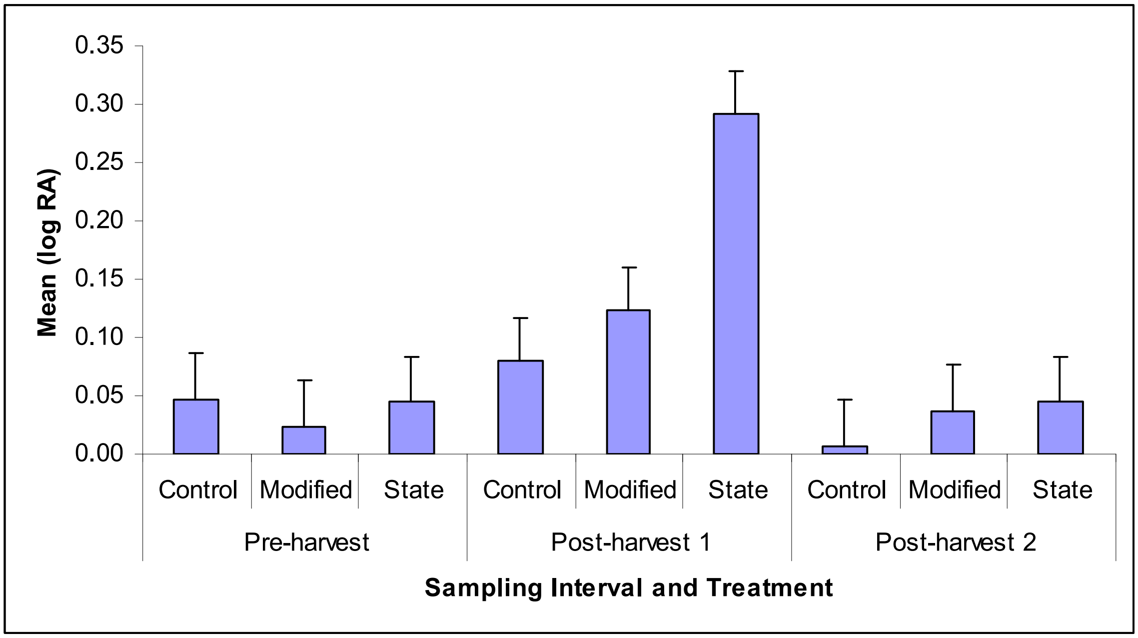

Figure 4-11. Mean relative abundance (log RA; +1SE) of *Zapus trinotatus* on riparian transects at the control, modified, and state treatments during each sampling interval. Pre-harvest (SI 1) = 1992/1993; Post-harvest 1 (SI 2) = 1995/1996; Post-harvest 2 (SI 3) = 2003/2004. The mean RA of *Z. trinotatus* was greater on riparian habitats at the state treatment compared to both the modified and control treatments during Post-harvest 1.

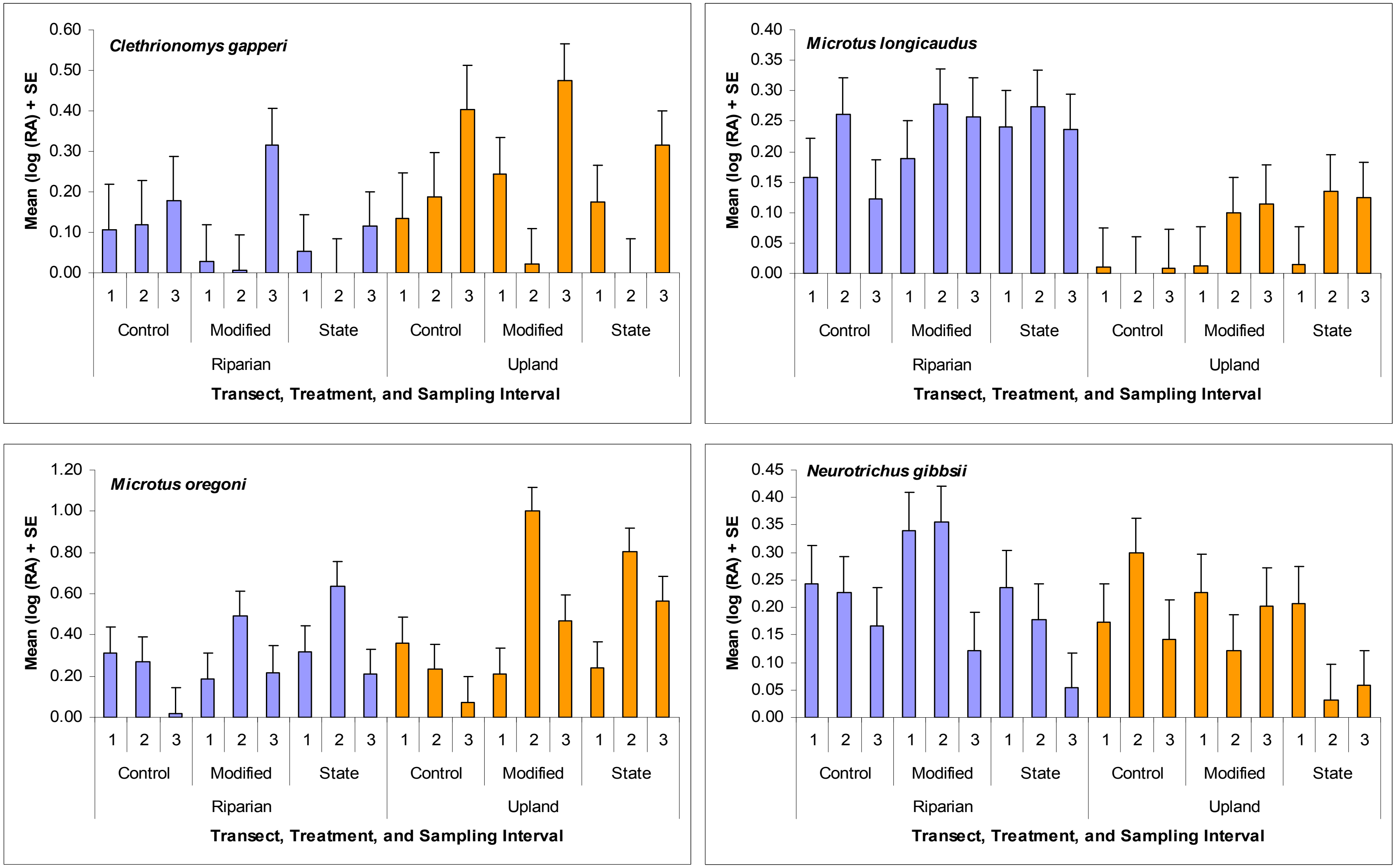

Figure 4-12. Mean relative abundance (log RA; +SE) for both riparian and upland transects for 12 of the more common species within treatments by sampling interval. 1 = Pre-harvest (1992/1993); 2 = Post-harvest 1 (1995/1996); 3 = Post-harvest 2 (2003/2004).

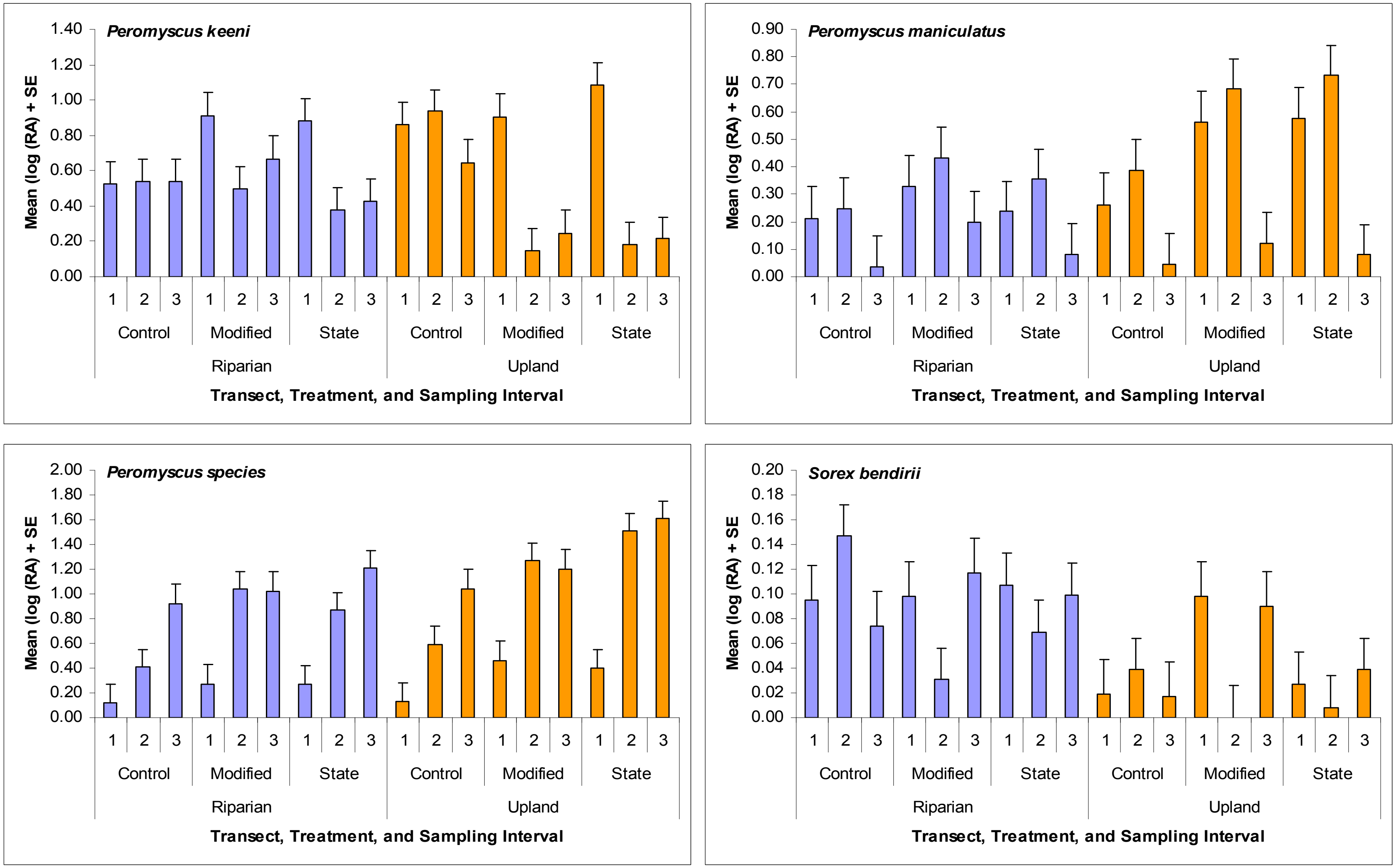

Figure 4-12 continued

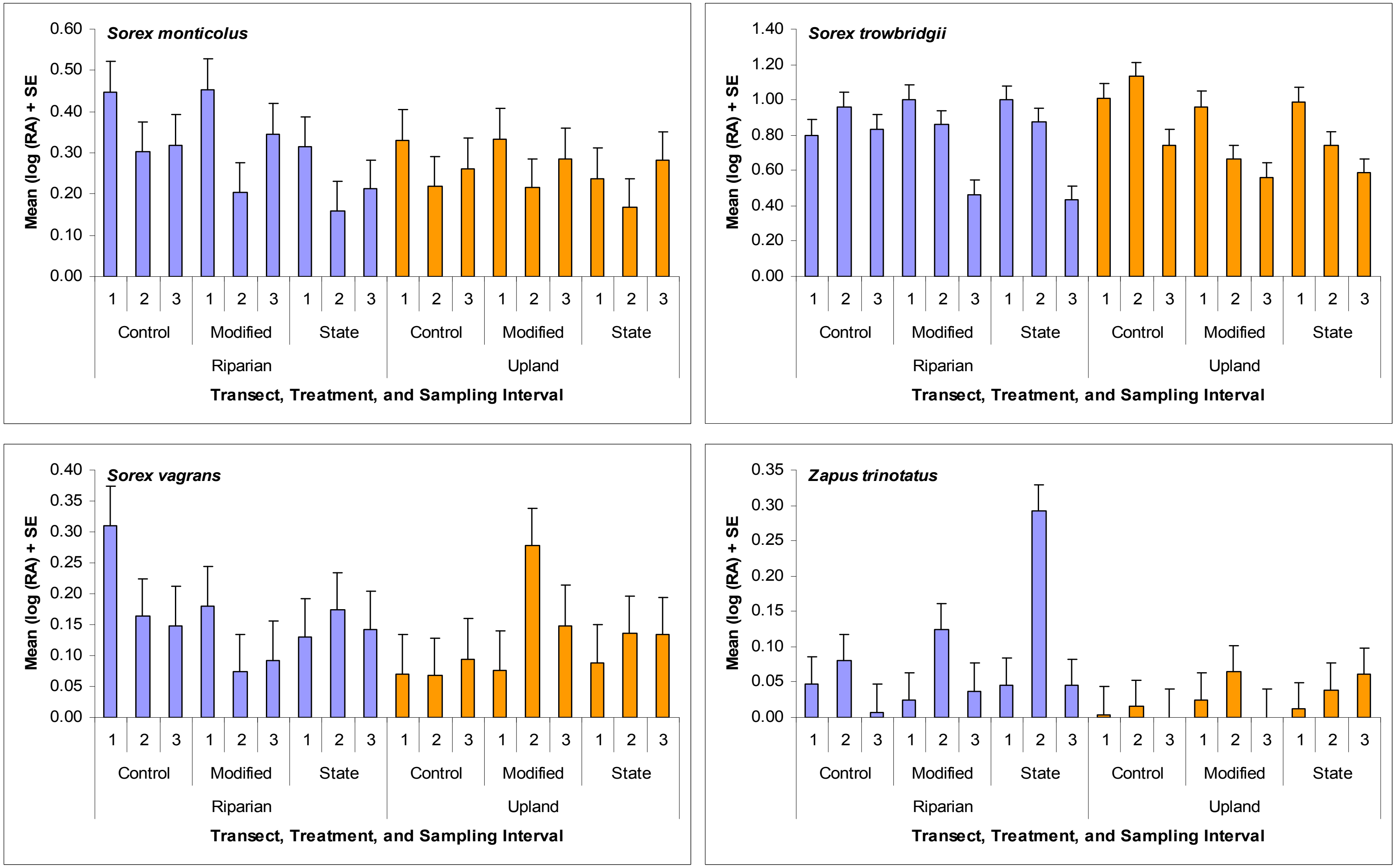

Figure 4-12 continued

Species Comparisons

The total number of species captured on riparian and upland transects combined was similar among treatments, with 15–18 species captured during pre-harvest, 17–19 species during the 2-year post-harvest period, and 18–20 species during the 10-year post-harvest period (Figure 4-13). The lowest number of species recorded (12) was on the upland transects at the control sites during pre-harvest, and the highest (19) was recorded on the upland transects at state buffer sites during the 10-year post-harvest period. Prior to logging, the number of species documented on riparian transects at the modified and state buffer treatments was equal. Following logging, there were more species documented on riparian transects at the modified buffer group than at the state buffer treatment. On upland transects, the number of species was initially higher (by 1) at the modified buffer treatment. After logging, there were 1 to 2 more species on upland transects at the state buffer treatment compared to the modified buffer and control groups in both post-harvest periods.

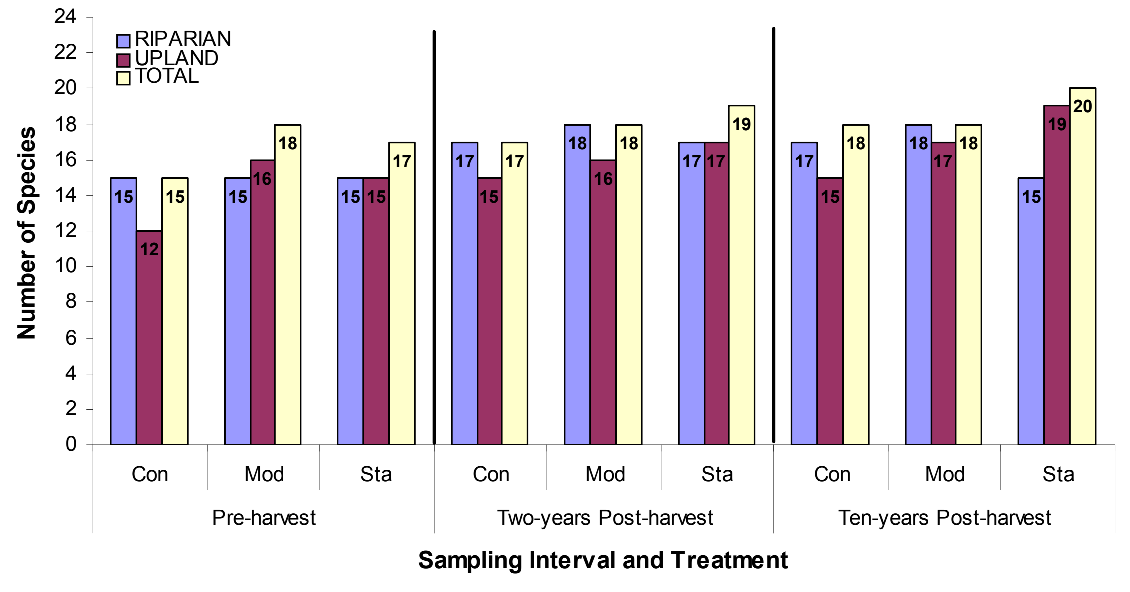

Figure 4-13. Total number of species detected per sampling interval, treatment, and transect at the western Washington study sites. Pre-harvest = 1992/1993; Two-years post-harvest = 1995/1996; Ten-years post-harvest = 2003/2004; Con = control; Mod = modified; Sta = state.

Plots of the mean number of species by transect type (i.e., riparian vs. upland) and treatment show that for each of the three sampling periods, variability around the mean for both riparian and upland transects was generally less for the state buffer than the modified buffer and control sites (Figure 4-14). The exception is the low variability in the upland modified buffer sites during the 2-year post-harvest period. Species richness increased slightly on riparian transects at both the control and modified buffer sites after logging but returned to pre-harvest levels 10 years after logging. There were no changes in species richness in riparian habitats at the state buffer sites among sampling intervals. For upland transects, species richness among treatments was similar for the pre- and 2-year post-harvest periods, but in the 10-year post-harvest period species richness declined at the modified buffer sites and increased at state buffer sites from the levels recorded in previous sampling periods. Differences in species richness between riparian and upland transects at the state buffer sites were significant in the pre-harvest (p = 0.03) and 2-year post-harvest (p = 0.012) intervals, whereas all other comparisons were non-significant.

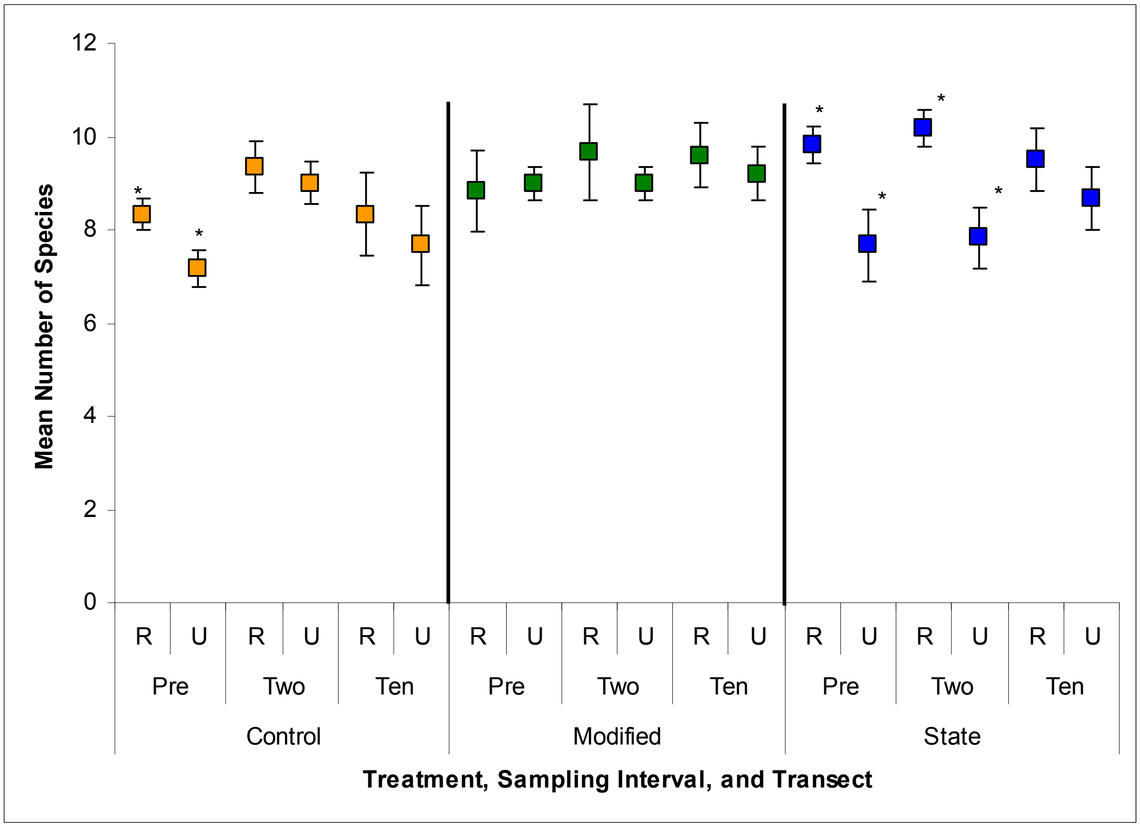

Figure 4-14. Mean number of species (±SE) per treatment, sampling interval, and transect. Asterisks denote significant differences (at α = 0.05) in mean number of species between paired habitat types within each sampling interval and treatment. Pre = pre-harvest (1992/1993); Two = two-years post-harvest (1995/1996); Ten = ten-years post-harvest (2003/2004); R = riparian; U = upland.

To assess how logging might influence the species richness of small mammals at the study sites, we pooled data from the state and modified buffer treatments and compared those data to the control sites during each sampling period for each habitat type (Figure 4-15). For the modified and state buffer sites combined (i.e., the logged sites), species richness was consistently greater on riparian than on upland transects among the sampling intervals, with the variability around the mean greatest during the 10-year post-harvest interval (Figure 4-15). Species richness at the modified and state buffer sites did not differ significantly from the control sites in any of the sampling intervals.

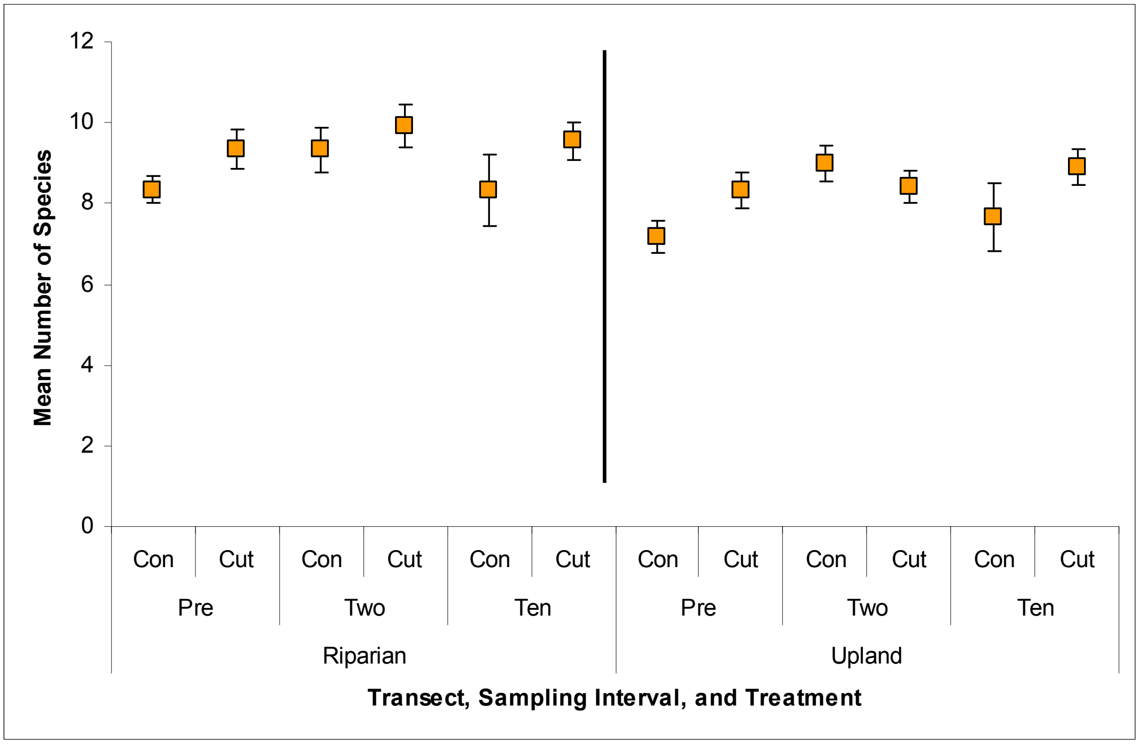

Figure 4-15. Mean number of species (±SE) for each transect and sampling period for the control group relative to the logged treatments (state and modified data combined). Pre = pre-harvest (1992/1993); Two = two-years post-harvest (1995/1996); Ten = ten-years post-harvest (2003/2004); Con = control; Cut = modified and state.

Community and Diversity Comparisons

Contrasting community similarity (Morisita's *C*) between pre-harvest and the 2- and 10-year post-harvest intervals revealed that, for the control group, community similarity remained relatively unchanged on the riparian transects across time (Figure 4-16). A notable decline in similarity was evident in upland habitats 10 years after logging relative to the pre-harvest condition at all three treatment types. Two years after logging, the small-mammal communities on riparian and upland transects at the control sites were similar. Likewise, the small-mammal communities present on riparian transects at the modified and state buffer treatments had a high degree of similarity to the communities recorded prior to logging, although some differences were indicated by a Morisita's coefficient of 0.72 for both treatments.

Immediately following logging, the small-mammal communities documented on upland transects at both the modified and state buffer treatments were dissimilar to the pre-harvest communities, with the largest divergence at the state buffer treatment. The upland community at the modified buffer treatment was different from the pre-harvest condition, with an overall reduction in similarity of 0.4 (*C* = 0.95 pre-harvest; 0.55 2-years post-harvest), pointing to a possible treatment effect. A similar trend was observed for upland habitats at the state buffer treatment 2 years after harvest, with similarity decreasing by 0.52 (*C* = 0.95 pre-harvest; 0.43 2-years post-harvest), again pointing to a possible treatment effect.

Ten years after logging, the community similarity for all treatments and transects was dissimilar to pre-harvest conditions, with declines in similarity for upland habitats at the modified buffer, state buffer, and control treatments (Figure 4-16). These changes point to a possible regional effect that was not observed during the 2-year post-treatment interval but that became apparent 10 years after logging, as evidenced by the overall reduction in community similarity on the upland transects at the controls.

Within the two post-harvest sampling periods, community similarity was high and stable across all treatments and habitat types (Figure 4-16). This does not indicate that the possible regional effect observed 10 years after logging is not real, because although the small mammal communities are similar in each post-harvest sampling interval, they are not equivalent, with the abundance of certain species increasing or decreasing over time (Figure 4-12).

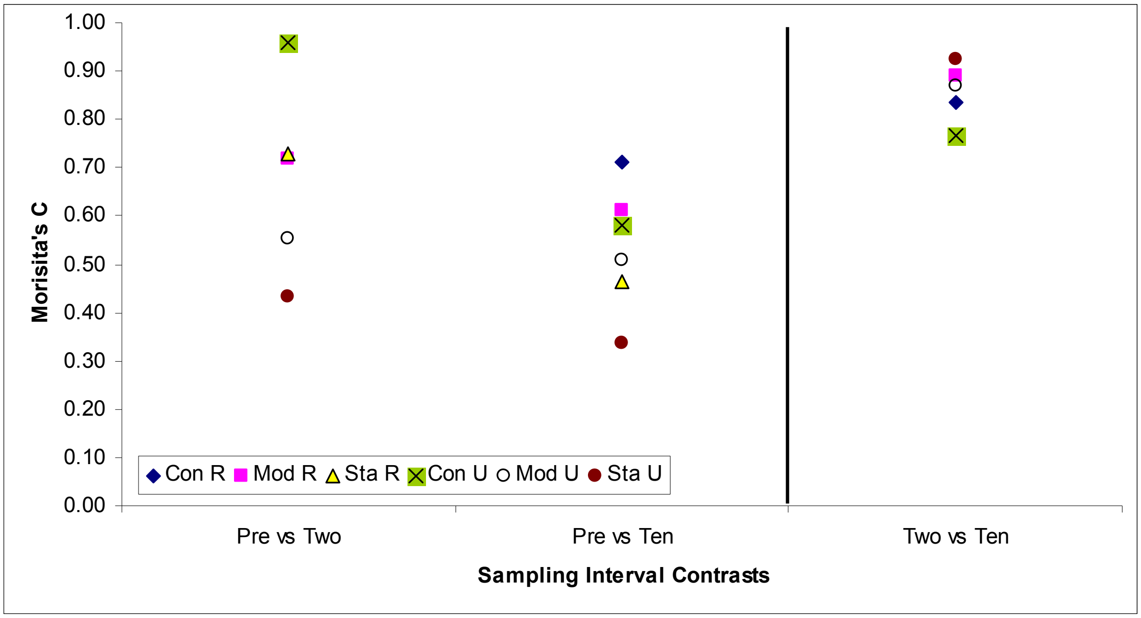

Figure 4-16. Morisita's *C* calculated for each transect contrasted against sampling interval. Pre = pre-harvest (1992/1993); Two = two-years post-harvest (1995/1996); Ten = ten-years post-harvest (2003/2004); Con = control; Mod = modified; Sta = state; R = riparian; U = upland.

There was considerable variation in species diversity for some sites, indicated by Shannon's diversity values (H') calculated for riparian and upland transects within each treatment group and sampling interval (Table 4-13). Also, there were considerable differences in species diversity between sites of the same treatment group, with Elbe Hills and Hotel Creek consistently showing higher diversity than most other sites for the duration of study.

Table 4-13. Shannon diversity values (*H'*) for each site within each treatment and sampling interval. Shaded cells emphasize variation in *H'* across all sampling intervals. Values in bold emphasize variation in *H'* within a treatment group and sampling interval for certain transects.

		Pre-harvest				Two-years Post-harvest				Ten-years Post-harvest			
		1992		1993		1995		1996		2003		2004	
Site	Treatment	R	U	R	U	R	U	R	U	R	U	R	U
Abernathy	Control			1.98	1.53	1.83	1.41	1.64	1.46	1.86	1.58	1.77	1.09
Elbe Hills	Control			1.91	1.66	**2.22**	1.95	**2.07**	1.91	2.15	1.81	1.87	1.68
Hotel Creek	Control	1.81	1.62	1.87	1.69	1.90	1.69	**1.58**	1.51	1.76	1.70	1.69	1.15
Porter Creek	Control	1.85	1.04	2.05	1.07	2.03	1.37	2.07	1.55	1.90	1.59	1.77	1.38
Taylor Creek	Control	1.80	1.63	1.67	1.65	1.97	1.86	1.60	1.61	1.64	1.83	1.00	1.73
Vail	Control	1.52	1.10	1.59	1.63	**1.16**	1.40	**1.49**	1.67				
West Fork	Control									1.24	1.32	1.77	1.66
Blue Tick	Modified			1.76	2.08	1.73	1.83	1.66	1.50	**1.08**	1.68	1.76	1.63
Eleven 31	Modified	1.47	1.14	1.77	1.54	1.80	1.29	1.94	1.67	1.95	1.80	1.69	1.50
Griffen Creek	Modified	1.65	1.53	1.69	1.79	1.74	1.75	1.66	1.66				
Ms Black	Modified	1.84	1.77	2.20	1.93	1.57	1.74	1.83	1.42	**2.30**	1.90	1.58	1.52
Ryderwood 860	Modified	1.71	1.61	1.64	1.86	1.84	1.23	1.93	1.62	2.04	1.26	1.56	1.02
Side Rod	Modified			1.54	1.64	1.55	1.58	2.33	2.33	1.77	1.86	1.86	1.94
Eleven 32	State	1.51	1.36	1.74	1.51	1.79	1.45	2.02	1.70	1.40	1.17	1.68	0.92
Kapowsin	State	0.91	1.65	1.83	1.78	1.28	1.33	1.54	1.56	1.24	1.80	1.49	1.70
Night Dancer	State			1.87	2.11	2.05	0.93	1.86	1.36	1.85	1.74	2.02	1.61
Potpourri	State	1.92	1.54	1.97	1.53	1.83	1.39	1.78	1.32	1.98	2.05	1.13	1.36
Ryderwood 1557	State	1.57	1.14	1.70	1.46	1.76	1.51	2.09	1.13	1.82	1.42	1.44	1.29
Simmons Creek	State	1.56	1.35	1.89	1.33	1.87	1.50	1.78	1.28	1.38	0.97	1.29	0.95

Plots of mean Shannon diversity values by habitat type within treatments and sampling intervals (Figure 4-17) showed that, with the exception of the upland control sites, species diversity was very similar between riparian and upland habitats during the pre-harvest period (Table 4-10). However, during both post-harvest periods (Table 4-10), mean Shannon diversity indices were consistently greater in riparian than upland transects, with the difference being most pronounced at the state buffer sites during the 2-year post-harvest period.

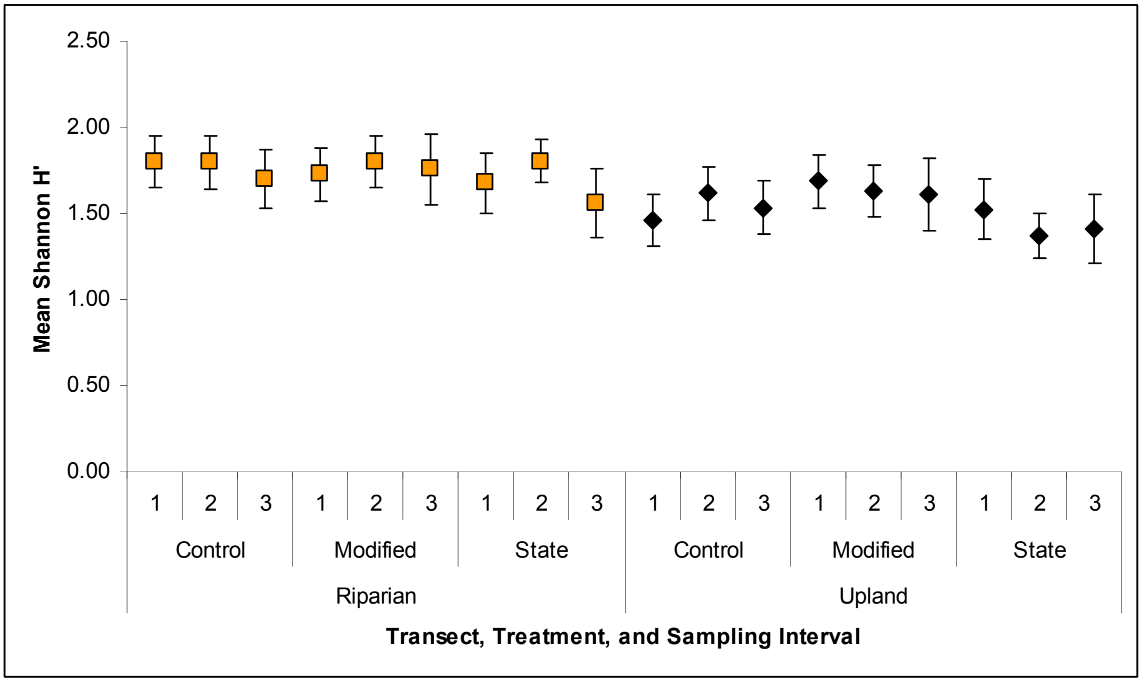

Figure 4-17. Mean Shannon diversity values (± 95% CI) by habitat type within treatments and sampling intervals. 1 = Pre-harvest (SI 1; 1992/1993); 2 = Two-years post-harvest (SI 2; 1995/1996); 3 = Ten-years post-harvest (SI 3; 2003/2004).

Species–Habitat Associations

Pre-harvest

During the pre-harvest period, when habitat conditions were similar between treatments, several species exhibited strong habitat associations. The pre-harvest riparian biplot reveals an environmental gradient tending toward mixed coniferous–deciduous forests with high structural complexity and increasing forest openness along axis 1, while the gradient associated with axis 2 describs increasing forest openness and increasing herbaceous growth (Figure 4-18). Most of the variation in the pre-harvest riparian data is explained by axis 1 (64%) and axis 2 (17%). The riparian biplot reveals that the abundance of *Peromyscus* spp. (all are considered generalists) was not strongly correlated with either axis 1 or axis 2. Their position on the biplot indicates that their abundance was more likely to be higher on habitats characterized by both increasing habitat complexity and increasing canopy cover. Furthermore, the biplot indicates a correlation between depth of deciduous leaf litter and the presence of large conifers, whereas the abundance of *Microtus oregoni*, another generalist species, was correlated with areas comprising both small and large deciduous trees and grass cover. Two species of *Sorex* (*S. vagrans* and *S. monticolus*) were correlated with axis 1, indicating that structural complexity was important. *M. longicaudus* was positively correlated with axis 1, suggesting that the abundance of this species will be higher in habitats consisting of high herbaceous growth and decreasing tree-canopy cover.

In contrast to the above, the biplot for upland habitat pre-harvest describs an environmental gradient of increasing structural complexity and decreasing canopy cover along axis 1 and a forest becoming increasingly dominated by Douglas-fir (Figure 4-18). Both axes 1 and 2 explain a majority of the variation in the data (axis 1: 39%; axis 2: 32%). *Peromyscus* spp. were negatively associated with axis 2, indicating a relationship between increasing abundance and deciduous stands; whereas

Sorex spp. (*S. vagrans* and *S. monticolus*) were positively correlated with axis 1, suggesting an association with increasing habitat complexity (especially shrub and grass cover) and areas of high fern cover. *M. oregoni* showed a relatively strong (positive) correlation with axis 2, indicating that higher abundances of this species will occur in forests dominated by large Douglas-fir.

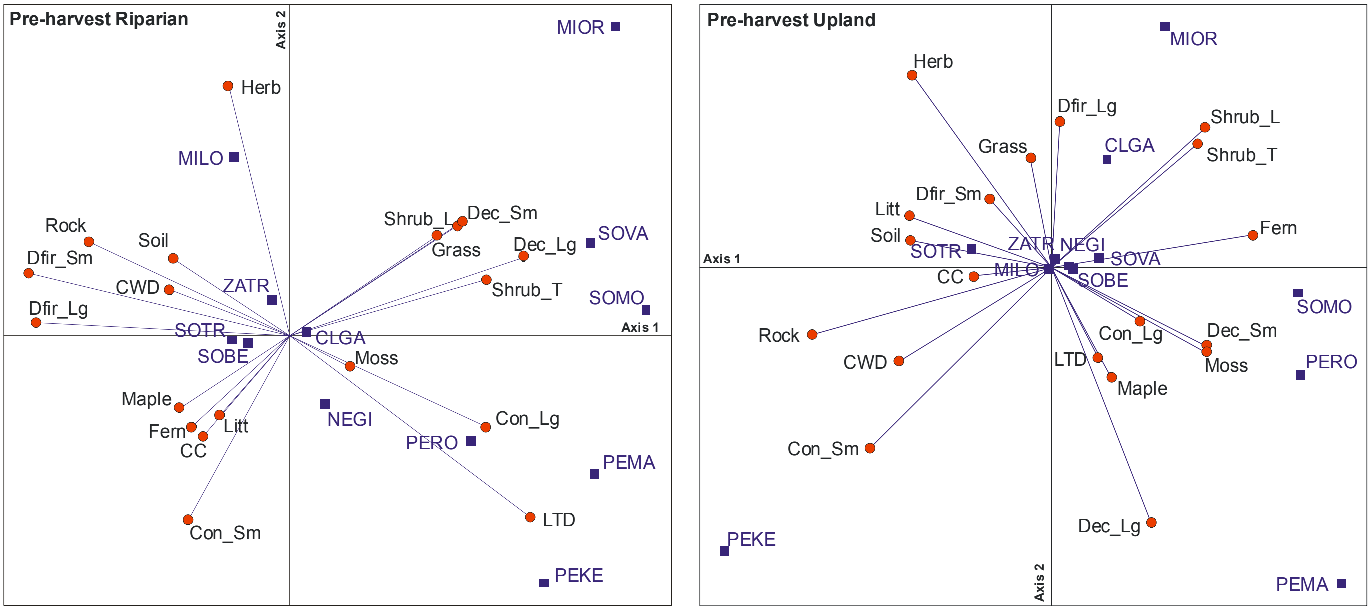

Figure 4-18. Biplot of small-mammal–habitat relationships on riparian and upland transects pre-harvest, data from all sites combined. Shrub_T = tall shrubs; Shrub_L = low shrubs; CC = overstory canopy cover; CWD = coarse woody debris; Fern = fern cover; Grass = grass cover; Herb = herbaceous cover; Litt = deciduous leaf-litter cover; LTD = deciduous leaf -litter depth; Moss = moss cover; Rock = rock cover; Soil = exposed soil; Maple = vine maple; Dec_Sm = bigleaf maple and red alder <1.3 m; Dec_Lg = bigleaf maple and red alder >1.3 m; Con_Sm = western redcedar and western hemlock <1.3 m; Con_Lg = western redcedar and western hemlock >1.3 m; Dfir_Sm = Douglas-fir <1.3 m; Dfir_Lg = Douglas-fir >1.3 m. CLGA = *Clethrionomys gapperi*; MILO = *Microtus longicaudus*; MIOR = *Microtus oregoni*; NEGI = *Neurotrichus gibbsii*; PEKE = *Peromyscus keeni*; PEMA = *Peromyscus maniculatus*; PERO = *Peromyscus* species; SOBE = *Sorex bendirii*; SOMO = *Sorex monticolus*; SOTR = *Sorex trowbridgii*; SOVA = *Sorex vagrans*; ZATR = *Zapus trinotatus*.

During the 2-year post-harvest period, when habitat conditions differed markedly between control and logged sites, differences existed in species–habitat associations between riparian habitats at the control sites and at the modified and state buffer sites. The biplot for riparian habitats at the control sites (Figure 4-19) describes a coniferous forest with decreasing structural complexity and the amount of deciduous cover decreasing along axis 1. Axis 2 suggests a gradient of decreasing forest openness and reduced herbaceous growth. Axes 1 and 2 describe 81.7% of the variance (axis 1: 48.8%; axis 2: 32.9%). Under these conditions, *S. monticolus* and *P. keeni* were negatively correlated with axis 1, and their abundance appears to have been correlated with increasing habitat diversity and a deciduous forest. *Microtus* spp. were negatively associated with axis 2, indicating that their abundance was related to habitat attributes associated with the forest floor, such as herbaceous growth, coarse woody debris, and grass cover.

At the modified buffer treatment, the environmental gradients described by axes 1 and 2 account for 68.7% of the variation in the data. Axis 1 describes a gradient of increasing habitat and structural complexity associated with an increase in the number of large conifers and, to a lesser extent, of large deciduous trees. Axis 2 reveals a gradient of decreasing forest openness, with increasing cover of small deciduous trees, grass, ferns, and exposed soil (Figure 4-19). Based on these gradients, it appears that the abundance of *P. keeni* was strongly correlated with axis 1, *P. monticolus* was an intermediary species (not strongly correlated with either axis 1 or 2), and *Peromyscus* spp. were strongly correlated with axis 2. The abundance of *M. oregoni* was positively correlated with axis 2 and was associated with small deciduous trees and soil and grass cover. Although not strongly correlated with either axis, *Neurotrichus gibbsii* showed a prominent association with areas of large deciduous trees in stands with increasing canopy cover (Figure 4-19).

Axis 1 of the state buffer treatment (Figure 4-19) describes a habitat condition characterized by a decreasing representation of coniferous trees, while canopy cover and grass cover increased. This situation reflects the gradient described by axis 2, which indicates a gradient from small coniferous trees to deciduous trees, increasing in size. Related to this was an increase in the amount of exposed soil and a decrease in shrub cover. Under these conditions, the abundance of *Zapus trinotatus* (a generalist species) was positively correlated with axis 1, suggesting that the abundance of this species was related to stands characterized by large deciduous trees, increasing areas of exposed soils, and decreasing shrub cover. In contrast, *M. oregoni* was positively correlated with axis 2, indicating a strong correlation between the abundance of this species and decreasing forest openness, which was consistent with the pre-harvest condition (Figure 4-18). The abundance of the remaining species were correlated with both axis 1 and axis 2 in an intermediate manner, suggesting that the abundance of these species was not strongly tied to the presence of a particular suite of habitat attributes. That being said, there are some notable relationships evident in Figure 4-19. First, the abundance of *P. maniculatus* and *S. monticolus* was associated with large Douglas-fir and the presence of ferns, which was consistent with the pre-harvest condition (Figure 4-18); although, in the pre-harvest condition, *P. maniculatus* was associated with large conifers and not specifically Douglas-fir. *S. vagrans*, previously associated with increasing habitat complexity (especially shrub and grass cover) and areas of high fern cover, appears to have been associated with the presence of vine maple and large deciduous trees, suggesting that this species had altered its habitat preference or that this species is a habitat generalist and uses whatever habitats are available.

To assess how species–habitat associations varied in upland habitats 2-years post-logging, we pooled data from state and modified upland transects and compared them to the upland habitats at the control sites (Figure 4-20). This was done because the uplands of both the modified and state buffer treatments were logged according to the same silvicultural prescriptions and because none of the species tested showed significant differences in relative abundance relative to riparian or upland habitats at any of the treatments (Table 4-12). For the control group, axis 1 (51% of variance) describes an environmental gradient of decreasing representation of coniferous (cedar and hemlock) trees, along with decreasing canopy cover and coarse woody debris cover, suggesting a forest tending toward an older seral stage. Axis 2 (28% of variance) describes a forest dominated by Douglas-fir, with fern and herb cover increasing, again suggesting a maturing forest. Within these conditions, *Peromyscus* spp. were negatively correlated with axis 2, suggesting that species diversity and structural complexity was related to increased abundance of these species. The abundance of *N. gibbsii* was strongly (positively) correlated with Douglas-fir–dominated forests.

For the logged treatment (i.e., modified and state buffer uplands). the environmental gradient described by axis 1 (59% of variation) indicates habitats decreasing in both structural and habitat complexity and increasing in rock cover. Axis 2 (23% of variation) reveals a gradient of decreasing large trees and increasing coarse woody debris cover, as would be expected post-logging. Under these conditions, *M. oregoni* showed a negative correlation with axis 1, suggesting that abundance of this species in upland habitats after logging was more closely associated with areas of increasing structural complexity. The abundance of *N. gibbsii* was higher in stands with increasing canopy cover and large coniferous and deciduous trees, as indicated by the negative association with axis 2, and mirrors the condition reported for riparian habitats at the modified buffer treatment (Figure 4-19) and on upland habitats at the control sites (Figure 4-20). This was similar to the strong correlation with axis 2. Following harvest, some species tended to correlate with a greater suite of variables at the upland logged sites than at the control sites. For example, *M. oregoni* at upland logged sites was associated with small deciduous and coniferous growth, in addition to several cover variables consisting of deciduous litter, ferns, herbaceous vegetation, mosses, and grasses. In contrast, in upland habitats at control sites, *M. oregoni* was mainly associated with areas of large deciduous trees and vine-maple cover, indicating that *M. oregoni* is a habitat generalist that is able to adapt to large-scale habitat alteration.

Ten-years Post-harvest

Ten years after logging, the riparian habitats at the control sites continued to exhibit characteristics typical of a dynamic riparian system. For example, axis 1 (55.2% of variation) reveals an environmental gradient of increasingly large trees, multilayered vegetation (=structural complexity), while axis 2 (22.3% of variation) indicates a gradient of increasingly large trees and decreasing canopy cover, typical of a dynamic, maturing system (Figure 4-21). *Sorex monticolus* and *P. keeni* and *Peromyscus* spp. showed a strong negative correlation with axis 2, indicating a relationship between the abundance of these species (or groups) and forests with high canopy cover and smaller trees typical of forests in early seral stages.

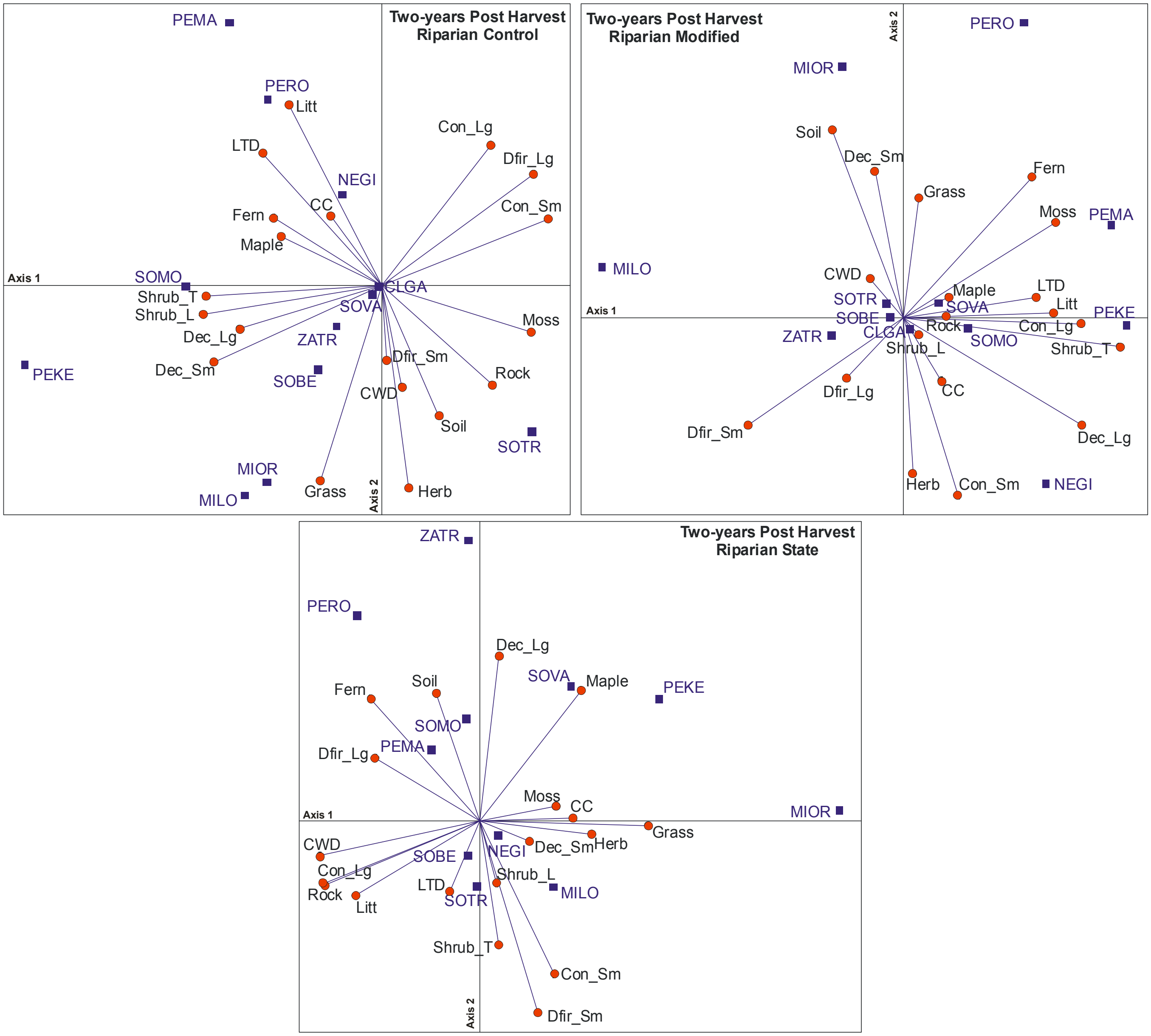

Figure 4-19. Biplot of small-mammal–habitat relationships on riparian transects of the control, modified, and state treatments two-years post-harvest, data from all sites combined. Shrub_T = tall shrubs; Shrub_L = low shrubs; CC = overstory canopy cover; CWD = coarse woody debris; Fern = fern cover; Grass = grass cover; Herb = herbaceous cover; Litt = deciduous leaf-litter cover; LTD = deciduous leaf-litter depth; Moss = moss cover; Rock = rock cover; Soil = exposed soil; Maple = vine maple; Dec_Sm = bigleaf maple and red alder <1.3 m; Dec_Lg = bigleaf maple and red alder >1.3 m; Con_Sm = western redcedar and western hemlock <1.3 m; Con_Lg = western redcedar and western hemlock >1.3 m; Dfir_Sm = Douglas-fir <1.3 m; Dfir_Lg = Douglas-fir >1.3 m. CLGA = *Clethrionomys gapperi*; MILO = *Microtus longicaudus*; MIOR = *Microtus oregoni*; NEGI = *Neurotrichus gibbsii*; PEKE = *Peromyscus keeni*; PEMA = *Peromyscus maniculatus*; PERO = *Peromyscus* species; SOBE = *Sorex bendirii*; SOMO = *Sorex monticolus*; SOTR = *Sorex trowbridgii*; SOVA = *Sorex vagrans*; ZATR = *Zapus trinotatus*.

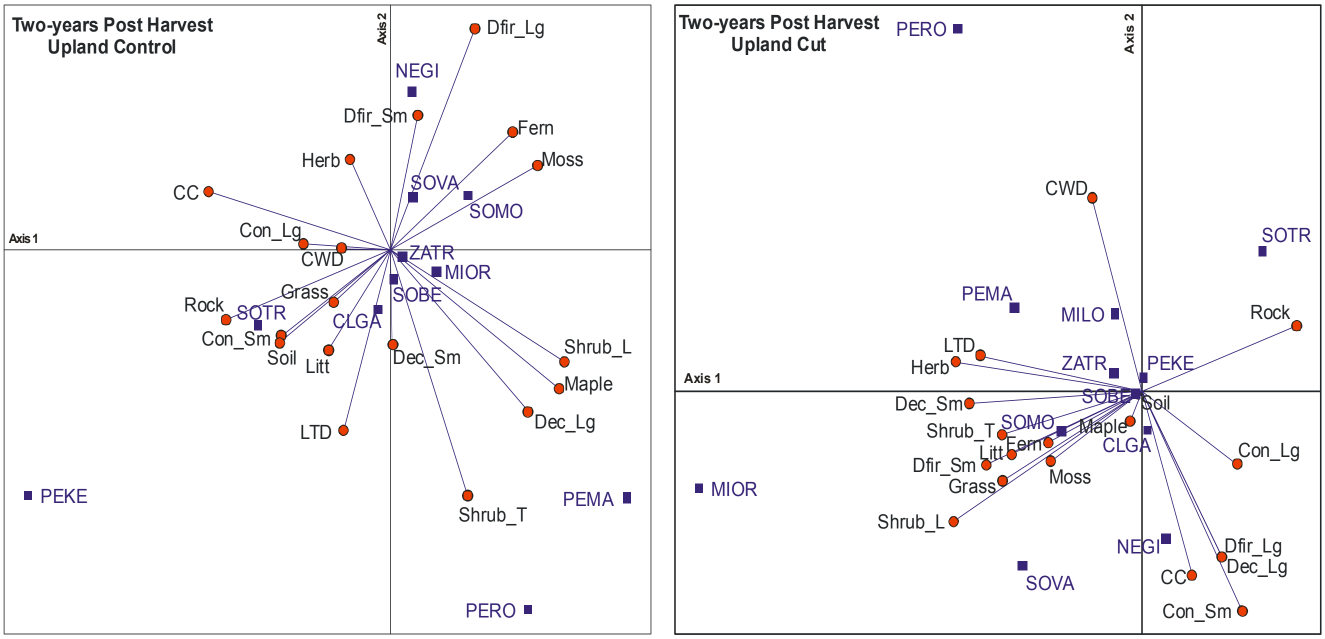

Figure 4-20. Biplot of small-mammal–habitat relationships on upland transects for the control treatment (a) and modified and state treatment combined (b) two-years post-harvest. Shrub_T = tall shrubs; Shrub_L = low shrubs; CC = overstory canopy cover; CWD = coarse woody debris; Fern = fern cover; Grass = grass cover; Herb = herbaceous cover; Litt = deciduous leaf-litter cover; LTD = deciduous leaf - litter depth; Moss = moss cover; Rock = rock cover; Soil = exposed soil; Maple = vine maple; Dec_Sm = bigleaf maple and red alder <1.3 m; Dec_Lg = bigleaf maple and red alder >1.3 m; Con_Sm = western redcedar and western hemlock <1.3 m; Con_Lg = western redcedar and western hemlock >1.3 m; Dfir_Sm = Douglas-fir <1.3 m; Dfir_Lg = Douglas-fir >1.3 m. CLGA = *Clethrionomys gapperi*; MILO = *Microtus longicaudus*; MIOR = *Microtus oregoni*; NEGI = *Neurotrichus gibbsii*; PEKE = *Peromyscus keeni*; PEMA = *Peromyscus maniculatus*; PERO = *Peromyscus* species; SOBE = *Sorex bendirii*; SOMO = *Sorex monticolus*; SOTR = *Sorex trowbridgii*; SOVA = *Sorex vagrans*; ZATR = *Zapus trinotatus*.

The biplot of species–habitat associations at the state and modified riparian buffers indicates dynamic habitats, with structural and habitat complexity varying along axis 1 and habitat attributes (e.g., canopy cover) changing along axis 2 (Figure 4-21). In these habitats, canopy cover was decreasing (at the state buffer treatment) or increasing (at the modified buffer treatment). Based on the riparian conditions available, certain species appeared to be correlated with different habitat attributes. For example, the Southern Red-backed Vole (*Clethrionomys gapperi*) had a strong positive correlation with axis 1 at the state buffer treatment, indicating a relationship between abundance and a forest containing large Douglas-fir, small deciduous trees in the understory, and a forest floor covered by ferns. Similarly, *C. gapperi* had a strong but negative correlation with axis 1 at modified buffer sites, and the abundance of this species was again associated with a forest containing large Douglas-fir, small deciduous trees in the understory, and a forest floor covered by ferns. Interestingly, *C. gapperi* was not detected at the state buffer treatment 2-years post-logging and occurred in low numbers at the modified buffer treatment during the 2-year post-harvest interval (Figure 4-19). During the 10-year post harvest interval, the relative abundance of *C. gapperi* increased at both the state and modified buffer treatments, suggesting that the suitability of riparian habitat may have been influenced by changes in the adjacent upland forest.

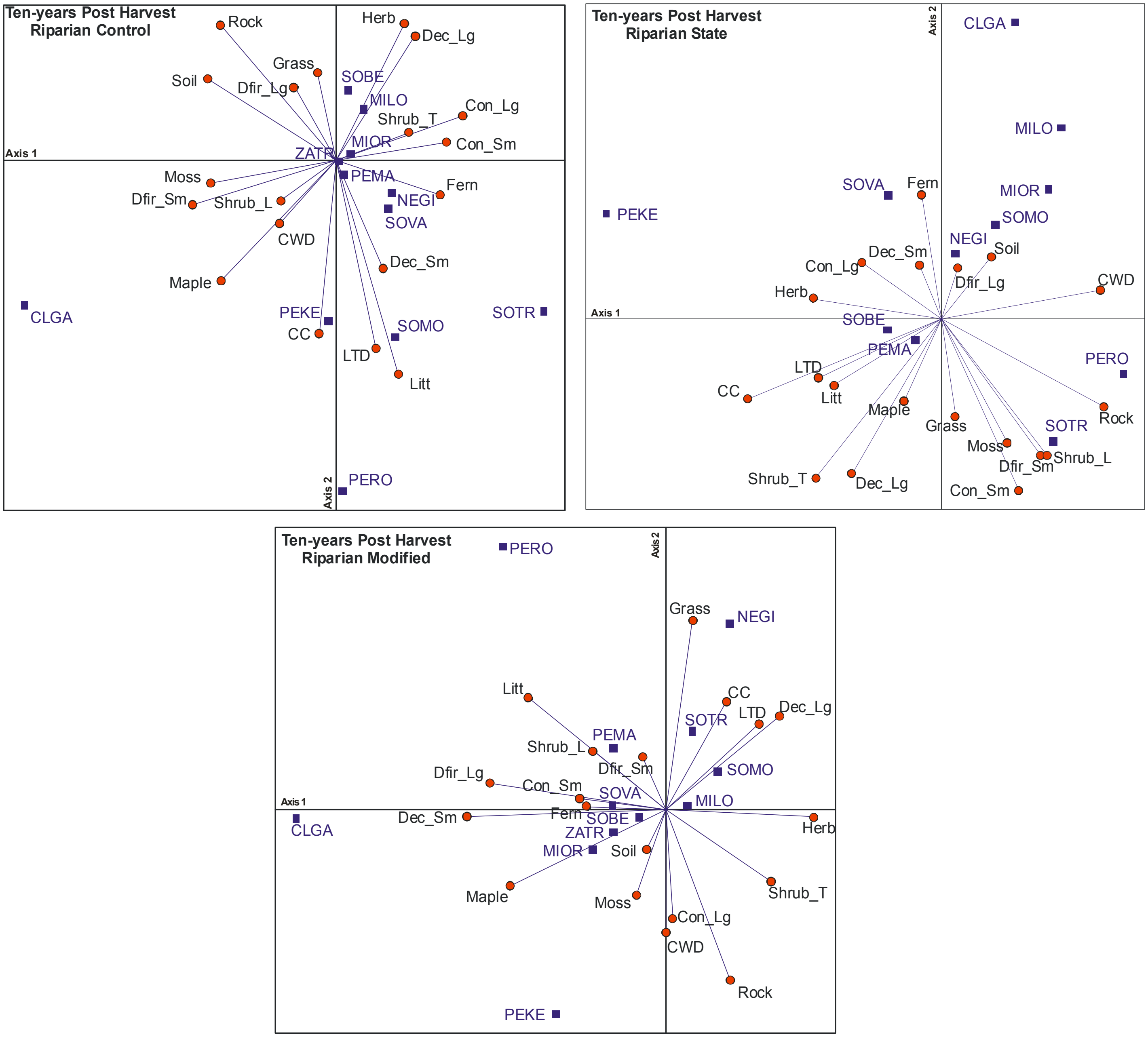

Figure 4-21. Biplot of small-mammal–habitat relationships on riparian transects at each treatment ten-years post-harvest, data from all sites combined. Shrub_T = tall shrubs; Shrub_L = low shrubs; CC = overstory canopy cover; CWD = coarse woody debris; Fern = fern cover; Grass = grass cover; Herb = herbaceous cover; Litt = deciduous leaf-litter cover; LTD = deciduous leaf-litter depth; Moss = moss cover; Rock = rock cover; Soil = exposed soil; Maple = vine maple; Dec_Sm = bigleaf maple and red alder <1.3 m; Dec_Lg = bigleaf maple and red alder >1.3 m; Con_Sm = western redcedar and western hemlock <1.3 m; Con_Lg = western redcedar and western hemlock >1.3 m; Dfir_Sm = Douglas-fir <1.3 m; Dfir_Lg = Douglas-fir >1.3 m. CLGA = *Clethrionomys gapperi*; MILO = *Microtus longicaudus*; MIOR = *Microtus oregoni*; NEGI = *Neurotrichus gibbsii*; PEKE = *Peromyscus keeni*; PEMA = *Peromyscus maniculatus*; PERO = *Peromyscus* species; SOBE = *Sorex bendirii*; SOMO = *Sorex monticolus*; SOTR = *Sorex trowbridgii*; SOVA = *Sorex vagrans*; ZATR = *Zapus trinotatus*.

Ten years after logging, some differences as well as similarities in species–habitat associations existed between the upland logged (modified and state combined) and control sites (Figure 4-22a, b). At logged sites, the abundance of *C. gapperi* was strongly correlated with axis 1, which describes an environmental gradient of decreasing tree height, increasing cover of tall shrubs, and small Douglas-fir, a condition characterized by an 8- to 10-year-old regenerating coniferous forest. At the control sites, *C. gapperi* was not strongly correlated with either axis, but its abundance does appear to be related to the presence of stands with increasing structural understory complexity (abundant shrub and small tree cover, both coniferous and deciduous). The abundance of *Peromyscus* in uplands of logged sites was strongly correlated with grassy sites with some small conifers. A similar trend was observed for this species group in upland habitats at the control sites: *Peromyscus* were negatively correlated with axis 1, which describes a gradient of increasing forest age and type (primarily coniferous).

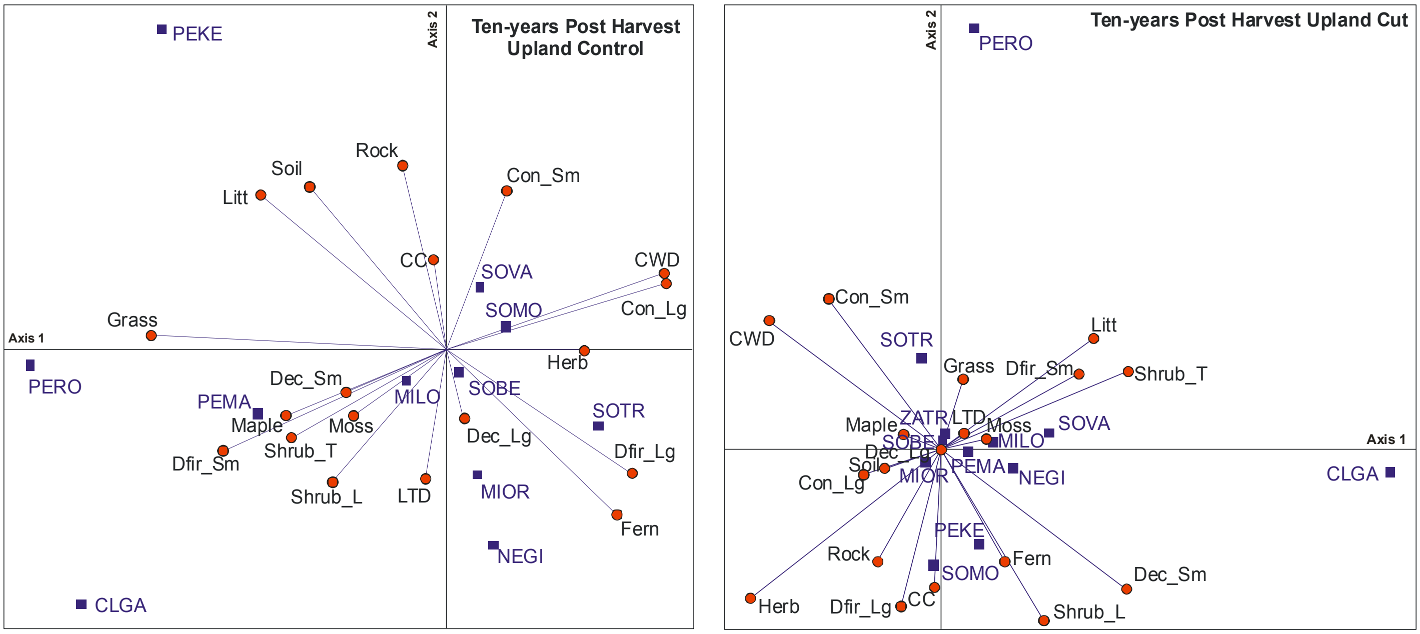

Figure 4-22. Biplot of small-mammal–habitat relationships on upland transects for the control treatment (a) and modified and state treatment combined (b) ten-years post-harvest. Shrub_T = tall shrubs; Shrub_L = low shrubs; CC = overstory canopy cover; CWD = coarse woody debris; Fern = fern cover; Grass = grass cover; Herb = herbaceous cover; Litt = deciduous leaf-litter cover; LTD = deciduous leaf-litter depth; Moss = moss cover; Rock = rock cover; Soil = exposed soil; Maple = vine maple; Dec_Sm = bigleaf maple and red alder <1.3 m; Dec_Lg = bigleaf maple and red alder >1.3 m; Con_Sm = western redcedar and western hemlock <1.3 m; Con_Lg = western redcedar and western hemlock >1.3 m; Dfir_Sm = Douglas-fir <1.3 m; Dfir_Lg = Douglas-fir >1.3 m. CLGA = *Clethrionomys gapperi*; MILO = *Microtus longicaudus*; MIOR = *Microtus oregoni*; NEGI = *Neurotrichus gibbsii*; PEKE = *Peromyscus keeni*; PEMA = *Peromyscus maniculatus*; PERO = *Peromyscus* species; SOBE = *Sorex bendirii*; SOMO = *Sorex monticolus*; SOTR = *Sorex trowbridgii*; SOVA = *Sorex vagrans*; ZATR = *Zapus trinotatus*.

In theory, there were to be two distinct riparian treatments to compare to a control or reference condition; however, this was not the case. By proxy, the assignment of treatment type to a site should define buffer width (i.e., state = narrow, uniform; modified = wider, more variable). Plotting the normalized score of the principle components analysis (PCA) on the habitat-attribute table relative to the treatments shows that the state and modified buffer sites were not measurably different after logging (Figure 4-23). Prior to logging (blue squares), the treatments occupied somewhat different positions in space. Although all sites within each treatment were selected based on specific criteria (see Methods), the riparian zones within each stand were likely different due to the dynamic nature of riparian forests. Two years after logging, the riparian habitats at both the state and modified buffer treatments diverged from their original pre-treatment positions, but they did not separate relative to each other in space, and both treatments could generally be described by the same suite of habitat variables. During the 10-year post harvest sampling interval, the modified and state buffer treatments diverged from their 2-year post-harvest position; however, they occupied a similar position in space relative to each other. If the riparian treatments were distinctly different, one would expect the two treatments immediately after logging to be positioned far apart in space; over time, although the riparian habitats would change, the separation in space should continue to be apparent and measurable, with some degree of convergence due to the growing of adjacent upland forest. While some of the habitat attributes varied relative to treatment and transect, the vegetation structure and complexity of treatments was not great enough to produce two distinct treatments.

Similarly, and perhaps more importantly, the plotting of upland habitats shows that, pre-harvest, all treatments are tightly clustered in the same space. Two years after logging, the control sites maintain a position close to the pre-treatment condition and the treated groups diverge. This pattern is repeated 10-years post-harvest, with the control sites maintaining a position close to the pre-treatment and 2-year post-treatment positions; and the treatments diverge, both away from the 2-year post-treatment and pre-treatment conditions. The distribution of upland transects of the modified and state buffer treatments provides a visual indication of forest succession and of treatments varying equally across time.

The position of the modified buffer and state buffer treatments after logging varies somewhat between the riparian and upland habitats. Two years after logging, the modified buffer and state buffer treatments are similar on the both the riparian and upland habitats, occupying similar positions in space. Ten years after logging, the modified buffer and state buffer treatments again occupy a similar position in space on the upland habitats but diverge on the riparian habitats. This suggests that the two treatments were similar 2 years after logging. However, in the 10-year post-harvest interval the differences between the two treatments were large enough (they occupy different positions in space) that measurable differences in the abundance, richness, and diversity of species should have been detectable. Also of note is how the riparian habitats of the modified treatment group approach the control condition 10 years after logging. This suggests that the prescriptions used to develop the modified buffer width are such that the riparian habitat can approach pre-harvest conditions over a relatively short time. Furthermore, the similarity of upland sites was expected, as both the modified and state treatments were clearcut in the uplands, with only the width of the riparian buffer varying.

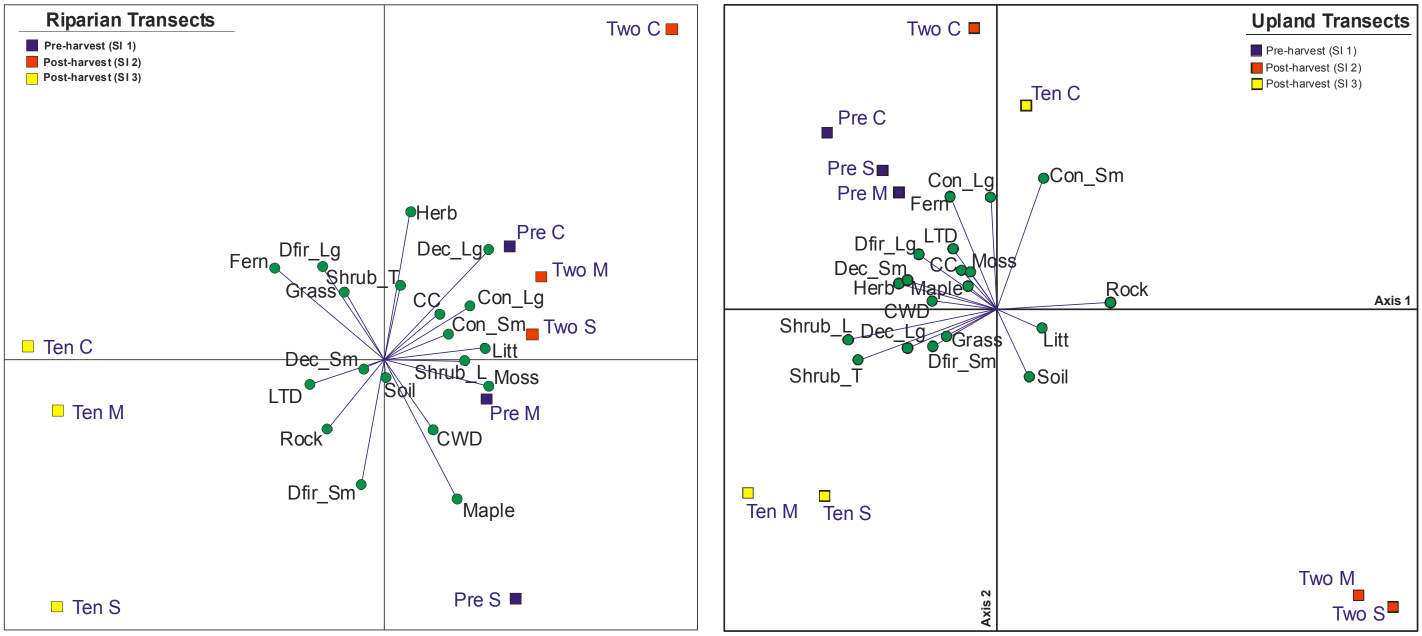

Figure 4-23. Biplot of treatment groups relative to the averaged (pooled) habitat condition for the riparian and upland habitats sampled in western Washington, 1993–2004. Pre C = Pre-harvest control; Pre M = Pre-harvest modified; Pre S = Pre-harvest state; Two C = Two-year post-harvest control; Two M = Two-year post-harvest modified; Two S = Two-year post-harvest state; Ten C = Ten-year post-harvest control; Ten M = Ten-year post-harvest modified; Ten S = Ten-year post-harvest state. Shrub_T = tall shrubs; Shrub_L = low shrubs; CC = overstory canopy cover; CWD = coarse woody debris; Fern = fern cover; Grass = grass cover; Herb = herbaceous cover; Litt = deciduous leaf-litter cover; LTD = deciduous leaf-litter depth; Moss = moss cover; Rock = rock cover; Soil = exposed soil; Maple = vine maple; Dec_Sm = bigleaf maple and red alder <1.3 m; Dec_Lg = bigleaf maple and red alder >1.3 m; Con_Sm = western redcedar and western hemlock <1.3 m; Con_Lg = western redcedar and western hemlock >1.3 m; Dfir_Sm = Douglas-fir <1.3 m; Dfir_Lg = Douglas-fir >1.3 m. CLGA = *Clethrionomys gapperi*; MILO = *Microtus longicaudus*; MIOR = *Microtus oregoni*; NEGI = *Neurotrichus gibbsii*; PEKE = *Peromyscus keeni*; PEMA = *Peromyscus maniculatus*; PERO = *Peromyscus* species; SOBE = *Sorex bendirii*; SOMO = *Sorex monticolus*; SOTR = *Sorex trowbridgii*; SOVA = *Sorex vagrans*; ZATR = *Zapus trinotatus*.

Discussion

This study provides a measure of the short- (2-year) and medium-term (10-year) responses of small-mammal populations to the effects of logging of second-growth forest. It also evaluates and compares the efficacy of riparian buffer widths in providing habitat attributes necessary to maintain the diversity, richness, and abundance of small mammals. Our findings suggest that there is no global response to riparian management zone (RMZ) width by small mammals in western Washington. Rather, species showed individual responses that varied over time, treatments, and transects. There is some suggestion of effects of buffer width and logging treatments for some species, but the analysis is confounded by patterns of natural population change over time (some local, some regional) and by likely differences among sites within treatment groups.

The works of several researchers (Gashwiler 1970; Hooven 1973; Martell and Radvanyi 1977; Kirkland 1990; McComb et al. 1993) show that logging usually results in changes in abundance of some species of small mammals in the community but not in species richness. The use of diversity indices to describe community similarity is contentious (Hurlbert 1971; Krebs 1999; Stirling and

Wilsey 2001). O'Connell et al. (2000) pointed out that measures of species richness are not particularly useful in assessing the effects of timber harvesting on the small-mammal community, because species favoring early successional and edge habitats are likely to replace species preferring closed-canopy areas. Such species interactions may give a net result of no change in species numbers in a given area, when in fact the species composition may have changed considerably. As such, we are inclined to rely on changes in capture rates of individual species rather than on species richness between pre- and post-harvest periods in assessing the effects of logging on the small-mammal communities.

Of the 24 species caught, deer mice (*Peromyscus* spp.), shrews (*Sorex* spp.), and voles (mainly *Microtus* spp.) dominated the communities in each sampling period. Temporally, major changes in abundance were recorded for some species within the control sites, particularly *Peromyscus* spp., *Sorex* spp. and *Clethrionomys gapperi*, and may have been due in part to the occurrence of regional fluctuations among populations (Aubry et al. 1998; West 1998). On the other hand, several species showed little change in abundance over time at the control sites. Species for which abundance levels were relatively stable for the duration of study included *M. longicaudus*, *S. vagrans,* and *Zapus trinotatus* in upland areas, and *C. gapperi*, *P. keeni*, and *Neurotrichus gibbsii* in riparian areas.

Species that tend to decline in numbers with clearcutting include shrew moles (Cross 1985; Cole et al. 1998) and red-backed voles (Martell 1983), whereas deer mice and creeping voles tend to increase (Hooven and Black 1976; Martell and Radvanyi 1977). These findings are consistent with those of this study, with both shrew moles and red-backed voles declining in abundance, while creeping voles increased at upland logged sites following logging. However, the responses by deer mice were inconsistent between species.

In general, *Peromyscus* spp. are regarded as habitat generalists, capable of occupying a wide variety of forested habitats (Baker 1968; Martell 1983; Sullivan et al. 1999). However, several studies have shown that the Forest Deer Mouse (*P. keeni*) is more abundant in older, continuous forested stands in the Pacific Northwest. For example, Lomolino and Perault (2000) reported that the density of *P. keeni* increased with forest age, with the lowest densities in clearcuts. In this study, we saw a marked decline in abundance of *P. keeni* in upland modified and state buffer sites during both post-harvest periods, providing an indication of a logging effect, as the species' abundance at the control sites was both greater and stable over that period. These findings appear to corroborate Lomolino and Perault's findings. However, Carey and Harrington (2001) found *P. keeni* dominant in stands as young as 34 years, and Carey and Johnson (1995) found the species in stands ranging from 40 to 70 years old. This suggests that, because clearcuts usually develop rapidly into forests, the temporal impacts on *P. keeni* are probably relatively short-lived if forests are managed for biodiversity and the logging rotation exceeds 90 years (Bayne and Hobson 1998). However, in western Washington, forest are managed on a 40- to 50-year average rotation (with some rotations longer, some shorter), and continuing to manage second-growth forests following existing rotation frequencies could have long-term detrimental impacts to populations of *Peromyscus* spp. The lack of significant recovery in abundance 10 years after logging may be related to factors other than logging, as the species' abundance also declined at the control sites during that period. This suggests that, at our study sites, the effects of logging were likely exacerbated by regional effects.

In contrast, there is no evidence that *P. maniculatus* or deer mice assigned to the group *Peromyscus* (i.e., either *P. keeni* or *P. maniculatus*) were adversely affected by logging; the significant decline in abundance of *P. maniculatus* in both riparian and upland transects of modified and state buffer sites during the 10-year post-harvest period is consistent with the decline at the control sites and is almost certainly unrelated to logging. Likewise, although the abundance of the unidentified species of *Peromyscus* increased appreciably in riparian and upland habitats at modified and state buffer treatments during the post-harvest periods, these changes are consistent with those at the control sites and are therefore probably unrelated to logging.

Other species that appear to have been affected by logging include the Shrew Mole (*N. gibbsii*), two species of shrews (*S. bendirii* and *S. trowbridgii*), and possibly in the long-term the Creeping Vole (*M. oregoni*). It is not entirely clear whether the changes in abundance of *M. oregoni* over time are related to logging or due to other factors. Its increase in abundance in upland habitats (and to some extent in riparian) at the modified and state buffer sites during the 2-year post-harvest period appears to be logging related, whereas the species' decline at these sites during the 10-year post-harvest period is consistent with the decline at the control sites and unlikely due to logging. Changes in the relative abundance of *M. oregoni* may be related to successional change in vegetative cover, with the logged sites going from relatively barren areas with snags immediately following harvest to areas covered with small coniferous trees, some small deciduous trees, and relatively dense ground cover 10 years after harvesting. Because *M. oregoni* prefers drier slopes with grasses, ferns, mosses, and other herbaceous forms of cover, changes in vegetative cover related to logging and forest regrowth were probably responsible for the observed changes in relative abundance noted 2 and 10 years after logging.

Three species of shrews—*S. monticolus*, *S. vagrans,* and *Sorex* sp.—appear to have been minimally affected by logging. The decline in abundance of *S. monticolus* on riparian and upland transects during the 2-year post-harvest period is consistent for all sites (including the control sites) and is not likely related to logging. Similarly, the decline in abundance of the unidentified *Sorex* species on riparian and upland transects, followed by a significant rise during the 2- and 10-year post-harvest periods, was consistent for all sites and most probably unrelated to logging. Although *S. vagrans* generally shows little evidence of logging effects, its significant increase in abundance at upland modified buffer sites during the 2-year post-harvest period is inconsistent with that of the control sites and may be related to logging. *S. vagrans* is regarded as a generalist species, preferring areas with small deciduous trees, Douglas-fir, and ground cover of grasses, mosses, ferns, deciduous leaf litter, and herbaceous growth.

Our findings provide no evidence that wider RMZs supported a greater abundance or more diverse community of small mammals than did narrow RMZs. Small-mammal abundance was highest at the modified buffer treatment during all three sampling intervals; however, when the total catch for riparian and upland transects was plotted separately, the differences between modified and state buffer sites were minor, with the catch generally being greater on upland transects, particularly in the 10-year post-harvest period. Moreover, differences in mean catch rates between modified and state buffer sites for 11 of the more common species of small mammals were non-significant for both riparian and upland transects in all sampling periods. Only the catch of *Z. trinotatus* differed significantly between modified and state buffer sites: it was greater at the state buffer sites following logging for reasons that are not apparent.

In light of the minor physical differences between the modified and state buffer RMZs (e.g., differences in deciduous leaf-litter depth, rock cover, and the number of small-diameter deciduous trees), the lack of significant differences in the response of small mammals to logging between modified and state buffer sites is not surprising. Greater differences between the riparian silvicultural treatments might have resulted in more pronounced effects on small-mammal populations in relation to logging. In a recent experimental study on the effects of clearcutting on small-mammal populations in southwestern British Columbia, Cockle and Richardson (2003) found that where riparian buffers were retained along streams in cut areas, species richness and abundance of two species of voles were similar to those of their control sites. This led them to conclude that 30 m–wide riparian buffers reduced the short-term impact of logging but did not eliminate impacts altogether. Cross (1985) also found that species richness of small mammals in riparian areas adjacent to logged sites was similar to that of unlogged sites. Several authors, including Darveau et al. (2001), have suggested that small mammals operate on a spatial scale in which 30 m–wide riparian strips are sufficient.

Overall, species richness of small mammals was similar between riparian and upland areas at both the modified and state buffer sites during the pre-harvest period. At the control sites, however, species richness was significantly greater in riparian areas and may largely have been due to regional differences in vegetation and to distribution and abundance of these populations. Varying results have been reported by others on species-richness levels of small mammals in riparian versus upland habitats. As examples, Rector (1990) found no difference in species richness between riparian and upland areas along the Nisqually River, whereas both Doyle (1990) and McComb et al. (1993) reported higher species richness in riparian than in upland areas. The findings of several studies indicate that small-mammal diversity in upland forested areas either increases or remains unchanged after logging (Cross 1985; Kirkland 1990), whereas in riparian areas it tends to decrease (Cross 1985; Cockle and Richardson 2003). In this study, species diversity in riparian areas at logged sites increased slightly after logging and returned to near pre-harvest levels 10 years after logging.

Although small-mammal species richness did not vary significantly across time, species composition did. The changes in species composition can be partially explained using standard conceptual models. Coniferous forests typically display a pattern of species richness depicted in Figure 4-24, which can be attributed to both plants and animals. In this pattern, species richness is typically highest during the earliest and latest seral stages. As the tree canopy closes (around stage 4), the understory is shaded out, resulting in a decline in habitat suitability for many species. As succession progresses, the structural diversity and richness of plant species in the forest increases. This brings about an improvement in habitat suitability, which in turn facilitates an increase in wildlife-species richness. Although wildlife-species richness in early and late successional stages can be similar and can have some species in common, the makeup of the wildlife communities is notably different. Thus, increases in relative abundance (at upland sites) and species diversity following forest harvesting of mid-seral second-growth stands in the present study area was expected; this is because, pre-harvest, stands in the present study were at a seral stage of ~4–5 (i.e., second-growth stands dominated by Douglas-fir ~65–75 years old), when natural levels of diversity and abundance are typically lowest (Figure 4-24). By harvesting sites and then allowing succession to proceed to stage ~2, an increase in species diversity is expected. Similarly, over time the seral stage of the buffers will advance, resulting in further increases in species diversity. Though not a novel finding by any means, this study clearly shows that the habitat suitability for a given species of small mammal is tied to measures of the physical environment that can be altered both directly and indirectly by forest harvesting.

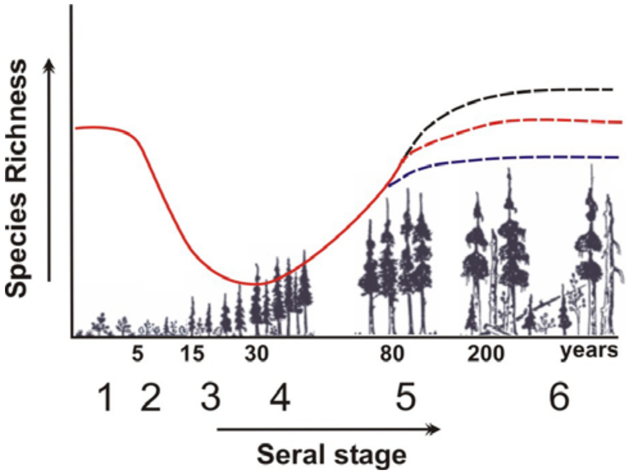

Figure 4-24. Conceptual model of the relative change in species richness (number of species) for wildlife communities as a function of forest seral stage, showing three possible outcomes (dashed lines) of lesser, similar, and greater species richness during the earliest and latest stages. Source: adapted from Franklin (1990) and Bunnell et al. (1999).

Habitat Relationships

Co-inertia analysis (COIA) has been evaluated extensively (e.g., Dray et al. 2003), and although it not used frequently in ecological studies compared to other multivariate methods, it is a rigorous method producing easily interpretable results. The visual comparison of the relationship between a species and the environmental variables should be accompanied by a meaningful biological interpretation, which requires knowledge of the species habits and site characteristics. Caution, however, needs to be exercised when assessing species–habitat relationships, because sample size can influence the analysis.

COIA proved useful in describing how the relative abundance of small mammals was related to the habitat structure of riparian and upland habitats at each treatment. In this regard, we were able to incorporate the measure of relative abundance (catch per unit effort) into an assessment of how that index varied over time relative to habitat alteration. This enabled a description of habitat associations between and among juxtaposed habitats that provided an indication of how species responded to large-scale habitat alterations and to the two different riparian buffer prescriptions.

From the COIA, it is evident that a high proportion of the small mammals in the western Washington study areas were generalist species. These species occupy a space close to the origin of both axes. Generalists are opportunists and capable of occupying a wider range of habitat conditions than more specialized species (Krebs and Wingate 1976). Generalists are likely to cope with clearcutting and associated early successional habitat changes (Cockle and Richardson 2003), whereas species preferring closed-canopy areas are likely to move into adjacent riparian habitat, which may lead to increased competition for resources in these areas. Assessing the effects of logging on the small-mammal community may, for some species, be compounded by the indirect effects of species interactions, in addition to that of successional changes in space and time.

Some species of small mammals prefer habitats close to water, with dense ground cover, and are therefore not likely to be seriously impacted by logging in upland areas. One such species is *Microtus longicaudus*, which appears to have been unaffected by logging. This vole species was not abundant at any of the upland sites (including the controls) over the duration of study; it comprised from about 2.4% to 6.5% of the total catch in riparian habitats where it was associated with areas of small (<25 cm diameter at breast height [DBH]) and large-diameter (>25 cm DBH) Douglas-fir and small deciduous trees, with ground cover consisting of herbaceous growth, grasses, and coarse woody debris.

Summary of Logging Effects

An assessment of the short- and long-term (i.e., 2- and 10-year post-harvest) effects of logging on small-mammal populations in riparian and upland habitats at each of the three treatments is summarized by species in Table 4-14. A brief description of this table is necessary to interpret the results. The assessment of logging effects is based on comparing species-abundance values in riparian and upland habitats at the modified and state buffer sites with those of the controls. In assigning whether the population increased (▲), declined (▼), or remained stable (–) over time, the abundance values at each of the three treatments in the 2- and 10-year post-harvest periods are compared against those of the pre-harvest and 2-year post-harvest periods, respectively. For 11 species, the available data are inadequate (blanks) for assessment of logging effects using this approach. For the 13 species for which the data are adequate, the following procedure was used to assess whether logging had a positive (+), negative (-), or nil (O) effect on the population. If the abundance at the cut-site during the 2- and 10-year post-harvest periods was a minimum of 10% greater or less than that at the control in the pre- and 2-year post-harvest periods, respectively, then the effect of logging was rated as an increase (+) or decrease (-). In situations where the difference in abundance (either an increase or a decrease) between cut-sites and controls was <10%, logging effect was rated as nil (O).

The main findings of the short- and long-term effects of logging on small-mammal populations summarized in Table 4-14 are as follows:

1. Two years after logging, 5 species were negatively affected by logging and were less abundant in riparian habitats at the logged sites (*Clethrionomys gapperi*, *Neurotrichus gibbsii*, *Peromyscus keeni*, *Sorex bendirii*, and *S. trowbridgii*). The abundance of these same 5 species plus *Sorex* sp. also declined on upland transects at logged sites.
2. Ten years after logging, the number of species negatively affected by logging declined relative to the 2-year post-harvest interval. In addition, the abundance of *Peromyscus* spp. and *S. vagrans* was lower in riparian habitats at logged sites (i.e., modified and state treatments) relative to the control sites. A similar trend was observed in upland habitats at logged sites for *Peromyscus* spp., *S. vagrans*, and *S. monticolus*.
3. The abundance of 8 species increased during one or both of the post-harvest sampling intervals, with 3 species continuing to have higher abundance on riparian transects at logged sites 10 years after logging. Five species had higher abundance in either the 2-year or 10-year post-harvest sampling interval, with the abundance of 3 species higher 10 years after logging.
4. Several species showed no measurable population effect (either negative or positive) from logging, ranging from 7 species at the modified buffer treatment to 4 species at the state buffer treatment 2 years after logging. In the 10-year post-harvest sampling interval, the number of species seemingly unaffected by logging increased to 10 at modified buffer sites and 11 at state buffer sites.
5. The long-term negative effects of logging were most evident on the Vagrant Shrew (*S. vagrans*), whose population declined at the upland logged sites and increased at the control sites likewise, this species declined in abundance at riparian state buffer sites but remained stable at riparian control sites. The decline of this generalist species at the 10-year post-

harvest interval may be related to successional change in habitat, as the species was abundant at the logged sites 2 years after logging.

6. The abundance of *Peromyscus* spp. at upland and riparian modified buffer sites declined 10 years after logging, while those at the control sites increased. Reasons for the decline of this generalist species, while the populations at the controls increased, are not apparent.

Table 4-14. Summary of the effects of logging on small-mammal populations in riparian and upland habitats in each of the three treatments for the two- and ten-year post-harvest periods. ▲= species increased in abundance; ▼ = species decreased in abundance; ▬ = species abundance remained stable; + = logging had a beneficial effect on the species; - = logging had a negative effect on the species; O = logging had no effect on the species. Species codes are explained in Appendix 4-C.

Species Code	Riparian						Logging Effect				Upland						Logging Effect			
	Control		Modified		State		Modified		State		Control		Modified		State		Modified		State	
	2-yr	10-yr	2-yr	10-yr	2-yr	10-yr	2-yr	10-yr	2-yr	10-yr	2-yr	10-yr	2-yr	10-yr	2-yr	10-yr	2-yr	10-yr	2-yr	10-yr
CLGA	▲	▲	▼	▲	▼		–	O	–		▲	▲	▼	▲	▼		–	O	–	
GLSA					▬	▼						▬			▼					
MILO	▲	▼	▲	▼	▲	▼	O	O	O	O	▼		▲	▲	▲	▬	+		+	
MIOR	▼	▼	▲	▼	▲	▼	+	O	+	O	▼	▼	▲	▼	▲	▼	+	O	+	O
MISP	▬	▼	▲	▼	▲	▼	+	O	+	O			▲	▼		▼				
MITO			▲	▼										▼						
MUER		▬		▼		▼						▼		▼		▲		O		+
MUMU																				
NEGI	▬	▼	▲	▼	▼	▼	+	O	–	O	▲	▼	▼	▲	▼	▲	–	+	–	+
PEKE	▬	▬	▼	▲	▼	▬	–	+	–	O	▲	▼	▼	▲	▼	▲	–	+	–	+
PEMA	▬	▼	▬	▼	▲	▼	O	O	+	O	▲	▼	▲	▼	▲	▼	O	O	O	O
PERO	▲	▲	▲	▬	▲	▲	O	–	O	O	▲	▲	▲	▬	▲	▲	O	–	O	O
SCOR			▼		▬	▬										▲				
SCTO																				
SOBE	▲	▼	▼	▲	▼	▲	–	+	–	+	▲	▼	▼		▼	▲	–		–	+
SOCI	▼					▼						▼			▬	▼				O
SOMO	▼	▲	▼	▲	▼	▲	O	O	O	O	▼	▲	▼	▬	▼	▲	O	–	O	O
SOPA			▲	▼	▼								▼		▼					
SOREX	▼	▲	▼	▲	▲	▲	O	O	+	O	▲	▲	▼	▲	▬	▲	–	O	–	O
SOTR	▲	▼	▼	▼	▼	▼	–	O	–	O	▲	▼	▼	▼	▼	▼	–	O	–	O
SOVA	▼	▬	▼	▲	▲	▼	O	+	+	–	▬	▲	▲	▼	▲	▬	+	–	+	–
TASP																				
TATO				▼		▼						▼		▼		▲		O		+
ZATR	▲	▼	▲	▼	▲	▼	O	O	O	O		▼	▲	▼		▲		O		+

Management Implications

Extensive forestry management in the Pacific Northwest has historically included clearcutting, prescribed fire, regeneration by seed trees, protection from forest fires, and salvage of timber killed by windthrow or disease (Carey and Harrington 2001). In the last 20 years, planting of Douglas-fir has increased, with most forest stands >40 years old originating from seed. Current management practices include the retention of seed trees and coarse woody debris and the planting of seedlings. Implicit in the shift to ecosystem management has been the retention of riparian management zones (RMZs) around perennial fish- and non–fish-bearing streams, which have been required in some form since 1988.

The RMZ width necessary to preserve species richness, diversity, and abundance of small mammals has not been adequately addressed for most ecological systems. However, Cockle and Richardson (2003) and Cross (1985) found that RMZs function to ameliorate the effects of upland logging on small mammals. There were species that had lower relative abundance at one logged treatment relative to the other immediately after logging. For example, *Neurotrichus gibbsii* had lower relative abundance in riparian habitats at state buffer sites relative to modified buffer sites; however, this difference did not exist 10 years after logging. Overall, we were unable to identify a link between riparian buffer width and species richness, diversity, or abundance. When species abundance was lower on logged sites, the magnitude of change was equivalent at both the modified and state buffer treatments, suggesting that the width of the riparian zone did not influence the small-mammal communities in upland habitats. The physical width of the RMZ, at least in this study, was not indicated in preserving species richness, diversity, or abundance. At the same time, although RMZs provide structural heterogeneity and complexity, the production of homogenous, simplified, even-aged stands of second-growth Douglas-fir forests in upland habitats may, over the long term, lead to localized extirpations of species like the Forest Deer Mouse.

For species that decline in abundance because of logging, management strategies to mitigate these declines have been proposed. For example, Carey and Harrington (2001) concluded that management-induced homogeneity and simplification (i.e., reduction of complexity) posed a real danger of small-mammal communities becoming non-supportive of predator populations. They proposed that active management for complexity on long rotations (>90 years) may be necessary to conserve the diversity attributed to forests. Because small-mammal communities are inherently linked to many forest ecosystem processes, the development of management strategies must be considered at a broader scale and should not be based on single-species conservation. The introduction of heterogeneity into homogenous, even-aged stands has been shown to have positive effects on biodiversity and ecosystem function in the short term (<5 years) (Carey 2003).

Since this study's inception, there have been changes to the Washington Forest Practices Rules and Regulations that have implications for our findings. The Forest Practices Rules of 1988 required a riparian buffer that met the state configuration (i.e., not less than 7.5 m from the ordinary high-water mark) for both fish-bearing (Type 3) and non–fish-bearing (Type 4) perennial streams. The development of the modified riparian prescription was an alterative to the state buffer that provided for greater structural complexity, heterogeneity, and ultimately greater diversity of both flora and fauna. The Forest Practices Rules of 2001, aside from changing how watercourses are defined, are site-specific, with many options. For example, there are now 3 zones of management for Type F streams (Type 3): (1) the core, (2) inner, and (3) outer zones, with various levels of tree removal, thinning, hardwood conversion, and other activities permitted within these zones depending on the site class of land, the management harvest option, and the bankfull width of the stream. In general, the site class will form the RMZ prescription. Therefore, riparian management has become (necessarily) site-specific. This implies that applying a one-size-fits-all approach to riparian management has been set aside in favor of a more intensive approach that considers the specifics of the site. For Type Np streams, or perennial non–fish-bearing streams (Type 4), the RMZ rules are slightly less complicated; however, there is ample room for site-specific management. Considering that the RMZ rules have evolved to embrace a site-specific approach, it seems only fitting that forest management in general adopt the same principals when considering biodiversity. Managing for biodiversity on a regional scale does not account for variation at the landscape or site level within the region, which could have dire consequences for species that are limited geographically, that occur at naturally low densities, or that are particularly sensitive to habitat change. As an alternative to the status quo, management that focuses on stand heterogeneity (as opposed to tree-species monocultures with reduced diversity) should produce landscapes with high biocomplexity and biodiversity at multiple scales, and therefore these landscapes should be resilient in the face of disturbance (Holling 2001).

Conclusions

It appears that the value of riparian zones for small mammals varies with species, as is to be expected. For species that readily adapt to large-scale habitat alterations, such as deer mice (Darveau et al. 2001), riparian zones may not be necessary; but for other species, riparian zones provide important connectivity in the landscape as well as being areas from which such species can recolonize regenerating forest stands (Cockle and Richardson 2003). For example, *Microtus oregoni* was consistently documented in higher abundance in the riparian habitats at all treatments during each sampling interval, indicating that this species would benefit from the retention of riparian zones around creeks in western Washington forests.

The efficacy of RMZs in providing similar habitat conditions capable of supporting equal species richness and community composition 2 and 10 years after logging is clearly related to the species and not to small mammals as a group. In terms of whether or not the state buffer performed differently than the modified buffer, we need to again assess the species composition of small mammals in these two nominally distinct conditions for each post-logging sampling interval. Our analyses demonstrate that, 2 years after logging, the species composition changed and the relative abundance of certain species increased or decreased on both the riparian and upland transects at the control treatment (Figure 4-5). The fact that the relative abundance of small mammals varied at the controls over time hints at a probable regional effect, which was likely observed at all three treatment groups (control, modified, and state). Therefore, the ability to assign significant changes in relative abundance (Table 4-9) to a particular treatment was weak at best. This is substantiated, at least in part, by the results of the contrasts analyses, which show for each sampling interval a lack of significant changes in the relative abundance of 12 of the more common species between the state and modified treatment groups in riparian habitats (the exception is *Zapus trinotatus* in the immediate post-logging sampling interval) There are more significant differences between the relative abundance of small mammals in the upland habitats (e.g., *M. oregoni* and *Sorex trowbridgii* decrease, *Peromyscus keeni* increases); however, these species also declined in the upland habitats of the control group, pointing again to a regional effect that may have been exacerbated to some degree by logging.

The treatment groups varied markedly in terms of habitat structure (Figure 4-23), and over time the modified treatment appears to be converging with the control group more rapidly than the state treatment. This is likely due to the more variable and complex structure of the riparian buffer retained at sites within the modified treatment. The width of the modified buffer likely provided greater stability for the riparian habitats by limiting the effects of wind, which would reduce the potential for blowdown. The heavier canopy and increased complexity of the vegetation community was also probably aligned with the conditions observed at the control sites and enabled the modified treatment to approach the control condition more quickly than the state treatment. In contrast, the state treatment was typically devoid of canopy and vegetative complexity because of extensive blowdown, which was observed at the state buffer sites during the 10-year post-harvest sampling interval.

Given that the relative abundance of small mammals did not show any clear pattern of change over time relative to treatment or transect, and that any changes observed cannot necessarily be linked to the retention of one buffer width over another, it appears (with few exceptions) that buffer width does not influence small-mammal communities in western Washington to a degree that is measurable on a consistent basis. Clearcut logging in upland habitats, on the other hand, does appear to affect both the relative abundance and community structure of small-mammal populations, with community similarity decreasing in upland habitats 2-years and 10-years post-logging, regardless of buffer width (Figure 4-16). In contrast, community similarity of riparian habitats remained virtually unchanged across time, again indicating that a particular buffer width without other factors does not generally influence the community structure of small mammals in western Washington.

Knowing which species occur in a given area is the first step toward recognizing the diversity of a particular site for small mammals. Management, however, requires an understanding of how those species interact with their environment and whether or not any (or all) of the species require specific habitat attributes or are riparian- or upland-associated species. Because it is possible to show that the abundance of certain species is correlated with particular structural elements or seral stages, it is therefore possible to model the potential effects of forest removal and riparian-zone creation on small mammals. This, coupled with the ability of forest managers to creatively retain habitat patches, implement RMZs, and to leave structural elements such as coarse woody debris on the landscape after logging, suggests that landscapes can be effectively managed for small-mammal diversity and abundance. While site prescriptions are necessary to deal with stand-level conservation objectives, they areonly part of the overall management plan developed for a landscape unit or watershed. Further, conserving the multitude of other taxonomic groups (i.e., plants, invertebrates, amphibians, reptiles, and songbirds) that warrant consideration will often require management actions very different from those that would be prescribed only for small mammals.

This is one of a very few studies utilizing a Before-After-Control-Impact (BACI) design to evaluate the longer-term efficacy of RMZs in providing habitat for small mammals. Moreover, it appears that this is the only study to span a period of 10 years and that was not limited to a small geographic area. The findings of this study can therefore be applied across a wide geographical area, which in this case consists of western Washington.

This study also specifically assessed how the relative abundance of small mammals was correlated with habitat variables present in riparian and upland habitats of each treatment group to determine if the habitat variables measured could be used as surrogates to predict small-mammal relative abundance and diversity. The methods used to describe the relationship between relative abundance of small mammals and habitat features are underused in ecology (Dray et al. 2003), and this study shows the utility of such methods in providing descriptive habitat relationships for many species of small mammals.

A number of factors potentially influence the veracity of this study's results or limit the extent to which the conclusions can be used to aid in managing and conserving small mammals in the coniferous forests of the Pacific Northwest. The purpose of presenting these limitations is to make the reader aware of (1) factors that could have influenced the results of the study and (2) factors that should be considered if this study is to be used to guide formal policy development. Perhaps most important is the accuracy of the data collected. This study was carried out by various primary investigators and the data handled and treated with various degrees of care and accuracy. The data used in the analyses likely contain errors that were not detectable, and we acknowledge that the database assimilated over the course of this study contains errors to some unknown extent. Although steps were taken to reduce this bias, its potential influence on the results must be considered.

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Appendices

Appendix 4-A. Ecological description of each site sampled in western Washington, 1992–2004. Refer to Figure 4-1 for distribution of study sites. – data not available; OP/SW = Olympic Peninsula and SW Washington; PT = Puget Trough; WSC = western slopes and crest, Washington Cascades physiographic provinces.

			Physiographic	Elevation (m ASL)			Stand Age			Stream Type		Bankfull		Riparian	
Site Name	Treatment	County	Province	Min	Max	Slope (%)	Year 1	Year 14	Dominant Tree	1988	2000	Width (m)	Shape	CC (%)	Lithology
Abernathy	Control	Cowlitz	OP/SW	362	491	16.0	55-65	68-78	Douglas-fir	3	F	5.04	Sloping	71.0	Basalt flows
Elbe Hills	Control	Pierce	WSC	554	609	10.0	55-67	68-80	Douglas-fir	4	F	3.20	V-shaped	91.0	Volcanclastic deposits or rocks
Hotel Creek	Control	King	PT	277	303	4.0	55-71	68-84	Douglas-fir	4	F	2.38	V-shaped	99.0	Continetal glacial outwash - Fraser Age
Porter Creek	Control	Thurston	PT	285	290	3.0	55-75	68-88	Douglas-fir	3	F	5.98	V-shaped	95.0	Basalt flows and flow breccias
Taylor Creek	Control	King	PT	415	424	2.0	55-81	68-94	Douglas-fir	2	F	6.47	V-shaped	93.8	Continetal glacial till - Fraser Age
Vail	Control	Lewis	PT				55-82	68-95	Douglas-fir	5	N				Andesite Flows
West Fork Falls Creek	Control	Thurston	PT	379	463	20.0	55-83	68-96	Douglas-fir	3	F	3.26	V-shaped	92.0	Basalt flows and flow breccias
Bluetick	Modified	Grays Harbor	PT	110	151	7.0	55-66	68-79	Douglas-fir	4	F	2.02	Undercut	94.0	Marine sedimentary rocks
Eleven Creek 31	Modified	Lewis	PT	493	561	6.0	55-68	68-81	Douglas-fir	4	N	3.30	V-shaped	92.5	Andesite flows
Griffen Creek	Modified	King	WSC	157	188	5.0	55-70	68-83	Douglas-fir	2	F	4.06	V-shaped	93.0	Continetal glacial till - Fraser Age
Ms. Black	Modified	Thurston	PT	67	92	3.0	55-73	68-86	Douglas-fir	3	F	1.82	Undercut	83.0	Basalt flows and flow breccias
Ryderwood 860	Modified	Cowlitz	OP/SW	303	337	8.0	55-78	68-91	Douglas-fir	3	F		V-shaped	75.0	Nearshore sedimentary rocks
Side Rod	Modified	Pierce	WSC	520	548	6.0	55-79	68-92	Douglas-fir	4	N	2.70	V-shaped	90.0	Volcanclastic deposits or rocks
Eleven Creek 32	State	Lewis	PT	415	501	10.0	55-69	68-82	Douglas-fir	4	N	1.32	V-shaped	98.0	Andesite flows
Kapowsin	State	Pierce	WSC	443	450	2.0	55-72	68-85	Douglas-fir	3	N	3.08	V-shaped	51.0	Basaltic andesite flows
Night Dancer	State	Grays Harbor	PT	125	201	11.0	55-74	68-87	Douglas-fir	5	F	1.44	V-shaped	96.0	Marine sedimentary rocks
PotPourri	State	Thurston	PT	239	249	7.0	55-76	68-89	Douglas-fir	3	F	5.18	V-shaped	74.0	Basalt flows and flow breccias
Ryderwood 1557	State	Cowlitz	OP/SW	333	376	7.0	55-77	68-90	Douglas-fir	4	N	3.30	V-shaped	77.0	Nearshore sedimentary rocks
Simmons Creek	State	Lewis	WSC	422	471	8.0	55-80	68-93	Douglas-fir	3	F	3.22	V-shaped	80.0	Basaltic andesite flows

Appendix 4-B. Site-specific monitoring at each of the western Washington study sites. A “1” indicates the site was sampled in a given year; blanks indicate no sampling.

		Year								
Treatment	Site Name	1992	1993	1995[a]	1996	1997	1998	2003	2004	Years Sampled
Control	Abernathy		1	1	1	1[b]	1[b]	1	1	7
	Elbe Hills		1	1	1			1	1	5
	Hotel Creek	1	1	1	1			1	1	6
	Porter Creek	1	1	1	1			1	1	6
	Taylor Creek	1	1	1	1			1	1	6
	Vail	1	1	1	1[c]					4
	West Fork Falls Creek[c]							1	1	2
Modified	Blue Tick		1	1	1			1	1	5
	Eleven 31	1	1	1	1			1	1	6
	Griffen Creek[d]	1	1	1	1					4
	Ms. Black	1	1	1	1			1	1	6
	Ryderwood 860	1	1	1	1			1	1	6
	Side Rod		1	1	1			1	1	5
State	Eleven 32	1	1	1	1			1	1	6
	Kapowsin	1	1	1	1			1	1	6
	Night Dancer		1	1	1			1	1	5
	Potpourri	1	1	1	1			1	1	6
	Ryderwood 1557	1	1	1[b]		1[e]	1[e]	1	1	7
	Simmons Creek	1	1	1	1			1	1	6
	Sites per Year	13	18	18	17	2	2	17	17	

[a] First year of post-harvest monitoring at the modified and state treatments

[b] Data excluded from analysis

[c] Vail was logged in 1998, therefore West Fork Falls Creek was added in 2003

[d] Not sampled in 2003 or 2004

[e] Ryderwood 1557 was logged in 1996, therefore 1997 and 1998 data were used for SI 2 in all analyses

Appendix 4-C. Codes and scientific and common names of small mammals referred to in the text.

Code	Scientific Name	Common Name
CLGA	*Clethrionomys gapperi*	Southern Red-backed Vole
GLSA	*Glaucomys sabrinus*	Northern Flying Squirrel
MILO	*Microtus longicaudus*	Long-tailed Vole
MIOR	*Microtus oregoni*	Creeping Vole
MISP	*Microtus* spp.	
MITO	*Microtus townsendii*	Townsend's Vole
MUER	*Mustela erminea*	Ermine
MUMU	*Mus musculus*	House Mouse
NEGI	*Neurotrichus gibbsii*	Shrew Mole
PEKE	*Peromyscus keeni*	Keen's Mouse
PEMA	*Peromyscus maniculatus*	Deer Mouse
PERO	*Peromyscus* spp.	
SCOR	*Scapanus orarius*	Coast Mole
SCTO	*Scapanus townsendii*	Townsend's Mole
SOBE	*Sorex bendirii*	Pacific Water Shrew
SOCI	*Sorex cinerus*	Masked Shrew
SOMO	*Sorex monticolus*	Dusky Shrew
SOPA	*Sorex palustris*	Common Water Shrew
SOREX	*Sorex* spp.	
SOTR	*Sorex trowbridgii*	Trowbridge's Shrew
SOVA	*Sorex vagrans*	Vagrant Shrew
TASP	*Tamias* spp.	
TATO	*Tamias townsendii*	Townsend's Chipmunk
ZATR	*Zapus trinotatus*	Pacific Jumping Mouse

Appendix 4-D. Habitat codes and definitions referred to in the co-inertia biplots.

Code	Definition	Measure	Notes
Shrub_T	Tall Shrubs	% Cover Tall Shrubs	1–3 m in height
Shrub_L	Low Shrubs	% Cover Low Shrubs	<1 m tall
CC	Canopy Cover	% Cover overstory	
CWD	Coarse Woody Debris	% Cover	
Fern	Fern Cover	% Cover	
Grass	Grass Cover	% Cover	
Herb	Herb Cover	% Cover	
Litt	Deciduous Leaf-litter Cover	% Cover	
LTD	Deciduous Leaf-litter Depth	mm	
Moss	Moss Cover	% Cover	
Rock	Rock Cover	% Cover	
Soil	Exposed Soil	% Cover	
Maple	Vine-maple Cover	Number	*Acer circinatum*
Dec_Sm	Small Deciduous Trees (<50 cm DBH)	Number of trees	
Dec_Lg	Large Deciduous Trees (>50 cm DBH)	Number of trees	
Con_Sm	Small Conifers (Hemlock and Cedar; <50 cm DBH)	Number of trees	
Con_Lg	Large Conifers (Hemlock and Cedar; >50 cm DBH)	Number of trees	
Dfir_Sm	Small Douglas-fir (<50 cm DBH)	Number of trees	
Dfir_Lg	Large Douglas-fir (>50 cm DBH)	Number of trees	

Chapter 5 WESTERN WASHINGTON TERRESTRIAL AMPHIBIANS

Virgil C. Hawkes

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Abstract

We assessed the efficacy of two different riparian buffer widths in providing habitat for terrestrial amphibians using a Before-After-Control-Impact (BACI) approach. No global response by terrestrial amphibians to forest harvest or the retention of riparian management zones (RMZs) was found in western Washington. Rather, species showed individual responses that varied over time and between treatments and transects. The influence of site effects was evident for some species (e.g., Western Red-backed Salamander and Ensatina), while for others some suggestion of treatment effects existed (e.g., Coastal Giant Salamander and Coastal Tailed Frog); but population changes at both local and regional scales confounded the analysis. A co-inertia analysis revealed that habitat associations did not change considerably over time and showed that the biological response to treatments did not differ from one another. For most species, a buffer width of 14 m was adequate in preserving pre-harvest occupancy and/or abundance patterns. However, for other species (e.g., Coastal Giant Salamander) a buffer width of 30 m was required. In even-aged, second-growth stands of Douglas-fir, a riparian buffer that approximates the conditions provided by the modified prescription would maintain relative abundance and diversity values of all terrestrial amphibians at levels commensurate with those detected prior to logging. The benefits of retaining riparian forest are identified and discussed in the context of maintaining biodiversity and conserving terrestrial amphibians in western Washington.

Introduction

Over the past 15–20 years, there have been numerous reports of amphibian declines and extirpations from many regions of the world (Banks et al. 2006; Cushman 2006; Collins and Storfer 2003). Some of the most commonly reported causes for these declines are increased UV-B irradiation, disease, predation, and acid rain (Kolozsvary and Swihart 1999; Houlahan et al. 2000). However, the leading cause of decline, as is true for most organisms, is habitat loss and/or fragmentation (Kolozsvary and Swihart 1999; Cushman 2006; Karraker and Welsh 2006). In the Pacific Northwest, one cause of habitat change and/or fragmentation is timber harvesting and associated activities, such as road building. The significance of these changes with respect to changes in amphibian communities has not been adequately studied and is the major focus of this study.

The impact of timber harvesting in riparian and adjacent upland habitats varies with the type of harvest and characteristics of the watershed. For example, clearcut logging might have a greater negative impact in riparian habitats than single-tree selection (Oakley et al. 1985). The results of previous research suggest that timber harvesting in riparian and upland habitats adjacent to streams in the Pacific Northwest can either have little influence on the physical attributes of streams (Carlson et al. 1990) or can contribute to major changes to in-stream habitat structure (Harr 1976; Harr et al. 1979). In recent years, forest management strategies have included the retention of forested buffer strips adjacent to the edges of streams (i.e., a riparian management zone; RMZ) to ameliorate the potential impacts to in-stream function attributable to timber harvesting. Riparian management originated largely to protect fish habitat and water quality but now considers other environmental values, including riparian and wildlife (Castelle et al. 1994; Vesely and McComb 2002; Richardson 2003).

Because riparian buffers are typically linear (i.e., aligned parallel to the stream edge), they are influenced by the natural edge effect of the stream and the edge effect resulting from the removal of adjacent upland forest (Yahner 1988; Mills 1995). Although edge effects caused by timber harvest are often temporary, the cumulative effects of creating edges on a managed landscape is likely to be increased when target stands are logged partway through the rotation age of the adjacent maturing stands (Harris 1984), a common practice in the Pacific Northwest. The ecology of forested edges is characterized by changes in biotic and abiotic elements, both of which have been associated with negative effects documented in bird and plant communities and for some species of terrestrial amphibians (deMaynadier and Hunter 1998).

Of particular interest is the concept that RMZs provide habitat refuges for wildlife. RMZs often contain the only remnants of older forests that persist on landscapes managed for timber. These zones are critical for the preservation of species diversity and abundance and have been identified as important for habitat connectivity between upland and riparian habitats (Machtans et al. 1996; Gibbs 1998). While the retention of riparian forest appears to provide the functions listed above, and several studies have examined the relationship between riparian buffer widths and amphibian abundance (Biek et al., 2002; Vesely and McComb 2002; Crawford and Semlitsch 2007), the efficacy of RMZs retained on landscapes managed for timber harvest in providing habitat for amphibians over a longer time period has been inadequately addressed (deMaynadier and Hunter 1995; Cushman 2006).

Recent studies of the effects of timber harvest on amphibians have focused primarily on changes in abundance relative to clearcutting and not on how management strategies serve to ameliorate those effects (Dupuis et al. 1995; Aubry 2000; Butts and McComb 2000; Grialou et al. 2000; Dupuis and Waterhouse 2001; Greenberg 2001; Welsh and Lind 2001; Biek et al. 2002; Russell et al. 2002; MacCracken 2004; Karraker and Welsh 2006). The results of those studies are equivocal, particularly with respect to species-specific responses to timber harvesting and the associated changes to habitat structure and availability. For example, Aubry (2000) found no evidence that variation in amphibian abundances was strongly influenced by the amount of coarse woody debris on the forest floor, which is contrary to the findings of Dupuis et al. (1995) and Butts and McComb (2000). Despite these reported differences, some generalizations about the relationship between timber harvesting and amphibians can be made from these studies. For example, the bulk of the literature attests to the importance of a higher proportion of older stands on the landscape, which has been positively associated with higher species richness, increased biomass, and higher species abundance. The establishment of riparian habitat adjacent to streams contributes to the proportion of the landscape retained in older stands.

Many studies of the relationships between amphibian populations and logging continue to lack pre-treatment data, making it impossible to determine if the reported changes in abundance are treatment effects, site effects, or population fluctuations unrelated to site or treatment that coincided with the period of study (deMaynadier and Hunter 1995; Cushman 2006; Karraker and Welsh 2006). Most previous studies have also been limited in duration, and none of them evaluated the longer-term, post-treatment interval effects of the retention of RMZs in providing habitat for wildlife. The only temporal studies of amphibian diversity and abundance in the Pacific Northwest that we found, and that were based on a Before-After-Control-Impact (BACI) design, were an earlier, largely unpublished version of the present study (O’Connell et al. 2000) and a short-term study by Grialou et al. (2000). Hannon et al. (2002) reported the temporal results of a similar study on RMZs around lakes in Alberta, Canada; and more recently, Perkins and Hunter (2006) reported on the temporal changes of amphibians relative to logging adjacent to headwater streams in Maine. However, the temporal component of these studies was insufficient to determine the longer, post-treatment effects of riparian management strategies on amphibians, and none of these studied occurred in Washington State. All other studies reviewed used sites and treatments as proxies for pre-treatment conditions, but they did not repeatedly measure diversity or abundance at the same sites over time, making it difficult to draw inferences about the effects of clearcut logging on amphibian abundance and diversity (Vesely and McComb 2002).

Western Washington RMZs have been required, at least in some form, since 1976. Between 1976 and 1988, the Washington Forest Practices Rules and Regulations required "streamside management zones" (SMZ), which were intended to provide "stream bank integrity and temperature control" (Washington State Forest Practices Board 1976, 1982). SMZ regulations prescribed riparian buffers 7.5–15 m wide, with all nonmerchantable vegetation and "sufficient merchantable timber, if any, necessary to retain 50–75 percent of the summer mid-day shade of the water surface" left within the SMZ (Washington State Forest Practices Board 1976). In 1988, the RMZ regulations were strengthened considerably, requiring riparian buffers up to 30 m wide (Washington State Forest Practices Board 1988). The change in regulations was a result of the Timber, Fish, and Wildlife (TFW) Agreement reached in 1987 among representatives of the Washington State tribes, forest-products industry, environmental community, and natural-resource agencies. The agreement sought to resolve conflicts between these diverse groups by recognizing the common goal of preserving natural resources and simultaneously maintaining a viable timber industry.

In 1990, the Wildlife Steering Committee of the Cooperative Monitoring, Evaluation, and Research Committee (CMER) of the TFW Agreement initiated a study to evaluate the efficacy of RMZs in providing habitat for wildlife. The study was designed to evaluate the abundance and diversity of wildlife in RMZs by comparing two different buffer configurations at unlogged control sites. The buffer configurations were based on the RMZ guidelines mandated at the time (Washington State Forest Practices Board 1988) and on modifications to the 1988 guidelines. The wildlife groups selected for study were terrestrial and aquatic amphibians, riparian-associated songbirds, small mammals (rodents and insectivores), and bats. A precursor to this study (O'Connell et al. 2000) compared the diversity and abundance of these wildlife groups before and immediately after logging to identify the short-term temporal effects of logging both within and between treatments. In 2001, CMER initiated a follow-up study to evaluate the long-term effects of RMZs on wildlife. This study is the only one in Washington to evaluate the long-term post-treatment effects of riparian management strategies on wildlife based on data collected at the same sites.

This study evaluated the long-term post-treatment effects of upland forest harvest and RMZ retention on the abundance and diversity of terrestrial amphibians[8] in managed forests of western Washington. The pre-treatment and 2-year post-treatment data were provided by the Washington Department of Natural Resources; the 10-year post-treatment data were collected by LGL Limited environmental research associates. The overarching goal of this study was to evaluate the efficacy of two different RMZ prescriptions in providing habitat for terrestrial amphibian populations in western Washington 2 years and 10 years after upland clearcut logging. Related to the overall goal were five objectives:

1. To determine if the RMZs retained along third- and fourth-order streams provided habitat attributes necessary to retain amphibian diversity and abundance;
2. To determine if there was a difference in the abundance and/or diversity of amphibians in RMZs of two different widths;
3. To determine if either RMZ width maintained relative abundance and diversity of amphibians that was equal to the unlogged controls;
4. To determine if terrestrial amphibian relative abundance and diversity changed in adjacent upland habitats as a result of timber harvest; and

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[8] Stream amphibians were excluded from the 10-year post-harvest sampling regime due to high variability in the data and a lack of power to conduct the analyses.

5. To identify how habitat structure and complexity varied over time relative to treatment type to determine if habitat structure could explain differences in relative abundance or species diversity.

Methods

This study was a component of a larger study evaluating the efficacy of RMZs in providing habitat for selected groups of wildlife. The larger study was implemented in 1991, with pre-treatment sampling in 1992 and 1993. The first post-treatment sampling occurred in 1995 and 1996, with a 10-year post-treatment sample (this study) in 2003 and 2004. Effective analysis of the results of this study required that sampling techniques used in the pre-harvest and the 2-year post-harvest sampling intervals be repeated precisely in the 10-year post-harvest sampling interval.

The study design was based on a split-plot repeated-measures design, with 18 sites assigned to one of three treatments. Each site was split into two distinct habitat types: (1) terrestrial riparian and (2) upland. Terrestrial riparian habitats occurred within 5 m of the ordinary high-water mark adjacent to a perennial stream and extended to the point where the vegetation visibly shifted to an upland type. Upland habitats extended upland from the zone of transition from riparian to upland vegetation. Upland sampling occurred 100 m upslope from terrestrial riparian habitat.

The 18 sites were assigned to one of three treatments:

1. **Control**: This treatment consisted of continuous stands of maturing second growth (65–75 years old) dominated by coniferous trees, with no upland harvesting for the duration of the study.
2. **State**: This treatment consisted of a logged upland forest with a narrow forested buffer retained in the riparian zone. The RMZ was based on the Washington Forest Practices Rules and Regulations in place in 1988 concerning buffer width and number of leave trees for Type 3 (fish-bearing, perennial) streams(Table 5-1). Streams of Type 2, 4, and 5 that were structurally similar to the Type 3 streams were also used in the study but were buffered according to Type 3 regulations to keep the experimental treatments uniform (Appendix 5-A). The RMZ extended from the ordinary high-water mark to the line where vegetation changes from riparian to upland but was not less than 15 m wide. The number and arrangement of leave trees was modified to accommodate the logging operation, and the RMZ width was expanded as necessary to include swamps, bogs, marshes, or ponds adjacent to the stream.
3. **Modified**: This treatment consisted of a logged upland forest with a wider, more variable forested buffer retained in the riparian zone. The RMZ was based on harvest prescriptions developed by O’Connell et al. (2000). The prescription for the modified buffer departed from the 1988 Forest Practices Rules and Regulations in three ways:

   i) The 1988 guidelines specified a minimum canopy cover based on stream temperature classification and the elevation at the midpoint of the stream. If the cover requirement is met, selective cutting is allowed in the no-entry zone (i.e., within 7.5 m of the ordinary high -water mark). The modified prescription differs from the state prescription in that harvest is prohibited within the no-entry zone. Cover requirements are the same as those indicated in the 1988 rules (WAC-222-30-040 [2] Temperature Control; Washington State Forest Practices Board 1988). Specifically, WAC-222-30-040 (2) states that:

   > “All nonmerchantable vegetation that provides mid-summer and mid-day shade of the water surface should be retained; and

> Sufficient merchantable timber, if any, necessary to retain 50% of the summer mid-day shade of the water surface, except when ambient water temperatures exceed 15.5° C for a 7-day period. In this case, 75% of the shade should be retained."

ii) The modified prescription applies 1988 guidelines for selective harvest in state RMZs to a variable width located 7.5–22.7 m from the ordinary high-water mark. This prescription will produce a buffer that is at least 7.5 m from the ordinary high-water mark along the riparian/upland boundary.

iii) The modified prescription increases the number of wildlife reserve trees, recruitment trees, and downed logs left for each acre logged. State Forest Practices Rules and Regulations required only three wildlife reserve trees, two green recruitment trees, and two downed logs left for each logged acre. Unless the wildlife tree violated Washington Department of Labor and Industry requirements, all wildlife reserve trees were retained at modified sites. Wildlife reserve trees are defined as follows:

- **Type 1**: Live tree with defective or deformed sound tops, trunks, and roots.
- **Type 2**: Dead tree with a sound top, trunk, and roots.
- **Type 3**: Live or dead tree with unstable tops or upper portions.
- **Type 4**: Live or dead tree with unstable trunk or roots, with or without bark. This includes "soft" snags as well as live trees with unstable roots caused by root rot or fire. Type 4 reserve trees are the most dangerous.

Table 5-1. Leave-tree requirements per water type and average riparian management zone (RMZ) width for western Washington (WAC 222-30-020 [4c]) using January 1988 or November 1988 rules.

Water type / Avg. width (m)	RMZ Max. width (m)	Ratio of conifers: Deciduous / Size of leave trees	# Trees / 300 m (each side) by bed material	
			Gravel/Cobble	Boulder/Bedrock
1 & 2 / ≥22	30	Representative of stand	50	25
1 & 2 / ≤22	22	Representative of stand	100	50
3 / ≥1.5	15	2:1 / 30 cm or next largest available	75	25
3 / ≤1.5	7.5	1:1 / 15 cm diameter or next largest available	25	25

Study Area

The 18 study sites were distributed throughout western Washington in Cowlitz, Grays Harbor, King, Lewis, Pierce, and Thurston counties (west of the Cascade Mountains on the Olympic Peninsula, in southwestern Washington, the Puget Trough, and on the western slopes and crest, or the Washington Cascades physiographic provinces) (Figure 5-1).

Study sites were established in 1992 and 1993 and were selected to minimize variation in forest age and composition, elevation, moisture condition, and water type. Initially, 18 sites were selected for this study based on the following criteria:

1. Low elevation (<620 m).
2. Second-growth forest (55–66 years old) dominated by Douglas-fir (*Pseudotsuga menziesii*).
3. Type 3 water according to state forest regulations. Type 2, 4, or 5 was selected if streams differed in only the presence of salmonids (Appendix 5-A). Water types are defined in Table 5-2.
4. Predominantly coniferous riparian canopy with deciduous tree component.
5. At least 500 m in stream length.
6. Road access within 500 m.

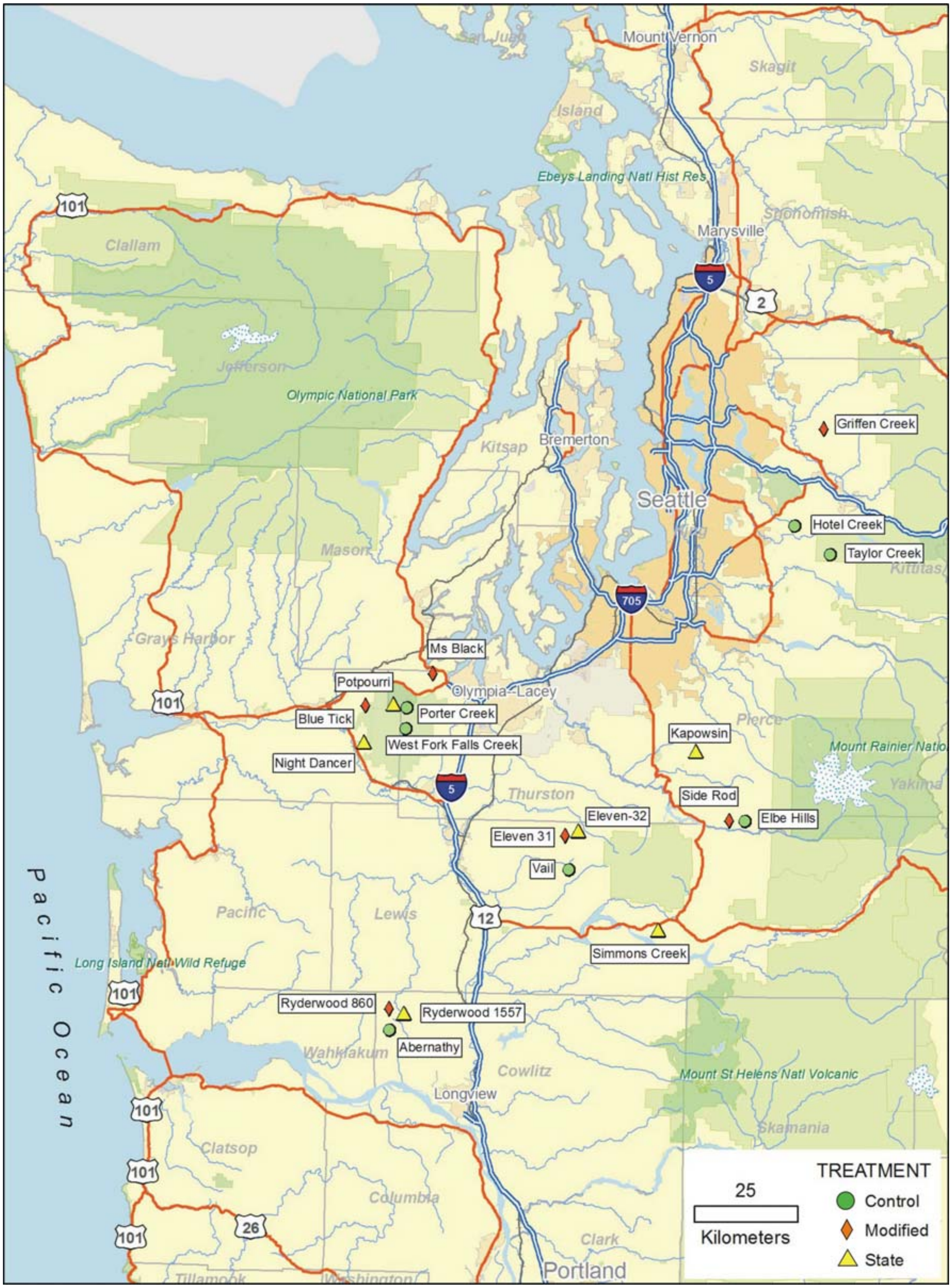

Figure 5-1. Distribution of western Washington study sites. West Fork Falls Creek replaced Vail in the ten-year post-harvest sampling interval.

Table 5-2. Water-typing criteria (WAC 222-16-030) used in Washington in 1988. Source: Washington State Forest Practices Board (1988).

	Water Type				
Parameter	**1**	**2**	**3**	**4**	**5**
Channel Width	N/A	≥ 6 m between OHWM	**Anadromous:** > 1.5m between OHWM Resident Game: > 3 m between OHWM	> 0.6 m between OHWM	> 0.6 m between OHWM
Gradient	N/A	< 4%	**Anadromous:** < 12% not upstream of a falls > 3m in height Resident Game: < 12%	N/A	N/A
Flow	N/A	N/A	**Anadromous:** N/A Resident Game: > 0.3 CFS at summer low flow	N/A	N/A
Impoundment	N/A	Water surface area of < 0.4 ha at seasonal low flow	**Anadromous:** Water surface area of < 0.4 ha at seasonal low flow Resident Game: Water surface area of < 0.2 ha at seasonal low flow	N/A	N/A
Fisheries	N/A	Used by substantial numbers of anadromous or resident game fish for spawning and rearing and migration	Used by significant numbers of anadromous or resident game fish for spawning and rearing and migration	Not used by significant numbers of fish	Not used by significant numbers of fish
Diversion	N/A	Domestic use for > 100 residences or campsites. Includes upstream reach of 450 m or until the drainage area is < 50%, whichever is less	Domestic use for > 10 residences or campsites. Includes upstream reach of 450 m or until the drainage area is < 50%, whichever is less	N/A	N/A
Other	All water within their OHWM inventoried as “Shoreline of the State” excluding related wetlands	Streams flowing through campgrounds available to public having ≥ 30 campsites	Contributes > 20% of flow to Type 1 or 2 water. Anadromous fish impoundments have outlet to stream with anadromous fish	N/A	All natural waters not classified as Type 1, 2, 3 ,or 4 or seepage areas, ponds and drainways having short run-off periods

Terrestrial Amphibian Sampling

Paired transects (Ruggiero et al. 1991) were established in riparian and upland habitats at each site (Figure 5-2). On one side of the stream, pitfall traps (Corn and Bury 1991; Kelsey 2000) were placed in the ground at 15 m intervals, with 18 traps in the riparian habitat and 18 traps in the adjacent upland habitat approximately 100 m away. Traps were placed in the ground such that the top of the trap was level with the ground and the area around each trap (within approximately 30 cm) was cleared of sticks and other debris that could fall into the trap (Figure 5-3). Drift fences were not used in conjunction with the pitfall traps to maintain consistency with sampling methods used at the sites in the pre-harvest and first post-harvest sampling intervals. For many locations, the ground around the trap was built up to create a smooth transition from the ground into the trap. Each trap location was marked using a blue flag stake in the ground and blue flagging tape tied to a tree above the trap. UTM coordinates were obtained for most pitfall traps using a Garmin GPS12 handheld receiver. Riparian traps were established approximately 5 m from the ordinary high-water mark. In certain cases, old pitfall-trap locations could not be found, so new sites and traps were required. New pitfall traps consisted of a single can measuring 35 cm deep with a diameter of 15 cm. Plastic inserts were used in the opening of the cans to reduce amphibian escape from traps (Figure 5-3). When previous upland transects were not found, their position on the landscape was approximated and a new transect was established. In many cases, traces of the previous transect were uncovered when establishing the new transect.

Not all sites were sampled in each year (Table 5-3). For example, 13 of 18 sites were sampled in 1992, while all 18 sites were sampled in 1993, 1995, and 1996. Some time after 1996, one of the control sites was logged, which necessitated the addition of a new unlogged site in 2003. The new site selected fit the criteria used in 1992. Also, in 2003, one of the modified buffer sites (Griffen Creek) received an upland silvicultural treatment (brushing and thinning in the upland) that was not applied to all modified sites. As a result, this site no longer varied through time equally with the other sites and was excluded from the study.

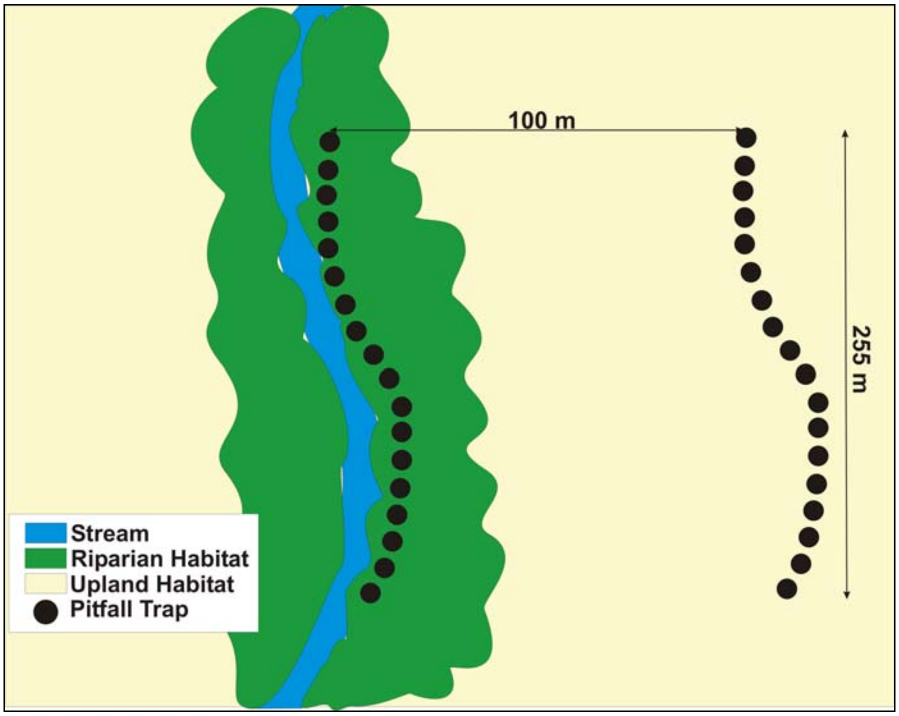

Figure 5-2. Schematic of a typical site set up to sample terrestrial amphibians using pitfall traps.

Table 5-3. Western Washington terrestrial amphibian sampling period for each sampling interval (SI).

SI	Year	Start	End	Days	Sites
1	1992	30-Oct-92	14-Dec-92	45	13
	1993	6-Oct-93	3-Nov-93	28	18
2	1995	14-Oct-95	16-Nov-95	33	18
	1996	13-Oct-96	14-Nov-96	32	18
3	2003	16-Oct-03	1-Dec-03	46	17[a]
	2004	4-Oct-04	16-Nov-04	43	17[a]

[a] The Griffen Creek modified site was not sampled because of modifications to the upland transects that did not occur at all sites

Figure 5-3. Photograph of a typical pitfall tap used at each site in all sampling intervals. The yellow object is the plastic insert used to prevent escapes.

Traps remained open for a period of 28 consecutive nights (per site) and were checked every 2 to 5 days. Approximately 5 cm of water was put into each trap to prevent amphibian desiccation. We also placed a small sponge, a small moss mat, or a small piece of woody debris in each trap so that captured amphibians would have a way to remain out of the water and not drown.

O'Connell et al. (2000) checked pitfall traps approximately weekly, removing all animals from the pitfall traps and the field. Any live amphibians captured were removed to a laboratory refrigerator and retained until the completion of the trapping period, after which they were returned to their original site of capture. Unfortunately, this approach caused mortality rates in excess of 50%, which may have had implications for local populations of certain species. We elected to increase the frequency of trap checking to facilitate the capture of live amphibians. The increased frequency of checking (approx. 2 times per week) may have resulted in a different capture rate for certain species (e.g., Northern Red-legged Frog); however, this number is expected to be negligible and inconsequential to the ensuing analyses. We assumed that certain species (e.g., Western Red-backed Salamander and Ensatina) could not escape from the pitfall traps (because of the plastic inserts); therefore, the increase in checking would not have affected sample size.

For the 10-year sampling interval, we elected to toe-clip amphibians (no more than three toes per individual) and release them near the pitfall trap. Each animal was given a unique clip to ensure that animals were not double-counted. In addition to toe-clipping, we photographed the dorsal side of most amphibians captured, which for certain species can function as a secondary mark; this is especially true for *Plethodon vehiculum* (Davis and Ovaska 2001) and *Rana aurora* (Hawkes, unpublished data). The literature on the utility and effects of toe-clipping is divided. For example, Parris and McCarthy (2001) and McCarthy and Parris (2004) reported on the potential influence of toe-clipping on the return rates of marked amphibians. while other studies (e.g., Hartel and Nemes 2006) attributed low recapture rates to study logistics and not to toe-clipping. Furthermore, Funk et al. (2005), provided a good argument for the use of toe-clipping. Based on the available literature, we assumed that clipping up to three toes per individual would not affect survivorship of clipped individuals, and as this was not a mark-recapture study, the potential for reduced return rates would not compromise the count data. Through toe-clipping and releasing captured animals at the site of capture, we ensured that animals were counted only one time per capture session, which aligns with the work done by O'Connell et al. (2000). Individuals captured in subsequent sampling sessions were also identified, which would not have been possible in O'Connell et al. (2000).

Because it was not possible to identify the direction of travel a particular amphibian had taken prior to falling into a pitfall trap, each amphibian was released approximately 2 m downstream of the trap in which it was captured, typically under or near a suitable cover object. Dead amphibians were measured and assigned to an age class (e.g., adult, juvenile) and preserved in formalin if they had not been partially consumed by small mammals or were not too decomposed.

Vegetation and Habitat Structure Sampling

Vegetation sampling occurred once during each sampling interval and coincided with the primary period of productivity and when most plants would be readily identifiable, which in this case was mid- to late summer. Vegetation data from the pre-treatment and 2-year post treatment sampling intervals were provided by the Washington Department of Natural Resources. Vegetation data for the 10-year post-harvest sampling interval were collected by LGL Limited environmental research associates.

Riparian and upland habitats were sampled at each site within a series of plots: 12 plots were established in the riparian habitat and 10 plots were established in the upland habitat. Each plot consisted of four 10 x 8 m large quadrats (Figure 5-4). Riparian plots extended 8 and 16 m from the ordinary high-water mark, which was usually within 5 m of the stream edge. Upland habitats were sampled in plots approximately 100 m upslope from the riparian transects. Within each large quadrat, we established 2 x 2 m and 1 x 1 m small quadrats for the collection of ground-cover measurements (Figure 5-4). These small quadrats were located 1, 4, 7, and 10 m from the streamside edge of the large quadrats.

At each small quadrat, we estimated the percentage cover of herbaceous and woody vegetation, rock, litter, and bare soil at 1, 4, 7, and 10 m from the streamside edge. The percentage cover of shrubs was estimated in large quadrats 2 and 3 at each plot in the riparian and upland sampling areas. Shrubs were grouped into three categories: (1) berry-producing, (2) evergreen, or (3) other deciduous.

Percentage of downed wood was estimated from large quadrats 2 and 3 at each plot. Wood was considered down if its angle of incidence with the ground was <45°. Each piece of downed wood was categorized by diameter (cm; 2 classes) and decay class (3 classes; Table 5-4).

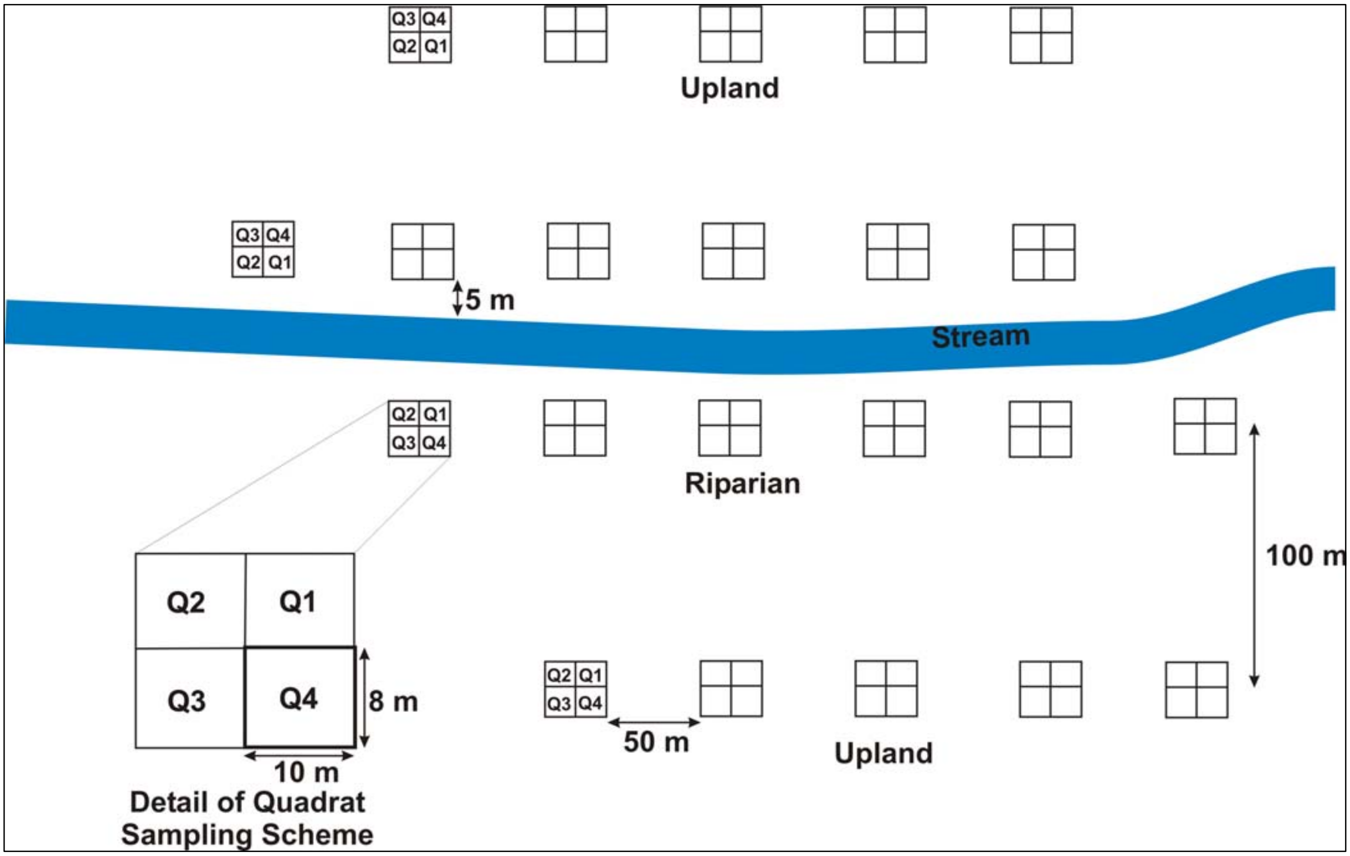

Figure 5-4. Schematic of the vegetation sampling scheme using 10 x 8 m large quadrats situated on each side of the stream at each study site.

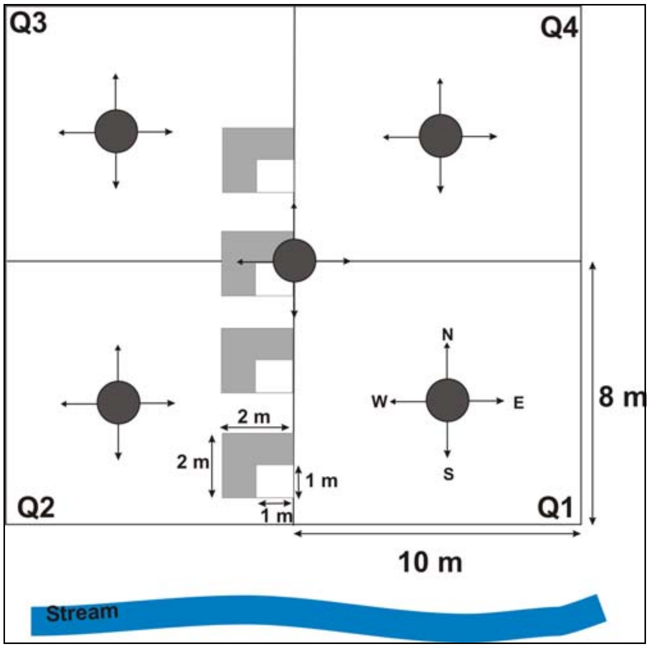

Figure 5-5. Schematic of vegetation sampling design showing the location of the 2 x 2 m and 1 x 1 m small quadrats at 1, 4, 7, and 10 m from the edge of the plots and the locations from which canopy-cover data were collected (black circles).

Table 5-4. Diameter- and decay-class categories used for downed-wood measurements in large quadrats 2 and 3.

Diameter Class (cm)		Decay Class	
1	10–30	1	Structurally sound wood with intact limbs
2	>30	2	Reduced structural integrity and some limb loss
		3	Minimal structural integrity and presence of epiphytes

Snags were counted in all 4 large quadrats and grouped according to quadrat location. Snags were classified as short (<1.5 m), medium (1.5–15 m), and tall (>15 m) in three diameter classes: Class 1 = all limbs attached and structurally sound; Class 2 = losing limbs and showing reduced structural integrity; and Class 3 = about to fall down due to minimal structural integrity.

Percentage cover of sapling trees was estimated from large quadrats 2 and 3. Saplings between 1 and 3 m in height were included regardless of whether they originated from the ground, a stump, or downed wood. Each sapling was identified to species, and we estimated percentage cover for each species encountered in each quadrat. We did not include tree species in our analyses that were <1 m tall.

Trees >3 m in height were counted in all 4 large quadrats and grouped according to quadrat (1, 2, 3, or 4) and diameter size (10 cm, 10–50 cm, 50–100 cm, >100 cm diameter at breast height [DBH]). All trees with split boles were counted as more than one tree (with the exception of vine maple) if the split occurred below breast height. Trees with more than half of the bole outside the quadrat were not counted.

Riparian buffer width was measured from five plots on either side of the stream and was measured from the normal high-water mark to the outermost edge of forest. Slope distance rather than horizontal distance from the outermost tree to the ordinary high-water mark was measured.

To estimate canopy cover we used a spherical, convex densiometer at the outer corners of each quadrat and at the center point where the 4 large quadrats met. At each of the five points, we took four readings: (1) facing the stream, (2) away from the stream, (3) downstream, and (4) upstream (Figure 5-5). We estimated canopy cover from each of the five locations in each plot. Other data recorded at each site included air and soil temperature, relative humidity, slope, and aspect.

Environmental Conditions

Because environmental conditions can affect the surface activity of amphibians, daily weather data for four stations in western Washington (Elma, Longview, Packwood, and Landsburg) were obtained from the Western Regional Climate Center Desert Research Institute, Reno, Nevada, for the period 1 January 1990–31 January 2005. The four stations were selected because they were centrally located among the study sites (Table 5-5). The weather data were obtained to evaluate whether or not environmental conditions could have influenced species activity, thereby affecting detectability and measures of species richness and relative abundance.

Table 5-5. Weather stations in western Washington queried for daily temperature and precipitation data, 1990–2004.

Station	Sites	General Area
Elma 452531	Blue Tick Ms Black Night Dancer Porter Creek Potpourri West Fork Falls Creek	Capitol State Forest
Longview 454769	Abernathy Ryderwood 1557 Ryderwood 860	SW Washington
Packwood 456262	Elbe Hills Eleven 31 Eleven 32 Kapowsin Side Rod Simmons Creek Vail	West-central Washington
Landsburg 454486	Griffin Creek Hotel Creek Taylor Creek	Cedar River Watershed

Statistical Analyses

Statistical analyses were performed using SAS V9.1 (© 2002–2003), R V2.2.1 (© 2005), and Microsoft Excel 2002 (© 1985–2003). Prior to performing all analyses, data were assessed to see if they fit a normal distribution using the Analyst Application in SAS, which produces four tests of normality (Shapiro–Wilk, Kolomogerov–Smirnov, Cramer–von Mises, and Anderson–Darling). Comparisons of relative abundance were done using log-transformed means ($\log[x + 1]$), which approximated a normal distribution. The critical value of alpha was set at 0.1 and beta at 0.8 (*a priori*) for all analyses, which is appropriate for landscape-based studies and when analyzing data obtained from populations that are highly variable.

Species Richness and Community Similarity

We used the number of species found in each habitat type (i.e., riparian or upland) of each site during each sampling interval as a measure of species richness. Species richness data from riparian transects were used to test for differences relative to buffer width. To test whether logging influenced species richness or diversity, we combined data from the buffer treatment groups (i.e., state and modified riparian buffers), because both groups received similar upland treatments (clearcut logging). Prior to pooling these data, we analyzed each treatment in isolation. With no detectable difference with respect to treatment, data from the treatments were pooled. Pooling the treatment data provided a sense of whether or not the retention of riparian habitat, regardless of width, provides for the persistence of terrestrial amphibian species richness.

We used Morisita’s coefficient of similarity (C) as a measure of species diversity because of its superior utility, relative to other measures, in pairwise comparisons of communities (Krebs 1999). Morisita's index (Morisita 1959; Horn 1966; Brower et al. 1990) measures community similarity and is based on Simpson's index of dominance (Simpson 1949). It calculates the probability that specimens randomly drawn from two sites will be of the same species, relative to the probability that specimens randomly drawn from the same site will be of the same species. This index is desirable because sample size and diversities of the samples have little influence on its calculation (Morisita 1959; Wolda 1981). Morisita's index returns a value from 0.0 (no similarity) to 1.0 (identical) and was chosen because it is affected little by large variations in sample size.

Relative Abundance

The catch of individuals per 100 trap nights, adjusted for missed nights or non-functional traps (Nelson and Clark 1973), was used as a measure of relative abundance for each species. Comparisons were made between mean relative abundance measures, transformed as $\log(x + 1)$, to approximate a normal distribution. We tested variation in relative abundance only for the most common species; sample sizes were too small for analyses of other species. Using PROC MIXED, we ran a repeated-measures ANOVA to test hypotheses about variation in relative abundance, followed by orthogonal and non-orthogonal contrasts (Table 5-6) of main effects (treatment, transect, sampling interval) and their interactions. The main experimental units for which measurements were repeatedly collected were the individual transects in riparian and upland habitats; and the data from each transect were pooled to derive a treatment mean for each of the three sampling intervals.

Contrasts were grouped into two sets of questions: differences *among* and differences *within* sampling intervals. In each question set, non-orthogonal contrasts were used to answer questions about the mean relative abundance of the more common species detected. Based on the objective of the overall study, the sampling intervals, treatments, and transects were grouped to best answer the questions of differences between the state and modified buffer treatments relative to the control sites over time. The comparisons derived were based on the following assumptions:

1. All sites selected for this study were similar enough that results derived for one site could be extrapolated across all sites of a particular treatment; as such, pooling of sites into treatments was acceptable.
2. The upland habitats at the state and modified treatments represented the same treatment and thus could be pooled into a post-cut condition, defined by clearcut logging followed by replanting.
3. A comparison of the control uplands with the pooled state and modified uplands (i.e., pre-cut vs. post-cut) would address logging effects (i.e., does logging uplands affect the relative abundance of terrestrial amphibians?).
4. A comparison of the control riparian transects to the pooled state and modified condition would determine if logging affects relative abundance of terrestrial amphibians irrespective of buffer width.
5. A comparison between the modified and state riparian transects would provide a measure of the effect of a wide buffer versus a narrow buffer, and the effect that buffer width has on the relative abundance of terrestrial amphibians.

The contrasts in Table 5-6 consist of 2 question sets. Question set 1 consists of 24 contrasts that compare the relative abundance of terrestrial amphibians across time to determine if there are temporal relationships due to treatment or habitat type (transect). The 18 contrasts in question set 2 evaluate relative abundance within each sampling interval relative to treatment and habitat type. This is particularly important for the pre-treatment sampling interval to establish a baseline of similarity or difference prior to treatment application. Non-orthogonal contrasts ask specific questions and are a compromise between a “fishing expedition” (all possible pairwise *a posteriori* comparisons) and completely orthogonal (and therefore independent) contrasts. The maximum number of pairwise comparisons that could have been made for each species was 153, of which we made 42; 22 of which are orthogonal. The 20 non-orthogonal contrasts are designed to ask pertinent questions of the data to determine if there are measurable treatment effects on the relative abundance of amphibians across time. Specifically, we wanted to know if the relative abundance of terrestrial amphibians had a relationship with one or more treatments and/or habitat types. Contrasts are preferred to multiple ANOVAs with post hoc multiple range tests to control FWE, because contrasts can reduce the number of questions asked of the data (and therefore avoid data mining) and because, in many cases, many of the comparisons of the means are not of interest.

There is little clarity in the literature regarding the “right” way to approach the use of non-orthogonal contrasts. Some authors have indicated that non-orthogonal contrasts are acceptable provided the redundancy is noted. Others have commented that orthogonality is desirable but not essential, because a reasonable approximation of the probabilities exist for non-orthogonal contrasts. Miller (1981) indicated that because multiple comparisons are being made (regardless of orthogonality) the results should be corrected for experiment-wise error using a correction factor such as Scheffé, Bonferroni, or Sidak. Miller (1981) discusses methods for pairwise comparison that can be adapted for general contrasts, suggesting that Bonferroni be used for additive inequality and Sidak for multiplicative inequality and that these results need to be corrected when the number of comparisons is large. The term “large” is vague and conveys nothing about when to account for experiment-wise error when using contrasts analyses in a repeated-measures ANOVA. We did not consider the number of comparisons to be large relative to the number of comparisons that were possible for each species. In addition, because we were analyzing a temporal data set that included three sampling intervals with 17 contrasts per sampling interval, we could have generated up to 51 orthogonal contrasts. Based on the ambiguity in the literature, we elected not to adjust the critical value of alpha.

Table 5-6. Orthogonal and non-orthogonal contrasts used to identify differences in relative abundance (RA) among and within sampling intervals (SI) relative to treatment (control, modified, state) and transect (riparian, upland). SI 1 = 1992/1993; SI 2 = 1995/1996; SI 3 = 2003/2004.

	Sampling Interval, Transect, and Treatment Type	Question
Among	CONTROL RIPARIAN - SI 1 vs. SI 2	Is RA different in SI 1 compared to SI 2 for control riparian transects?
	CONTROL RIPARIAN - SI 1 vs. SI 3	Is RA different in SI 1 compared to SI 3 for control riparian transects?
	CONTROL RIPARIAN - SI 2 vs. SI 3	Is RA different in SI 2 compared to SI 3 for control riparian transects?
	CONTROL RIPARIAN - pre-cut vs. post-cut	Is RA pre-harvest different from post-harvest for control riparian transects?
	MODIFIED RIPARIAN - SI 1 vs. SI 2	Is RA different in SI 2 compared to SI 3 for modified riparian transects?
	MODIFIED RIPARIAN - SI 1 vs. SI 3	Is RA different in SI 1 compared to SI 2 for modified riparian transects?
	MODIFIED RIPARIAN - SI 2 vs. SI 3	Is RA different in SI 1 compared to SI 3 for modified riparian transects?
	MODIFIED RIPARIAN - pre-cut vs. post-cut	Is RA pre-harvest different from post-harvest for modified riparian transects?
	STATE RIPARIAN - SI 1 vs. SI 2	Is RA different in SI 1 compared to SI 2 for state riparian transects?
	STATE RIPARIAN - SI 1 vs. SI 3	Is RA different in SI 1 compared to SI 3 for state riparian transects?
	STATE RIPARIAN - SI 2 vs. SI 3	Is RA different in SI 2 compared to SI 3 for state riparian transects?
	STATE RIPARIAN - pre-cut vs. post-cut	Is RA pre-harvest different from post-harvest for state riparian transects?
	CONTROL UPLAND - SI 1 vs. SI 2	Is RA different in SI 1 compared to SI 2 for control upland transects?
	CONTROL UPLAND - SI 1 vs. SI 3	Is RA different in SI 1 compared to SI 3 for control upland transects?
	CONTROL UPLAND - SI 2 vs. SI 3	Is RA different in SI 2 compared to SI 3 for control upland transects?
	CONTROL UPLAND - pre-cut vs. post-cut	Is RA pre-harvest different from post-harvest for control upland transects?
	MODIFIED UPLAND - SI 1 vs. SI 2	Is RA different in SI 1 compared to SI 2 for modified upland transects?
	MODIFIED UPLAND - SI 1 vs. SI 3	Is RA different in SI 1 compared to SI 3 for modified upland transects?
	MODIFIED UPLAND - SI 2 vs. SI 3	Is RA different in SI 2 compared to SI 3 for modified upland transects?
	MODIFIED UPLAND - pre-cut vs. post-cut	Is RA pre-harvest different from post-harvest for modified upland transects?
	STATE UPLAND - SI 1 vs. SI 2	Is RA different in SI 1 compared to SI 2 for state upland transects?
	STATE UPLAND - SI 1 vs. SI 3	Is RA different in SI 1 compared to SI 3 for state upland transects?
	STATE UPLAND - SI 2 vs. SI 3	Is RA different in SI 2 compared to SI 3 for state upland transects?
	STATE UPLAND - pre-cut vs. post-cut	Is RA pre-harvest different from post-harvest for state upland transects?
Within	SI 1 RIPARIAN - Control vs. cut	In SI 1, was riparian RA at the controls different from the treatments?
	SI 1 RIPARIAN - Control vs. State	In SI 1, was riparian RA different between the control and state?
	SI 1 RIPARIAN - Modified vs. State	In SI 1, was riparian RA different between the modified and state?
	SI 1 RIPARIAN - Control vs. Modified	In SI 1 was riparian RA different between the modified and control?
	SI 2 RIPARIAN - Control vs. cut	In SI 2, was riparian RA at the controls different from the treatments?
	SI 2 RIPARIAN - Control vs. State	In SI 2, was riparian RA different between the control and state?
	SI 2 RIPARIAN - Modified vs. State	In SI 2, was riparian RA different between the modified and state?
	SI 2 RIPARIAN - Control vs. Modified	In SI 2, was riparian RA different between the controls and modified?
	SI 3 RIPARIAN - Control vs. cut	In SI 3, was riparian RA at the controls different from the treatments?
	SI 3 RIPARIAN - Control vs. State	In SI 3, was riparian RA different between the control and state?
	SI 3 RIPARIAN - Modified vs. State	In SI 3, was riparian RA different between the modified and state?
	SI 3 RIPARIAN - Control vs. Modified	In SI 3, was riparian RA different between the controls and modified?
	SI 1 UPLAND - Control vs. cut	In SI 1, was upland RA at the controls different from the treatments?
	SI 1 UPLAND - Modified vs. State	In SI 1, was upland RA different between the modified and state?
	SI 2 UPLAND - Control vs. cut	In SI 2, was upland RA at the controls different from the treatments?
	SI 2 UPLAND - Modified vs. State	In SI 2, was upland RA different between the modified and state?
	SI 3 UPLAND - Control vs. cut	In SI 3, was upland RA at the controls different from the treatments?
	SI 3 UPLAND - Modified vs. State	In SI 3, was upland RA different between the modified and state?

Co-inertia Analysis

The program R V2.2.1 (© 2005) was used to investigate habitat relationships between abundant or common species and the habitat attributes measured at each site and treatment. For this study, a co-inertia analyses (COIA) was used and was based on the ade4 package (Chessel et al. 2005). COIA is a multivariate statistical technique that provides for the visualization of the structure of ecological data via ordination. COIA is very general and some existing methods appear as special cases of it (Dray et al. 2003). For example, interbattery analysis (Tucker 1958) is mathematically equivalent to a simultaneous principle components analysis (PCA) on two ecological tables followed by a COIA (i.e., PCA–PCA COIA). When table X contains qualitative variables and Y contains species numbers, it is usual to cross tables X and Y to obtain a matrix containing the distribution of species among the categories of environmental variables. A simple canonical analysis (CA) of this new table allows one to ordinate the species and the environmental classes (analysis of ecological profiles; Montana and Greig-Smith 1990). Binary discriminant analysis (Strahler 1978), which has been used in ecology (Del Moral 1982; Huang and Del Moral 1988), is mathematically equivalent to Romane's CA (Dray et al. 2003). Although this approach allows plotting of species and environmental classes (Ben-Shahar 1987; Ben-Shahar and Skinner 1988), no information about ordination of sites is available.

COIA is similar to other ecological ordination techniques, such as a canonical correspondence analysis (CCA; ter Braak 1986) and redundancy analysis (RDA; Wollenberg 1977); however, COIA is a general coupling method that maximizes the covariance between the variables of two tables. Separate tables X and Y can be analyzed by various methods, leading to different coupling methods. This approach aims to find a site score that is a linear combination of environmental variables maximizing the variance of species centroids (i.e., separation of species niches). Unlike CCA, COIA does not have a regression step (Thiolouse et al. 2004). This difference has important implications for ecological data analyses: the regression step implies that in CCA and RDA the number of samples must be high compared to the number of environmental variables; COIA does not have this constraint. CCA also has an additional constraint (the total variance must be equal to 1) and must be avoided in the case of numerous environmental variables, which is common in biological studies and is the case in this study.

COIA is a paired ecological table analysis that relies on the covariance matrix produced from (in this case) a simultaneous PCA of the environmental and species data (Figure 5-6). Each table is related through a common number of rows, which in this case represents the number of sites. The output of the PCA can be visualized as two different representations of the sites in space: one relating to the relative abundance of species, the other to the habitat attributes. The covariance matrices produced by the PCA are then joined via COIA so that the relationship between species relative abundance and habitat data can be viewed. The resulting biplot allows for visualization of how the two initially unrelated data sets covary at the same sites. From the biplot, a description of the habitat variables can be derived for species of interest. The ability to quickly assign habitat relationships based on the relative position of species to habitat variables provides an effective way to describe the habitat relationships for each species in each sampling interval and for each treatment.

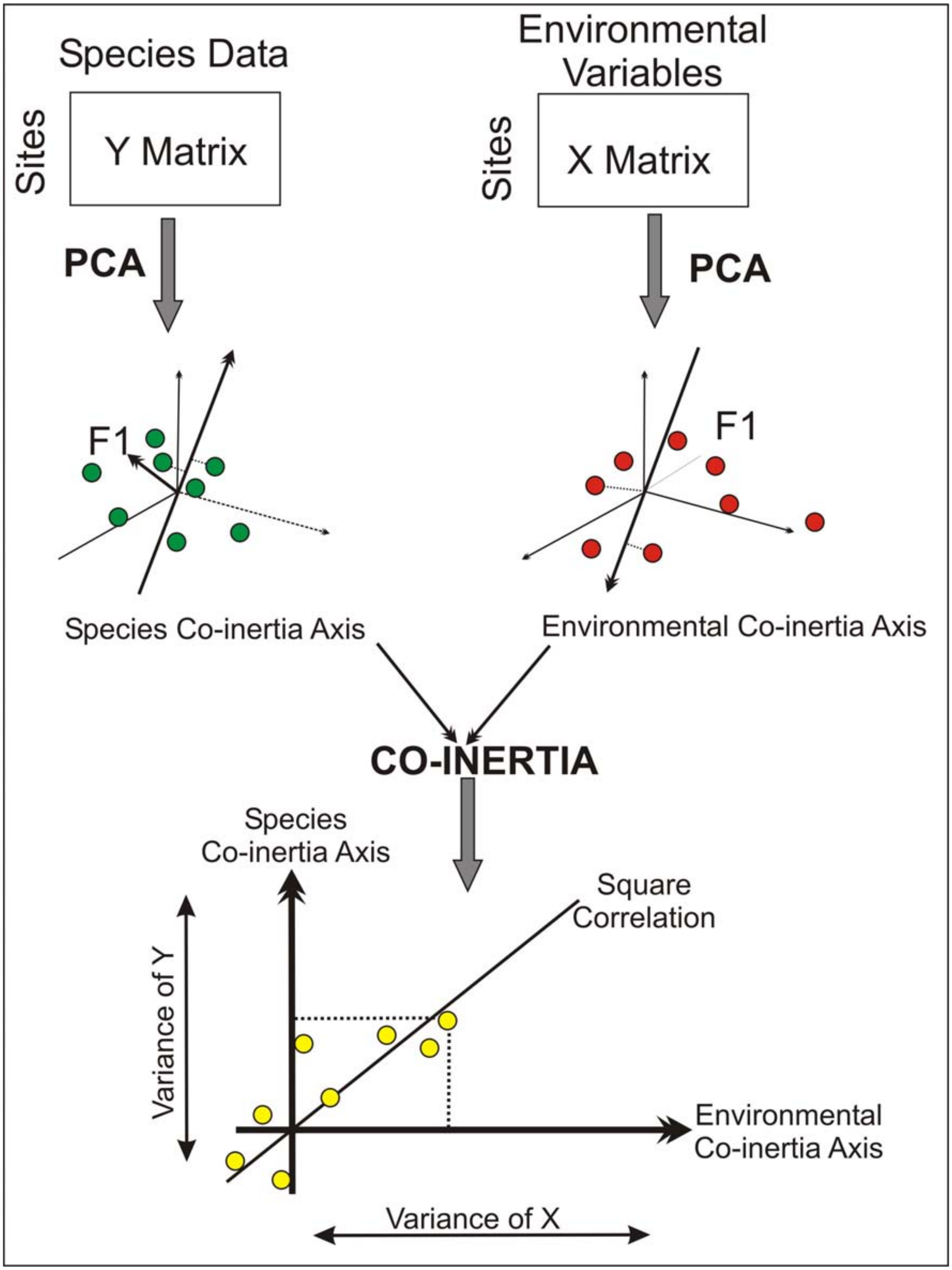

Figure 5-6. Schematic showing the principles of co-inertia analysis. The two ecological data tables X and Y produce two representations of the sites in two hyperspaces. Separate analyses find axes maximizing inertia (=variation) in space (F1 [first factorial axis]). Co-inertia analysis maximizes the square covariance between the projections of the sites on the co-inertia axes. Source: modified from Dray et al. (200).

The relationship between species relative abundance and environmental variables can be viewed on a biplot. One approach is to view the species relative abundance and environmental variable biplots side by side to look for relationships. Alternatively, and as we have done, the species relative abundance biplot can be superimposed onto the environmental variable biplot for easier interpretation. When this is done, an environmental gradient can be described for each axis, with the horizontal axis (axis 1) representing the primary environmental gradient and the vertical axis (axis 2) describing a secondary gradient. Species that occur closer to the origin are indicative of (a) species with too few detections to infer habitat associations, (b) habitat generalists, or (c) species that respond to intermediate conditions along the gradients described by the axes. Species that are plotted farther away from the origin can be regarded as correlated with a particular habitat variable or suite of variables. The length of the vector from the origin to the location of the habitat variable in space is indicative of the strength of the correlation between the habitat variable and the gradient described for each axis. Unlike the more commonly used multivariate analyses, such as CCA (ter Braak 1986), for which the number of environmental variables is limited to the number of sites minus one, COIA has no limitations on the number of environmental variables that can be used regardless of the number of sites sampled (Thioulouse et al. 2004).

A typical biplot resulting from a COIA in shown in Figure 5-7. In this example, the COIA combined information on amphibian relative abundance and habitat variables measured along upland habitats at the control and logged treatments 10 years after logging. The biplots provide an opportunity to compare how the relative abundance of selected species is associated with habitat attributes and how the relative abundance of those species varies relative to treatment. The correlation of each environmental variable with axes 1 and 2 is indicated by its position relative to each axis, with variables closer to axis 1 or 2 more strongly correlated with the particular axis; variables occupying space between are correlated with both axes. Vector length is directly proportional to the degree of correlation: positive correlation to the right of the origin for axis 1 and above the origin for axis 2; negative correlation to the left of the origin along axis 1 and below the origin along axis 2. Species farther from the origin are more abundant in environments that are dominated by particular variables.

For example in Figure 5-7, Coastal Tailed Frog (ASTR) was correlated with large Douglas-fir in upland habitats at the control site, and its abundance was low and not correlated with any particular habitat features at logged sites. At logged sites, Western Red-backed Salamander (PLVE), Ensatina (ENES), or Coastal Tailed Frog were not directly affiliated with the environmental gradients described by axis 1 or 2. The relative abundance of Western Red-backed Salamander and Ensatina was higher when the upland habitats contained large conifers with abundant coarse woody debris and increasing rock availability. Coastal Tailed Frog was more abundant when large Douglas-fir persisted in the upland habitats. In this study, Coastal Tailed Frog was not captured in the uplands of any site after the site was logged. However, it continued to be captured in the upland habitats at the unlogged control sites, suggesting that the relative abundance of Coastal Tailed Frog is negatively affected by logging. Western Red-backed Salamander was abundant in upland habitats and associated with coarse woody debris; and Ensatina was abundant, with more plasticity in terms of habitat selection in unlogged and logged upland habitats.

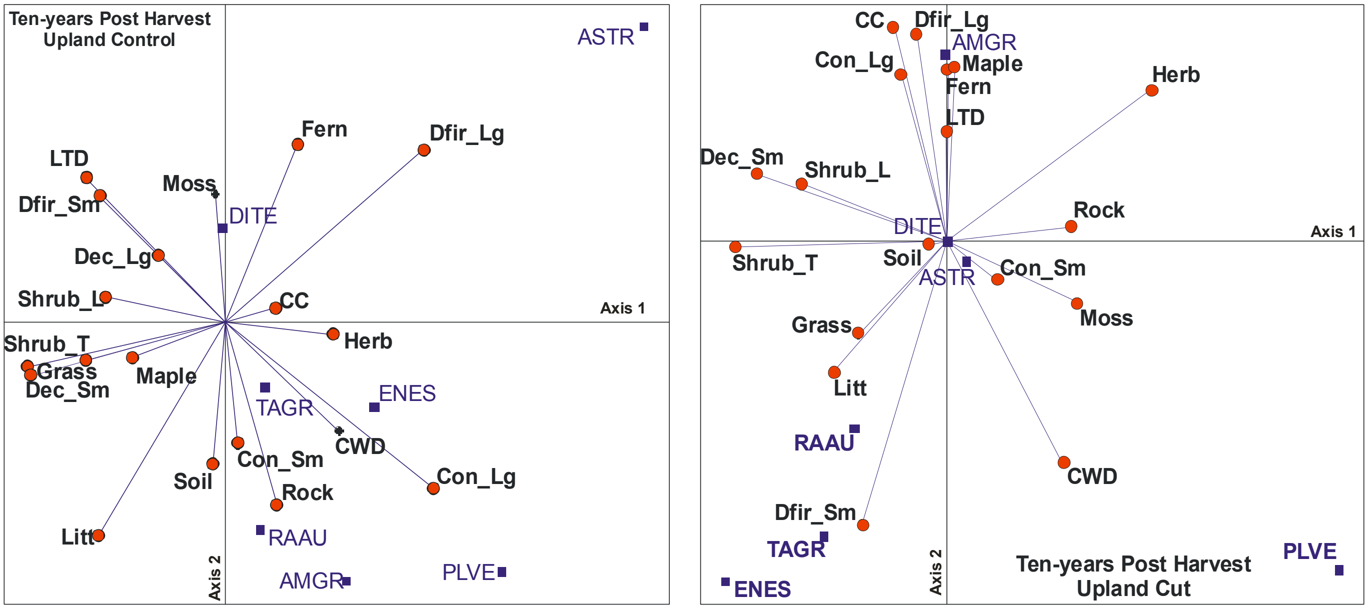

Figure 5-7. Typical biplots of amphibian–habitat relationships in upland habitats at the control treatment relative to logged sites (modified and state) ten-years post-logging. Shrub_T = tall shrubs; Shrub_L = low shrubs; CC = overstory canopy cover; CWD = coarse woody debris; Fern = fern cover; Grass = grass cover; Herb = herbaceous cover; Litt = deciduous leaf-litter cover; LTD = deciduous leaf-litter depth; Moss = moss cover; Rock = rock cover; Soil = exposed soil; Maple = vine maple; Dec_Sm = bigleaf maple and red alder <1.3 m; Dec_Lg = bigleaf maple and red alder >1.3 m; Con_Sm = western redcedar and western hemlock <1.3 m; Con_Lg = western redcedar and western hemlock >1.3 m; Dfir_Sm = Douglas-fir <1.3 m; Dfir_Lg = Douglas-fir >1.3 m. AMGR = *Ambystoma gracile*; ASTR = *Ascaphus truei*; DITE = *Dicamptodon tenebrosus*; ENES = *Ensatina eschscholtzii*; PLVE = *Plethodon vehiculum*; RAAU = *Rana aurora*; TAGR = *Taricha granulosa*.

Results

Prior to completing all statistical analyses, the data were partitioned in two ways: (1) only those sites sampled in all three sampling intervals; and (2) all sites sampled, including sites used as replacement sites, in all three sampling intervals. The results of all analyses were consistent between both data sets, and the results reported below are based on the analyses that included all sites sampled. We assumed that the inclusion of West Fork Falls Creek would not introduce bias into the data, and the two parallel analyses verified that assumption. The regional scale of this study lends itself to this type of site-level replacement, because all sites within the region are considered part of the available sampling population. Given that the replacement site was selected using the same criteria as the original sites, it is representative of the region in which sampling occurred.

Field Sampling

A total of 100,140 trap nights (riparian = 50,066; upland = 50,074) were amassed for all sites and sampling intervals. The highest trap-night total was obtained at the state buffer sites (n = 34,668), and the 2-year post-logging sampling interval had a higher number of trap nights (n = 36,288) than both the pre-harvest and 10-year post harvest intervals. Overall, the number of trap nights per treatment and transect did not vary markedly across time, with the exception of the modified treatment in the 10-year post-treatment sampling interval (Table 5-7) because of the removal of Griffen Creek from the study. The remainder of the trap-night variation was largely related to the number of non-functional traps encountered at each site within each sampling interval. The number of site visits increased in the 10-year post-harvest sampling interval (2003/2004) because of the desire to reduce trap mortalities.

Sampling intensity was measured as the number of pitfall-trap nights for each site and treatment (Table 5-8). Site- and trap-specific data were not available for either the pre-treatment or 2-year post-treatment sampling periods (only the total number of trap nights was available). The number of nights pitfall traps were non-functional (i.e., not able to trap animals) during the 10-year post-treatment sampling interval was assessed. A correction factor was derived and was applied to the number of pitfall-trap nights calculated for the pre-harvest and 2-year post-harvest sampling intervals. In most cases, the adjustment to the number of pitfall-trap nights was minimal or nil. For all pre-treatment sites and 2-year post-treatment control sites, the average trap nights calculated for the 10-year post-treatment sampling interval control sites were used. For modified and state treatments in the 2-year post-treatment interval, the average number of trap nights obtained for each treatment in the 10-year post-treatment interval was applied. Values in Table 5-8 were used in all subsequent analyses.

Table 5-7. Total trap nights by treatment, transect, and sampling interval. SI 1 = 1992/1993; SI 2 = 1995/1996; SI 3 = 2003/2004; Rip = riparian; Upl = upland.

	SI 1			SI 2			SI 3			Totals		
Treatment	Rip	Upl	Total	Rip	Upl	Total	Rip	Upl	Total	Rip	Upl	Tot
Control	5040	5040	10080	6048	6048	12096	5802	5825	11627	16890	16913	33803
Modified	5040	5040	10080	6048	6048	12096	4754	4739	9493	15842	15827	31669
State	5544	5544	11088	6048	6048	12096	5742	5742	11484	17334	17334	34668
	15624	15624	31248	18144	18144	36288	16298	16306	32604	50066	50074	100140

Table 5-8. Pitfall-trap nights calculated for each site during each sampling interval. Values for SI 1 and SI 2 are based on averages of values obtained during SI 3. SI 1 = 1992/1993; SI 2 = 1995/1996; SI 3 = 2003/2004; Rip = riparian; Upl = upland.

		SI 1				SI 2				SI 3			
	Pitfall Traps	1992		1993		1995		1996		2003		2004	
Treatment	Site Name	Rip	Upl	Rip	Upl	Rip	Upl	Rip	Upl	Rip	Upl	Rip	Upl
Control	Abernathy			504	504	504	504	504	504	486	486	486	486
Control	Elbe Hills			504	504	504	504	504	504	486	486	462	465
Control	Hotel Creek	504	504	504	504	504	504	504	504	486	486	468	468
Control	Porter Creek	504	504	504	504	504	504	504	504	468	468	486	486
Control	Taylor Creek	504	504	504	504	504	504	504	504	486	482	468	468
Control	Vail[a]	504	504	504	504	504	504	504	504				
Control	West Fork Falls Creek[b]									558	558	462	486
Modified	Blue Tick			504	504	504	504	504	504	486	461	486	423
Modified	Eleven 31	504	504	504	504	504	504	504	504	432	432	468	468
Modified	Griffen Creek[c]	504	504	504	504	504	504	504	504				
Modified	Ms Black	504	504	504	504	504	504	504	504	524	570	486	513
Modified	Ryderwood 860	504	504	504	504	504	504	504	504	432	432	486	486
Modified	Side Rod			504	504	504	504	504	504	486	486	468	468

		SI 1				SI 2				SI 3			
	Pitfall Traps	1992		1993		1995		1996		2003		2004	
State	Eleven 32	504	504	504	504	504	504	504	504	432	432	468	468
State	Kapowsin	504	504	504	504	504	504	504	504	504	504	468	468
State	Night Dancer			504	504	504	504	504	504	486	486	486	486
State	Potpourri	504	504	504	504	504	504	504	504	468	468	486	486
State	Ryderwood 1557	504	504	504	504	504	504	504	504	486	486	486	486
State	Simmons Creek	504	504	504	504	504	504	504	504	504	504	468	468

[a] Vail control site was logged after SI 2
[b] West Fork Falls Creek replaced Vail
[c] Griffen Creek dropped from study in summer 2003 due to silvicultural practices that occurred there but nowhere else

Weather Conditions

Temperature and precipitation data were plotted for the months of October and November (pooled) for each year of each sampling period to determine if environmental conditions were similar (Figure 5-8; Figure 5-9). Mean temperature varied among years and stations; however, the variation was not enough to affect surface activities of amphibians and was within the range of temperature conditions considered suitable for amphibian sampling in the Pacific Northwest (Blaustein et al. 1995). The relative frequency of rain days (i.e., days with >1 mm of rain) did not differ among years or weather stations ($x_{15}^2 = 9.91$; $p = 0.83$).

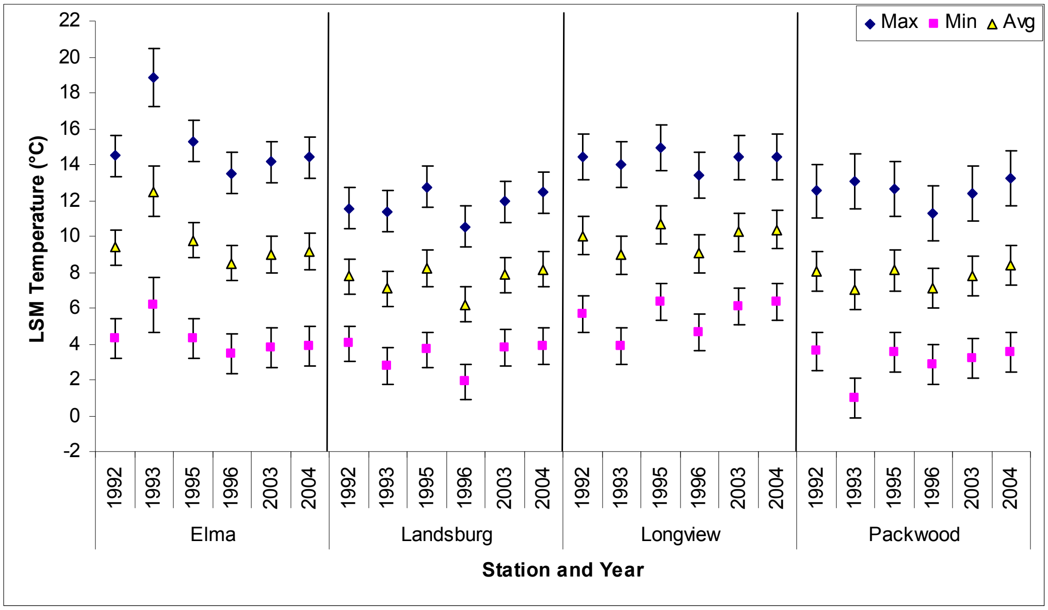

Figure 5-8. Maximum, minimum, and average temperatures (±95% CI) for the period 1 October–30 November for each year of amphibian sampling in western Washington.

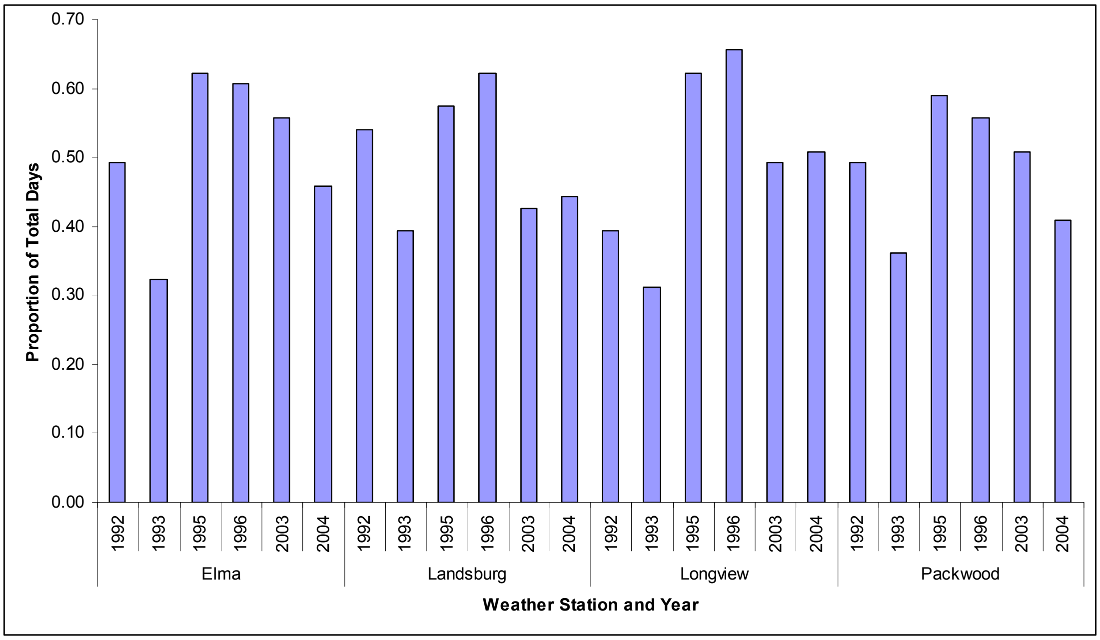

Figure 5-9. Proportion of rain days (i.e., >1 mm rain) for the period 1 October–30 November for each year of amphibian sampling in western Washington. Only data from October were available for Elma in 1993.

Species Richness

Fourteen species of amphibians were documented across the study sites and sampling intervals (Table 5-9). Species richness did not differ relative to treatment, habitat type, and sampling interval ($F_{1,4} = 0.25$; $p = 0.64$) (Figure 5-10). Certain species (e.g., *Bufo boreas* and *Rana cascadae*) were detected in only one sampling interval while other species (e.g., *Plethodon vehiculum* and *Ensatina eschscholtzii*) were detected in every sampling interval and on all transects at all sites (Table 5-9). In addition, no consistent patterns of change in species richness were observed over all three sampling intervals (Figure 5-10).

Table 5-9. Amphibian species detected per sampling interval, treatment, and transect. R = riparian; U = upland; T = treatment; P = presence; blank = not detected.

	Pre-harvest									Two-years Post-harvest									Ten-years Post-harvest								
	Control			Modified			State			Control			Modified			State			Control			Modified			State		
Species	**R**	**U**	**T**	**R**	**U**	**T**	**R**	**U**	**T**	**R**	**U**	**T**	**R**	**U**	**T**	**R**	**U**	**T**	**R**	**U**	**T**	**R**	**U**	**T**	**R**	**U**	**T**
Ascaphus truei	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P
Bufo boreas																										P	P
Hyla regilla					P	P				P		P					P	P				P		P			
Rana aurora	P		P	P	P	P	P	P	P	P	P	P	P	P	P	P		P	P	P	P	P	P	P	P	P	P
Rana cascadae										P		P															
Ambystoma gracile	P	P	P	P	P	P	P	P	P	P	P	P	P		P	P	P	P	P	P	P	P	P	P	P	P	P
Ambystoma macrodactylum	P		P	P	P	P							P	P	P							P	P	P			
Dicamptodon tenebrosus	P	P	P	P	P	P	P	P	P	P	P	P	P		P	P	P	P	P	P	P	P		P	P		P
Ensatina eschscholtzii	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P
Plethodon dunni	P		P	P		P	P		P	P	P	P				P		P		P	P						
Plethodon vehiculum	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P
Rhyacotriton cascadae				P		P											P	P				P		P			
Rhyacotriton kezeri	P		P		P	P					P	P	P		P				P	P	P	P	P	P	P	P	P
Taricha granulosa				P	P	P	P	P	P	P	P	P		P	P		P	P	P	P	P	P	P	P	P		P
Species Richness	**9**	**5**	**9**	**10**	**10**	**12**	**8**	**7**	**8**	**10**	**9**	**11**	**8**	**6**	**9**	**7**	**8**	**10**	**8**	**9**	**9**	**11**	**8**	**11**	**8**	**7**	**9**

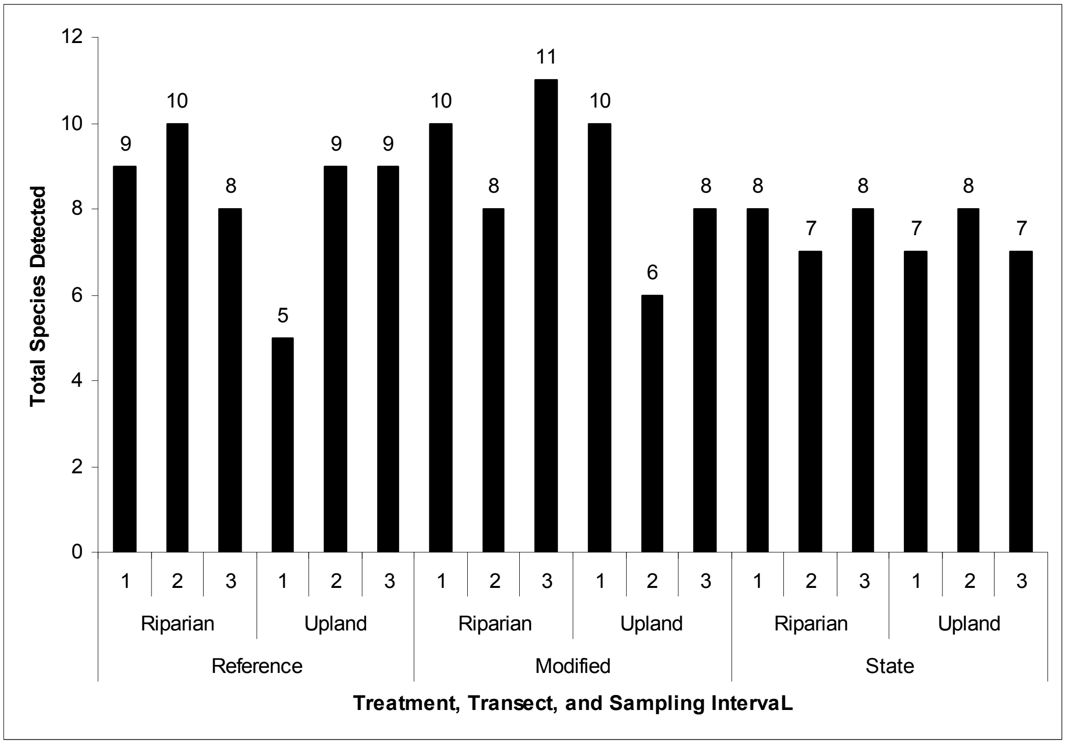

Figure 5-10. Total species detected per treatment, transect, and sampling interval. 1 = 1992/1993; 2 = 1995/1996; 3 = 2003/2004.

During the pre-treatment sampling interval (1992/1993), 12 species of amphibians were captured at all sites (n = 18); 11 species were captured on riparian transects, and 10 species were captured on upland transects (Table 5-9). Mean number of species (±SE) did not differ significantly between riparian (4.1 ± 0.35) and upland habitats (all sites combined) (3.4 ± 0.35; $F_{1,34} = 2.16$; $p = 0.1513$), nor did they differ significantly within assigned treatment types on riparian or upland transects (Figure 5-11). The mean number of species (riparian and upland data combined) at the modified buffer sites (4.5 ± 0.4) was significantly higher than at the control (3.7 ± 0.4) and state buffer sites (3.1 ± 0.4; $F_{2,33} = 3.02$; $p = 0.06$) (Figure 5-11). The modified treatment had the highest number of species on both the riparian and upland transects (Figure 5-11).

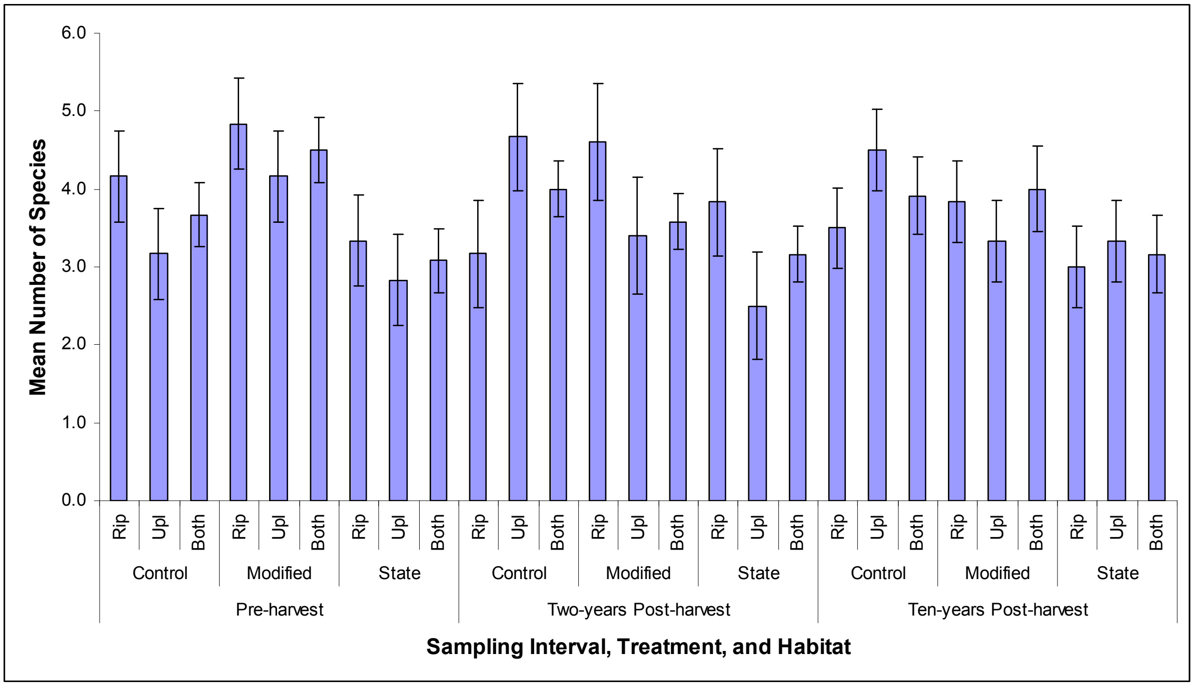

Figure 5-11. Mean number (±SE) of species by sampling interval, treatment, and habitat type. Rip = riparian; Upl = upland.

During the 2-year post-treatment sampling interval (1995/1996), 13 amphibian species were detected (riparian, n = 12; upland, n = 12) (all sites combined; Table 5-9). The number of species detected on riparian transects ranged from 7 at state sites to 10 at the control sites; 11 species were documented on riparian transects at modified sites (Figure 5-10). The number of species detected per treatment and transect did not vary among treatments ($F_{2,33}$ = 1.31; p = 0.28). Similar to the pre-harvest period, the modified buffer sites had the highest number of species (on average) on riparian transects, with the highest number of species detected on upland transects at the control sites (Figure 5-11).

The 10-year post-harvest data followed a trend similar to the 2-year post-harvest data, with the average number of species detected per site within each sampling interval not varying significantly relative to transect ($F_{2,31}$ = 0.8, p = 0.46; Figure 5-11). On riparian transects, the highest average number of species (3.8 ± 0.51; Figure 5-11) occurred at the modified treatment. In upland habitats, the control group had the highest average number of species (4.5 ± 0.51; Figure 5-11).

The most obvious trend noted across time for all treatments was that the riparian habitats at the modified buffer sites always had the highest mean number of species; and, for two of the three sampling intervals, the highest mean number of species in upland habitats was documented at the control sites.

Community and Diversity Comparisons

Community similarity (Morisita's *C*) remained relatively constant in riparian habitats at the control sites across time, with similar amphibian communities before and after logging (Figure 5-12). Similarity between riparian habitats in the 2- and 10-year post-harvest sampling intervals was also high (*C* = 0.82), providing an indication of community stability over time (Figure 5-13). The slight reduction in similarity noted for the 2- and 10-year post-harvest amphibian communities is attributable to the non-detection of Pacific Treefrog, Dunn's Salamander, and Cascade Frog in the 10-year post-harvest sampling interval. The number of Ensatina and Western Red-backed Salamander was lower in the 10-year post-harvest sampling interval, which would have influenced the results. One species, Columbia Torrent Salamander, was not detected 2 years after logging, with only two captures 10 years after harvest.

Contrasting community similarity in riparian and upland habitats at the modified treatment across time revealed that amphibian communities 2 and 10 years after logging were similar to pre-harvest conditions. Conversely, comparing the two post-harvest sampling periods revealed that the communities had changed slightly in both riparian (*C* = 0.69) and upland habitats (*C* = 0.72). The changes are again attributable to differences in total captures of certain species (e.g., Western Red-backed Salamander, Ensatina, and Northern Red-legged Frog), or the detection of species in one sampling interval but not the other. The amphibian communities sampled during the 2-year and 10-year post-harvest sampling periods on both riparian and upland transects were similar to the pre-harvest interval; the community sampled in 1995/1996 was equal to that sampled in 2003/2004 (Figure 5-12).

Although there was little change in community similarity among sampling intervals for riparian and upland habitats, there were some notable changes at the treatment level within and across time (Figure 5-13). First, amphibian communities on riparian and upland transects at the state treatment were the least similar and were less abundant than at the modified and control treatments prior to logging. Second, amphibian communities on riparian and upland transects became more similar at all treatments 2 years after logging. Third, the amphibian communities at the control and modified buffer sites became more dissimilar 10 years after logging relative to the pre-harvest and 2-year post-logging intervals, with the largest change observed for the control sites. Last, the 10-year post-harvest amphibian communities of the state buffer sites were equal to the 2-year post-harvest period and more similar than the pre-harvest communities (Figure 5-13). Pre-logging and immediate post-logging (1995/1996) community similarity between riparian and upland habitats was highest for the modified buffer sites; similarity was highest for the state buffer sites during the 10-year post-harvest period. Amphibian communities in riparian and upland habitats at the control sites had the lowest similarity coefficient in the third sampling interval.

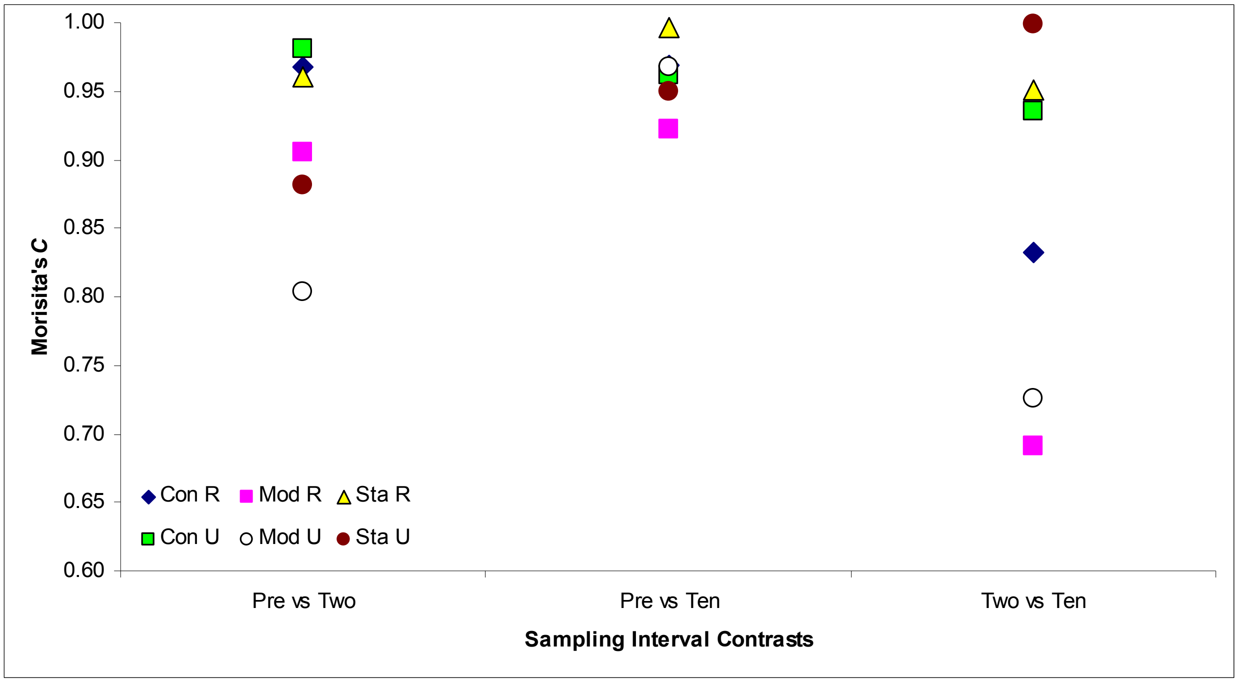

Figure 5-12. Amphibian community similarity coefficients (Morisita's *C*) for each transect contrasted against sampling interval. Pre = pre-harvest (1992/1993); Two = two-years post-harvest (1995/1996); Ten = ten-years post-harvest (2003/2004). Con = control; Mod = modified; Sta = state; R = riparian; U = upland.

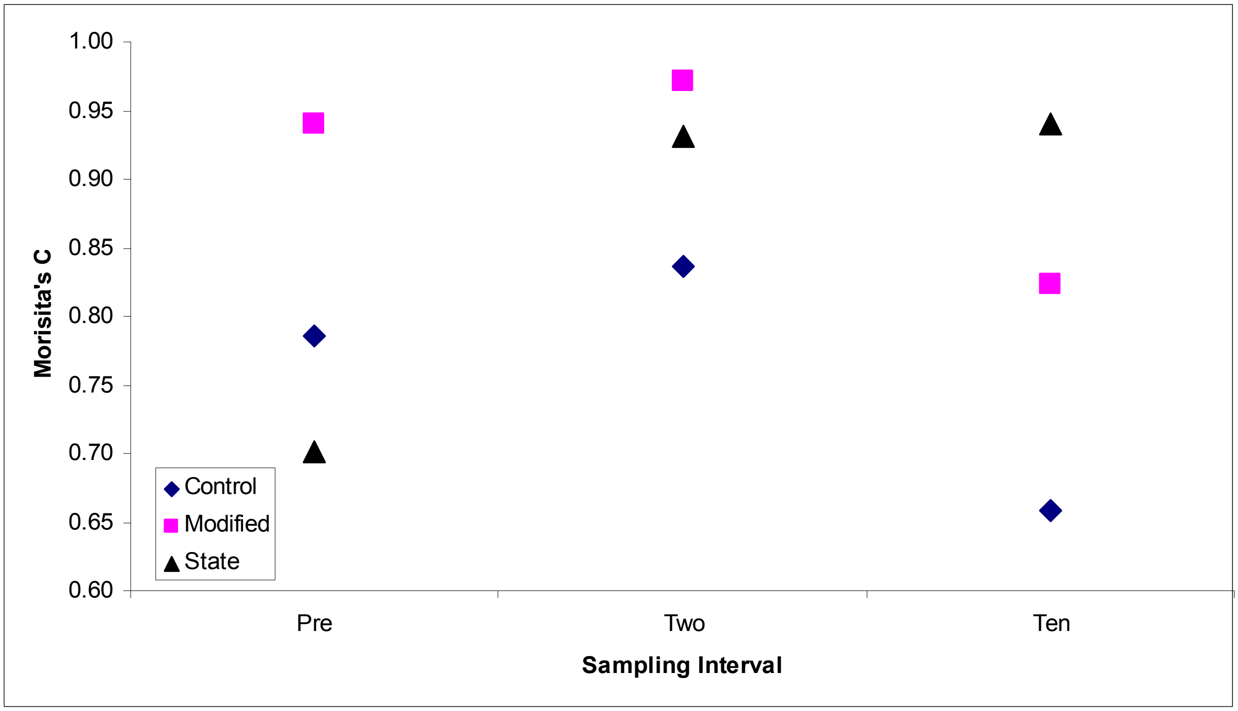

Figure 5-13. Amphibian community similarity coefficients (Morisita's *C*) for comparisons of riparian and upland communities for each treatment within each sampling interval. Pre = pre-harvest (1992/1993); Two = two-years post-harvest (1995/1996); Ten = ten-years post-harvest (2003/2004).

Relative Abundance

The total catch for the pre-harvest and the 2- and 10-year post-harvest periods was 605, 869, and 409 individuals, respectively. In each sampling period, the total catch was dominated by Western Red-backed Salamander (*Plethodon vehiculum*), Ensatina (*Ensatina eschscholtzii*), and Coastal Tailed Frog (*Ascaphus truei*). Overall, these three species constituted 81% of all animals captured, with Western Red-backed Salamanders making up 48.1%, Ensatina 25%, and Coastal Tailed Frog 8%. During the pre-harvest sampling interval, 79% of the sample consisted of these three species, 89% during the 2-year post-harvest sampling interval, and 68% 10 years after logging. Within each sampling interval, Western Red-backed Salamander was the most abundant animal captured on riparian transects in all three sampling intervals and on upland transects during the 2-year post-treatment interval. During the pre-treatment and 10-year post-harvest intervals, Ensatina was the most abundant species captured on upland transects. Some of the less common species captured included Western Toad (*Bufo boreas*), Cascades Frog (*Rana cascadae*), and Cascade Torrent Salamander (*Rhyacotriton cascadae*), all of which comprised less than 1% of the total sample.

The only species with a significantly higher abundance on one of the transects was Ensatina, which was consistently more abundant in upland transects during all three sampling intervals (pre: $p = < 0.01$; 2-years: $p = < 0.01$; 10-years: $p = < 0.01$). Western Red-backed Salamander was more abundant on upland transects during the 2-year post-harvest sampling interval, although only slightly ($p = 0.07$). Conversely, the number of Coastal Giant Salamander was higher on riparian transects compared to the uplands; however, the difference was only marginally significant ($p = 0.07$).

Relative Abundance within Sampling Intervals

Although the goal of this study was to evaluate the utility of RMZs in providing habitat for terrestrial amphibians, it was important to look at differences in relative abundance within each sampling interval to determine if any of the temporal changes in relative abundance observed were a product of site effects. Of particular interest were differences in relative abundance observed prior to timber harvest between treatments and transects. The relative abundance of Ensatina was higher on upland transects at all treatments compared to riparian transects ($t = 2.8$; $p = 0.002$; Figure 5-14). The relative abundance of Coastal Tailed Frog was marginally higher on riparian transects at the control sites compared to both the modified ($F_{1,35.70} = 3.17$; $p = 0.083$) and state buffer sites ($F_{1,34.50} = 2.92$; $p = 0.096$). On upland transects, Coastal Tailed Frog was slightly more abundant at the control sites compared to the state buffer sites ($F_{1,34.50} = 3.42$; $p = 0.072$). Coastal Giant Salamander was more abundant on riparian transects at the state buffer sites compared to both the modified buffer sites ($F_{1,155.0} = 3.23$; $p = 0.074$) and controls ($F_{1,155.1} = 4.72$; $p = 0.031$). The relative abundance of Northern Red-legged Frog was higher at the state and modified buffer sites compared to the control sites on both riparian and upland transects, but these differences were not significant. The proximity of still-water habitats to our sampling sites likely influenced the presence of species like Northern Red-legged Frog, Northwestern Salamander, Long-toed Salamander, and Rough-skinned Newt. Because the proximity to still-water habitats is unknown, we do not know the degree to which this affected the presence of these species at our sites. The relative abundance of all other species did not vary significantly among treatments or transects prior to harvest (Figure 5-14). These pre-existing differences in relative abundance were considered when determining possible treatment effects.

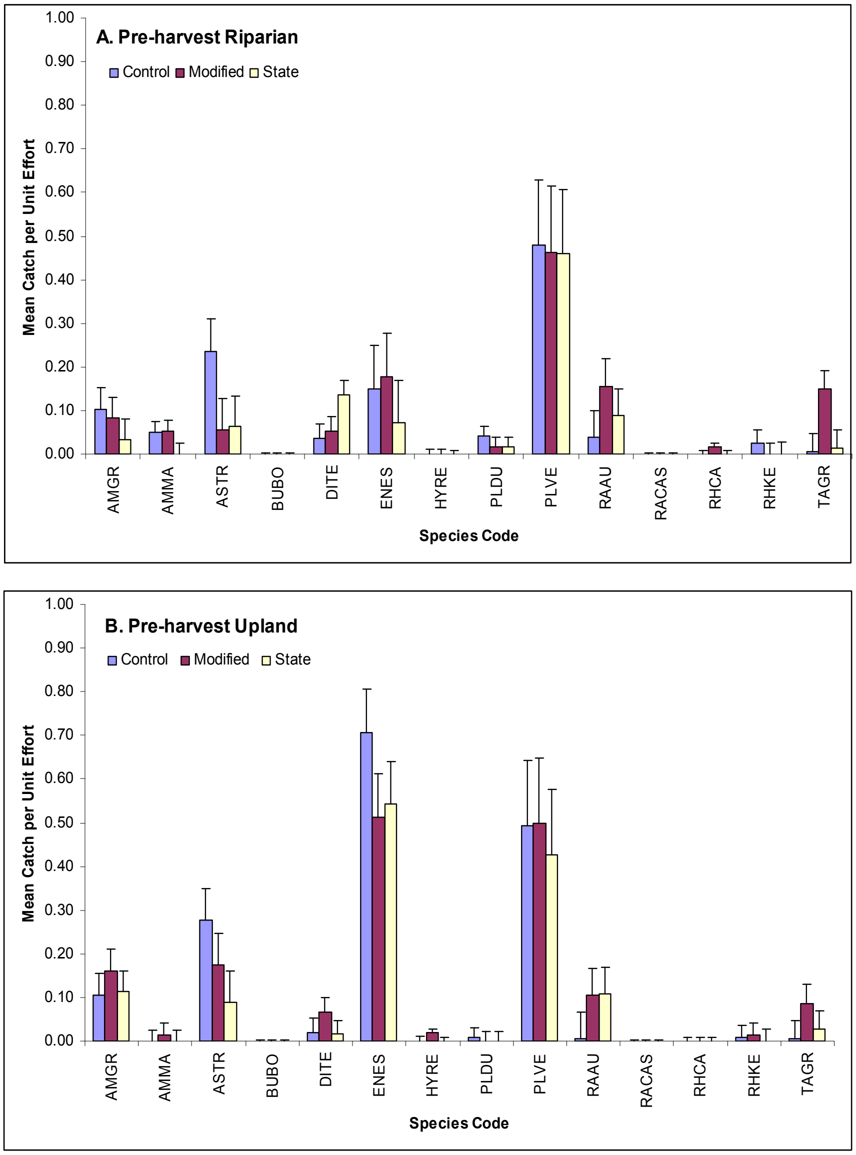

Figure 5-14. Mean catch per unit effort (+SE) (A = Riparian; B = Upland) for each species of terrestrial amphibian caught at the western Washington control sites during the pre-harvest sampling period (1992/1993). AMGR = *Ambystoma gracile*; AMMA = *A. macrodactylum*; ASTR = *Ascaphus truei*; BUBO = *Bufo boreas*; DITE = *Dicamptodon tenebrosus*; ENES = *Ensatina eschscholtzii*; HYRE = *Hyla regilla*; PLDU = *Plethodon dunni*; PLVE = *P. vehiculum*; RAAU = *Rana aurora*; RACAS = *R. cascadae*; RHCA = *Rhyacotriton cascadae*; RHKE = *R. kezeri*; TAGR = *Taricha granulosa*.

Natural Variation at the Control Sites

One of the functions of the control group is to allow for an interpretation of naturally occurring changes in the relative abundance of amphibians over time. The mean numbers of animals captured in riparian and upland control sites was plotted by species for each of the three sampling intervals (Figure 5-15a, b). From these plots, it is evident that large changes in relative abundance occurred for some species but not others. Changes in the relative abundance of amphibians on riparian transects can be categorized in one of three ways: (1) no change; (2) an increase in the first post-harvest sampling interval followed by a decrease; or (3) a decrease followed by an increase 10 years after logging. The relative abundance of Long-toed Salamander, Western Toad, Pacific Treefrog, Cascade

Frog and Cascade Torrent Salamander did not change over time, or they occurred in such low numbers that statistical comparisons were not meaningful (Figure 5-15a). The relative abundance of 5 species (Coastal Giant Salamander, Ensatina, Dunn's Salamander, Western Red-backed Salamander, and Rough-skinned Newt) increased in the first post-harvest sampling interval and then decreased 10 years after logging (Figure 5-15a). Of these, only the relative abundance of Western Red-backed and Dunn's salamanders changed significantly. Western Red-backed Salamander increased by approximately 30% immediately after logging, followed by a 62% reduction in the 10-year post-harvest sampling interval (Figure 5-15a). The increase in relative abundance was not significant, but the decrease was (increase: $F_{1,154.9} = 1.79$; $p = 0.181$; decrease: $F_{1,154.0} = 9.71$; $p = 0.002$). Similarly, the relative abundance of Dunn's Salamander increased by 33% immediately after logging ($F_{1,154.0} = 0.85$; $p = 0.356$) and decreased significantly in the 10-year post-harvest sampling interval ($F_{1,151.5} = 7.81$; $p = 0.005$). Conversely, numbers of Northwestern Salamander, Coastal Tailed Frog, Northern Red-legged Frog, and Columbia Torrent Salamander decreased in the first post-harvest sampling interval and then increased 10 years after logging (Figure 5-15a). Of these species, only the 56.5% decrease in the relative abundance of Coastal Tailed Frog in the first post-harvest sampling interval was significant ($F_{1,156.0} = 4.89$; $p = 0.028$).

The patterns of change were more variable on upland transects, with relative abundance not changing at all, increasing, or decreasing in both post-harvest sampling intervals; increasing in the first post-harvest sampling interval and then decreasing in the second post-harvest sampling interval; or first decreasing then increasing (Figure 5-15b). The relative abundance of 7 species (Long-toed Salamander, Western Toad, Pacific Treefrog, Dunn's Salamander, Cascade Frog, Cascade Torrent Salamander, and Rough-skinned Newt) did not change, or they were captured in such low numbers that statistical comparisons were not meaningful. The relative abundance of 2 species (Northwestern Salamander and Columbia Torrent Salamander) increased in both post-harvest sampling intervals. For both species, the increases in relative abundance recorded for the first post-harvest sampling interval were not significant (Northwestern: $F_{1,170.3} = 0.02$; $p = 0.864$; Columbia Torrent: $F_{1,169.3} = 0.05$; $p = 0.812$). The increases measured 10 years following logging were not significant for Northwestern Salamander ($F_{1,169.5} = 0.17$; $p = 0.672$), but were marginally significant for Columbia Torrent Salamander ($F_{1,168.9} = 3.20$; $p = 0.075$). Ensatina was the only species that consistently decreased in numbers over time (Figure 5-15B), first by 10% ($F_{1,156.7} = 0.6$; $p = 0.46$) in the first post-harvest sampling interval and then by an additional 50% in the second post-harvest sampling interval ($F_{1,154.9} = 9.21$; $p = 0.002$). The mean relative abundance recorded 10 years after logging was significantly lower than the pre-harvest means ($F_{1,156.7} = 13.1$; $p < 0.001$).

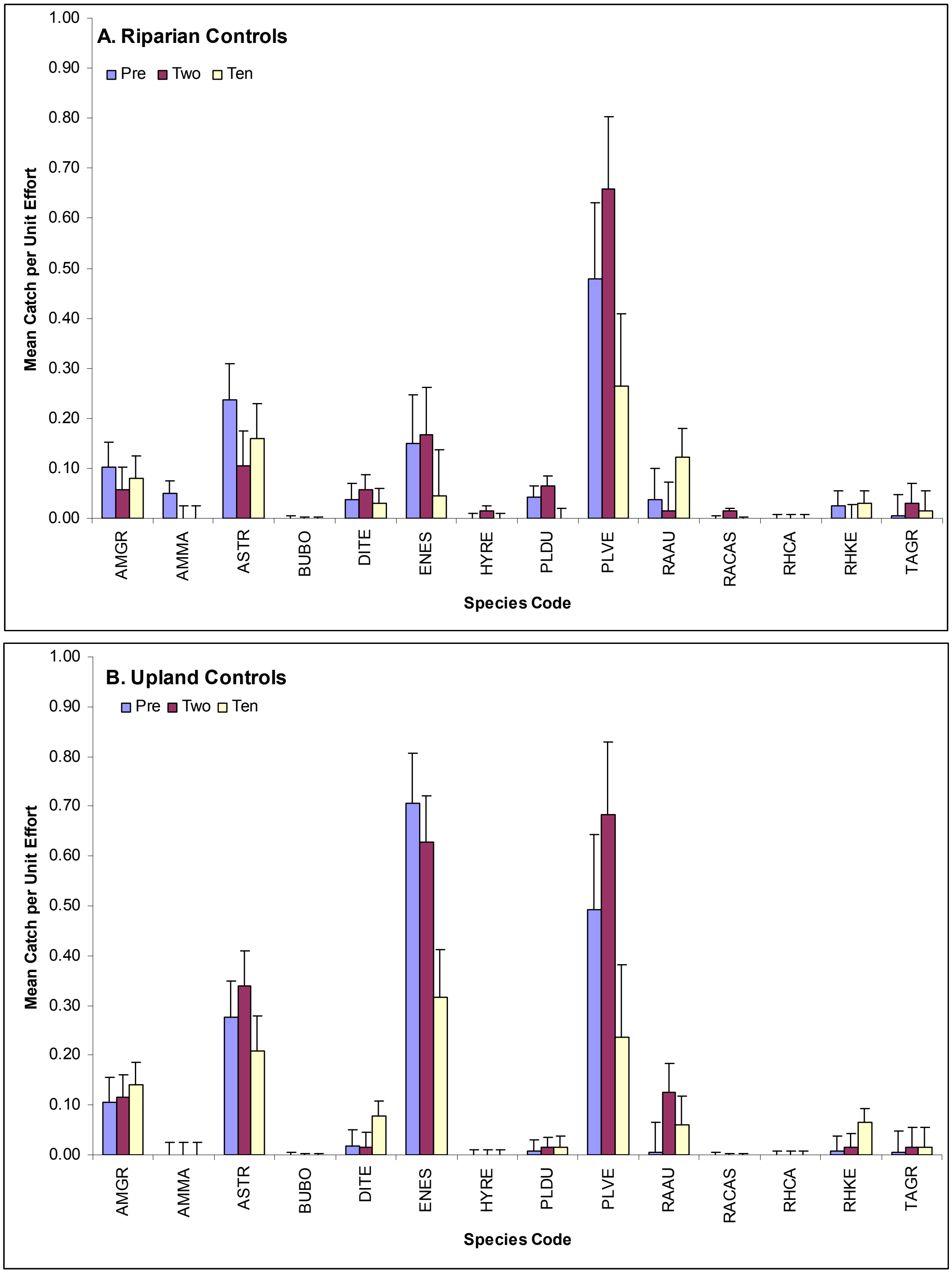

Figure 5-15. Mean catch per unit effort (+SE) (A = Riparian; B = Upland) for each species of terrestrial amphibian caught at the western Washington control sites 1993–2004. Pre = pre-harvest (1992/1993); Two = two-years post-harvest (1995/1996); Ten = ten-years post-harvest (2003/2004). AMGR = *Ambystoma gracile*; AMMA = *A. macrodactylum*; ASTR = *Ascaphus truei*; BUBO = *Bufo boreas*; DITE = *Dicamptodon tenebrosus*; ENES = *Ensatina eschscholtzii*; HYRE = *Hyla regilla*; PLDU = *Plethodon dunni*; PLVE = *Plethodon vehiculum*; RAAU = *Rana aurora*; RACAS = *R. cascadae*; RHCA = *Rhyacotriton cascadae*; RHKE = *R. kezeri*; TAGR = *Taricha granulosa*.

Western Red-backed Salamander, Coastal Tailed Frog, and Northern Red-legged Frog all increased in the first post-harvest sampling interval, followed by a decrease 10 years after logging. The relative abundance of Western Red-backed Salamander and Coastal Tailed Frog increased by 29.2% and 18.2%, respectively, relative to pre-harvest values. In both cases, relative abundance did not increase significantly (Western Red-backed Salamander: $F_{1,154.9} = 2.04$; $p = 0.154$; Coastal Tailed Frog: $F_{1,156.0} = 1.08$; $p = 0.299$). The relative abundance of Northern Red-legged Frog increased in the first post-harvest sampling interval but only marginally ($F_{1,169.6} = 3.21$; $p = 0.074$). Ten years after logging, the relative abundance of Western Red-backed Salamander had decreased by 66% ($F_{1,154.0} = 12.4$; $p < 0.001$), Coastal Tailed Frog by 39% ($F_{1,154.7} = 5.35$; $p = 0.022$), and Northern Red-legged Frog by 50% ($F_{1,169.1} = 1.02$; $p = 0.312$) compared to the 2-year post-harvest means. Finally, the relative abundance of Coastal Giant Salamander decreased in the first post-harvest sampling interval and then increased in the second post-harvest sampling interval. The initial decrease was not significant ($F_{1,171.4} = 0.005$; $p = 0.941$) nor was the subsequent increase ($F_{1,169.8} = 2.17$; $p = 0.142$).

Riparian Habitats at Buffer Treatments

The relative abundance of terrestrial amphibians clearly changed overtime, and these changes were species-specific and could be loosely categorized (see Natural Variation at the Control Sites). For riparian habitats, we evaluated the efficacy of the modified and state buffer sites in providing habitat to maintain relative abundance of terrestrial amphibians (relative to the controls) by plotting the change in mean relative abundance (+SE) for each sampling interval for each buffer treatment (Figure 5-16a, b).

The changes in relative abundance observed at the modified and state buffer sites cannot be categorized as easily as they were for the control sites. Only the changes in relative abundance of Northwestern and Western Red-backed Salamanders followed a consistent pattern among all three treatments (Figure 5-15; Figure 5-16). At the state and modified buffer sites, the relative abundance of Northwestern Salamander decreased immediately following logging and then increased in the 10-year post-harvest sampling interval. The decrease immediately following logging was not significant at either treatment, nor was the increase on state buffer sites; however, the increase at the modified buffer sites was significant ($F_{1,172.4} = 4.23$; $p = 0.040$). The changes observed at the buffer sites were consistent with those observed at the controls, providing an indication of buffer width efficacy. Western Red-backed Salamander also changed consistently over time, with a large increase in relative abundance in the first post-harvest sampling interval, followed by a decrease in the second post-harvest sampling interval (Figure 5-15; Figure 5-16). The changes in relative abundance were significant for both sampling intervals at the state buffer sites (increase: $F_{1,154.5} = 7.08$; $p = 0.008$; decrease: $F_{1,154.0} = 11.5$; $p < 0.001$), as was the decrease at the modified buffer sites ($F_{1,157.3} = 4.84$; $p = 0.029$); and all changes were consistent with those observed at the controls. At all treatments, the relative abundance of Western Red-backed Salamander was lower 10 years after logging compared to the pre-harvest means, but not significantly.

Trends in relative abundance were variable for the remaining species and inconsistent among treatments. For example, the relative abundance of Ensatina decreased at the modified buffer sites immediately after logging but increased at state buffer sites, a change that was consistent with the controls; and neither change was significant. Ten years after logging, Ensatina decreased on state buffer sites, again consistent with the controls, but increased at the modified buffer sites. None of the changes were significant, and the changes observed for Ensatina exemplify the situation of most species: change in relative abundance was variable and independent of treatment type. The one exception was the Coastal Giant Salamander. At modified buffer sites, this species decreased following logging but increased in the 10-year post-harvest sampling interval, and neither of the changes were significant (decrease: $F_{1,171.5} = 0.28$; $p = 0.592$; decrease: $F_{1,174.1} = 1.08$; $p = 0.298$). Similarly, the relative abundance of this species decreased following logging on the state buffer sites by 23% and decreased a further 70% in the 10-year post-harvest sampling interval. The change from the first to second post-harvest sampling interval was significant ($F_{1,169.8} = 3.57$; $p = 0.060$), with the overall reduction in relative abundance 10 years after logging significantly different from the pre-harvest means ($F_{1,170.7} = 6.08$; $p = 0.014$). This suggests that the state buffer does not provide suitable habitat for this species over time and that the wider buffer (i.e., modified) does a better job of preserving relative abundance 10 years after logging.

For species with enough captures to make meaningful statistical comparisons (Northwestern Salamander, Coastal Tailed Frog, Coastal Giant Salamander, Ensatina, Western Red-backed Salamander, Northern Red-legged Frog, and Rough-skinned Newt), the buffer treatments preserved relative abundance at levels similar to, or consistent with, the control sites. Again, the only exception was the Coastal Giant Salamander, which appears to have declined steadily at state buffer sites.

Upland Habitats at Buffer Treatments

The relative abundance of 5 species (Northwestern Salamander, Coastal Tailed Frog, Coastal Giant Salamander, Ensatina, and Northern Red-legged Frog) declined over time in upland habitats at the buffer sites, and the decline was not always the same at both treatments (Figure 5-17), nor was the change consistent with the change at the controls (Figure 5-15b). The only species with consistent changes in relative abundance was the Western Red-backed Salamander. At all treatments, its relative abundance first increased immediately after logging and then decreased in the 10-year post-harvest sampling interval, a pattern that was consistent with the changes observed on riparian transects (Figure 5-16). The initial increase in relative abundance was significant at both the state ($F_{1,154.5} = 26.3$; $p < 0.001$) and modified ($F_{1,155.0} = 4.32$; $p = 0.039$) buffer sites. The subsequent decrease in relative abundance was also significant at both buffer treatments (state: $F_{1,154.0} = 27.4$; $p < 0.001$; modified: $F_{1,157.3} = 8.09$; $p = 0.005$). The change in relative abundance of Ensatina in upland habitats at the modified buffer sites was consistent with the changes observed at the controls, decreasing steadily over time, with the decrease immediately after logging marginally significant ($F_{1,156.8} = 3.24$; $p = 0.073$). Mean relative abundance 10 years after logging was significantly lower than pre-harvest means ($F_{1,163.7} = 4.34$; $p = 0.038$). On state buffer sites, the relative abundance of Ensatina first increased significantly immediately following logging ($F_{1,155.9} = 4.40$; $p = 0.04$) and then decreased significantly in the 10-year post-harvest sampling interval ($F_{1,154.9} = 20.9$; $p < 0.001$), with the 10-year means significantly lower than the pre-harvest means ($F_{1,155.9} = 5.63$; $p = 0.02$). On state buffer sites, the 10-year post-harvest means were consistent with both the modified buffer and control sites (i.e., they were lower than the pre-harvest means), suggesting a regional effect rather than a treatment response.

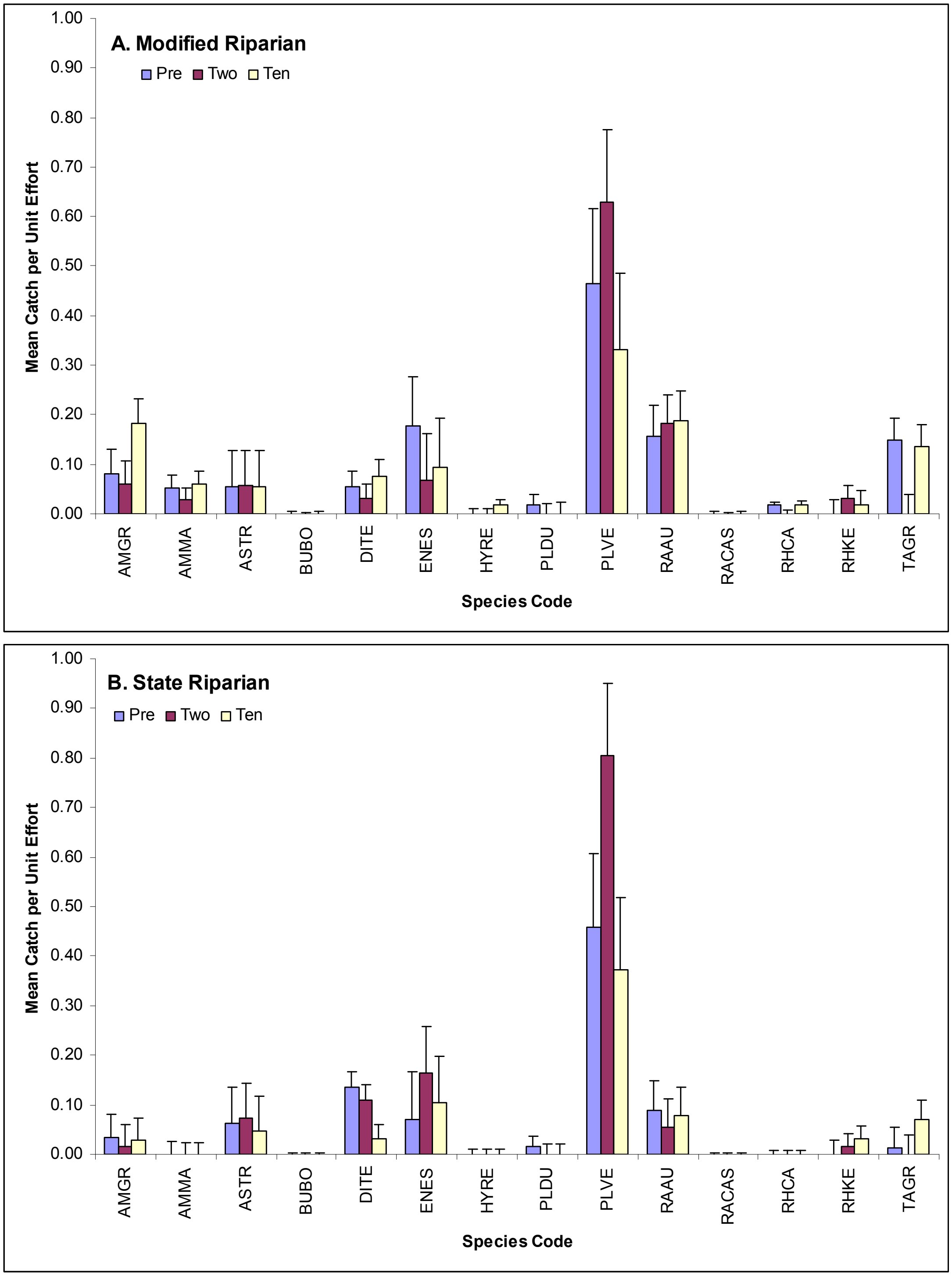

Figure 5-16. Mean catch per unit effort (+SE) (A = Modified; B = State) for each species of terrestrial amphibian caught on riparian transects at the western Washington buffer sites, 1993–2004. Pre = pre-harvest (1992/1993); Two = two-years post-harvest (1995/1996); Ten = ten-years post-harvest (2003/2004). AMGR = *Ambystoma gracile*; AMMA = *A. macrodactylum*; ASTR = *Ascaphus truei*; BUBO = *Bufo boreas*; DITE = *Dicamptodon tenebrosus*; ENES = *Ensatina eschscholtzii*; HYRE = *Hyla regilla*; PLDU = *Plethodon dunni*; PLVE = *P. vehiculum*; RAAU = *Rana aurora*; RACAS = *R. cascadae*; RHCA = *Rhyacotriton cascadae*; RHKE = *R. kezeri*; TAGR = *Taricha granulosa*.

Coastal Giant Salamander and Coastal Tailed Frog were likely adversely affected by logging the uplands at both buffer treatments. Both species were documented prior to harvest (Figure 5-17). Immediately after logging, numbers of Coastal Giant Salamander increased slightly at the state buffer sites, and this species was not detected at the modified buffer sites. Ten years after logging, Coastal Giant Salamander was not detected in the upland habitats at either treatment, which is contrary to the pattern observed at the controls, where numbers of Coastal Giant Salamander increased 10 years after logging (Figure 5-15b). Similarly, the relative abundance of Coastal Tailed Frog declined over time at both the state and modified buffer sites (Figure 5-17). The change in relative abundance was not significant at the state buffer sites but was close to being significant at the modified buffer sites immediately after logging ($F_{1,156.1} = 3.09$; $p = 0.08$). Furthermore, 10 years after logging, the relative abundance of Coastal Tailed Frog was significantly lower than the pre-harvest value ($F_{1,161.2} = 7.38$; $p = 0.007$). Although pre-harvest mean relative abundance values were lower at the state and modified buffer treatments compared to the controls, the temporal effects on Coastal Tailed Frog relative abundance were not consistent; the 10-year post-harvest means at the control was not significantly different from the pre-harvest means ($F_{1,156.0} = 1.33$; $p = 0.250$).

Logging the uplands at the state buffer sites may have adversely affected the relative abundance of Northwestern Salamander. This species declined in numbers immediately after logging, with the decline continuing in the 10-year post-harvest sampling interval. The immediate post-logging decline was also observed on upland transects of the modified buffer but not at the controls. Although the initial decline at the state buffer sites was not significant, the difference between the 10-year post-harvest means and the pre-harvest values was close to significant ($F_{1,169.9} = 2.86$; $p = 0.092$). The change at the state buffer sites indicates that the riparian buffer may have some degree of influence on the upland habitats at logged sites. In this case, the narrower state buffer may not ameliorate the effects of logging in the upland as well as the modified, wider buffer does.

The relative abundance of the remaining species varied relative to treatment and sampling interval. There were no patterns to the changes observed, indicating that site effects were likely responsible for some of the change in relative abundance observed in the upland habitats at all treatments.

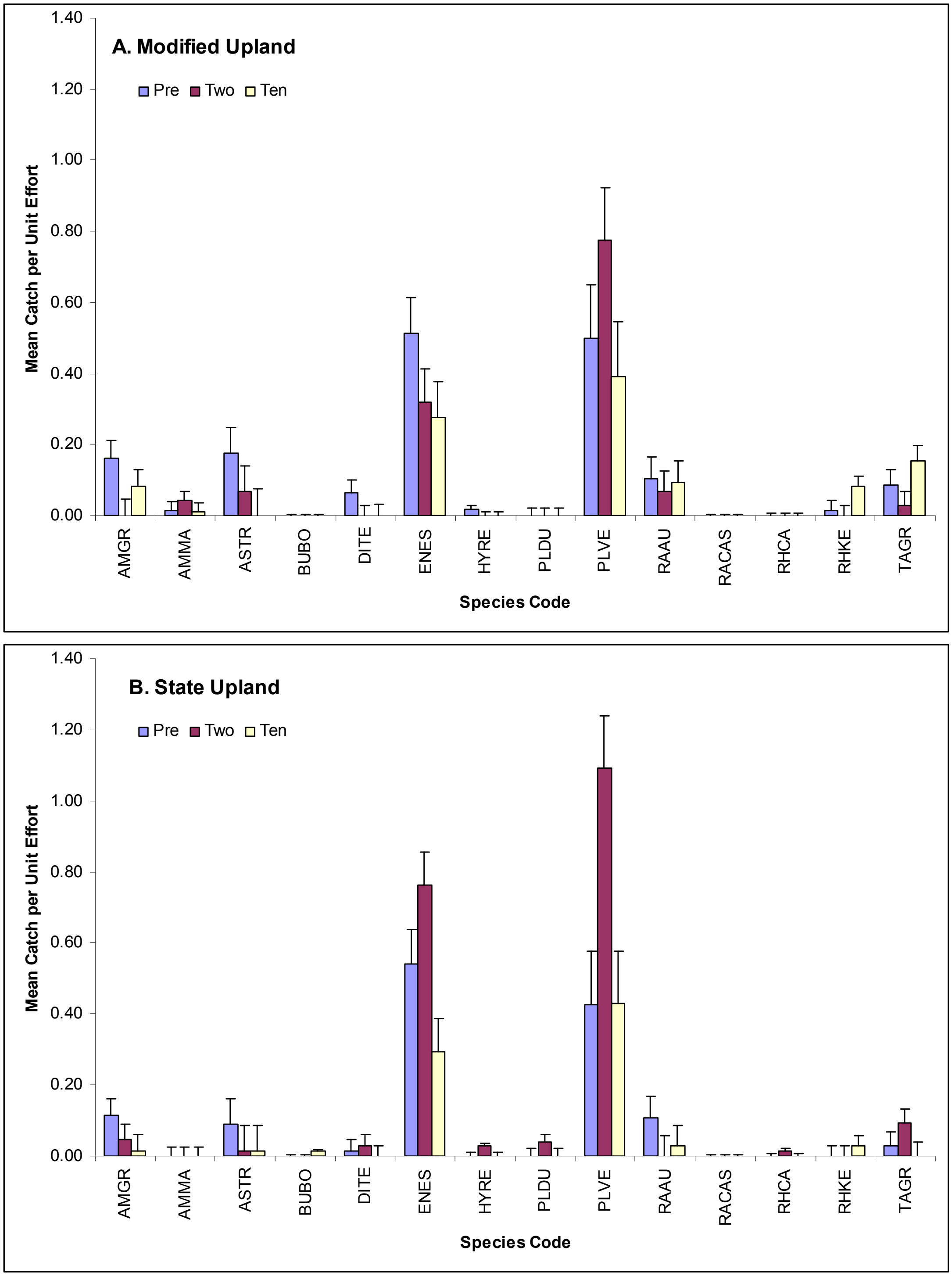

Figure 5-17. Mean catch per unit effort (+SE) (A = Modified; B = State) for each species of terrestrial amphibian caught on upland transects at the western Washington buffer sites, 1993–2004. Pre = pre-harvest (1992/1993); Two = two-years post-harvest (1995/1996); Ten = ten-years post-harvest (2003/2004). AMGR = *Ambystoma gracile*; AMMA = *A. macrodactylum*; ASTR = Ascaphus truei; BUBO = *Bufo boreas*; DITE = *Dicamptodon tenebrosus*; ENES = *Ensatina eschscholtzii*; HYRE = *Hyla regilla*; PLDU = *Plethodon dunni*; PLVE = *P. vehiculum*; RAAU = *Rana aurora*; RACAS = *R. cascadae*; RHCA = *Rhyacotriton cascadae*; RHKE = *R. kezeri*; TAGR = *Taricha granulosa*.

Amphibian–Habitat Relationships

Amphibian habitat associations were considered in the context of common, widely distributed species (Western Red-backed Salamander, Ensatina, and Coastal Tailed Frog) and species that were less commonly encountered at our study sites (Coastal Giant Salamander, Rough-skinned Newt, Northwestern Salamander, and Northern Red-legged Frog). By grouping species into these categories, habitat relationships were considered in the context of common, abundant species typical of many seral stages (Western Red-backed Salamander and Ensatina) and a species that is relatively common but apparently restricted to older forests dominated by Douglas-fir (Coastal Tailed Frog). This group of amphibians was used to assess whether or not common species were plastic with respect to large-scale habitat perturbations and to determine if the Coastal Tailed Frog was closely associated with maturing coniferous trees, as documented in the literature. In general, forest habitat relationships have not been described for the second group of amphibians (i.e., the less commonly encountered or narrowly distributed group), and the co-inertia analysis (COIA) provided an opportunity to describe some of the general relationships these species have with habitat attributes, measured in a changing landscape.

Riparian and upland forests are functionally and structurally different; therefore, we chose to treat the analyses of habitat relationships separately for each distinct habitat type but to comment on how the RMZ may influence the upland forest at logged sites (and vice versa). Habitat associations are discussed in a temporal context to provide an appreciation of how habitat associations changed over time at the unlogged controls and at the buffer sites. As with relative abundance, it was necessary to first establish pre-harvest habitat relationships and then to document how habitat associations changed across time at the controls; this gave a sense of the variation associated with habitat use by each species as it related to forest succession. Finally, a comparison of the state and modified buffer sites was made with the pre-harvest associations and with the controls to determine if habitat associations changed in riparian buffers or in logged upland habitats.

COIA is a multivariate statistic that enables description of the relationships between species data and environmental variables. We used relative abundance data as the species data and habitat attribute data as the environmental variable data in all analyses. Because COIA does not contain a regression step, the results presented below cannot be used in a predictive manner and are therefore necessarily descriptive. Only necessary figures are provided in the results section. All biplots for each sampling interval, treatment, and transect are provided in Appendix 5-B and Appendix 5-C.

Riparian Habitat Associations

Pre-harvest

The characterization of habitat associations prior to harvest served as a baseline for comparing post-harvest habitat associations in riparian forests at the controls and at both buffer treatments. The riparian forest at the control sites in all three sampling intervals was characterized by high structural complexity and increasing heterogeneity, which was expected. The pre-harvest biplot (Figure 5-18) combines relative abundance and habitat attribute data obtained from all 18 sites (i.e., regardless of treatment, because treatments had not yet been applied). It reveals a gradient tending toward larger Douglas-fir trees, abundant woody debris and small deciduous and coniferous trees in the understory along axis 1, which was typical of the maturing riparian forests at the study sites. Axis 2 describes an environmental gradient tending away from bare soil to a more structurally complex habitat, with large coniferous trees (cedar and hemlock) and large deciduous trees (bigleaf maple) comprising a secondary component of the overstory. In this case, axis 2 reflects (to a large degree) the environmental gradient observed on axis 1, and both axes describe an environmental gradient of increasing forest age and structural complexity. The majority of the variation in the data is described by axis 1 (65.7%), with an additional 12.6% explained by axis 2.

The relative abundance of the three common and widely distributed species was strongly associated with these gradients: Western Red-backed Salamander abundance was negatively correlated with axis 1, indicating that this species was more abundant at sites dominated by habitat components typical of younger seral forests and in general was more abundant when the volume of deciduous canopy cover increased, perhaps because deciduous leaf letter provided thermally suitable, secure cover for this species. Leaf litter likely also provided suitable foraging habitat. Ensatina was correlated negatively with axes 1 and 2 and had an association with large deciduous trees and structural habitat complexity at the shrub level (shrubs ≤1.3 m in height). The relative abundance of Coastal Tailed Frog was positively correlated with axis 2, suggesting a strong correlation between increasing relative abundance and increasing forest age and increasing canopy cover (decreasing forest openness). These relationships are generally aligned with previously documented habitat associations for these species, although prior to harvest Ensatina was not strongly correlated with coarse woody debris as has been found in other studies.

Northern Red-legged Frog was negatively correlated with axis 2, indicating that the relative abundance of this species was associated with increasing canopy cover and decreasing tree cover and that the species was more abundant in habitats containing ferns. Rough-skinned Newt and Northwestern Salamander were negatively associated with both axes, suggesting that these species were likely to be more abundant in habitats characterized by decreasing forest openness (increasing canopy cover) with fewer large trees. Fern cover appears to have been more important for Northwester Salamander than it was for Rough-skinned Newt. Coastal Giant Salamander was most abundant in habitats with large Douglas-fir trees and rocky soils. Increasing moss and fern cover was also positively associated with the relative abundance of Coastal Giant Salamander.

Post-harvest Controls

Immediately following logging, the level of variation in habitat associations at the controls for the common species was minimal and consistent with the pre-harvest descriptions for Western Red-backed Salamander, Ensatina, and Coastal Tailed Frog, which was likely related to the short time period between sampling. Ten years after logging, Western Red-backed Salamander and Ensatina continued to occupy similar habitats (relative to each other), and the relative abundance of these two species was strongly associated with increasing numbers of large coniferous and deciduous trees and with increasing numbers of large Douglas-fir. The habitat relationships described for Western Red-backed Salamander represented a change from the pre-harvest and 2-year post-harvest sampling intervals and is likely explained by forest succession and the dynamic nature of riparian forests. Ten years after logging, the relative abundance of Coastal Tailed Frog was not correlated with large conifers or Douglas-fir trees. Rather, increasing rock and exposed soil appeared to be more highly correlated with numbers of Coastal Tailed Frog than was the presence of large Douglas-fir (Figure 5-18).

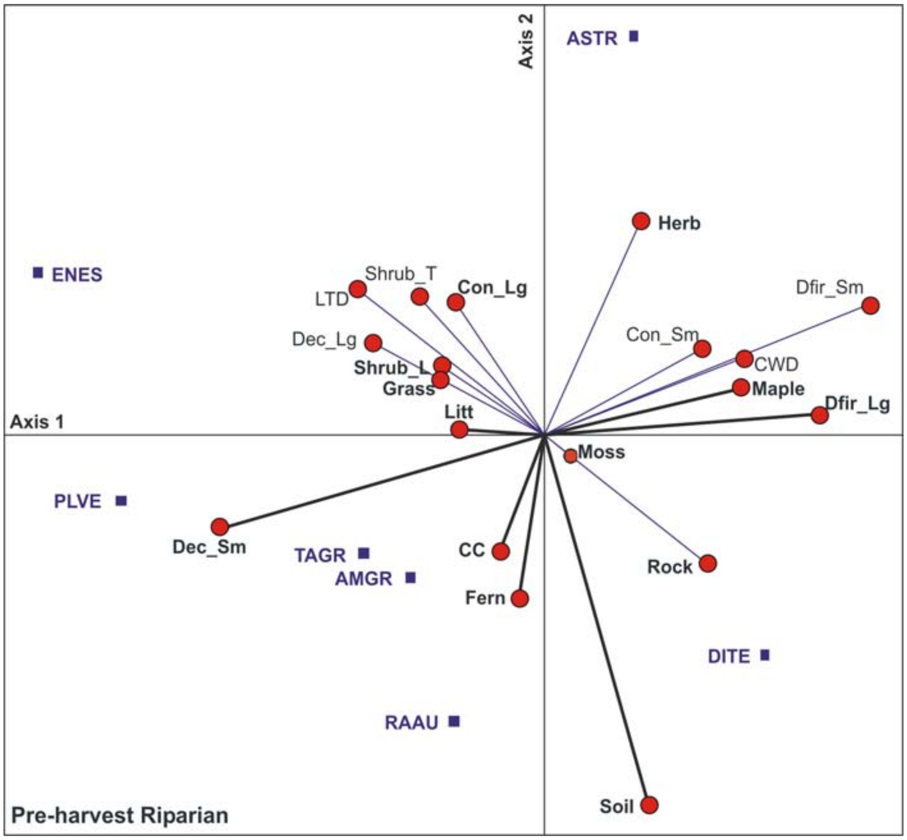

Figure 5-18. Amphibian–habitat relationships in riparian habitats prior to logging. Thick black lines and bold text indicate habitat variables that describe the gradient along each axis. Shrub_T = tall shrubs; Shrub_L = low shrubs; CC = overstory canopy cover; CWD = coarse woody debris; Fern = fern cover; Grass = grass cover; Herb = herbaceous cover; Litt = deciduous leaf litter cover; LTD = deciduous leaf litter depth; Moss = moss cover; Rock = rock cover; Soil = exposed soil; Maple = vine maple; Dec_Sm = bigleaf maple and red alder <1.3 m; Dec_Lg = bigleaf maple and red alder >1.3 m; Con_Sm = western redcedar and western hemlock <1.3 m; Con_Lg = western redcedar and western hemlock >1.3 m; Dfir_Sm = Douglas-fir <1.3 m; Dfir_Lg = Douglas-fir >1.3 m. AMGR = *Ambystoma gracile*; ASTR = *Ascaphus truei*; DITE = *Dicamptodon tenebrosus*; ENES = *Ensatina eschscholtzii*; PLVE = *Plethodon vehiculum*; RAAU = *Rana aurora*; TAGR = *Taricha granulosa*.

Overall the changes in species–habitat associations changed minimally over time and did not appear to be affected to any large degree by forest succession, suggesting that Western Red-backed Salamander, Ensatina, and Coastal Tailed Frog (although possibly to a lesser degree) maintain high habitat-attribute affinity in riparian forests of 65–75 years of age.

For species not commonly encountered, some interesting patterns were observed in riparian habitats at the control sites over time. The relative abundance of Northwestern Salamander was always strongly associated with increasing fern cover, and in each sampling interval Northwestern Salamander always occupied a similar position in space. In the first post-harvest sampling interval, the relative abundance of Rough-skinned Newt, Northern Red-legged Frog, and Coastal Giant Salamander was too low to derive any meaningful habitat relationships. However, in the 10-year post-harvest sampling interval, the relative abundance of all three species was strongly associated with increasing numbers of large coniferous trees, large deciduous trees, and large Douglas-fir, which was consistent with both Western Red-backed Salamander and Ensatina. The presence of large coniferous and deciduous trees in a heterogeneous, structurally complex forest was positively correlated with increasing relative abundance of terrestrial and semi-aquatic amphibians in western Washington.

Post-harvest Buffer Sites

Two years after logging, there was no consistency in the habitat associations indicated by the biplots for each of the three treatments for Western Red-backed Salamander, Ensatina, or Coastal Tailed Frog. The relative abundance of Coastal Tailed Frog was associated with increasing numbers of large conifers at state buffer sites and with increasing cover of coarse woody debris at modified buffer sites. Although there was a difference in habitat association between treatments, the correlation with these habitat types post-harvest was consistent with the pre-harvest and control habitat associations. Ten years after logging, numbers of Coastal Tailed Frog were correlated with increasing deciduous leaf-litter cover and depth and with increasing canopy cover; and at both treatments Coastal Tailed Frog occurred in very low numbers. At both buffer treatments, numbers of Ensatina and Northwestern Salamander were positively correlated with increasing canopy cover and increasing numbers of maple, an association that differed from the habitat associations noted prior to logging and at the controls. Those associations persisted into the 10-year post-harvest sampling interval, although the relative abundance of both species was also positively associated with increasing volumes of coarse woody debris.

One possible explanation for this observation is that these species sought out habitats with increasing cover because such habitats closely approximated those occupied prior to harvest and at the control sites, characterized by high shrub cover and large deciduous trees. In the spring and summer, these habitats would have provided excellent shade from the sun, which in turn would produce higher-quality thermal habitat for Ensatina and Northwestern Salamander. With a reduction in shading caused by the removal of the adjacent upland forest, large deciduous trees and tall shrubs may not have provided the same quality of thermal protection required by these two species. Although the forest had started to grow back by the 10-year post-harvest sampling interval, the forest in the uplands would not have contributed to shading the riparian habitat because it was not tall enough.

At the modified and state buffer sites, the relative abundance of Western red-backed Salamander was associated with small coniferous trees (state) and with moss and rock cover (modified) in both the 2-year and 10-year post-harvest sampling intervals, representing a departure from the pre-harvest and post-harvest control associations. This perceived change in habitat association may be indicative of this species' ability to adapt to a changing landscape. Logging in the adjacent upland is expected to have affected the riparian habitats, but the degree of impact was not measured. Despite the perceived changes in habitat associations, the relative abundance of Western Red-backed Salamander was not affected. This suggests that habitat use by this species is variable and associated with certain habitat features, namely moss, ferns, coarse woody debris, rocky slopes, and deciduous leaf litter.

The relationship between the relative abundance of Northern Red-legged Frog and habitat attributes was not consistent across time at the buffer sites, nor was relative abundance correlated with a specific habitat attribute. This suggests that Northern Red-legged Frog is either a generalist species; that Northern Red-legged Frog abundance is linked to proximity to breeding habitat,; or that, because Northern Red-legged Frog is mobile at relatively large scales, its relative abundance at our study sites was unrelated to the dimension and positioning of sites in this study. There was some correlation between the habitat associations at the state and modified treatments, but the differences observed may be due to habitat variability at the site level and not necessarily indicative of real habitat differences. Where Northern Red-legged Frog occurs, it is generally abundant and not seemingly associated with a particular habitat attribute. Rather, the overall habitat condition includes high moisture, access to escape habitat, and increasing levels of shade that dictate where this species occurs. Connectivity with upland habitats also increases the value of riparian habitat for Northern Red-legged Frog. Rough-skinned Newt was not documented in the 2-year post-harvest sampling interval, so habitat associations cannot be described. Ten years after logging, the relative abundance of this species was positively associated with increasing numbers of large Douglas-fir and with increasing rock cover, a pattern consistent with the associations described for the pre-harvest riparian sites and control sites in the 10-year post-harvest period.

Upland Habitat Associations

Pre-harvest

The characterization of habitat associations prior to harvest in upland habitats served as a baseline for the comparison of post-harvest habitat associations at the controls and at the state and modified buffer treatments. Upland habitats in the pre-harvest period were characterized by coniferous tree dominance, with a component of large deciduous trees and smaller coniferous trees in the understory. In general, upland habitats were even-aged stands dominated by Douglas-fir, with little heterogeneity and low species diversity. Axis 1 of the pre-harvest upland biplot describes a gradient of increasing numbers of large conifers in both the overstory and understory, suggesting a maturing and relatively homogenous second-growth upland forest with abundant coarse woody debris (Figure 5-19). Axis 2 describes a gradient of declining deciduous representation in the forest, followed by an opening of the forest floor due to increasing canopy cover (Figure 5-19). Axis 1 explains 62.4% of the total variation in the data, and axis 2 explains a further 16.3%. Under those conditions, Western Red-backed Salamander was most abundant in upland habitats of increasing forest age with greater volumes of coarse woody debris, and was more abundant in habitats characterized by increasing canopy cover, increasing herbaceous growth, and increasing area of exposed soil. Increasing litter depth was also associated with increasing Western Red-backed Salamander abundance. The relative abundance of Ensatina was strongly correlated with axis 2, suggesting that habitats with increasing canopy cover and herbaceous growth would support higher populations of this species; tree species did not appear to be important, but the upland forests were dominated by coniferous trees, primarily Douglas-fir. The upland forests were also characterized by high volumes of coarse woody debris, a habitat attribute known to be positively correlated with the relative abundance of Ensatina. Coastal Tailed Frog occupied a position between the two gradients and was more abundant in habitats dominated by forests with increasing canopy cover, rock cover, and coarse woody debris.

Rough-skinned Newt, Northwestern Salamander, and Northern Red-legged Frog were negatively associated with axis 2, suggesting that the relative abundance of these species was correlated with stands dominated by large and small deciduous trees and decreasing canopy cover. Coastal Giant Salamander was not as abundant on upland transects, but when it was present it was associated with large conifers, maple, and moss.

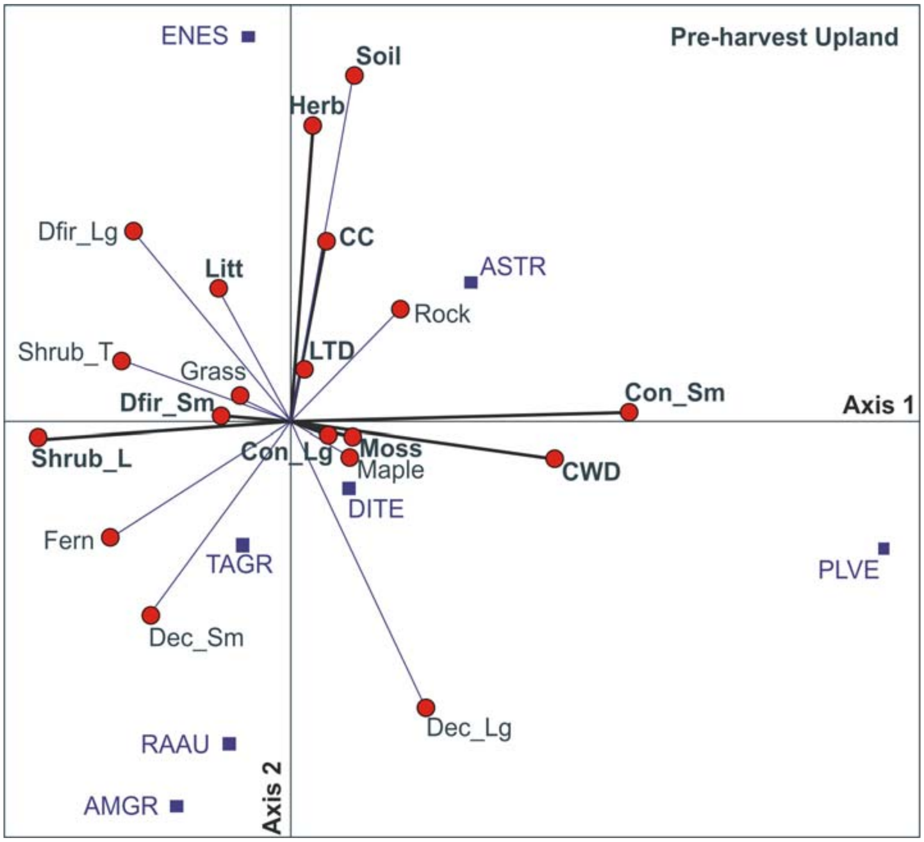

Figure 5-19. Amphibian–habitat relationships in upland habitats prior to logging. Thick black lines and bold text indicate habitat variables that describe the gradient along each axis. Shrub_T = tall shrubs; Shrub_L = low shrubs; CC = overstory canopy cover; CWD = coarse woody debris; Fern = fern cover; Grass = grass cover; Herb = herbaceous cover; Litt = deciduous leaf litter cover; LTD = deciduous leaf litter depth; Moss = moss cover; Rock = rock cover; Soil = exposed soil; Maple = vine maple; Dec_Sm = bigleaf maple and red alder <1.3 m; Dec_Lg = bigleaf maple and red alder >1.3 m; Con_Sm = western redcedar and western hemlock <1.3 m; Con_Lg = western redcedar and western hemlock >1.3 m; Dfir_Sm = Douglas-fir <1.3 m; Dfir_Lg = Douglas-fir >1.3 m. AMGR = *Ambystoma gracile*; ASTR = *Ascaphus truei*; DITE = *Dicamptodon tenebrosus*; ENES = *Ensatina eschscholtzii*; PLVE = *Plethodon vehiculum*; RAAU = *Rana aurora*; TAGR = *Taricha granulosa*.

Post-harvest Controls

The upland habitats at the control sites remained constant over time, with few changes observed. Upland habitats continued to be characterized by homogenous, even-aged forests dominated by Douglas-fir and with low species diversity. Two years after logging, the relative abundance of Western Red-backed Salamanders and Ensatina was associated with an increasing number of large conifers, increasing canopy cover, and small conifers in the understory. Coastal Tailed Frog, on the other hand, was more abundant in upland forest dominated by Douglas-fir. These relationships persisted through the 10-year post-harvest sampling interval. The amphibian–habitat associations were most similar in the 2- and 10-year post-harvest sampling intervals; both differed slightly from those observed prior to harvest, with the difference likely attributable to forest succession. The relative abundance of Northern Red-legged Frog, Coastal Giant Salamanders, Northwestern Salamander, and Rough-skinned Newt was loosely associated with the same habitat attributes as for Western Red-backed Salamander and Ensatina. Ten years after logging, numbers of Coastal Giant Salamander were closely associated with increasing moss cover. This species was also correlated with increasing fern cover, increasing leaf-litter depth, and increasing numbers of large Douglas-fir trees in the overstory.

Post-harvest Buffer Sites

The environmental conditions of the upland habitats at all logged sites combined reflect gradients expected immediately after logging: Axis 1 (85.8%) is characterized by few small coniferous trees, increasing herbaceous cover, limited fern cover, and a few large Douglas-fir (leave trees) (Figure 5-20). Axis 2 (11.2%) describes an environment of few shrubs, little coarse woody debris, and no large or small deciduous trees. In this case, axis 2 reflects to a large degree the gradient described by axis 1. Under these conditions, the relative abundance of Western Red-backed Salamander was negatively correlated with axis 1 and was associated with small conifers, rocky substrates, and exposed soil. The relative abundance of Ensatina was correlated with both axes 1 and 2 and appears to have been associated with increasing canopy cover and large conifers. Coastal Tailed Frog showed a positive correlation with both axes, with some association with large Douglas-fir, although this relationship is weak owing to the paucity of large trees in the uplands of logged sites. The Northern Red-legged Frog was negatively correlated with axis 2, and its relative abundance was related to the presence of large deciduous trees and maple, which were more prominent in riparian habitats (Figure 5-18). Rough-skinned Newt was more abundant in upland habitats where deciduous leaf-litter depth was greater and where rock and litter cover were higher. Given that these conditions would have been associated with an increase in maple cover, and that the position of Rough-skinned Newt in Figure 5-20 indicates a negative relationship with axis 2, which was characterized by an increase in maple cover, it stands to reason that this species was not commonly encountered in upland habitats 2 years after logging. The relative abundance of Northwestern Salamander and Coastal Giant Salamander was too low to describe any meaningful habitat associations.

Ten years after logging, the upland habitats at the buffer sites had changed considerably, with abundant regenerating Douglas-fir, coniferous trees, vine maple, and ferns dominating the landscape (Figure 5-21). These changes were expected because the upland sites were replanted with Douglas-fir shortly after logging. Other tree species growing in the upland sites likely grew from seed that originated from adjacent second-growth forests. The habitat associations of terrestrial amphibians changed because of the change in seral stage. In the 10-year post-harvest sampling interval, abundant coarse woody debris, higher moss cover, and more small coniferous trees were associated with higher numbers of Western Red-backed Salamander. The relative abundance of Ensatina was associated with increasing numbers of small Douglas-fir and increasing deciduous leaf-litter depth. Coastal Tailed Frog numbers were lower in upland habitats 10 years after logging, which is why the position of Coastal Tailed Frog in Figure 5-21 is close to the origin of both axes.

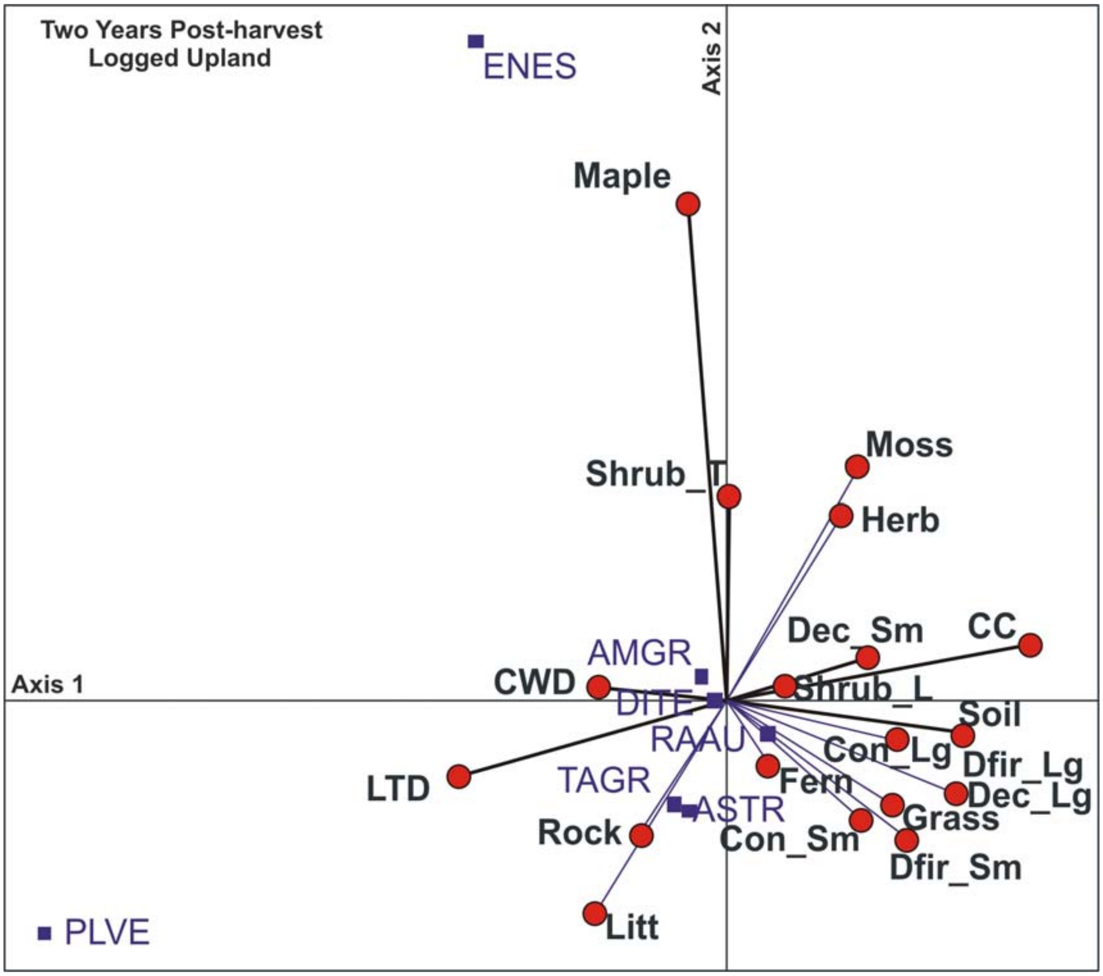

Figure 5-20. Amphibian–habitat relationships in upland habitats two-years post-logging. Thick black lines and bold text indicate habitat variables that describe the gradient along each axis. Shrub_T = tall shrubs; Shrub_L = low shrubs; CC = overstory canopy cover; CWD = coarse woody debris; Fern = fern cover; Grass = grass cover; Herb = herbaceous cover; Litt = deciduous leaf litter cover; LTD = deciduous leaf litter depth; Moss = moss cover; Rock = rock cover; Soil = exposed soil; Maple = vine maple; Dec_Sm = bigleaf maple and red alder <1.3 m; Dec_Lg = bigleaf maple and red alder >1.3 m; Con_Sm = western redcedar and western hemlock <1.3 m; Con_Lg = western redcedar and western hemlock >1.3 m; Dfir_Sm = Douglas-fir <1.3 m; Dfir_Lg = Douglas-fir >1.3 m. AMGR = *Ambystoma gracile*; ASTR = *Ascaphus truei*; DITE = *Dicamptodon tenebrosus*; ENES = *Ensatina eschscholtzii*; PLVE = *Plethodon vehiculum*; RAAU = *Rana aurora*; TAGR = *Taricha granulosa*.

Coastal Giant Salamander was not documented in upland habitats 10 years after logging. Its position at the origin of both axes of Figure 5-21 is indicative of this. Northwestern Salamander was positively associated with increasing maple cover, increasing numbers of large Douglas-fir, and increasing fern cover. Both Northern Red-legged Frog and Rough-skinned Newt were more abundant when upland habitats had deeper leaf litter and more small Douglas-fir trees.

The changes in habitat associations in upland habitats are not surprising given the changes in seral stages related to clearcut logging, followed by the subsequent replanting of upland forests. By the 10-year post-harvest sampling interval, the upland forest had started to regrow, and at most sites the regenerating trees were approaching 3–4 m in height. The persistence of terrestrial amphibians on the landscape is likely related to the relatively fast regrowth observed at the uplands of the logged sites. The cover of 3–4 m trees provided substantial ground shading, which likely produced adequate protection from the sun.

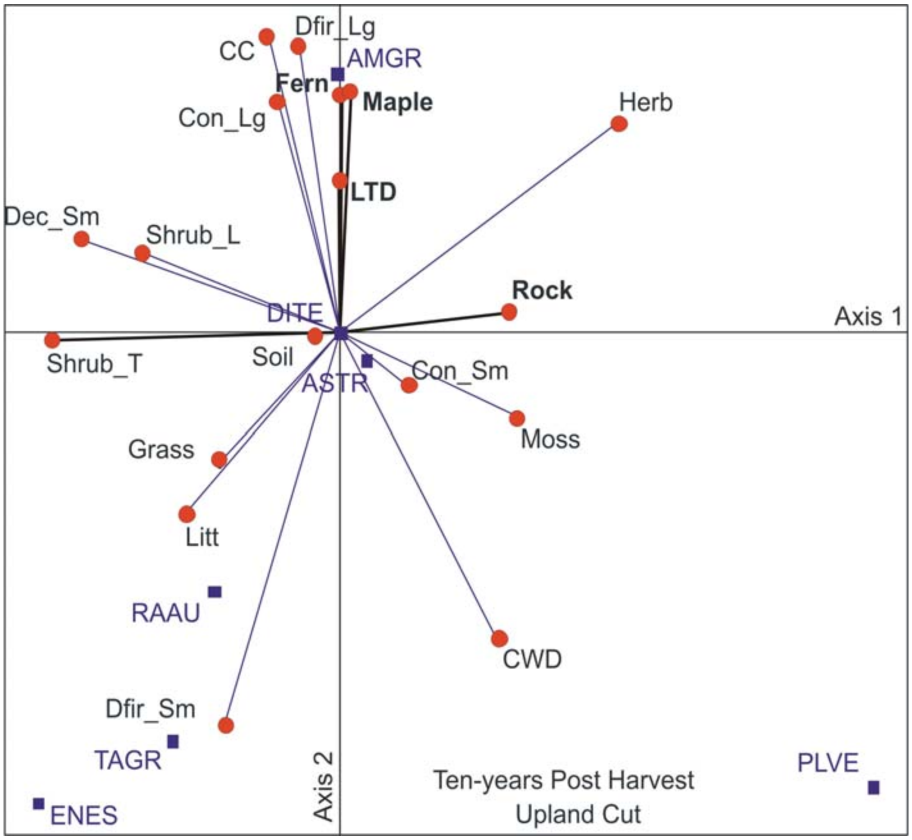

Figure 5-21. Amphibian–habitat relationships in upland habitats ten-years post-logging. Thick black lines and bold text indicate habitat variables that describe the gradient along each axis. Shrub_T = tall shrubs; Shrub_L = low shrubs; CC = overstory canopy cover; CWD = coarse woody debris; Fern = fern cover; Grass = grass cover; Herb = herbaceous cover; Litt = deciduous leaf litter cover; LTD = deciduous leaf litter depth; Moss = moss cover; Rock = rock cover; Soil = exposed soil; Maple = vine maple; Dec_Sm = bigleaf maple and red alder < 1.3 m; Dec_Lg = bigleaf maple and red alder >1.3 m; Con_Sm = western redcedar and western hemlock <1.3 m; Con_Lg = western redcedar and western hemlock >1.3 m; Dfir_Sm = Douglas-fir <1.3 m; Dfir_Lg = Douglas-fir >1.3 m. AMGR = *Ambystoma gracile*; ASTR = *Ascaphus truei*; DITE = *Dicamptodon tenebrosus*; ENES = *Ensatina eschscholtzii*; PLVE = *Plethodon vehiculum*; RAAU = *Rana aurora*; TAGR = *Taricha granulosa*.

Habitat Structure: Treatments

In theory, there were to be two distinct riparian treatments to compare to a control condition; however, this was not the case. By proxy, the assignment of treatment type to a site should define buffer width (i.e., state = narrow, uniform; modified = wider, more variable). Plotting the normalized score of the principle components analysis (PCA) on the habitat-attribute table relative to the treatments provides a visual indication that the state and modified buffer sites were not measurably different after logging (Figure 5-22). Prior to logging (blue squares), all three treatments occupied somewhat different positions in space. Although all sites within each treatment were selected based on specific criteria, the riparian zones within each stand were likely different due to the dynamic nature of riparian forests. Two years after logging, the riparian habitats at both the state and modified treatments diverged from their original pre-treatment positions; but they did not separate relative to each other in space, and both treatments could generally be described by the same suite of habitat variables. The only real differences were the greater number of large deciduous trees at the modified buffer treatment and more moss growing at the state buffer sites. In the 10-year post-harvest sampling interval, the modified and state treatments diverged from their 2-year post-harvest position; however, they occupied a similar position in space relative to each other. If the riparian treatments were distinctly different, the expectation would be that immediately following logging the two treatments would be positioned far apart in space, and that over time, although the riparian habitats changed, the separation in space would continue to be apparent and measurable, with some degree of convergence due to the growing of adjacent upland forest. While some of the habitat attributes varied relative to treatment and transect, the vegetation structure and complexity of treatments was not great enough to produce two distinct treatments.

The riparian habitats at the control sites varied across time and in association with both the modified and state buffer treatments. Because the controls continued to occupy a position in space similar to the buffer sites, with respect to the habitat attributes measured, comparisons of relative abundance among treatments was justified. Comparisons of the habitat associations of terrestrial salamanders among treatments were also validated.

Examination of habitat conditions for the upland transects at each treatment type shows that the pre-harvest conditions at the modified and state sites were very similar to the control sites, and that the control sites remained relatively constant across time (Figure 5-23). Clustering of the two post-treatment interval data sets for the modified and state sites provides evidence that the logging prescriptions produced ecologically similar results with regard to habitat structure over time. This is what would be expected considering that forest-harvest prescriptions along upland transects were essentially the same at modified and state buffer sites. Immediately after harvesting, the upland transects were largely devoid of vegetation. By ~10 years later, modified and state sites showed clear signs of succession, supporting an increased abundance of Douglas-fir, deciduous trees, and berry-producing shrubs (Figure 5-23). As expected, control sites did not change as much as the logged sites.

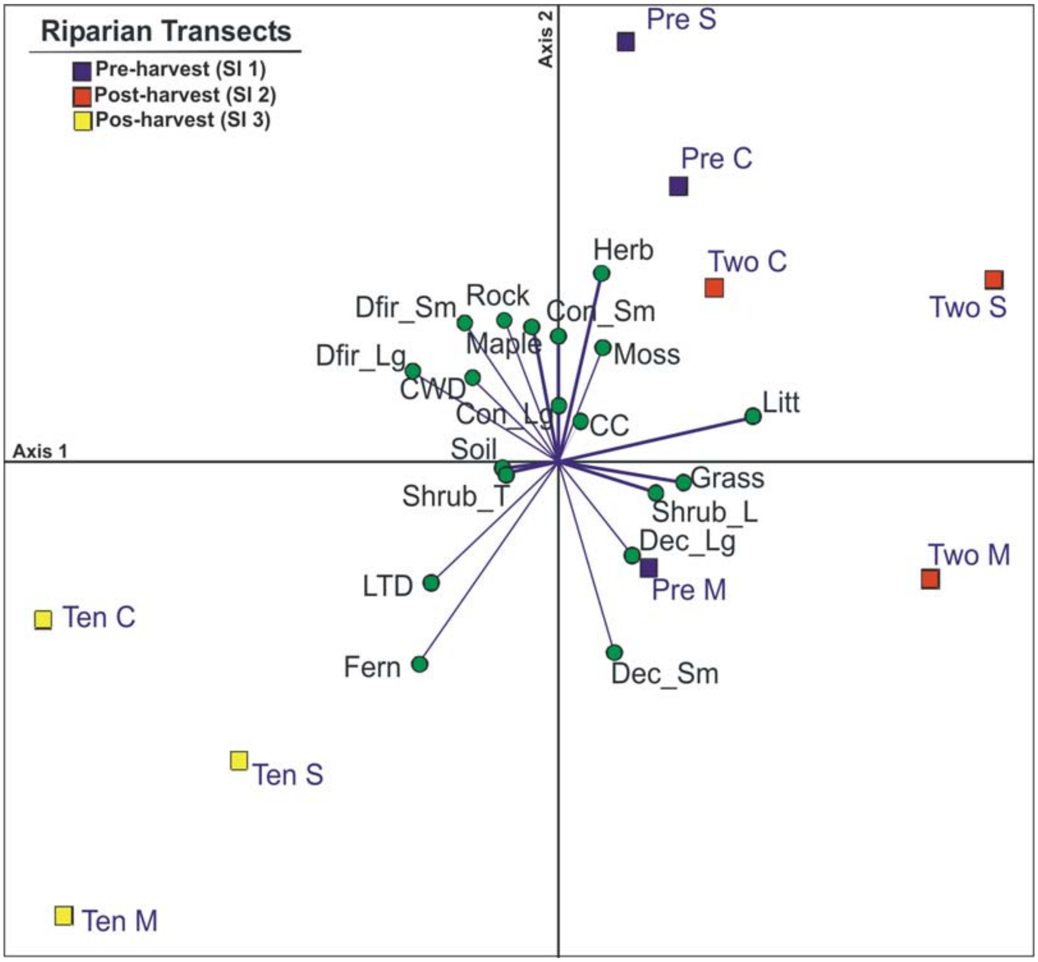

Figure 5-22. Biplot of riparian transect habitat condition by treatment and time interval. The data in this figure represent the mean condition as derived from the normalized score of the PCA on the habitat data matrix. Thick lines denote variables that describe the gradients along each axis. SI 1 = 1992/1993; SI 2 = 1995/1996; SI 3 = 2003/2004; Pre C = Pre-harvest control; Pre M = Pre-harvest modified; Pre S = Pre-harvest state; Two C = Two-year post-harvest control; Two M = Two-year post-harvest modified; Two S = Two-year post-harvest state; Ten C = Ten-year post-harvest control; Ten M = Ten-year post-harvest modified; Ten S = Ten-year post-harvest state; Shrub_T = tall shrubs; Shrub_L = low shrubs; CC = overstory canopy cover; CWD = coarse woody debris; Fern = fern cover; Grass = grass cover; Herb = herbaceous cover; Litt = deciduous leaf litter cover; LTD = deciduous leaf litter depth; Moss = moss cover; Rock = rock cover; Soil = exposed soil; Maple = vine maple; Dec_Sm = bigleaf maple and red alder <1.3 m; Dec_Lg = bigleaf maple and red alder > 1.3 m; Con_Sm = western redcedar and western hemlock <1.3 m; Con_Lg = western redcedar and western hemlock >1.3 m; Dfir_Sm = Douglas-fir <1.3 m; Dfir_Lg = Douglas-fir >1.3 m.

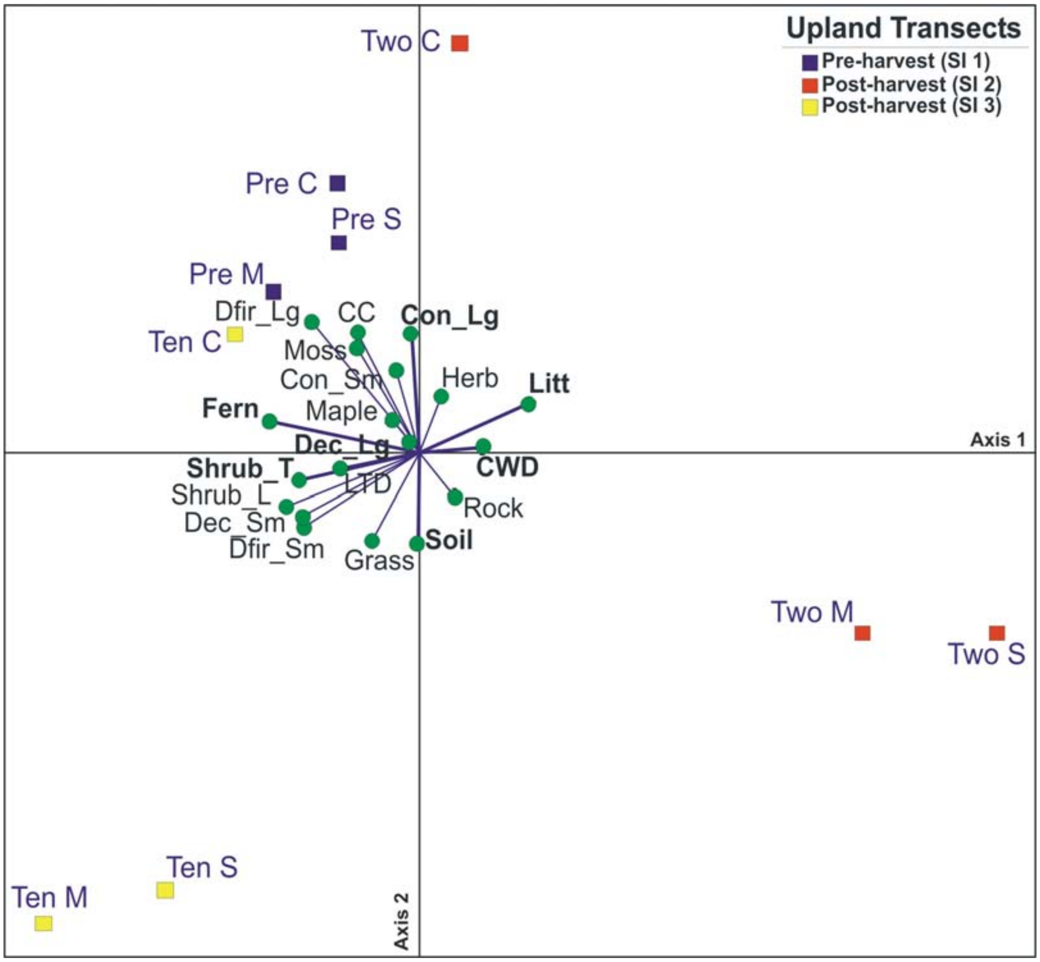

Figure 5-23. Biplot of upland transect habitat condition by treatment and time interval. The data in this figure represent the mean condition as derived from the normalized score of the PCA on the habitat data matrix. Thick lines denote variables that describe the gradients along each axis. SI 1 = 1992/1993; SI 2 = 1995/1996; SI 3 = 2003/2004; Pre C = Pre-harvest control; Pre M = Pre-harvest modified; Pre S = Pre-harvest state; Two C = Two-year post-harvest control; Two M = Two-year post-harvest modified; Two S = Two-year post-harvest state; Ten C = Ten-year post-harvest control; Ten M = Ten-year post-harvest modified; Ten S = Ten-year post-harvest state; Shrub_T = tall shrubs; Shrub_L = low shrubs; CC = overstory canopy cover; CWD = coarse woody debris; Fern = fern cover; Grass = grass cover; Herb = herbaceous cover; Litt = deciduous leaf litter cover; LTD = deciduous leaf litter depth; Moss = moss cover; Rock = rock cover; Soil = exposed soil; Maple = vine maple; Dec_Sm = bigleaf maple and red alder <1.3 m; Dec_Lg = bigleaf maple and red alder >1.3 m; Con_Sm = western redcedar and western hemlock <1.3 m; Con_Lg = western redcedar and western hemlock >1.3 m; Dfir_Sm = Douglas-fir <1.3 m; Dfir_Lg = Douglas-fir >1.3 m.

Discussion

Species Richness, Diversity, and Relative Abundance

This study provides a measure of the short- (2-year) and medium-term (10-year) responses of terrestrial amphibian populations to the effects of clearcut logging of second-growth forest. It also evaluates and compares the efficacy of riparian buffer widths in providing habitat attributes necessary to maintain the diversity and abundance of terrestrial amphibians over time. Our findings suggest that there is no uniform response by terrestrial amphibians to timber harvest or to the retention of riparian management zones (RMZs) in the Pacific Northwest. Rather, species showed individual responses that varied over time and between treatments and transects. The influence of site effects was evident for some species, while for others there is some suggestion of treatment effects, but the analyses were confounded by patterns of natural population change at both local and regional scales.

For most species of terrestrial amphibians, the retention of RMZs, regardless of width, ameliorated the impacts of clearcut logging, with no differences in relative abundance attributable to RMZ width. Only the relative abundance of the Coastal Giant Salamander was adversely affected on the state buffer sites, indicating that the narrow buffer width was not effective in maintaining the relative abundance of this species at levels comparable to the pre-harvest means. Moreover, because the relative abundance of this species increased on riparian habitats at the modified buffer sites, there is reason to suspect that the width and configuration of the modified buffer sites is sufficient to maintain the relative abundance of this species, at least in riparian habitats.

Logging upland forests have been implicated in the decline of amphibians in the Pacific Northwest (deMaynadier and Hunter 1995; Collins and Storfer 2003; Karraker and Welsh 2006); however, we found no evidence to suggest that logging directly affected the species richness, community similarity, or relative abundance of most species of amphibians captured. Two species (Coastal Giant Salamander and Coastal Tailed Frog) were negatively affected by timber removal in the upland habitats, and neither RMZ was able to offset the impacts of upland logging on the relative abundance of these species. Both species were either absent, or the relative abundance significantly lower, in upland habitats at the state and modified buffer sites 2 and 10 years after logging relative to the pre-harvest values. For Coastal Giant Salamander, this was associated with significant declines in the riparian habitats at the state buffer sites. The relative abundance of Coastal Tailed Frog was not affected in riparian habitats at either buffer treatment, suggesting that Coastal Tailed Frog persistence in riparian forest is not directly tied to higher canopy-cover values in the adjacent upland forest. However, logging of upland forest appears to have limited the dispersal of this species into upland habitat 2 and 10 years after logging, a trend that has been reported elsewhere (e.g., Corn and Bury 1989; Dupuis and Steventon 1999; Ashton et al. 2006).

The non-detection of Coastal Giant Salamander in upland habitats at the logged sites may be related to the effects of logging on the movement patterns of this species. In their retrospective study, Johnston and Frid (2002) found that logging restricted the movement patterns of Coastal Giant Salamander and that, after logging, salamanders remained significantly closer to the stream, spent more time in subterranean refuges, and had smaller home ranges than those in forested sites. Johnston and Frid (2002) also found that movement of Coastal Giant Salamander in riparian buffers was similar to that observed in forested sites and significantly different from clearcut sites. Our findings corroborate this: the mean relative abundance of Coastal Giant Salamander was higher in riparian buffers compared to upland habitats at logged sites and not markedly different compared to the upland habitats at the controls. Over time, the relative abundance of this species remained higher in upland habitats at control sites, where salamanders may have had larger home ranges and may not have been restricted in their movement.

Not only was the relative abundance of most species not affected by logging or measurably different on either buffer treatment, RMZ width was not associated with increased or different species richness or community similarity between riparian zones of the state and modified buffer sites and the controls or between the upland habitats at either treatment. In general, it appears that riparian buffer width, or even clearcut logging, did not affect the persistence of common species at each treatment, which is contrary to some literature (Vesely and McComb 2002) and congruent with others (Raphael 1991; Aubry 2000). Not all of these studies occurred in climactically equivalent regions of the Pacific Northwest, but they did involve similar species assemblages. The variation in results likely points to differences in species–habitat relationships that follow a north–south gradient. As such, the habitats (i.e., riparian or upland) used by terrestrial amphibian species at lower latitudes, where precipitation is lower and temperature somewhat higher, may differ from those reported in this study.

In Vesely and McComb's study (2002), sites were not repeatedly sampled across time to remove any potential site effects from the interpretation of their results. In fact, most studies evaluating the effects of forestry management on amphibian richness and abundance have not been well replicated; have not resampled the same sites before and after logging (and therefore could not account for site effects); and have reported on data collected over a relatively short time period (see reviews in deMaynadier and Hunter [1995] and Cushman [2006]), which did not allow for an investigation of the longer-term effects of clearcut logging on terrestrial amphibians.

The data collected during this study provided an opportunity to examine the longer-term post-treatment effects of logging on terrestrial amphibians. In this study, we were able to plot changes in the relative abundance of both Western Red-backed Salamander and Ensatina that appeared to reflect regional population change, which could have the potential to confound the interpretation of our data. However, regional population changes are not the only possible explanation for the observed changes in relative abundance. For example, changes in relative abundance could also have been related to environmental conditions, habitat structure and function, or forest succession.

The surface activity of terrestrial amphibians is tied to climactic conditions (Pilliod et al. 2003; Davic and Welsh 2004), which varied annually at our study sites. Although we do not have site-specific environmental data for all three sampling intervals, the broader regional weather data (precipitation, maximum and minimum temperatures) did not indicate a significant difference among years of sampling (Figure 5-8; Figure 5-9), suggesting that the environmental conditions present when sampling occurred cannot be used to explain the observed changes in relative abundance. We did not investigate the relationship between the number of surface-active animals and total density within the soil column, as has been done in previous research (Dupuis et al. 1995). Kelsey (2000) surmised that during periods of high precipitation, soil moisture may reach higher levels in clearcuts because of decreases in overstory interception rates. While the regional weather data indicated no difference among sampling years, site-specific differences may have existed within and across each sampling interval such that the surface activity of terrestrial amphibians could have been affected.

Microhabitat availability could also potentially explain the differences in relative abundance of Ensatina observed in each sampling interval. Bury and Corn (1988) reported a significant positive correlation between Ensatina and advanced decay–class logs and coarse woody debris, a result not reported by Aubry (2000). Our findings align more with Aubry (2000), as we did not find a relationship between the relative abundance of Ensatina and coarse woody debris or any other specific habitat feature (Figure 5-20; Figure 5-21).

Bury and Corn (1988) reported that Ensatina was more abundant in young forests (30–76 years old) compared to intermediate-aged stands, and there may be an association between forest age and the relative abundance of Ensatina. Forest communities develop through a continuum that, for conceptual purposes, has been grouped into sequential stages (e.g., Thomas 1979; Bunnell et al. 1999). Figure 5-24 depicts 6 seral (structural) stages of forest succession[9] as they are conceptualized for a forest stand culminating in an old-growth climax. Stage 1 begins with a vegetation community dominated by non-woody plants such as rushes, grasses, and forbs. By stage 2, woody shrubs and shade-intolerant conifer seedlings are present. At stage 3, conifers begin to dominate the site as they

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[9] Other authors have used 4 stages (e.g., Franklin 1990; Klenner et al.. 2000).

compete with other plants for light, water, and nutrients. During stage 4, the conifer canopy closes, effectively shading out vegetation on a forest floor that is becoming increasingly barren. At stage 5, conifers are mature, and weaker trees begin to die. Those trees will stand as snags that will eventually fall to the forest floor, where they become coarse woody debris. Shade-tolerant species of conifers become established in the understory. During stage 6, shade-intolerant species of conifers are gradually outcompeted by the shade-tolerant ones. Those shade-tolerant species continue to germinate and grow up in the understory. The number of standing dead trees and downed logs increases. Dead and fallen trees create gaps in the canopy that, together with other stand-mediated changes to microclimatic conditions, create suitable conditions for shade-intolerant species. The increasing diversity of plant species—and their increasing density, size, age, and state of decay—promotes the structural heterogeneity of the forest during stage 6.

Of the four explanations provided for the observed changes in relative abundance, we suggest that we observed a regional-level change in populations of Western Red-backed Salamander and Ensatina, which may have been affected by site-specific environmental conditions that varied across time. Forest succession also likely contributed to the overall decline of Ensatina in upland habitats. Because little is known about the proportion of surface-active terrestrial salamanders like Western Red-backed Salamander and Ensatina, additional studies should be considered to evaluate what proportion of the population is surface-active and the degree to which terrestrial salamanders move in the soil column. This would help address uncertainties about why numbers of Western Red-backed Salamander increased significantly 2-years following logging and then declined significantly in the 10-year post-harvest sampling interval, and also why the relative abundance of Ensatina showed a precipitous decline over time in upland habitats.

Although species richness did not vary significantly across time (Figure 5-10), species composition and relative abundance did. For example, Coastal Tailed Frog and Coastal Giant Salamander were virtually non-existent in upland habitats at state and modified buffer sites 10 years after logging (Figure 5-17), and the abundance of Ensatina declined in upland habitats at all treatments (Figure 5-15; Figure 5-17). The changes in species composition can be partially explained using standard conceptual models of forest succession. Coniferous forests typically display the pattern of species richness depicted in Figure 5-24. In that pattern, species richness is comparatively higher during the earliest and latest seral stages. As the tree canopy closes (around stage 4), the understory is shaded out, resulting in a decline in habitat suitability for some species. As succession progresses, the structural diversity and richness of plant species in the forest increases. This brings about an improvement in habitat suitability, which in turn facilitates an increase in wildlife-species richness. Although wildlife-species richness in early and late successional stages can be similar and can have some species in common, the makeup of the wildlife communities is notably different. In this study, it appears that populations of Ensatina responded directly to this model of forest succession, and one would expect that once the stand reaches old-growth status the relative abundance of Ensatina in upland habitats would increase—a prediction partially supported by Bury and Corn (1988). If the model depicted in Figure 5-24 is accepted, we would expect the relative abundance of Ensatina to be highest in forests between 30–80 years old and older than approximately 250 years, with a decline in relative abundance between 80 and 200 years.

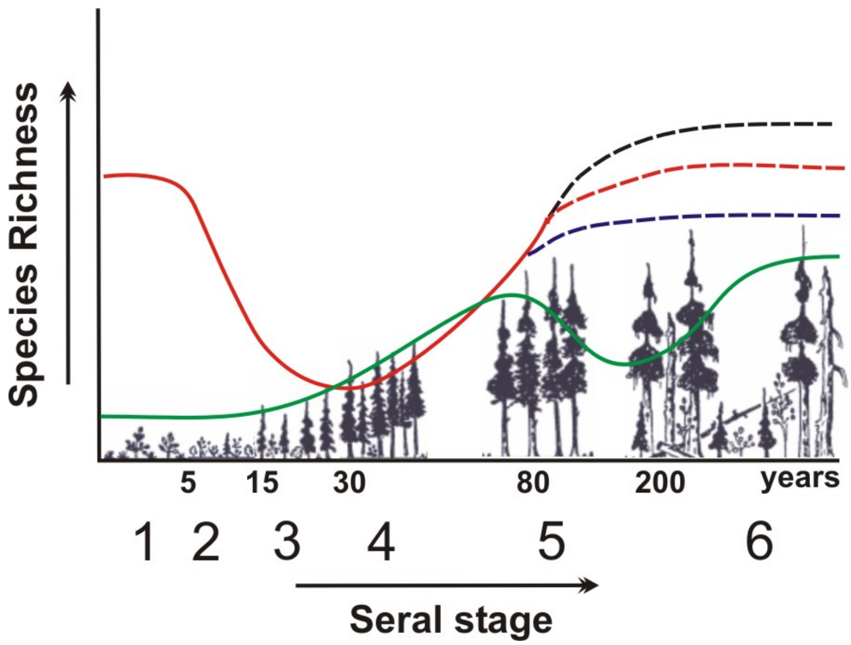

Figure 5-24. Conceptual model of the relative change in species richness (number of species) for wildlife communities as a function of forest seral stage, showing three possible outcomes (dashed lines) of lesser, similar, and greater species richness during the earliest and latest stages. The expected relative abundance of Ensatina is shown as a green line. Source: adapted from Franklin (1990) and Bunnell et al. (1999).

Several studies have suggested that buffers wider than those used in this study would be beneficial to amphibians. McComb et al. (1993) reported that a buffer width of 50 m adjacent to second- and third -order streams in Oregon would provide marginal habitat and that buffers should be >50 m wide to provide linkages between mature forests. Vesely and McComb (2002) estimated that RMZs 43 m wide would support total salamander abundance and that buffers of 47 m would support species richness at levels similar to unlogged sites. Recently, Crawford and Semlitsch (2007) found that a buffer width of 27 m would encompass 95% of the salamander assemblage and that an additional 50 m would ameliorate edge effects. They recommended that a total buffer of 92.6 m be used in southern Appalachian streams to maintain the biodiversity of amphibians.

Apparently, the minimum buffer width required to preserve amphibian diversity and relative abundance is related to geography. We found that RMZs of either 14 m (state buffer) or 30 m (modified buffer) around third- and fourth -order streams in western Washington function to preserve species richness and community similarity for most species of terrestrial amphibians and some species of semi-aquatic amphibians. In general, the 14 m–wide buffers were adequate for preserving the relative abundance of most species. In contrast, 30 m–wide buffers always preserved the relative abundance of all terrestrial and semi-aquatic amphibians assessed in this study. This suggests that for the preservation of species richness, community similarity, and relative abundance the modified buffer slightly outperformed the state buffer. The modified buffer configuration and associated habitat characteristics (i.e., the prescription) would serve as a better model for habitat mitigation strategies in forested landscapes of the Pacific Northwest and will likely aid in the preservation of biodiversity, at least for the biota included in this study. The variability of the modified buffer, both in terms of width and habitat characteristics, and the variability in species distribution and sensitivity to habitat change, dictates that RMZ delineation occur at a scale commensurate with the variability. For example, sites with lower species diversity and lacking sensitive species (i.e., those affected by habitat change) could have a riparian buffer that differs from a site with high species diversity and sensitive species. An understanding of amphibian diversity and distribution in western Washington would be invaluable when managing for amphibian biodiversity using RMZs.

Habitat Relationships

Despite continuing forest succession in riparian habitats and major changes in the upland forests, the habitat associations of the terrestrial amphibians sampled in this study remained surprisingly consistent over time. Co-inertia analysis (COIA) revealed that the species of terrestrial salamanders that we sampled (Western Red-backed Salamander and Ensatina) in western Washington are habitat generalists that are capable of adapting to significant habitat changes. Moreover, most species appeared to be able to seek out specific habitat attributes that provided suitable cover and protection, enabling them to persist on the landscape. Other species, such as Coastal Tailed Frog, however, were consistently associated with specific features on the landscape, indicating a higher degree of habitat specification.

The habitat associations identified for terrestrial amphibians at the western Washington study sites were generally consistent with the published literature, providing an indication that COIA can properly elucidate the relationships between habitat attributes and terrestrial amphibians in western Washington.[10] For example, the relative abundance of Western Red-backed Salamander was positively correlated with habitat attributes of a younger seral forest in all three sampling intervals. Vegetation diversity and cover, used to infer productivity and moisture regime of riparian habitats relative to upland forest, was important because populations of this species are often associated with rocky seeps, springs, and small streams in the northern Puget Trough and Cascade Range foothills in Washington (Ovaska and Gregory 1989; Leonard 1996; Petranka 1998). Coastal Tailed Frog was consistently associated with large coniferous trees (western redcedar, western hemlock, and Douglas-fir), a high percentage of canopy cover, and large amounts of coarse woody debris. This suggests that this species is associated with components of maturing forests, a notion that is corroborated by the literature (Dupuis et al. 1995; Dupuis and Waterhouse 2001). Coastal Giant Salamander was common but not abundant in riparian habitats, making it difficult to derive specific habitat correlates; however, this species was always captured in riparian habitats with relatively high canopy cover and abundant cover vegetation, such as ferns.

Other species present in riparian habitats did not show a consistent correlation with any particular habitat attributes, but because their relative abundance did not change over time, it is apparent that the riparian habitat structure and complexity retained at the state and modified treatments was sufficient to enable amphibians to persist at densities comparable to pre-harvest levels.

The riparian habitats in each of the treatments contained habitat components and heterogeneity that were suitable for the persistence of both amphibian abundance and diversity; however, because the state and modified treatments were not measurably different, it is not possible to comment on the effective RMZ width required to maintain terrestrial amphibian diversity and abundance. Given that the width of the riparian buffer averaged 14 m (state) and 30 m (modified), a buffer width within that range is sufficient to preserve habitat characteristics required to maintain species relative abundance up to 10 years after logging; however, these values should be used as a minimum width, because edge effects can be more pronounced when buffers are narrower (deMaynadier and Hunter 1998; Karraker and Welsh 2006). To ensure that species like Western Red-backed Salamander, Coastal Tailed Frog, and Coastal Giant Salamander persist in RMZs, it is imperative that these zones be large enough to be protected from wind events that could lead to extensive blowdown, which may result in

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[10] For the regional level of this study, COIA provided a good measure of habitat use. To examine species-specific habitat use at the site level, a study at the scale of use by the amphibian would be more appropriate.

reduced function of the riparian zone, and that the habitat structure of the riparian zone not be compromised by thinning or other silvicultural activities that could degrade the suitability of the RMZ for amphibians. In other words, once the RMZ is established, it should not be modified without additional research to determine further affects of riparian habitat modification on amphibians.

Populations of plethodontid salamanders (Western Red-backed and Ensatina) survived the large-scale habitat perturbation associated with clearcut logging; populations of both of these species were not adversely affected, as evidenced by the lack of significant changes in relative abundance attributable to treatment effects. The habitat associations of both of these species necessarily changed before and after logging because of the removal of the overstory and serious modification to the soil through compaction and scraping, and because of changes in forest-floor cover related to the removal of coarse woody debris, ferns, shrubs, and small trees. However, despite the magnitude of the change in habitat structure in upland habitats, both species persisted. This is due in part to their generalist habits and to their survival strategy. During periods of extreme environmental conditions, such as surface freezing temperatures or high temperatures, plethodontid salamanders will go underground and take refuge in small mammal burrows or natural crevices in the soil or talus (Stebbins 1954; Dumas 1956; Ovaska and Gregory 1989; Petranka 1998). It is conceivable that both Western Red-backed and Ensatina salamanders retreat to underground burrows during periods of large-scale habitat perturbations.

Unlike many Pacific Northwest plethodontid salamanders, Ensatina salamanders are often more abundant away from streams (McComb et al. 1993; Olson et al. 2007). In Washington State, Ensatina is the most dry-adapted of the four plethodontid salamanders that occur, and under relatively wet conditions this species tends to occupy the least-wet sites. In upland habitats of western Washington, Ensatina abundance was positively correlated with increasing canopy cover, large coniferous trees, and vine maple, suggesting that this species is associated with a variety of upland habitat attributes provided there is sufficient moisture. Western Red-backed Salamander occupied similar habitats in upland forest and its abundance was correlated with increasing cover of coarse woody debris. This helps explain why Western Red-backed Salamander populations were not affected by logging: clearcut sites typically have an abundance of coarse woody debris in the form of downed logs of small-diameter or non-merchantable timber. Additionally, although much of the coarse woody debris is piled and burned, there is an abundance of woody material left on the ground after logging. This likely provides suitable security and thermal habitat for Western Red-backed Salamander.

Some species were noticeably absent from upland habitats, and their absence can be readily explained through an assessment of their habitat relationships in upland forest prior to logging. Before logging, Coastal Tailed Frog relative abundance was positively correlated with increasing canopy cover. After logging, the canopy cover was effectively zero, as was the abundance of this species. Coastal Giant Salamander relative abundance was correlated with increasing numbers of large coniferous and deciduous trees in the overstory. After logging, both of these habitat components were removed, which in turn may have affected the movement of Coastal Giant Salamander, restricting it to riparian habitats (Johnston and Frid 2002). Grialou et al. (2000) found that Coastal Giant Salamander was absent from sites that were clearcut 2 years after logging.

In temporal studies evaluating the effectiveness of a treatment in providing wildlife with habitat to maintain biodiversity, it is necessary to be able to describe the habitat associations of those species prior to treatment application and to monitor how habitat associations vary across time. Although COIA continues to be an underused multivariate technique, it has been used recently to describe species–habitat associations (e.g., Diallo et al. 2006; Sirami et al. 2007). The utility of a COIA was evident in this study, as it provided an efficient way to visualize and describe the habitat associations of many species of terrestrial amphibians before and after logging and to compare the habitat attributes in riparian and upland habitats at two buffer treatments.

Forest managers require direction from ecologists to manage for biodiversity on landscapes managed for timber. This requires that ecological studies occur at broader scales so that regionwide patterns of diversity, relative abundance, and habitat associations can be detected. With an increase in scale comes an increase in cost and time required to complete the study. As a trade-off, ecologists are faced with reducing the number of sites and treatments or reducing the geographic scale of the study. If regionwide biodiversity management is the premise for a study, then a reduction in replication or treatments is often the only way to make the study economically viable. Under theses conditions, attempting to define species–habitat associations using common and widely reported methods of ordination would be erroneous, because a major assumption of those techniques would be violated: the number of samples relative to the number of environmental variables would be too low. The introduction of COIA to ecologists by Dolédec and Chessel (1994) provided a viable, robust method with which to describe habitat relationships at a broader scale that was not constrained by the need for many samples. As broader-scale studies are undertaken to establish better biodiversity management plans that include species–habitat relationships, COIA will likely increase in use because of its adaptability, ease of interpretation, and ability to properly elucidate habitat relationships.

The interpretation of our results is subject to a number of caveats that should be considered for future studies of this type. First, the regional scale of this study represents a dilemma in terms of comparing differences in species richness at the treatment level. Because not all amphibians in Washington State are ubiquitous or widespread, there should be differences in species richness that are independent of treatment. For example, certain species (Dunn's Salamander and Columbia Torrent Salamander) are restricted to the southwestern portion of the state (i.e., the Abernathy, Ryderwood 860 and 1557, and Vail sites; Figure 5-1), whereas other species (e.g., Western Red-backed Salamander, Ensatina, and Rough-skinned Newt) are ubiquitous in their distribution. Typically, the geographic range of a species does not need to be considered because of the small spatial scale at which most studies are conducted (deMaynadier and Hunter 1995; Cushman 2006). However, for regionwide studies, the inclusion of species that have small geographic ranges can skew the interpretation of species richness data.

Second, accurate differences in species richness would be noted only if all species could be caught using the same capture methods; however, this is not likely to be the case. Mackenzie et al. (2004) point out that many ecological studies evaluating the spatio-temporal patterns of species assume equal detection of all species present. This assumption can lead to incorrect inferences about patterns of species richness, because some species may be inaccurately labeled as absent when they simply were not detected. In most cases, the more abundant species were captured at each site (Table 5-10), but even the widely distributed species were not always detected (e.g., *Taricha granulosa*) using the capture methods employed during this study. Some species of amphibians (e.g., Pacific Treefrog) are not easily captured in pitfall traps, while others (e.g., Larch Mountain Salamander) may be so patchy in their distribution that even if a site occurred in their expected range, they might not be captured by pitfall trapping. Finally, the behavior of other species (e.g., Long-toed Salamander) may preclude detection through pitfall trapping. In most studies where pitfall tapping is used as the primary means for species detection, pitfall traps are used in conjunction with drift fences that increase the catchability of the trap (see Heyer et al. 1994). Drift fences were not used in this study, which may have contributed to lower than expected species richness at each site (and treatment) and, perhaps, to lower relative abundance values.

Third, to properly assess differences in species richness, sampling needs to coincide with species phenology, to occur via different sampling techniques, to consider geographic distribution, and to consider species density (as a proxy for rarity). Although amphibian species may be active during spring and fall, it is evident that the surface activity of certain species is greater in one season relative to the other and that different capture methods are better for certain species. As an example, spring amphibian data obtained through time-constrained searches and hand-capture from a subset of the western Washington study sites in the second post-harvest sampling interval show that, in spring, Northern Red-legged Frog was more abundant and more readily detected through time-constrained searches (i.e., hand-capture; Figure 5-25) than through fall pitfall trapping. Similarly, although Western Red-backed Salamander was the most abundant species detected in the fall in every sampling interval, it represented a larger proportion of the total sample detected in the spring using hand-catch methods. To obtain an accurate representation of species relative abundance and species richness, sites should be sampled for amphibians using a combination of methods throughout the year. This would likely result in more accurate species richness and total abundance values.

Table 5-10. Expected versus observed species presence in western Washington. Shaded cells with bold values indicate that the species was detected at all of the expected sites. SI 1 = 1992/1993; SI 2 = 1995/1996; SI 3 = 2003/2004; Obs = observed.

	Sites Expected			SI 1		SI 2		SI 3	
Species	SI 1	SI 2	SI 3	Obs	%	Obs	%	Obs	%
Ambystoma gracile	18	18	17	11	61	10	56	11	65
Ambystoma macrodactylum	18	18	17	5	28	3	17	2	12
Ascaphus truei	18	18	17	8	44	8	44	9	53
Bufo boreas	18	18	17	1	6	1	6	2	12
Dicamptodon tenebrosus	13	13	11	11	85	7	54	8	73
Ensatina eschscholtzii	18	18	17	13	72	16	89	15	88
Hyla regilla	18	18	17	2	11	3	17	2	12
Plethodon dunni	4	4	3	4	**100**	3	75	2	67
Plethodon larselli	4	4	4	0	0	0	0	0	0
Plethodon vandykei	7	7	7	0	0	0	0	0	0
Plethodon vehiculum	18	18	17	16	89	16	89	17	**100**
Rana aurora	18	18	17	9	50	9	50	10	59
Rana cascadae	13	13	11	0	0	1	8	0	0
Rhyacotriton cascadae	3	3	3	1	33	1	33	1	33
Rhyacotriton kezeri	3	3	3	2	67	2	67	3	**100**
Taricha granulosa	18	18	17	6	33	6	33	6	35

The utility of sampling sites in different seasons and with different methods to improve the accuracy of species richness values is exemplified by the number of species detected at the Blue Tick modified site in the third sampling period. In the spring, 5 species were detected in riparian habitats, whereas only 1 was documented in the upland for 5 unique species at the site. In the fall, 4 species were detected in the riparian, 3 in the upland for 4 unique species at the site. The richness measure used in this study was derived only from the fall data. If data from both sampling intervals had been combined, the total number of species detected would increase to 6. If spring sampling occurred at all sites, it is likely that different richness values would be derived for each site, habitat type, and treatment and could potentially influence the conclusions.

The selection of study sites for this study was constrained by finding enough sites of similar rotation age that had streams with similar characteristics. Wilkins and Peterson (2000) showed that habitat quality for headwater amphibians in southwestern Washington is influenced by landform characteristics, including basin lithology. Wilkins and Peterson (2000) further suggest that periodic landscape disturbances, such as clearcut logging, likely interact with the variation in landform to organize amphibian assemblages. This study occurred after the RMZ study was implemented in 1991; however, had regionally based data such as lithology been applied to the site selection process, the results of this longer-term monitoring study might have been different.

Finally, the use of count data (as a proxy for abundance) and species richness measures may provide misleading results when evaluating the temporal effects of timber harvest on terrestrial amphibians. Because relationships between counts and abundance are rarely demonstrated, it is difficult to calculate how well counts reflect temporal changes in population numbers. Furthermore, a thorough understanding of population dynamics depends not only on knowing abundance, but also requires knowledge of the fundamentals of population processes, such as survivorship; these are the parameters we need to study to determine whether and how populations are affected. The collection of pre-treatment data, including information on life-history parameters and movement patterns, are critical in determining the temporal and spatial response of terrestrial amphibians to forest harvesting (Maxcy and Richardson 2000). The generally limited dispersal and movement capabilities of many terrestrial amphibians, particularly terrestrial salamanders, may limit their ability to disperse to more suitable habitat following large-scale habitat perturbations, such as clearcut logging, which may contribute to effects of forest harvest on these species (Stebbins 1954; Ovaska 1988; Grialou et al. 2000). An understanding of home-range size and seasonal habitat use prior to logging would also help elucidate the effects of logging on terrestrial amphibians. Furthermore, because precipitation can influence the surface activity of terrestrial salamanders such as Western Red-backed and Ensatina, accurate precipitation data need to be collected from the sites being sampled to remove any potential environmentally induced bias from capture rates of terrestrial amphibians.

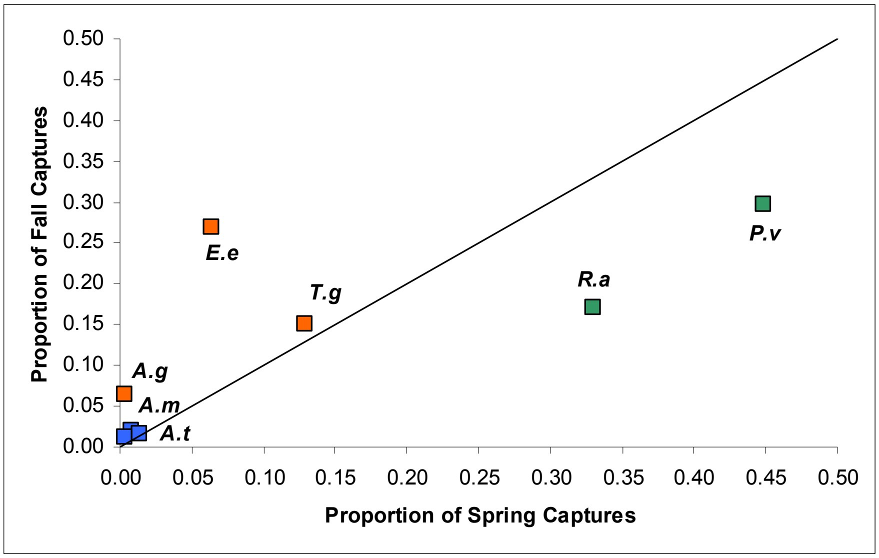

Figure 5-25. Proportional catch of amphibians detected in spring and fall from a subset of sites (*n* = 6) in western Washington. *A.g* = *Ambystoma gracile*; *A.m* = *Ambystoma macrodactylum*; *A.t* = *Ascaphus truei*; *E.e* = *Ensatina eschscholtzii*; *P.v* = *Plethodon vehiculum*; *R.a* = *Rana aurora*.

Conclusions

This study was the first longer-term study utilizing a Before-After-Control-Impact (BACI) design to evaluate the medium-term efficacy of RMZs in providing habitat for wildlife. Our results showed that the variability in response to upland clearcut logging by amphibians makes it difficult to define management strategies that are effective for all species present. In general, the relative abundance, richness, community similarity, and habitat associations of amphibians did not change as a result timber harvest or the retention of RMZs around third- and fourth-order streams in western Washington. However, this was not a global response, as the relative abundance of Coastal Giant Salamander was not maintained in the riparian habitats at the state buffer treatments, and both Coastal Giant Salamander and Coastal Tailed Frog declined in numbers in upland habitats at both the modified and state buffer sites. When the relative abundance of other species changed, it was attributable to local or regional population changes.

The COIA provided an indication of species–habitat associations over time, and no landscape feature appears to be missing from the state buffer sites that would influence the decline of Coastal Giant Salamander. The more likely reason for Coastal Giant Salamander decline is related to the reduced area within the RMZ, which may have forced the species to seek out larger stands of maturing forests so that it could occupy large enough areas to fulfill its life requirements. From a management perspective, this would suggest that a riparian buffer width of at least 30 m is required to maintain the relative abundance and richness of amphibians in western Washington.

The reason for the decline in numbers of Coastal Tailed Frog in upland habitats at the state and modified buffer sites was evident through the COIA. Two and 10 years after logging, there were no large Douglas-fir trees in the upland. Based on the habitat associations described for Coastal Tailed Frog during the pre-harvest sampling interval and at the controls over time, it appears that this species requires habitat characteristics associated with maturing coniferous forest dominated by Douglas-fir. When these conditions are not met, the numbers of Coastal Tailed Frog declines. Presumably, Coastal Tailed Frog moves to adjacent maturing stands of coniferous forest; however, a study comparing the relative abundance of the species in upland clearcuts relative to adjacent stands of maturing timber is required to test this hypothesis.

Although precipitation levels did not differ significantly on a regional scale during this study, rainfall is not always evenly distributed across a given area. It is possible that local levels of precipitation varied among our sample sites to the point where precipitation influenced the surface activity and movement patterns of amphibians. This may partially explain the differences in relative abundance observed for certain species; however, for species like Ensatina and Coastal Tailed Frog, it is more likely that forest succession and lack of optimal habitat were stronger determinants of changes in relative abundance. For Western Red-backed Salamander, the large increase in numbers 2 years after logging is difficult to explain and could have been related to localized levels of precipitation.

Management Implications

Extensive forestry management in the Pacific Northwest has historically included clearcutting, prescribed fire, regeneration by seed trees, protection from forest fires, and salvage of timber killed by windthrow or disease (Carey and Harrington 2001). In the last 20 years, planting of Douglas-fir has increased, with most forest stands >40 years old originating from seed. Current management practices include the retention of seed trees and coarse woody debris and the planting of seedlings. Implicit in the shift to ecosystem management has been the retention of RMZs around perennial fish- and non–fish-bearing streams, which have been required in some form since 1978.

During the past 10–15 years, increased attention has focused on defining and delineating riparian areas. McComb et al. (1993) reported that a buffer width of 50 m adjacent to second- and third-order streams in Oregon would provide marginal habitat and that buffers should be >50 m wide to provide linkages between mature forests. Vesely and McComb (2002) estimated that RMZs 43 m wide would support total salamander abundance and that buffers of 47 m would support species richness at levels similar to unlogged sites. Recently, Crawford and Semlitsch (2007) found that a buffer width of 27 m would encompass 95% of the salamander assemblage and that an additional 50 m would ameliorate edge effects. They recommended that a total buffer of 92.6 m be used in southern Appalachian streams to enable salamander biodiversity to persist. We found that a minimum buffer width of 30 m around third- and fourth-order streams in western Washington would maintain species richness and relative abundance values at levels similar to those recorded prior to logging, at least for those species sampled in this study. For less-common or sensitive amphibians species, or for species of amphibians that primarily occupy riparian habitats (e.g., *Plethodon dunni*, *Rhyacotriton* spp.), buffer widths may need to be larger than those established for more-common, generalist species that occur in western Washington.

Despite these recommendations for buffer width, the minimum width of an RMZ continues to be a geographic issue related to the implementation of regionwide management plans that are too general to account for local variability in species richness and diversity. Yet, the research on the benefits of RMZs is clear (deMaynadier and Hunter 1995; Vesely and McComb 2002; Cushman 2006), and most jurisdictions in Canada and the United Sates require the retention of riparian forest adjacent to upland logged habitats (Lee et al. 2004). The problem faced by land managers is that, far too often, management plans become specific to one group of organisms and do not use biodiversity preservation as the basis for management.

Although RMZs retain structural heterogeneity and complexity, the production of homogenous, simplified, even-aged stands of second-growth Douglas-fir forests in upland habitats may, over the long term, lead to localized extirpations of species like the Coastal Giant Salamander. Therefore, it is important to consider not only the geographic relationship to riparian management, which will provide an indication of species richness and abundance, but also to consider the frequency of upland rotation and the type of prescription applied to the upland forest. For example, Perkins and Hunter (2006) recommended that riparian management be a two-tiered approach, with no-cut zones in the riparian forest and a partial-cut zone in the adjacent upland. This approach would protect the riparian forest from blowdown and minimize edge effects (deMayandier and Hunter 1998).

For species that decline in abundance because of logging, management strategies to mitigate these declines have been proposed. Carey and Harrington (2001) concluded that management-induced homogeneity and simplification (i.e., reduction of complexity) posed a real danger to small-mammal communities becoming non-supportive of predator populations. They proposed that active management for habitat complexity on long rotations (>90 years) may be necessary to conserve the diversity attributed to maturing forests. Because terrestrial amphibian communities are inherently linked to many forest ecosystem processes, the development of biodiversity management strategies must be considered at a broader scale and should not be based on single-species conservation. The introduction of heterogeneity into homogenous, even-aged stands has been shown to have positive effects on biodiversity and ecosystem function in the short term (<5 years; Carey 2003), and longer-term studies are required to determine if the positive effects are long-lasting.

Since this study's inception in 1991, there have been changes to Washington Forest Practices Rules and Regulations that have implications for our findings. The Forest Practice Rules of 1988 required a riparian buffer that met the state configuration (i.e., not less than 7.5 m from the ordinary high-water mark) for both fish-bearing (Type 3) and non–fish-bearing (Type 4) perennial streams. The development of the modified riparian prescription was an alterative to the state buffer that provided for greater structural complexity, heterogeneity, and ultimately greater diversity of both flora and fauna. Lee et al. (2004) identified the current trend in riparian management prescriptions as tending towards site specific buffers that vary amongst broadly similar harvest areas to within a single harvest area. The Washington Forest Practice Rules of 2001 seem to fit this model, and aside from changing how watercourses are defined, are site-specific with many options. Lee et al. (2004) suggest that the primary benefit of this type of prescription is the identification of well-defined criteria to delineate the riparian zone. These criteria are specifically defined by the modifying factors selected by jurisdictions, and buffers are applied in a predictable response to these criteria. The caveat to tailor-made buffers is the greater complexity in guidelines. For example, Washington now describes 3 zones of management for Type F streams (Type 3): (1) the core, (2) inner, and (3) outer zones, with various levels of tree removal, thinning, hardwood conversion, and other activities permitted within these zones depending on the site class of land, the management harvest option, and the bankfull width of the stream. In general, the site class will form the RMZ prescription. It appears that riparian management in western Washington has become (necessarily) site-specific. This implies that applying a one-size-fits-all approach to riparian management has been set aside in favor of a more intensive approach that considers the specifics of the site, which is consistent with Lee et al. (2004). For Type Np streams, or perennial non–fish-bearing streams (Type 4), the RMZ rules are slightly less complicated; however, there is ample room for site-specific management. Considering that the RMZ rules have evolved to embrace a site-specific approach, it seems only fitting that forest management in general adopt the same principals when considering biodiversity. Managing for biodiversity on a regional scale does not account for variation at the landscape or site level within the region, which could have dire consequences for species that are limited geographically, that occur at naturally low densities, or that are particularly sensitive to habitat change. As an alternative to the status quo, management that focuses on stand heterogeneity (as opposed to tree-species monocultures with reduced diversity) should produce landscapes with high biocomplexity and biodiversity at multiple scales, and therefore these landscapes should be resilient in the face of disturbance (Holling 2001).

Several factors potentially influence the veracity of this study's results or limit the extent to which the conclusions can be used to aid in the management and conservation of terrestrial amphibians in coniferous forests of the Pacific Northwest. Many of these factors were unmeasured and are presented to promote thought and aid in the development of future studies of the interactions between amphibians and habitats in managed landscapes.

First, if the relative abundance of amphibians at logged sites is overestimated, the severity of the harvest on the amphibian population is underestimated and, conversely, underestimating the relative abundance of amphibians at treated sites results in overestimation of the severity of the harvest (Knapp 1999). Several factors can influence surface activity of amphibians, including precipitation (Gibbons and Bennett 1974; Johnston and Frid 2002), elevation, population size (Hairston 1980, 1986), and food availability (Mitchell et al 1996). Of these factors, the effects of upland timber harvest on amphibian food availability have not been researched in the Pacific Northwest.

Second, accurate comparisons of species richness values among treatments and transects requires that all species potentially present at a site be documented to avoid falsely labeling a species as not present. This requires the use of various methods of species detection and sampling that spans more than one season. The sampling used in this study was not suited to the detection of all species potentially present at each site and was restricted to the fall season. Expanding the sampling period to include the spring and including other sampling techniques would undoubtedly have increased the species richness values recorded during this study.

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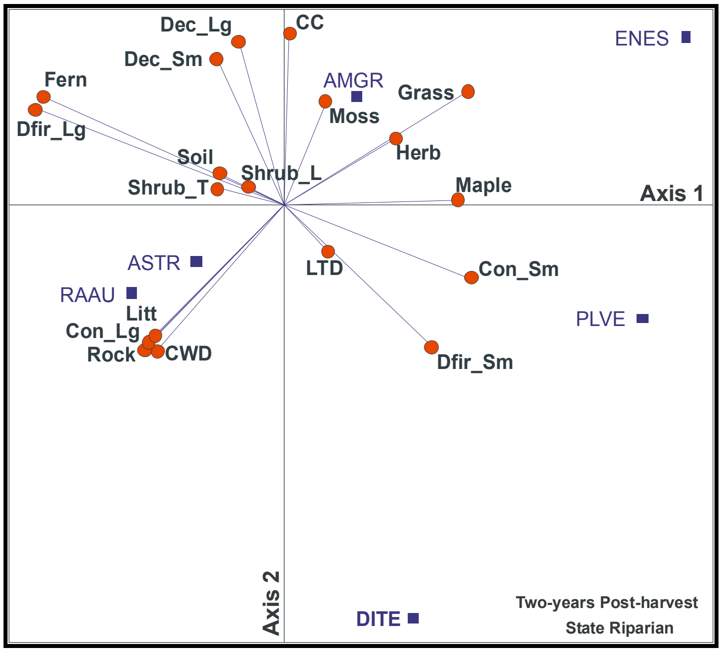
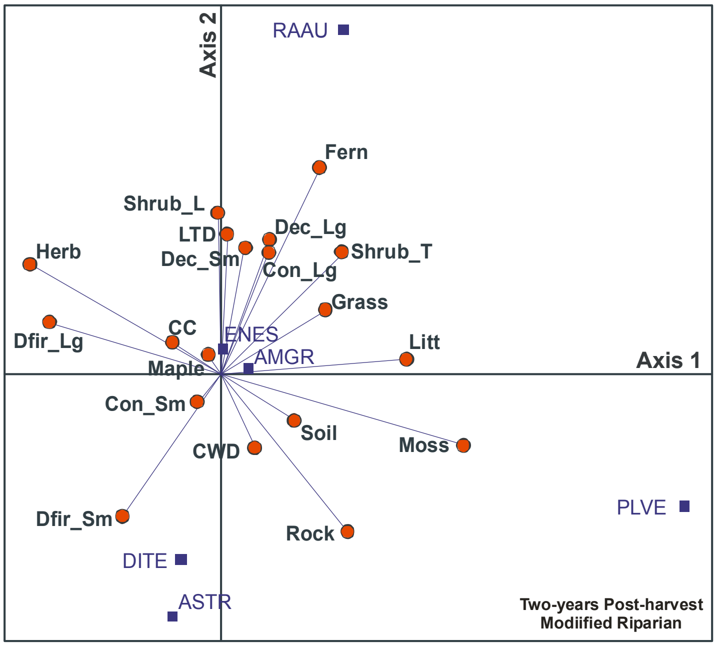
Appendices

Appendix 5-A. Ecological description of each site sampled in western Washington, 1992–2004. Refer to Figure 5-1 for distribution of study sites. – data not available; OP/SW = Olympic Peninsula and SW Washington; PT = Puget Trough; WSC = western slopes and crest, or Washington Cascades physiographic provinces.

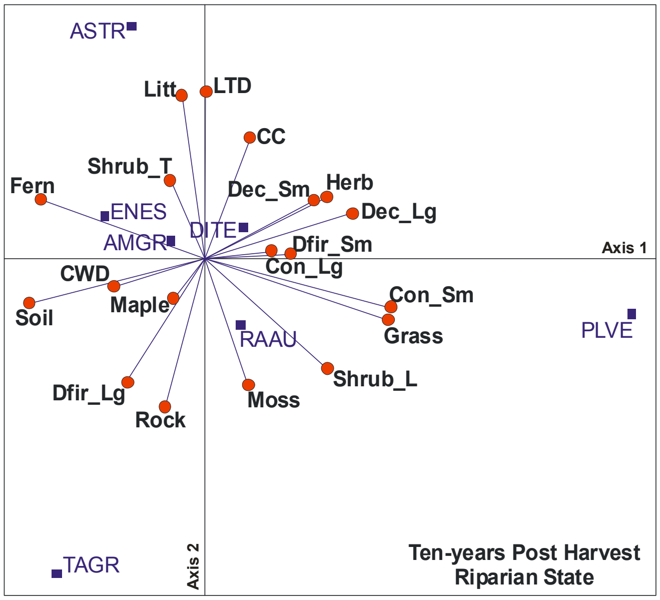
			Physiographic	Elevation (m ASL)			Stand Age			Stream Type		Bankfull		Riparian	
Site Name	Treatment	County	Province	Min	Max	Slope (%)	Year 1	Year 14	Dominant Tree	1988	2000	Width (m)	Shape	CC (%)	Lithology
Abernathy	Control	Cowlitz	OP/SW	362	491	16.0	55-65	68-78	Douglas-fir	3	F	5.04	Sloping	71.0	Basalt flows
Elbe Hills	Control	Pierce	WSC	554	609	10.0	55-67	68-80	Douglas-fir	4	F	3.20	V-shaped	91.0	Volcanclastic deposits or rocks
Hotel Creek	Control	King	PT	277	303	4.0	55-71	68-84	Douglas-fir	4	F	2.38	V-shaped	99.0	Contineta glacial outwash - Fraser Age
Porter Creek	Control	Thurston	PT	285	290	3.0	55-75	68-88	Douglas-fir	3	F	5.98	V-shaped	95.0	Basalt flows and flow breccias
Taylor Creek	Control	King	PT	415	424	2.0	55-81	68-94	Douglas-fir	2	F	6.47	V-shaped	93.8	Contineta glacial till - Fraser Age
Vail	Control	Lewis	PT				55-82	68-95	Douglas-fir	5	N				Andesite Flows
West Fork Falls Creek	Control	Thurston	PT	379	463	20.0	55-83	68-96	Douglas-fir	3	F	3.26	V-shaped	92.0	Basalt flows and flow breccias
Bluetick	Modified	Grays Harbor	PT	110	151	7.0	55-66	68-79	Douglas-fir	4	F	2.02	Undercut	94.0	Marine sedimentary rocks
Eleven Creek 31	Modified	Lewis	PT	493	561	6.0	55-68	68-81	Douglas-fir	4	N	3.30	V-shaped	92.5	Andesite flows
Griffen Creek	Modified	King	WSC	157	188	5.0	55-70	68-83	Douglas-fir	2	F	4.06	V-shaped	93.0	Contineta glacial till - Fraser Age
Ms. Black	Modified	Thurston	PT	67	92	3.0	55-73	68-86	Douglas-fir	3	F	1.82	Undercut	83.0	Basalt flows and flow breccias
Ryderwood 860	Modified	Cowlitz	OP/SW	303	337	8.0	55-78	68-91	Douglas-fir	3	F		V-shaped	75.0	Nearshore sedimentary rocks
Side Rod	Modified	Pierce	WSC	520	548	6.0	55-79	68-92	Douglas-fir	4	N	2.70	V-shaped	90.0	Volcanclastic deposits or rocks
Eleven Creek 32	State	Lewis	PT	415	501	10.0	55-69	68-82	Douglas-fir	4	N	1.32	V-shaped	98.0	Andesite flows
Kapowsin	State	Pierce	WSC	443	450	2.0	55-72	68-85	Douglas-fir	3	N	3.08	V-shaped	51.0	Basaltic andesite flows
Night Dancer	State	Grays Harbor	PT	125	201	11.0	55-74	68-87	Douglas-fir	5	F	1.44	V-shaped	96.0	Marine sedimentary rocks
PotPourri	State	Thurston	PT	239	249	7.0	55-76	68-89	Douglas-fir	3	F	5.18	V-shaped	74.0	Basalt flows and flow breccias
Ryderwood 1557	State	Cowlitz	OP/SW	333	376	7.0	55-77	68-90	Douglas-fir	4	N	3.30	V-shaped	77.0	Nearshore sedimentary rocks
Simmons Creek	State	Lewis	WSC	422	471	8.0	55-80	68-93	Douglas-fir	3	F	3.22	V-shaped	80.0	Basaltic andesite flows

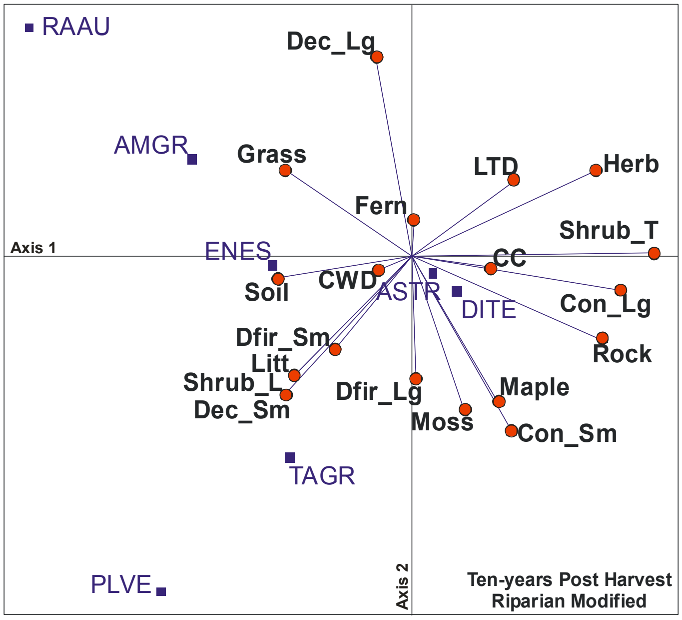
Appendix 5-B. Amphibian–habitat relationships in riparian habitats two-years post-logging. Shrub_T = tall shrubs; Shrub_L = low shrubs; CC = overstory canopy cover; CWD = coarse woody debris; Fern = fern cover; Grass = grass cover; Herb = herbaceous cover; Litt = deciduous leaf litter cover; LTD = deciduous leaf litter depth; Moss = moss cover; Rock = rock cover; Soil = exposed soil; Maple = vine maple; Dec_Sm = bigleaf maple and red alder <1.3 m; Dec_Lg = bigleaf maple and red alder >1.3 m; Con_Sm = western redcedar and western hemlock <1.3 m; Con_Lg = western redcedar and western hemlock >1.3 m; Dfir_Sm = Douglas-fir <1.3 m; Dfir_Lg = Douglas-fir >1.3 m. AMGR = *Ambystoma gracile*; ASTR = *Ascaphus truei*; DITE = *Dicamptodon tenebrosus*; ENES = *Ensatina eschscholtzii*; PLVE = *P. vehiculum*; RAAU = *Rana aurora*; TAGR = *Taricha granulosa*.





Appendix 5-C. Amphibian–habitat relationships in riparian habitats ten-years post-logging. Shrub_T = tall shrubs; Shrub_L = low shrubs; CC = overstory canopy cover; CWD = coarse woody debris; Fern = fern cover; Grass = grass cover; Herb = herbaceous cover; Litt = deciduous leaf litter cover; LTD = deciduous leaf litter depth; Moss = moss cover; Rock = rock cover; Soil = exposed soil; Maple = vine maple; Dec_Sm = bigleaf maple and red alder <1.3 m; Dec_Lg = bigleaf maple and red alder >1.3 m; Con_Sm = western redcedar and western hemlock <1.3 m; Con_Lg = western redcedar and western hemlock >1.3 m; Dfir_Sm = Douglas-fir <1.3 m; Dfir_Lg = Douglas-fir >1.3 m. AMGR = *Ambystoma gracile*; ASTR = *Ascaphus truei*; DITE = *Dicamptodon tenebrosus*; ENES = *Ensatina eschscholtzii*; PLVE = *P. vehiculum*; RAAU = *Rana aurora*; TAGR = *Taricha granulosa*.





Appendix 5-D. Codes and scientific and common names of amphibians referred to in the text.

Code	Scientific Name	Common Name
AMGR	*Ambystoma gracile*	Northwestern Salamander
AMMA	*Ambystoma macrodactylum*	Long-toed Salamander
ASTR	*Ascaphus truei*	Coastal Tailed Frog
BUBO	*Bufo boreas*	Western Toad
DITE	*Dicamptodon tenebrosus*	Coastal Giant Salamander
ENES	*Ensatina eschscholtzii*	Ensatina
HYRE	*Hyla regilla*	Pacific Tree Frog
PLDU	*Plethodon dunni*	Dunn’s Salamander
PLVE	*Plethodon vehiculum*	Western Red-backed Salamander
RAAU	*Rana aurora*	Northern Red-legged Frog
RACAS	*Rana cascadae*	Cascades Frog
RHCA	*Rhyacotriton cascadae*	Cascade Torrent Salamander
RHKE	*Rhyacotriton kezeri*	Columbia Torrent Salamander
TAGR	*Taricha granulosa*	Rough-skinned Newt

Chapter 6 WESTERN WASHINGTON CONCLUSIONS AND MANAGEMENT IMPLICATIONS

Virgil C. Hawkes

Extensive forestry management in the Pacific Northwest has historically included clearcutting, prescribed fire, regeneration by seed trees, protection from forest fires, and salvage of timber killed by windthrow or disease (Carey and Harrington 2001). In the last 20 years, planting of Douglas-fir has increased, with most forest stands >40 years old originating from seed. Current management practices include the retention of seed trees and coarse woody debris and the planting of seedlings. Implicit in the shift to ecosystem management has been the retention of riparian management zones (RMZs) around perennial fish- and non–fish-bearing streams, which have been required in Washington State in some form since 1978.

During the past 10–15 years, increased attention has focused on defining and delineating riparian areas. McComb et al. (1993) reported that a buffer width of 50 m adjacent to second- and third-order streams in Oregon would provide marginal habitat and that buffers should be >50 m wide to provide linkages between mature forests. Vesely and McComb (2002) estimated that RMZs 43 m wide would support total salamander abundance and that buffers of 47 m would support species richness at levels similar to unlogged sites. Recently, Crawford and Semlitsch (2007) found that a buffer width of 27 m would encompass 95% of the salamander assemblage and that an additional 50 m would ameliorate edge effects. They recommended that a total buffer of 92.6 m be used in southern Appalachian streams to enable salamander biodiversity to persist. We found that a minimum buffer width of 30 m around third-and fourth-order streams in western Washington would maintain species richness and relative abundance values at levels similar to those recorded prior to logging.

Despite these recommendations for buffer width, the minimum width of an RMZ continues to be a geographic issue related to the implementation of regionwide management plans that are too general to account for local variability in species richness and diversity. Yet, the research on the benefits of RMZs is clear (deMaynadier and Hunter 1995; Vesely and McComb 2002; Cushman 2006), and most jurisdictions in Canada and the United Sates require the retention of riparian forest adjacent to upland logged habitats (Lee et al. 2004). The problem faced by land managers is that, far too often, management plans become specific to one group of organisms and the plans do not always use biodiversity preservation as the basis for management.

Although RMZs retain structural heterogeneity and complexity, the production of homogenous, simplified, even-aged stands of second-growth Douglas-fir forests in upland habitats may, over the long term, lead to localized extirpations of species like the Coastal Giant Salamander. Therefore, it is important to consider not only the geographic relationship to riparian management, which will provide an indication of species richness and abundance, but also to consider the frequency of upland rotation and the type of prescription applied to the upland forest. For example, Perkins and Hunter (2006) recommended that riparian management be a two-tiered approach, with no-cut zones in the riparian forest and a partial-cut zone in the adjacent upland. This approach would protect the riparian forest from blowdown and minimize edge effects (deMayandier and Hunter 1998).

For species that decline in abundance because of logging, management strategies to mitigate these declines have been proposed. Carey and Harrington (2001) concluded that management-induced homogeneity and simplification (i.e., reduction of complexity) posed a real danger to small-mammal communities becoming non-supportive of predator populations. They proposed that active management for habitat complexity on long rotations (>90 years) may be necessary to conserve the diversity attributed to maturing forests. Because terrestrial amphibian communities are inherently linked to many forest ecosystem processes, the development of biodiversity management strategies must be considered at a broader scale and should not be based on single-species conservation. The introduction of heterogeneity into homogenous, even-aged stands has been shown to have positive effects on biodiversity and ecosystem function in the short term (<5 years; Carey 2003), and longer-term studies are required to determine if the positive effects are long-lasting.

Since this study's inception in 1991, there have been changes to the Washington Forest Practices Rules and Regulations that have implications for our findings. The Forest Practice Rules of 1988 required a riparian buffer that met the state configuration (i.e., not less than 7.5 m from the ordinary high-water mark) for both fish-bearing (Type 3) and non–fish-bearing (Type 4) perennial streams. The development of the modified riparian prescription was an alterative to the state buffer that provided for greater structural complexity, heterogeneity, and ultimately greater diversity of both flora and fauna. Lee et al. (2004) identified the current trend in riparian management prescriptions as tending toward site-specific buffers that vary among broadly similar harvest areas to within a single harvest area. The Forest Practice Rules of 2001 seem to fit this model and, aside from changing how watercourses are defined, are site-specific with many options. Lee et al. (2004) suggest that the primary benefit of this type of prescription is the identification of well-defined criteria to delineate the RMZ. These criteria are specifically defined by the modifying factors selected by jurisdictions, and buffers are applied in a predictable response to these criteria. The caveat to tailor-made buffers is the greater complexity in guidelines. For example, Washington now describes 3 zones of management for Type F streams (Type 3): (1) the core, (2) inner, and (3) outer zones, with various levels of tree removal, thinning, hardwood conversion, and other activities permitted within these zones depending on the site class of land, the management harvest option, and the bankfull width of the stream. In general, the site class will form the RMZ prescription. It appears that riparian management in western Washington has become (necessarily) site-specific. This implies that applying a one-size-fits-all approach to riparian management has been set aside in favor of a more intensive approach that considers the specifics of the site, which is consistent with Lee et al. (2004). For Type Np streams, or perennial non–fish-bearing streams (Type 4), the RMZ rules are slightly less complicated; however, there is ample room for site-specific management. Considering that the RMZ rules have evolved to embrace a site-specific approach, it seems only fitting that forest management in general adopt the same principals when considering biodiversity. Managing for biodiversity on a regional scale does not account for variation at the landscape or site level within the region, which could have dire consequences for species that are limited geographically, that occur at naturally low densities, or that are particularly sensitive to habitat change. As an alternative to the status quo, management that focuses on stand heterogeneity (as opposed to tree-species monocultures with reduced diversity) should produce landscapes with high biocomplexity and biodiversity at multiple scales, and therefore these landscapes should be resilient in the face of disturbance (Holling 2001).

Although we did not assess the efficacy of RMZs from a community ecology perspective, management strategies that are implemented at regional scales should be premised on the preservation of biodiversity and not predicated on single-species or single-wildlife-group preservation. Further, management strategies should consider habitat connectivity, and all components of the landscape should be managed in concert—not in isolation. Establishing RMZs as a means of preserving non-game wildlife populations may be adequate in even-aged, second-growth forests of western and eastern Washington, where diversity is typically low. However, the efficacy of RMZs as broad-scale, one-size-fits-all management tools needs to be considered for all groups of wildlife, not only for species with relatively small home ranges or specific habitat requirements. For example, the habitat needs of deer, elk, bears, marten, and fisher (as examples) must also be incorporated into landscape management strategies.

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Chapter 7 EASTERN WASHINGTON SONGBIRDS

Mike W. Demarchi and Virgil C. Hawkes

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Abstract

A replicated, experimental study was conducted in the mid-elevation, advanced (mid-seral) second-growth, conifer-dominated forests of northeastern Washington to investigate the efficacy of three different riparian treatments in providing habitat for songbirds: (1) no harvest, (2) upland harvest according to state guidelines, and (3) upland harvest according to a modified version of state guidelines.

Comparisons of relative abundance among all three sampling intervals for the most common songbird species did not reveal any significant differences over time within treatments and transects. However, differences in the relative abundance of all birds among treatments and transects were apparent during the second post-harvest sampling interval. Overall, forest harvesting led to increased bird abundance along both transect types at modified and state buffer sites. Relative abundance was lowest along riparian transects at the control sites and highest along upland transects at state buffer treatments. Differences in habitat structure resulting from forest harvesting, as elucidated by co-inertia analysis, are believed to be largely responsible for the observed differences in bird abundance, though site effects cannot be discounted entirely.

Both riparian and upland habitats contain important habitats for bird species. For 12 species, riparian habitats (transects) were used by >50% of the individuals within those species. Conversely, 30 species showed individuals that used riparian transects for <50% of the records. Upland transects of logged sites supported the greatest alpha diversity during the second post-harvest interval. Increased alpha diversity in logged upland areas likely contributed to increased alpha diversity in the adjacent riparian buffers of modified and state buffer treatment sites. Relatively low values and small ranges in indices of community similarity (Morisita's *C*) for comparisons between the riparian and upland transects suggests that the effects of any differences between treatment types on avian community structure were modest during the second post-harvest sampling interval. That is, community similarity between transect types appeared to be independent of treatment type. Relatively higher similarity values among comparisons of communities along riparian transects during the second post-harvest sampling interval suggested that riparian bird communities were not affected greatly by harvesting.

In conclusion, at upland locations forestry-induced changes to habitat structure and forest succession altered the abundance and alpha diversity of songbird communities, but no notable differences in key parameters of bird ecology between state and modified buffer prescriptions were revealed. This suggests that, despite undergoing greater timber removal, the state buffer treatment is no less capable than the modified buffer treatment of preserving the relative abundance, alpha diversity, and community structure of songbirds 10 years after logging in the riparian areas of coniferous forests of eastern Washington that are comparable to those examined in this study. Finally, a number of study limitations and management implications are presented.

Introduction

Coniferous forests of the Pacific Northwest, including those in northeastern Washington, are intensively managed for timber production, creating a landscape of forest stands that vary widely in age, structure, and composition (O'Connell et al. 2000). The presence or abundance of a wildlife species in a given area reflects a complex set of interactions among biophysical components of the environment. Because vegetation directly or indirectly provides a wildlife species with its food and cover[11] requirements, predictable associations between wildlife species and habitat conditions exist. Recognizing that wildlife–habitat associations can be affected by industrial forestry, applied forestry–wildlife research has sought ways of mitigating the adverse effects of forestry on wildlife habitat (e.g., no-harvest reserves, variable retention, selective harvesting). Contemporary forest managers bear an increasing level of responsibility for maintaining biological diversity in forested landscapes (Bunnell et al. 1999). Forest managers are tasked with harvesting a crop of trees and then, in time, returning the site to a condition that provides for most of the same timber and non-timber values as it did pre-harvest—or at least ensuring that comparable values always exist somewhere in the landscape (Tuchmann et al. 1996; Moore and Allen 1999).

Riparian areas are defined as "adjacent to a stream, river, lake, or wetland that supports vegetation that, due to the presence of water, is distinctly different from the vegetation of adjacent upland areas."[12] Such differences in vegetation communities often lead to differences in vertebrate assemblages. Riparian areas are characterized by increased primary productivity, higher levels of energy transport, and, often, more frequent natural disturbances than upland habitats.

Riparian zones in the temperate forests of western North America provide a number of ecological values. Although riparian zones occupy comparatively small portions of the forested landscape, they often support unique species and higher levels of biodiversity and species abundance compared to more abundant, upland forests (e.g., Sabo et al. 2005; Peak and Thompson 2006). Riparian zones also play critical roles in watershed hydrology. For these and many other reasons, the conservation and management of riparian habitats have garnered much attention in recent years (Naiman et al. 2000).

Riparian buffers are commonly used in an attempt to mitigate the effects of industrial forestry on biodiversity (Knopf et al. 1988). The width of such buffers has been the subject of considerable debate (Knutson and Naef 1997) and study because of the trade-offs between ecological and economic values (Berg 1995). If buffers are wider than necessary to provide their stated ecological values, economic revenue from timber harvesting could be unnecessarily foregone. If short-term economic values are maximized by minimizing buffer width, species conservation could be jeopardized. As a means of balancing these often competing objectives, guidelines for riparian buffer width are common throughout the political jurisdictions of North America (Lee et al. 2004). Riparian buffer width and breeding-bird (particularly passerine) responses to them have been studied in the forests of western North America and beyond (e.g., Kinley and Newhouse 1997; Hagar 1999; O'Connell et al. 2000; Pearson and Manuwal 2001; Shirley and Smith 2005; chapter 3 of the current study). Carey (1988) found no relationship between riparian stand and bird diversity. In a related study, Pearson and Manuwal (2001) found that within ~2 years post-harvest, riparian buffers retained after the adjacent upland was logged did not differ in terms of overall abundance relative to the unlogged control sites; but the number of species and average species turnover increased in narrow buffers (~7.5 m wide), while both species turnover and richness were similar to the control sites at wider buffers (~30 m wide). The longer-term, post-treatment efficacy of riparian management zones

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[11] "Cover" describes habitat that provides such values as security from predators, security from weather extremes, and sites for breeding.

[12] British Columbia Ministry of Forests and Range, "Glossary of Forestry Terms in British Columbia," March 2008, http://www.for.gov.bc.ca/hfd/library/documents/glossary.

(RMZs) in providing habitat for songbirds has not been studied in eastern Washington prior to this investigation.

In 1990, the Wildlife Steering Committee of the Cooperative Monitoring, Evaluation, and Research Committee (CMER) of the Timber, Fish, and Wildlife (TFW) Agreement initiated a study to evaluate the efficacy of RMZs in providing habitat for wildlife. The study was designed to evaluate the abundance and diversity of wildlife in riparian zones by comparing two different buffer configurations to unlogged control sites. The buffer configurations were based on the RMZ guidelines mandated at the time (Washington State Forest Practices Board 1988) and on modifications to the 1988 guidelines. The wildlife groups selected for study were terrestrial and aquatic amphibians, riparian-associated songbirds, small mammals (rodents and insectivores), and bats. A precursor to this study (O'Connell et al. 2000) compared the diversity and abundance of these wildlife groups before and immediately after logging to identify the short-term temporal effects of logging both within and between treatments. In 2001, CMER initiated a follow-up study to evaluate the longer-term effects of RMZs on wildlife. This study is the only one in Washington to evaluate the longer-term post-treatment effects of riparian management strategies on wildlife based on data collected at the same sites.

The efficacy of RMZs in providing habitat for songbirds was evaluated experimentally by comparing bird populations in riparian and adjacent upland habitats before and after a partial timber harvest in the forests of the Selkirk Mountains of northeastern Washington. The objectives of this study were to compare the relative abundance, alpha diversity, and community similarity of abundant species of breeding songbirds in the riparian and upland areas of replicate sites subject to one of three treatments: (1) no harvesting (i.e., control sites), (2) modified riparian buffer, and (2) state buffer. Where possible, comparisons would be across three sampling intervals: (1) pre-harvest, (2) 1–2 years post-harvest, and (3) 9–10 years post-harvest. Note that a lack of access to the raw data from the first and second sampling intervals constrained temporal analyses.

Methods

The methods used in this study were carried forward from a study by O'Connell et al. (2000). Seventeen previously established sites used in that study were resampled during this study (Figure 7-1). One of the sites sampled by O'Connell et al. (Calispell) could not be relocated on the ground and was excluded. The intent of this study was to evaluate the longer-term post-treatment efficacy of RMZs in providing habitat for songbirds by comparing data collected 10 years following logging, with both the pre-harvest and 2-year post-harvest data. However, pre-treatment and 2-year post-treatment data were not available; therefore, our analyses are primarily limited to the second post-harvest data set except where summary data presented in O'Connell et al. (2000) facilitated comparative analyses.

All sites were located in Stevens and Pend Oreille counties in the Okanagan Highlands physiographic province of northeastern Washington (Figure 7-1). Site selection in the original study (O'Connell et al. 2000) was based on five environmental criteria, which minimized the variation between sites. The criteria used for site selection (and thus the scope of sites to which the study results are applicable) included the following:

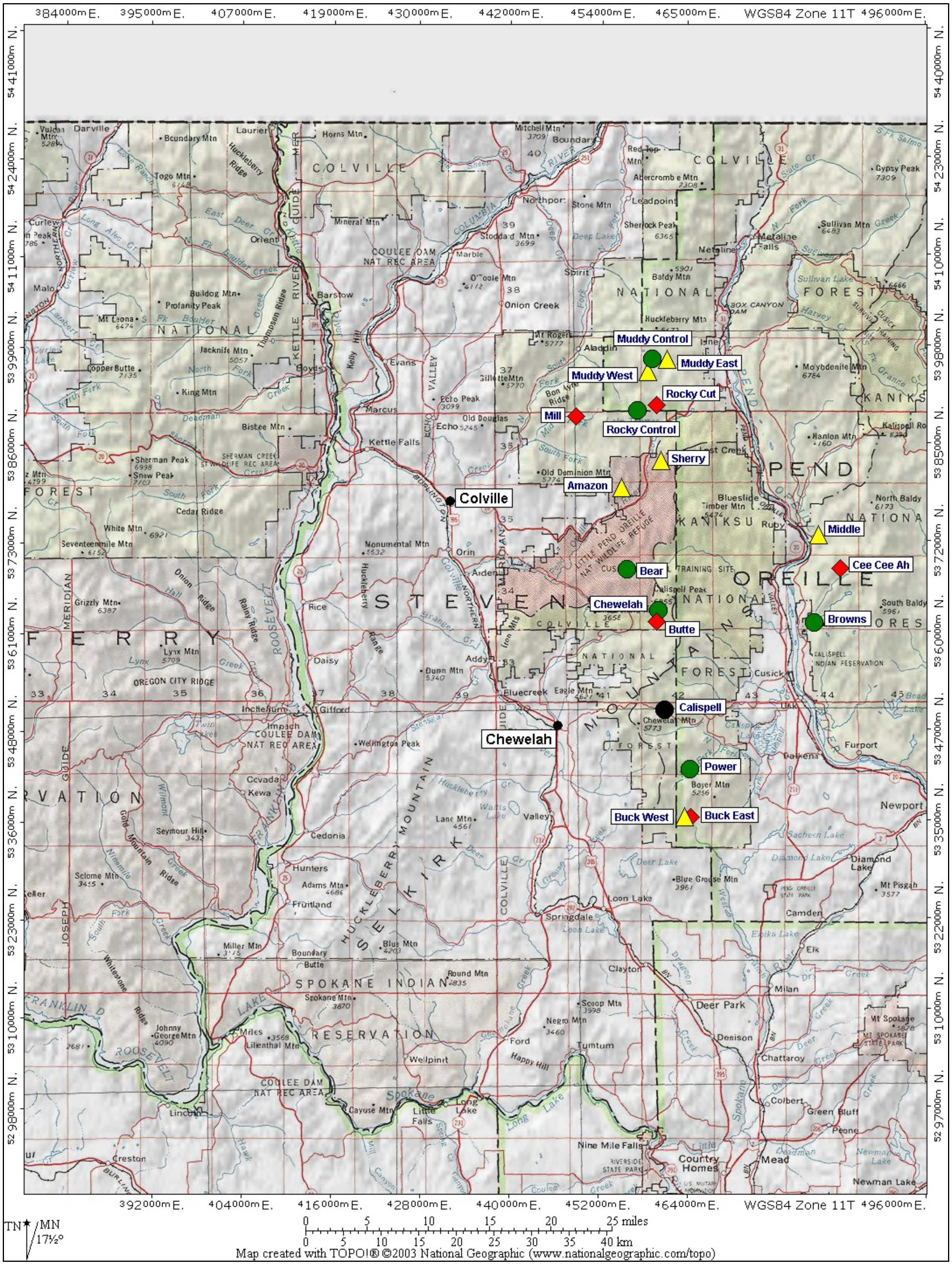
1. Managed forests of harvestable age.
2. ≥800 m reach of Type 3 or permanent Type 4 streams, as delineated by Washington's 1988 Forest Practices Rules and Regulations (Table 7-1).
3. 16.2 ha previously logged stands on either side of the stream.
4. Mixed-coniferous forests.

5. 600–1200 m elevation.

The selected sites were typical of mixed-coniferous forests in the Selkirk Mountains of northeastern Washington (O'Connell et al. 2000). The dominant tree species present at the sites were Douglas-fir (*Pseudotsuga menziesii* var. *glauca*), lodgepole pine (*Pinus contorta* var. *latifolia*), western redcedar (*Thuja plicata*), western hemlock (*Tsuga heterophylla*), western larch (*Larix occidentalis*), and grand fir (*Abies grandis*). Understory vegetation included gooseberry (*Ribes* spp.), devil's club (*Oplopanax horridus*), Oregon grape (*Mahonia* spp.), falsebox (*Paxistima myrsinites*), red-osier dogwood (*Cornus stolonifera*), mallow ninebark (*Physcarpus malvaceus*), spirea (*Spiraea* spp.), Saskatoon (*Amelanchier alnifolia*), roses (*Rosa* spp.), and huckleberry (*Vaccinium* spp.). Despite attempts at consistency among sites, some random environmental events such as wildfire and extensive blowdown altered habitat conditions at some sites relative to others (e.g., Rocky Cut and Muddy East).

Table 7-1. Water-typing criteria (WAC 222-16-030) used in Washington in 1988. Source: Washington State Forest Practices Board (1988).

Parameter	Water Type				
	1	2	3	4	5
Channel Width	N/A	≥ 6 m between Ordinary High Water Mark (OHWM)	**Anadromous:** > 1.5m between OHWM **Resident Game:** > 3 m between OHWM	> 0.6 m between OHWM	> 0.6 m between OHWM
Gradient	N/A	< 4%	**Anadromous:** < 12% not upstream of a falls > 3m in height **Resident Game:** < 12%	N/A	N/A
Flow	N/A	N/A	**Anadromous:** N/A **Resident Game:** > 0.3 CFS at summer low flow	N/A	N/A
Impoundment	N/A	Water surface area of < 0.4 ha at seasonal low flow	**Anadromous:** Water surface area of < 0.4 ha at seasonal low flow **Resident Game:** Water surface area of < 0.2 ha at seasonal low flow	N/A	N/A
Fisheries	N/A	Used by substantial numbers of anadromous or resident game fish for spawning and rearing and migration	Used by significant numbers of anadromous or resident game fish for spawning and rearing and migration	Not used by significant numbers of fish	Not used by significant numbers of fish
Diversion	N/A	Domestic use for > 100 residences or campsites. Includes upstream reach of 450 m or until the drainage area is < 50%, whichever is less	Domestic use for > 10 residences or campsites. Includes upstream reach of 450 m or until the drainage area is < 50%, whichever is less	N/A	N/A
Other	All water within their OHWM inventoried as "Shoreline of the State" excluding related wetlands	Streams flowing through campgrounds available to public having ≥ 30 campsites	Contributes > 20% of flow to Type 1 or 2 water. Anadromous fish impoundments have outlet to stream with anadromous fish	N/A	All natural waters not classified as Type 1, 2, 3, or 4 or seepage areas, ponds and drainways having short run-off periods



Map created with TOPO!® ©2003 National Geographic (www.nationalgeographic.com/topo)

Figure 7-1. Distribution of eastern Washington study sites. △ state buffers; ♦ modified buffers; ● control sites; • excluded control site.

Study Design

The study design was based on a split-plot repeated-measures design, with 18 sites assigned to one of three treatments (Table 7-2). Each site was split into two distinct habitat types: (1) terrestrial riparian and (2) upland. Terrestrial riparian habitats occurred within 8 m of the ordinary high-water mark adjacent to a perennial stream and extended to the point where the vegetation visibly shifted to an upland type. Upland habitats extended upland from the zone of transition from riparian to upland vegetation. Upland sampling occurred 100 m upslope from terrestrial riparian habitat.

Table 7-2. Eastern Washington study sites sampled in 2003 and 2004. T = Township; R = Range; S = Section.

Site Number	Site Name	Treatment	T	R	S
2	Bear	Control	T34N	R41E	12
3	Browns	Control	T34N	R44E	33
7	Calispell[a]	Control	N/A	N/A	N/A
9	Chewelah	Control	T34N	R42E	29
12	Muddy Control	Control	T37N	R42E	17 & 18
15	Power	Control	T32N	R43E	28 & 33
16	Rocky Control	Control	T27N	R41E	35
1	Amazon	State	T35N	R41E	2
5	Buck West	State	T31N	R42E	22
10	Middle	State	T35N	R44E	28
13	Muddy East	State	T37N	R42E	17
14	Muddy West	State	T37N	R42E	18
18	Sherry	State	T36N	R42E	28
4	Buck East	Modified	T31N	R42E	22 & 23
6	Butte	Modified	T34N	R42E	32
8	Cee Cee Ah	Modified	T34N	R44E	12
11	Mill	Modified	T36N	R41E	5 & 6
17	Rocky Cut	Modified	T37N	R41E	25

[a] Calispell was not sampled in 2003/2004.

The three treatments developed were based on their recent (or lack of) logging history and the RMZ prescription:

1. **Control sites**: (*n* = 7) These were tracts of unlogged second-growth coniferous forest that were used as reference sites.
2. **Modified buffer treatment sites**: (*n* = 5) At these sites, the riparian zone was logged in a modified way in conjunction with the objectives of this study as outlined in O'Connell et al. (2000) (Table 7-3).
3. **State buffer treatment sites**: (*n* = 6) At these sites, the riparian zone was logged according to RMZ guidelines in the Washington Forest Practices Rules of 1988 (Table 7-3).

The upland habitats at both the modified and state buffer treatment sites were logged with partial retention (40–60%) during 1993–1994. The modified and state buffer treatment sites differed in several ways (Table 7-3). Most notable were the configuration of the RMZ and the retention of wildlife reserve trees around the creek. Wildlife reserve trees are defined as follows:

- **Type 1**: Live trees with defective or deformed sound tops, trunks, and roots.
- **Type 2**: Dead tree with a sound top, trunk, and roots.
- **Type 3**: Live or dead tree with unstable tops or upper portions.
- **Type 4**: Live or dead tree with unstable trunk or roots, with or without bark. This includes "soft" snags as well as live trees with unstable roots caused by root rot or fire. Type 4 reserve trees are the most dangerous.

Table 7-3. Comparison of riparian harvest prescription for state and modified buffers.

Feature	State	Modified
Buffer	9-–5 m buffer	30 m zone of consideration
Leave Trees	Leave Type 3 & 4 reserve trees; no protection	Leave Type 1, 2, 3, & 4 reserve trees; buffer one Type 3 or 4 tree per 2 acres by 1.5x the tree height
Seeps	Protect from machinery	Buffer by 9 m no-entry zone extending to the stream
Deciduous Trees	2 large or 3 smaller trees per acre	Leave all live trees
Shrubs	Avoid disturbance	Leave all within 9–15 m of the stream
Coniferous Trees	Leave all <30 cm DBH	Single tree selection
Delineation of Riparian Management Zone (RMZ)	Extend RMZ to maximum width of riparian zone	Extend zone of consideration to 1/3:2/3 riparian:upland. If no specific habitat features present within zone of consideration, follow state RMZ.

Each site had two 750 m–long transects on the same side of the stream, one in the riparian zone approximately 8 m from the edge of the stream, and another in the upland zone approximately 100 m away from the riparian transect (Figure 7-2). Along each transect, songbird survey stations were situated at 50 m intervals.

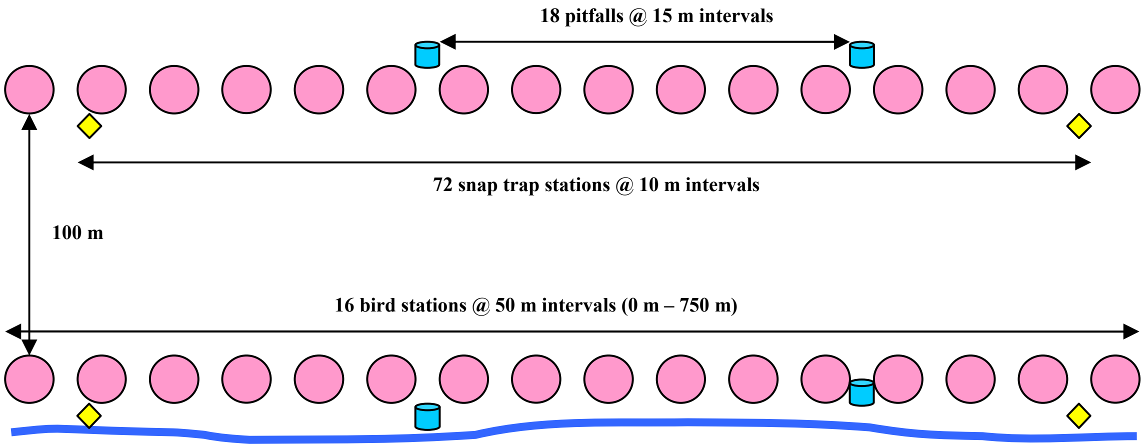


Figure 7-2. Eastern Washington study site layout. Pink circles = songbird station; Blue tubes = pitfall traps; Yellow diamonds = snap traps; Blue line = creek. Pitfall traps and snap traps refer to study components not related to the bird component.

Field Sampling

Bird surveys were conducted using modified transect encounter surveys. Each day, surveys would begin at approximately 05:00 and continue for ~5 hours. Along each transect, observers would start at either 0 m or 750 m (i.e., either end of transect), wait for 2 minutes, and then move along at a rate of ~50 m every 5 minutes, recording all bird detections. Prior to 2003, data were recorded using datasheets. In 2003/2004, data were recorded using iPAQ Pocket PCs (handheld computers running a version of Microsoft Windows) and Visual CE software. Table 7-4 summarizes the types of data collected during each survey. Flyovers (i.e., birds seen or heard flying over the tree canopy at a point count/transect but not directly associated with the terrestrial environment at that site) were identified as such. Every attempt was made to avoid counting individual birds more than once. Particular attention was paid to detections within a 30 m–wide strip along each transect, because those data formed the basis of most analyses: specifically, detections in the band of 0–8 m streamside of the riparian transects (0–8R; Table 7-4); those in the band of 0–22 m upland of the riparian transects (0–22R; Table 7-4); and those in the 0–15 m band along either side of the upland transects (0–15U; Table 7-4). Birds within any of these three distances and not counted as flyovers are referred to as "on-transect."

To confirm that bird surveys were conducted under optimal conditions (little to no rain, cool to warm temperatures, and little to no wind), ambient temperature, wind strength, and precipitation were recorded at each site and for all transects during 2003/2004. Wind speed was estimated using the scale in

Table 7-5, and the type of precipitation, if any, was recorded. Because wind speed varied greatly within each site (openings vs. forest; treetop vs. forest floor; upland vs. riparian; etc.), the data recorded are not that useful other than to suggest that higher winds might have hampered singing/detection. Sites surveyed in 2003/2004 when weather deteriorated after the start of the survey (e.g., snow or heavy rain) were resurveyed at a later date to ensure that each site was adequately surveyed six times.

A sampling schedule was developed that would enable the completion of six visits per site by ~30 June. The starting order of sampling sites was rotated and the sampling period spread out to ensure that each site was sampled near the beginning, middle, and end of the survey period. Observers were rotated among sites and transects, and starting points at either end of the transects were alternated to reduce bias.

All sites, regardless of treatment type, were surveyed six times per year during the pre-harvest sampling interval, before any forest harvesting occurred at the modified and state buffer sites. Following forest harvesting, sites were sampled six times during the first post-harvest sampling interval. With the exception of five sites that were monitored seven times in 2003 and 2004 (combined), all sites were monitored six times during the second post-harvest sampling interval.

Data from the first two sampling sessions (i.e., 1992–1996) were not available, and therefore all analyses were performed using the second post-treatment sampling interval data only (2003/2004). Furthermore, we were unable to fulfill the study's objective of conducting a comprehensive temporal analysis of the changes in species relative abundance and diversity. Where possible, we extracted data from O'Connell et al. (2000) for inclusion in this report. Temporal contrasts were limited to untested comparisons that could be done using summary statistics presented in O'Connell et al. (2000).

The following criteria were applied to data collected in the second post-harvest sampling interval for the purposes of analyzing bird data, except where noted otherwise:

- Songbirds detected within 0–8 m streamside of the riparian transect, 0–22 m upland side of the riparian transect, or within 15 m of the upland transect were included.
- All species detected ≥30 times were included.
- Flyovers were excluded.
- Data from partial surveys were excluded.

Table 7-4. Data collected at each point-count station during bird surveys.

Field	Definition	Value
Observer	Observer	Initials
Date	Date	Day, month, year
Site Number	Number of site	1–18
Site Name	Name of site	Insert site name
Point Count	Closest station to observation	1–16
Transect	Riparian or upland	R or U
Offset	Meters plus or minus from a point count	Number of meters
Location	Indicates side of creek relative to parking location	Streamside/upland/across stream
Species	Bird species	4-letter species code; "NONE" = no birds
Count	Number of birds for that sighting	Number
Sex	Male, female, unknown	M/F/U
Age	Adult, juvenile, unknown	A/J/U
Flyover	Was observation a flyover?	Y/N
Distance	Estimated distance of bird from transect (m)	0–8R; 0–22R; 22–50R; 50–100R; >100R; 0–15U; 15–50U; 50–100U; >100U; R for riparian transects, U for upland transects
Time Interval	Period of time when detection was made (min)	0–3; 3–5; 5–6
Detection	Type of detection	Sight/call/song/drum/wing
Habitat	Habitat type for specific detection	Buffer/logged/forest/unknown
Comments	Comments	Any other information
Cloud Cover	Cloud cover	1 = clear; 2 = cloudy <50%; 3 = cloudy >50% but <100%; 4 = 100% cloud cover
Air Temperature	Air temperature	Degrees C
Precipitation	Precipitation	Nil; fog; misty; drizzle; light rain; hard rain; hail; snow
Wind	Wind speed (estimate)	Km/h
Time Start	Survey start time (per site)	24-hour time
Time End	Survey end time (per site)	24-hour time

Table 7-5. Wind scale used in breeding-bird surveys.

Code	Definition
1	calm (<2 km/h)
2	light air (2–5 km/h)
3	leaves rustle (6–12 km/h)
4	leaves and twigs constantly move (13–19 km/h)
5	small branches move, dust rises (20–29 km/h)
6	small trees sway (30–39 km/h)
7	large branches moving, wind whistling (40–50 km/h)

Vegetation Sampling

Structural habitat characteristics were examined by establishing 16 x 20 m plots at 50 m intervals along both the riparian and upland transects. Fifteen plots were established on each transect for a total of 30 transects per site. Each plot was divided into four 8 x 10 m quadrats (Figure 7-3), and between each plot a 30 m point-intercept transect was established to evaluate floristic diversity. Fourteen point-intercept transects were established along each habitat transect (28 per site) (Figure 7-3). All vegetation field sampling (quadrats and point-intercept transects) was conducted during July and August 2004.

In each 16 x 20 m plot, we documented the composition and distribution of shrubs >0.5 m high, down wood and stumps, trees and snags, canopy cover, and tree regeneration. The following sections describe the methods used to collect data for each of these components.

Shrubs

From the center point of each plot, we measured the distance (m) to the nearest shrub (>0.5 m high) in each quadrat, identified the species, and measured the area of the shrub (length x width).

Down Wood and Stumps

In quadrats 1 and 3 (Figure 7-3), we recorded the size class and decay class of woody debris and stumps. Logs were assigned to one of four size classes and decay classes (Table 7-6), and stumps were identified as either natural or cut and were categorized into the same decay classes as logs.

Table 7-6. Size and decay classes assigned to woody debris and stumps in eastern Washington. L = length; D = diameter. Source: after O'Connell et al. (2000).

Size Class	Definition
1	>5 m L;<15 cm D
2	>5 m L; 16–24 cm D
3	>5m L; >25 cm D
4	<5m L; >25 cm D
Decay Class	
1	Freshly fallen tree; bark essentially intact; wood solid; no decomposition
2	Bark beginning to slough or almost completely gone; start of decomposition; sapwood softened; log generally firm
3	Wood softened and breaks into chunks; each chunk still has integrity
4	Essentially no integrity to log; wood decomposed to soil-like texture

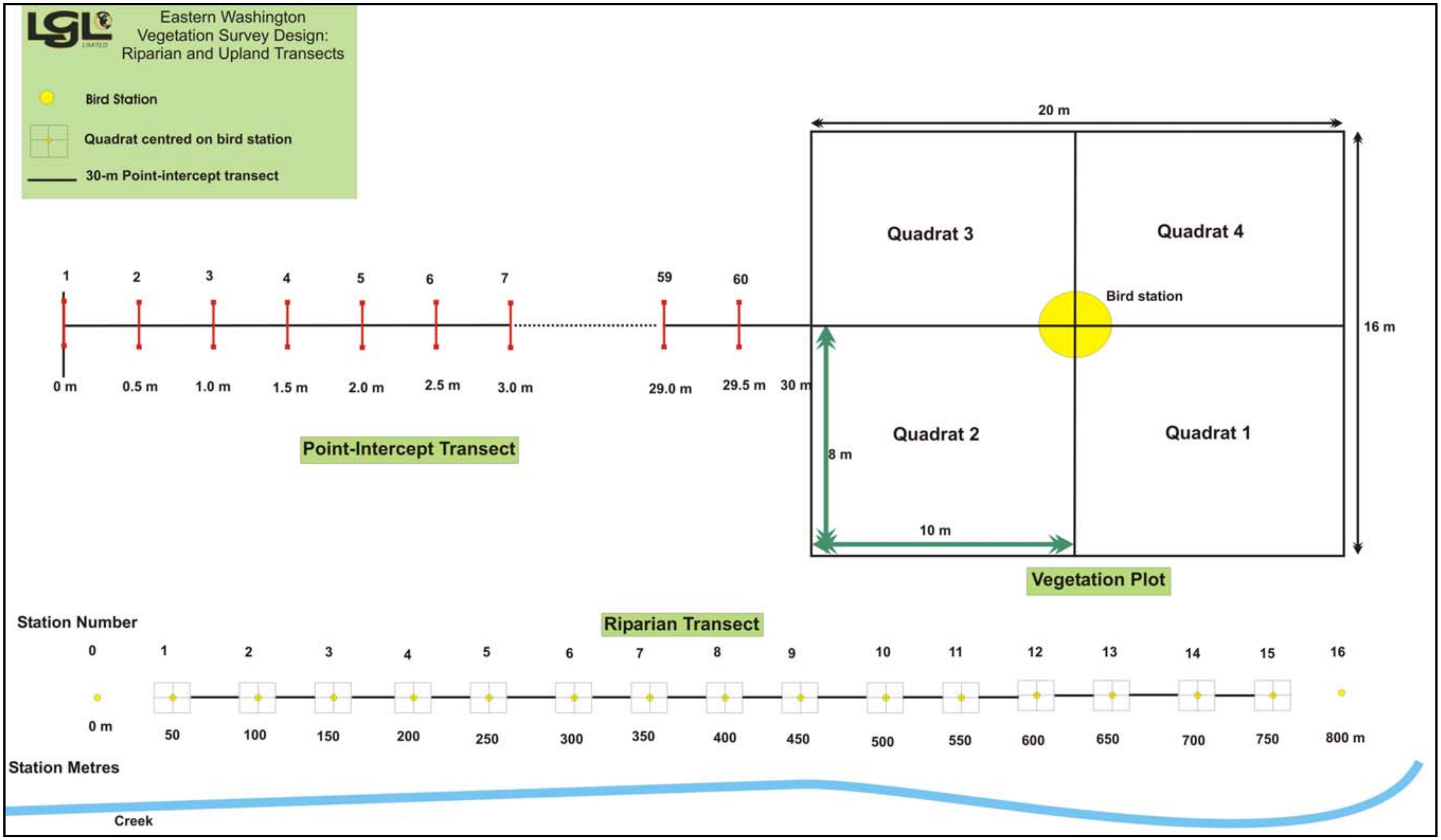


Figure 7-3. Schematic showing the distribution of 16 x 20 m vegetation plots in the riparian transect and the division of one plot into 4 quadrats. Point-intercept transects between each plot are also shown. The configuration was repeated in the upland transect.

Trees and Snags

Within each plot (all 4 quadrats), trees were identified to species and assigned to one of four diameter at breast height (DBH) categories: (1) 4–10 cm, (2) 11–25 cm, (3) 26–50 cm, (4) >50 cm. Snags were counted by condition class (condition 1: bark basically intact; condition 2: bark peeling off to absent) and assigned to the same DBH categories as trees.

In each plot, four average-sized trees and two average-sized snags (average size for the plot) were chosen at random and their crown heights were estimated using a clinometer (Figure 7-4).

For example:

- S1 = Sighting 1 = 60%
- S2 = Sighting 2 = -10%
- ∑S1S2 = 70% (sum of absolute values)
- Distance from observer to tree = 30 m
- Tree height = 70% x 30m = 21 m

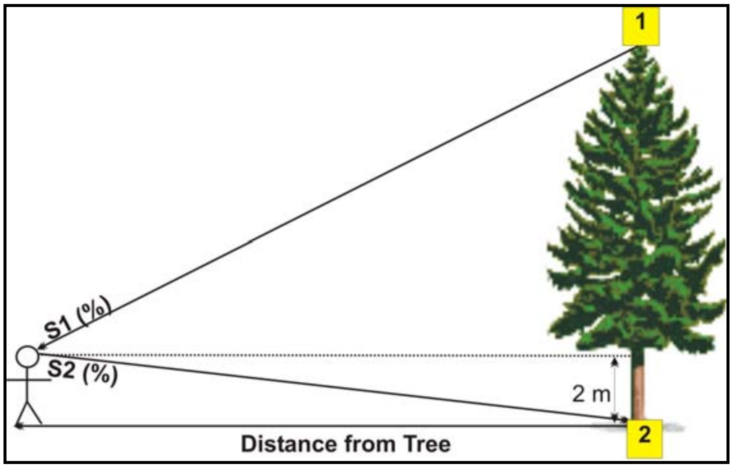


Figure 7-4. Estimating tree-crown height with a clinometer. S1 and S2 are sighting angles (in %).

Canopy Cover

We initially used a convex densiometer to measure overstory and understory canopy cover at the center of each plot and at the center of each quadrat. However, given the large amount of time required to obtain accurate measurements with the densiometer, and the very similar measurements obtained by ocular estimates, we elected to make most canopy-cover measurements by ocular estimation. The densiometer was used periodically to check the accuracy of ocular estimates.

Tree Regeneration

In quadrats 1 and 3, we recorded the number of regenerating coniferous trees (>0.5 m high and <4cm DBH).

Floristics

At 0.5 m increments along each 30 m transect between plots, a point-intercept rod was lowered perpendicular to the transect, and all vegetation, woody debris, and substrate that the rod contacted was recorded by height class (when relevant) (Table 7-7). Herbaceous plants, shrubs, ferns, and trees were identified to species, while grasses were recorded as present or absent and not identified to species.

Table 7-7. Height classes used on point-intercept transects in eastern Washington.

Height Class	Definition
1	1.5 m
2	1.0 m
3	0.5 m
4	0.25 m
5	0 m

Logs were assigned to one of six size classes (Table 15-5) and one of four decay classes (Table 7-8). Stumps were classified as either natural or cut and assigned to one of the same decay classes as logs. Stumps were differentiated from snags by height; dead trees were considered to be stumps when they were <1.37 m high. Litter depth was measured every 5 m on each floristics transect.

Table 7-8. Size classes used for woody debris encountered on floristics point-intercept transects in eastern Washington.

Size Class	Definition
1	<5 m L; <15 cm D
2	<5 m L; 16–24 cm D
3	>5 m L;<15 cm D
4	>5 m L; 16–24 cm D
5	>5m L; >25 cm D
6	<5m L; >25 cm D

L = Length; D = Diameter

Statistical Analyses

Statistical analyses were performed using SAS V9.1 (© 2002–2003), R V2.2.1 (© 2005), and Microsoft Excel 2002 (© 1985–2003). Prior to performing all analyses, data were assessed to see if they fit a normal distribution using the Analyst Application in SAS, which produces four tests of normality (Shapiro–Wilk, Kolomogerov–Smirnov, Cramer–von Mises, and Anderson–Darling). Data were also assessed for annual variation prior to being pooled. Given that data collected in 2003 did not differ from 2004 for any of the sites, data from the two years were pooled. Comparisons of relative abundance were done using log-transformed means ($\log[x + 1]$), which approximated a normal distribution. Where appropriate, post hoc multiple range tests were used to account for experiment-wise error and thereby reduce Type I errors (Shaffer 1995). When pairwise comparisons of means were made, the Bonferroni method was applied to control the family-wise error (FWE) rate. The critical value of alpha was set at 0.1 and beta at 0.8 (*a priori*) for all analyses, which is appropriate for landscape-based studies and when analyzing data obtained from populations that are highly variable.

Alpha Diversity and Community Similarity

We used the number of species found in each habitat type (i.e., riparian or upland) at each site during each sampling interval as a measure of alpha diversity (species richness). Species richness data from riparian transects were used to test for differences relative to buffer width. To test whether logging influenced alpha diversity, we combined data from the buffer treatment groups (i.e., state and modified riparian buffers). Because both groups received similar upland treatments, pooling the treatments should provide a sense of whether or not the retention of riparian habitat, regardless of width, permits the persistence of alpha diversity.

We used Morisita's coefficient of similarity (C) as a measure of community similarity because of its superior utility, relative to other measures, in pairwise comparisons of communities (Krebs 1999).

Morisita's index (Morisita 1959; Horn 1966; Brower et al. 1990) measures community similarity and is based on Simpson's index of dominance (Simpson 1949). It calculates the probability that specimens randomly drawn from two sites will be of the same species, relative to the probability that specimens randomly drawn from the same site will be of the same species. This index is desirable because sample size and diversities of the samples have little influence on its calculation (Morisita 1959; Wolda 1981). Morisita's index returns a value from 0.0 (no similarity) to 1.0 (identical) and was chosen because it is affected little by large variations in sample size.

Relative Abundance

We used detection rate as a measure of relative abundance. Detection rate was calculated as the number of detections of each species divided by the number of visits to a given habitat. For example, if riparian habitats were visited 6 times in one year and Golden-crowned Kinglet was documented 27 times, the detection rate would be 27 ÷ 6 = 4.5. For each species, an average detection rate was derived for each sampling interval; each sampling interval consisted of 2 years. Comparisons were made between mean relative abundance measures, transformed as log(x + 1) to approximate a normal distribution. We tested variation in relative abundance only for the most common species; sample size was too small for analysis of other species.

Using PROC MIXED, we ran a repeated-measures ANOVAs to test hypotheses about variation in relative abundance, followed by orthogonal and non-orthogonal contrasts (Table 7-9) of main effects (treatment and transect) and their interactions. The main experimental units for which measurements were repeatedly collected were the individual transects in riparian and upland habitats, and the data from each transect were pooled to derive a treatment mean.

Table 7-9. Orthogonal and non-orthogonal contrasts used to identify differences in relative abundance (RA) of songbirds within the second post-harvest sampling interval (SI; 2003/2004) relative to treatment (control, modified, state) and transect (riparian, upland).

Comparison	Question
CONTROL - Riparian vs. Upland	Does RA at control sites differ between riparian and upland habitats?
MODIFIED - Riparian vs. Upland	Does RA at modified buffer sites differ between riparian and upland habitats?
STATE - Riparian vs. Upland	Does RA at state buffer sites differ between riparian and upland habitats?
RIPARIAN - Control vs. cut	Does RA in riparian habitats differ between control and logged (cut; state and modified buffer combined) sites?
RIPARIAN - Modified vs. State	Does RA in riparian habitats differ between modified buffer and state buffer sites?
RIPARIAN - Control vs. Modified	Does RA in riparian habitats differ between control and modified buffer sites?
RIPARIAN - Control vs. State	Does RA in riparian habitats differ between control and state buffer sites?
UPLAND - Control vs. cut	Does RA in upland habitats differ between control and logged (cut; state and modified buffer combined) habitats?
UPLAND - Modified vs. State	Does RA in upland habitats differ between modified and state buffer sites?

All nine contrasts were used to answer questions about differences in mean relative abundance of the more common species detected. Based on the objective of the overall study, treatments and transects were grouped to best answer the questions of differences between the state and modified buffer treatments relative to the control sites. The comparisons derived were based on the following assumptions:

1. All sites selected for this study were similar enough that results derived for one site could be extrapolated across all sites of a particular treatment; as such, pooling of sites into treatments was acceptable.
2. The upland habitats at the state and modified treatments represented the same treatment and thus could be pooled into a post-cut condition, defined by clearcut logging followed by replanting.

3. A comparison of the upland habitats of the control sites with the pooled state and modified uplands (i.e., pre-cut vs. post-cut) would address logging effects (i.e., does logging uplands affect the relative abundance of songbirds?).
4. A comparison of the control riparian habitats to the pooled state and modified condition would determine if logging affects relative abundance of songbirds irrespective of buffer width.
5. A comparison between the modified and state riparian transects would provide a measure of the effect of a wide buffer versus a narrow buffer and the effect that buffer width has on the relative abundance of songbirds.

Co-inertia Analysis

The program R V2.2.1 (© 2005) was used to investigate habitat relationships between abundant or common species and the habitat attributes measured at each site and treatment. For this study, a co-inertia analyses (COIA) was used and was based on the ade4 package (Chessel et al. 2005). COIA is a multivariate statistical technique that provides for the visualization of the structure of ecological data via ordination. COIA is very general and some existing methods appear as special cases of it (Dray et al. 2003). For example, interbattery analysis (Tucker 1958) is mathematically equivalent to a simultaneous principle components analysis (PCA) on two ecological tables followed by a COIA (i.e., PCA–PCA COIA). When table X contains qualitative variables and Y contains species numbers, it is usual to cross tables X and Y to obtain a matrix containing the distribution of species among the categories of environmental variables. A simple canonical analysis (CA) of this new table allows one to ordinate the species and the environmental classes (analysis of ecological profiles; Montana and Greig-Smith 1990). Binary discriminant analysis (Strahler 1978), which has been used in ecology (Del Moral 1982; Huang and Del Moral 1988), is mathematically equivalent to Romane's CA (Dray et al. 2003). Although this approach allows plotting of species and environmental classes (Ben-Shahar 1987; Ben-Shahar and Skinner 1988), no information about ordination of sites is available.

COIA is similar to other ecological ordination techniques, such as a canonical correspondence analysis (CCA; ter Braak 1986) and redundancy analysis (RDA; Wollenberg 1977); however, COIA is a general coupling method that maximizes the covariance between the variables of two tables. Separate tables X and Y can be analyzed by various methods, leading to different coupling methods. This approach aims to find a site score that is a linear combination of environmental variables maximizing the variance of species centroids (i.e., separation of species niches). Unlike CCA, COIA does not have a regression step (Thiolouse et al. 2004). This difference has important implications for ecological data analyses: the regression step implies that in CCA and RDA, the number of samples must be high compared to the number of environmental variables; COIA does not have this constraint. CCA also has an additional constraint (the total variance must be equal to 1) and must be avoided in the case of numerous environmental variables, which is common in biological studies and is the case in this study.

COIA is a paired ecological table analysis that relies on the covariance matrix produced from (in this case) a simultaneous PCA of the environmental and species data (Figure 7-5). Each table is related through a common number of rows, which in this case represents the numbers of sites. The output of the PCA can be visualized as two different representations of the sites in space: one relating to the relative abundance of species; the other to the habitat attributes. The covariance matrices produced by the PCA are then joined via COIA so that the relationship between species relative abundance and habitat data can be viewed. The resulting biplot allows for visualization of how the two initially unrelated data sets covary at the same sites. From the biplot, a description of the habitat variables can be derived for species of interest. The ability to quickly assign habitat relationships based on the relative position of species to habitat variables provides an effective way to describe the habitat relationships for each species in each sampling interval and for each treatment.

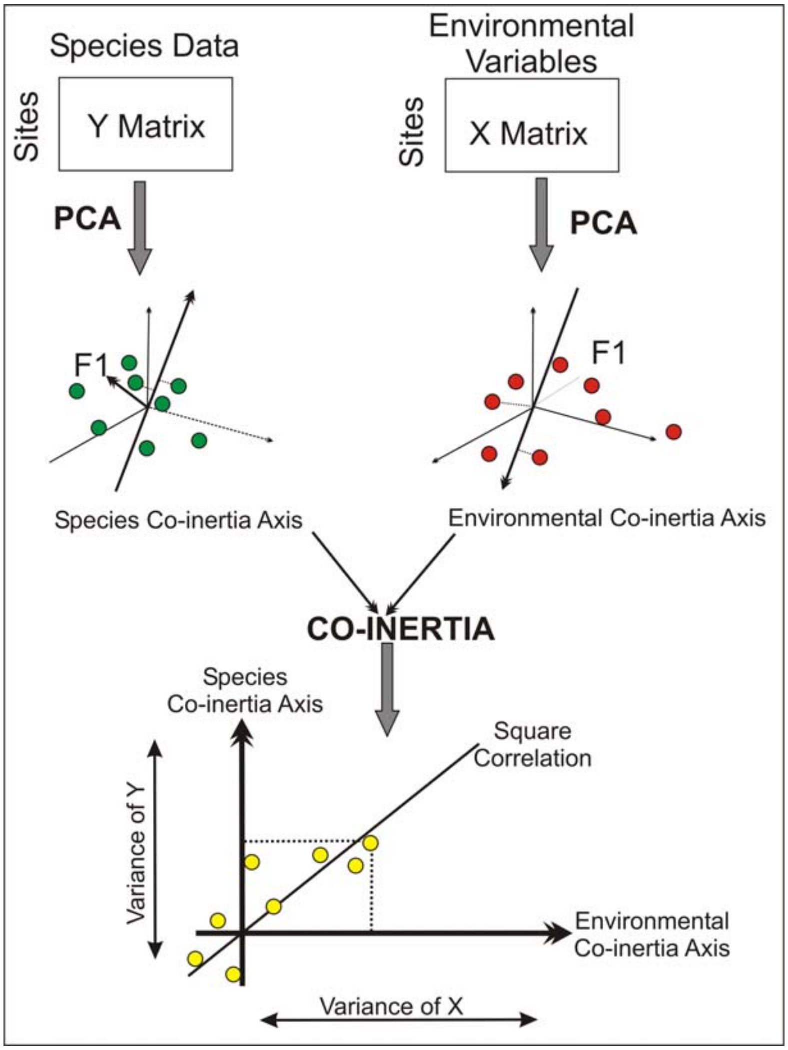


Figure 7-5. Schematic showing the principles of co-inertia analysis. The two ecological data tables X and Y produce two representations of the sites in two hyperspaces. Separate analyses find axes maximizing inertia (=variation) in space (F1 [first factorial axis]). Co-inertia analysis maximizes the square covariance between the projections of the sites on the co-inertia axes. Source: modified from Dray et al. (2003).

The output of a COIA can be viewed on a biplot. The biplots have vertical and horizontal axes, with their intersection being the origin. An environmental gradient can be described for each axis, with the horizontal axis (axis 1) representing the primary environmental gradient and the vertical axis (axis 2) describing a secondary gradient. Species that occur nearer the origin are indicative of (a) species with too few detections to infer habitat associations, (b) habitat generalists, or (c) species that respond to intermediate conditions along the gradients described by the axes. Species that plot farther away from the origin can be regarded as correlating with a particular habitat variable or suite of variables. The length of the vector from the origin to the location of the habitat variable in space is indicative of the strength of the correlation between habitat variable and the gradient described for each axis. Unlike the more commonly used multivariate analyses, such as CCA (ter Braak 1986), for which the number of environmental variables is limited to the number of sites minus one, COIA has no limitations on the number of environmental variables that can be used regardless of the number of sites sampled (Thioulouse et al. 2004). COIA has been evaluated extensively (e.g., Dray et al. 2003), and although it is not used frequently in ecological studies compared to other multivariate methods, it is a rigorous method producing easily interpretable results. The visual comparison of the relationship between a species and the environmental variables should be accompanied by a meaningful biological interpretation, which requires knowledge of the species habits and site characteristics. Caution, however, needs to be exercised when assessing species–habitat relationships, because sample size can influence the analysis.

Results

General

Survey effort during 1992–2004 is summarized in Table 7-10. Forest harvesting at the modified and state buffer treatment sites occurred between autumn 1993 and summer 1994. Bird surveys in 1994 overlapped with the period of forest harvesting, but we were unable to determine the extent to which those surveys occurred on logged and unlogged sites. O'Connell et al. (2000) did not document this aspect of the study.

Table 7-10. Number of complete breeding-bird surveys (site visits) in eastern Washington, 1992–2004. Surveys in 1992/1993 and an unknown fraction of surveys in 1994 comprise the pre-harvest sampling interval; an unknown fraction of surveys in 1994 and all surveys in 1995/1996 comprise the first post-harvest sampling interval; surveys in 2003/2004 comprise the second post-harvest sampling interval.

		Year						
Treatment	**Site Name**	**1992**	**1993**	**1994**[a]	**1995**	**1996**	**2003**	**2004**
Control	Bear	6	6	6	6	6	6	6
	Browns	6	6	6	6	6	6	6
	Calispell[b]	6	6	6	6	6	-	-
	Chewelah	6	6	6	6	6	6	6
	Muddy Control	6	6	6	6	6	6	6
	Power	6	6	6	6	6	6	6
	Rocky Control	6	6	6	6	6	6	6
Modified Buffer	Buck East	6	6	6	6	6	6	6
	Butte	6	6	6	6	6	6	6
	Cee Cee Ah	6	6	6	6	6	6	6
	Mill	6	6	6	6	6	6	6
	Rocky Cut	6	6	6	6	6	6	6
State Buffer	Amazon	6	6	6	6	6	6	6
	Buck West	6	6	6	6	6	6	6
	Middle	6	6	6	6	6	6	6
	Muddy East	6	6	6	6	6	6	6
	Muddy West	6	6	6	6	6	6	6
	Sherry	6	6	6	6	6	6	6

[a] Assumed
[b] Site could not be located during the 2003/2004 sampling period (second post-harvest sampling interval)

The vast majority of bird detections were of passerines (songbirds; Figure 7-6). Songbirds comprised 94.3% of all bird detections during the second post-harvest sampling interval.

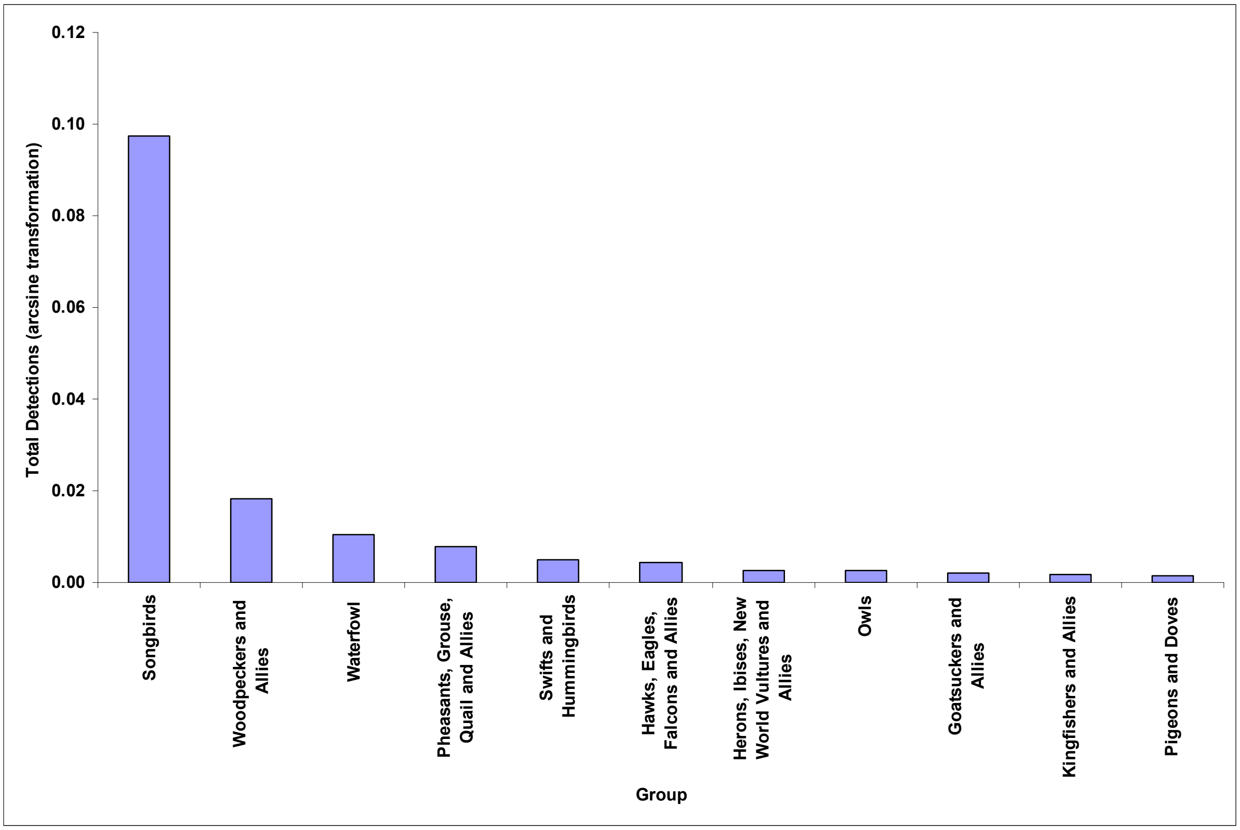


Figure 7-6. Proportion (arcsine transformed) of birds by taxonomic grouping, detected along the riparian and upland transects of study sites during breeding-bird surveys in eastern Washington (second post-harvest sampling interval, 2003/2004; includes flyovers, all distances).

Figure 7-7 shows attrition in the number of bird species and individuals detected (excluding flyovers) along the riparian and upland transects of the study sites during the second post-harvest sampling interval according to three sets of increasingly restrictive filtering criteria. The results were as follows:

- A total of 79 species comprising 10,412 individuals were detected during the 2003/2004 surveys.
- A total of 62 species comprising 4453 individuals were detected within the 30 m–wide survey boundaries (Appendix 7-B).
- A total of 45 species of songbirds comprising 4254 individuals were detected within the 30 m–wide survey boundaries.

Applying the additional species-selection conditions (i.e., species detected a grand total of ≥30 times during the second post-harvest sampling interval) to the 45 species of songbirds resulting from the third set of criteria above resulted in 23 species comprising 4028 individuals. This total is 38.7% of all birds detected, excluding flyovers.

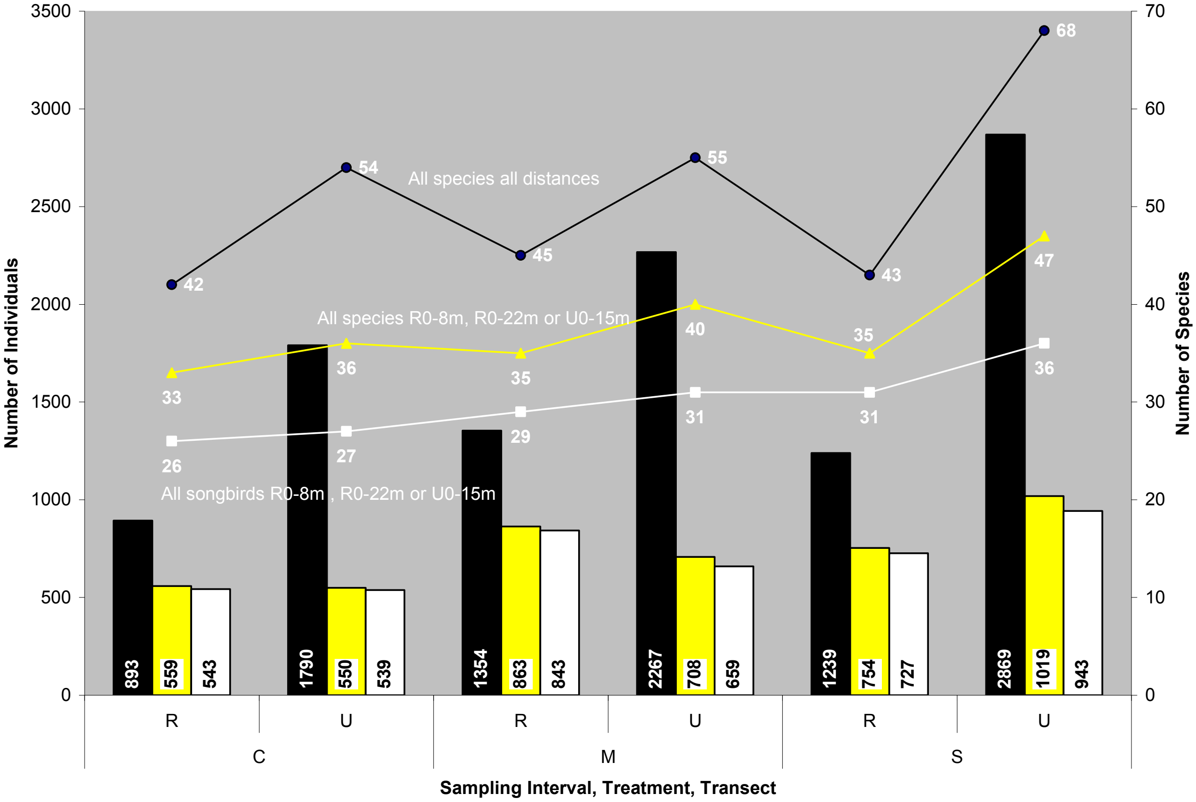


Figure 7-7. Number of species (points) and individuals (bars) according to three increasingly restrictive criteria detected along the riparian (R) and upland (U) transects of control (C), modified (M), and state (S) sites during breeding-bird surveys in eastern Washington during the second post-harvest sampling interval (2003/2004) according to three increasingly restrictive sets of criteria: (1) all bird species detected during the survey (black circles and bars), (2) as for 1 but only those within transect belts (yellow triangles and bars), (3) as for 2 except only songbirds (white squares and bars). Transect belts: R 0–8m = within 0–8 m on the streamside of riparian transect; R 0–22 m = within 22 m of the upland side of riparian transect; U 0–15 m = within 15 m either side of the upland transect. Lines connecting numbers of species are solely to aid in the display of data and do not reflect any trends or other relationships among numbers.

Relative Abundance Comparisons

Comparisons of relative abundance among all three sampling intervals for 17 songbird species (i.e., a subset of those considered common by O'Connell et al. [2000, Table 4, p. 6-39] *and* considered for more detailed analysis during the second post-harvest sampling interval; data presented in Appendix 7-C) did not reveal any significant differences over time as a function of treatment and transect (Table 7-11).

Table 7-11. Mean relative abundance (1 SD) and ANOVA results for selected[a] species of songbirds detected on-transect during surveys in eastern Washington in all three sampling intervals. Analyses are based on overall relative abundance for each transect, treatment, and sampling interval. Source: data for the pre-harvest and first post-harvest sampling intervals are from O'Connell et al. (2000, Table 4, p. 6-39). For all tests, df = 2, 48.

		Sampling Interval				
Transect	**Treatment**	**Pre-harvest**	**1st Post-harvest**	**2nd Post-harvest**	***F***	***p***
Riparian	Control	0.48 (0.71)	0.56 (0.20)	0.42 (0.56)	0.174	0.841
	Modified	0.62 (0.91)	0.54 (0.76)	0.74 (1.01)	0.213	0.810
	State	0.55 (0.89)	0.65 (0.75)	0.51 (0.60)	0.154	0.857
Upland	Control[b]	0.53 (0.74)	0.67 (0.73)	0.41 (0.44)	0.680	0.512
	Modified	0.45 (0.59)	0.51 (0.45)	0.55 (0.45)	0.058	0.944
	State	0.46 (0.49)	0.66 (0.75)	0.58 (0.54)	0.497	0.612

[a] The subset of species summarized by O'Connell et al. (2000, Table 4, p. 6-39) *and* those considered for analyses during the second post-harvest sampling interval (i.e., those listed in Table 7-13): American Robin, Brown Creeper, Chestnut-backed Chickadee, Chipping Sparrow, Dark-eyed Junco, Golden-crowned Kinglet, Hammond's Flycatcher, Hermit Thrush, Mountain Chickadee, Nashville Warbler, Pine Siskin, Red-breasted Nuthatch, Swainson's Thrush, Townsend's Warbler, Varied Thrush, Winter Wren, Yellow-rumped Warbler. Input data values used in the analyses are those plotted in Appendix 7-C.

[b] Also tested using Welch ANOVA for unequal variance and found to be insignificant (α = 0.05).

The relative abundance of all birds (on-transect) was significantly different among treatments and transects during the second post-harvest sampling interval (Welch ANOVA; $F_{5,182.4} = 12.7$; $p < 0.001$). Overall, relative abundance was lowest along riparian transects at the control sites and highest along upland transects at state buffer sites, but not all means were significantly different (Table 7-12). Both the untransformed and log transformed data were heteroscedastic according to Bartlett's test ($p < 0.001$; $F = 13.53$; df = 5; hence the use of the Welch ANOVA).

Table 7-12. Mean numbers of birds detected per survey (on-transect) during bird surveys in eastern Washington for each combination of treatment (control, modified, state) and transect (R = riparian, U = upland) during the second post-harvest sampling interval (2003/2004). Means were calculated from the total number of birds detected during the second post-harvest sampling interval by transect by survey divided by the total number of surveys; excludes flyovers. Double-underscoring connects means that are not significantly different (α = 0.05, Tukey–Kramer HSD). Sample sizes are indicated in parentheses.

Treatment and Transect					
Control-R	**Control-U**	**State-R**	**Modified-U**	**State-U**	**Modified-R**
(*n* = 72)	(*n* = 67)	(*n* = 72)	(*n* = 60)	(*n* = 71)	(*n* = 60)
7.76	8.24	10.47	11.82	14.37	14.38

Beyond patterns in relative abundance among the three treatment and two transect types during the second post-harvest sampling interval, individual species responded in a number of different ways, not necessarily consistent with overall responses at the control sites (Table 7-12). Orthogonal contrasts of the relative abundance of 23 selected species of songbirds (Table 7-13) detected during the second post-harvest sampling interval show that the relative abundance of individual species varied among treatment and transect type (Table 7-14). In descending order, the greatest numbers of significant differences were between the following:

- The riparian and upland transects of state buffer treatment sites;

- The riparian and upland transects of modified buffer treatment sites;
- The upland transects of modified and state buffer treatment sites;
- The upland transects of unlogged and logged sites; and
- The riparian and upland transects of control sites.

Fifteen species of songbirds exhibited significant differences in relative abundance for at least one of the nine contrasts tested; relative abundance of the other 8 species did not vary significantly relative to treatment or transect within or among treatments during the second post-harvest sampling interval (Table 7-14). The relative abundance of Dark-eyed Junco varied the most, with six of nine contrasts producing significant results. The relative abundance for Winter Wren, Hammond's Flycatcher, and Red-breasted Nuthatch also varied notably relative to transect and treatment (Table 7-14).

Mean relative abundance varied within each treatment relative to habitat type (Table 7-14). For the control group, three species (Chestnut-backed Chickadee: $F_{1,14} = 9.2$; $p < 0.01$; Red-breasted Nuthatch: $F_{1,27.9} = 29.7$; $p < 0.001$; Yellow-rumped Warbler: $F_{1,14} = 5.9$; $p = 0.003$) had significantly higher relative abundance on upland transects, while only one species (Winter Wren: $F_{1,14} = 113.5$; $p < 0.001$) had a higher relative abundance on riparian transects (Table 7-14). At the modified and state treatments, 11 species showed significant differences with respect to habitat type, with 5 of those species having significantly different relative abundance at both the modified and state treatments (Table 7-14). For species showing significant differences between riparian and upland habitats at the control, state buffer, and modified buffer treatments (Red-breasted Nuthatch, Winter Wren, Yellow-rumped Warbler), the differences were equal, with either higher relative abundance in riparian habitats at both treatments (Winter Wren) or higher relative abundance in upland habitats at both treatments (Red-breasted Nuthatch, Yellow-rumped Warbler). For each treatment and all 3 species, the magnitude of the observed difference was approximately the same, and the differences observed are a reflection of each species' habitat-use patterns.

Comparisons of relative abundance among treatments revealed that Winter Wren, Hammond's Flycatcher, and Warbling Vireo were significantly more abundant on riparian transects at the modified treatment relative to both the control and state buffer treatments (Table 7-14). Mean relative abundance of Dark-eyed Junco and American Robin was significantly higher in riparian habitats of the state buffer treatment compared to the control group (Table 7-14) but did not differ significantly from the modified buffer treatment. Pooling data from treated (i.e., logged: modified and state) sites, and contrasting the relative abundance for each species at logged sites against relative abundance at unlogged (i.e., control) sites, revealed that 2 species (Dark-eyed Junco, Hammond's Flycatcher) had significantly different relative abundance on riparian transects and both of these species were more frequently observed at logged sites (Table 7-14). The relative abundance of all other species tested did not vary appreciably among riparian habitats at each treatment (Table 7-14).

Significant differences in relative abundance between upland habitats at the modified and state buffer treatments and between the control group relative to logged sites (i.e., modified and state combined) were determined for 7 species, 3 of which were common to both comparisons (Table 7-14). Four of 5 species (Chipping Sparrow, Dark-eyed Junco, Hermit Thrush, Red-breasted Nuthatch) had higher relative abundance in upland habitats at the state treatment relative to the modified treatment. The relative abundance of the other species (Pine Siskin) was higher at the modified treatment. Contrasting logged sites against the control group indicates that relative abundance was always higher at logged sites for the 5 species showing a significant difference (Table 7-14).

Table 7-13. Species of songbirds detected in the second post-harvest sampling interval (2003/2004) that were included in comparison of relative abundance.

Species Code	Common Name	Scientific Name
DEJU	Dark-eyed Junco	*Junco hyemalis*
WIWR	Winter Wren	*Troglodytes troglodytes*
HAFL	Hammond's Flycatcher	*Empidonax hammondii*
RBNU	Red-breasted Nuthatch	*Sitta canadensis*
CHSP	Chipping Sparrow	*Spizella passerina*
PISI	Pine Siskin	*Carduelis pinus*
YRWA	Yellow-rumped Warbler	*Dendroica coronata*
GCKI	Golden-crowned Kinglet	*Regulus satrapa*
HETH	Hermit Thrush	*Catharus guttatus*
OCWA	Orange-crowned Warbler	*Vermivora celata*
WAVI	Warbling Vireo	*Vireo gilvus*
AMRO	American Robin	*Turdus migratorius*
CBCH	Chestnut-backed Chickadee	*Poecile rufescens*
MGWA	MacGillivray's Warbler	*Oporornis tolmiei*
NAWA	Nashville Warbler	*Vermivora ruficapilla,*
BRCR	Brown Creeper	*Certhia americana*
CAVI	Cassin's Vireo	*Vireo cassinii*
MOCH	Mountain Chickadee	*Poecile gambeli*
RECR	Red Crossbill	*Loxia curvirostra*
SWTH	Swainson's Thrush	*Catharus ustulatus*
TOWA	Townsend's Warbler	*Dendroica townsendi*
VATH	Varied Thrush	*Ixoreus naevius*
WETA	Western Tanager	*Piranga ludoviciana*

Table 7-14. Significance values (at α = 0.05) and interpretation for each species having a significantly different relative abundance for orthogonal contrasts of habitat type and treatment. Numerator degrees of freedom = 1. Denominator degrees of freedom = DenDF.

Species	Contrast	F	DenDF	*p*	Higher relative abundance
Chestnut-backed Chickadee	CONTROL - Riparian vs. Upland	9.19	14.00	0.009	Upland
Chipping Sparrow	STATE - Riparian vs. Upland	10.53	14.00	0.006	Upland
Dark-eyed Junco	STATE - Riparian vs. Upland	37.09	14.00	0.000	Upland
Dark-eyed Junco	MODIFIED - Riparian vs. Upland	9.28	14.00	0.009	Upland
Golden-crowned Kinglet	STATE - Riparian vs. Upland	5.62	14.00	0.033	Riparian
Golden-crowned Kinglet	MODIFIED - Riparian vs. Upland	4.58	14.00	0.050	Riparian
Hammond’s Flycatcher	MODIFIED - Riparian vs. Upland	18.90	14.00	0.001	Riparian
Hermit Thrush	STATE - Riparian vs. Upland	8.84	14.00	0.010	Upland
Nashville Warbler	MODIFIED - Riparian vs. Upland	5.50	14.00	0.034	Upland
Orange-crowned Warbler	STATE - Riparian vs. Upland	5.30	28.00	0.029	Upland
Pine Siskin	MODIFIED - Riparian vs. Upland	8.08	14.00	0.013	Upland
Red-breasted Nuthatch	STATE - Riparian vs. Upland	44.89	28.00	0.000	Upland
Red-breasted Nuthatch	MODIFIED - Riparian vs. Upland	10.87	28.00	0.003	Upland
Red-breasted Nuthatch	CONTROL - Riparian vs. Upland	29.70	28.00	0.000	Upland
Winter Wren	STATE - Riparian vs. Upland	81.65	14.00	0.000	Riparian
Winter Wren	MODIFIED - Riparian vs. Upland	139.41	14.00	0.000	Riparian
Winter Wren	CONTROL - Riparian vs. Upland	113.52	14.00	0.000	Riparian
Yellow-rumped Warbler	STATE - Riparian vs. Upland	21.87	14.00	0.000	Upland
Yellow-rumped Warbler	MODIFIED - Riparian vs. Upland	6.87	14.00	0.020	Upland
Yellow-rumped Warbler	CONTROL - Riparian vs. Upland	5.95	14.00	0.029	Upland
Chipping Sparrow	Per 3 UPLAND - Control vs. cut	5.86	28.00	0.022	Logged
Dark-eyed Junco	Per 3 UPLAND - Control vs. cut	28.76	27.30	0.000	Logged
MacGillivray’s Warbler	Per 3 UPLAND - Control vs. cut	4.96	27.85	0.034	Logged
Orange-crowned Warbler	Per 3 UPLAND - Control vs. cut	5.33	28.00	0.029	Logged
Pine Siskin	Per 3 UPLAND - Control vs. cut	5.77	23.51	0.025	Logged
Chipping Sparrow	Per 3 UPLAND - Modified vs. State	5.65	28.00	0.025	State
Dark-eyed Junco	Per 3 UPLAND - Modified vs. State	18.05	27.30	0.000	State
Hermit Thrush	Per 3 UPLAND - Modified vs. State	5.19	25.21	0.032	State
Pine Siskin	Per 3 UPLAND - Modified vs. State	4.59	23.51	0.043	Modified
Red-breasted Nuthatch	Per 3 UPLAND - Modified vs. State	6.99	28.00	0.013	State
Dark-eyed Junco	Per 3 RIPARIAN - Control vs. cut	5.19	27.30	0.031	Logged
Hammond’s Flycatcher	Per 3 RIPARIAN - Control vs. cut	5.82	26.87	0.023	Logged
Hammond’s Flycatcher	Per 3 RIPARIAN - Modified vs. State	8.96	26.87	0.006	Modified
Warbling Vireo	Per 3 RIPARIAN - Modified vs. State	4.23	25.11	0.050	Modified
Winter Wren	Per 3 RIPARIAN - Modified vs. State	22.70	27.07	0.000	Modified
Hammond’s Flycatcher	Per3 RIPARIAN - Control vs. Modified	12.41	26.87	0.002	Modified
Warbling Vireo	Per3 RIPARIAN - Control vs. Modified	6.16	25.11	0.020	Modified
Winter Wren	Per3 RIPARIAN - Control vs. Modified	9.08	27.07	0.006	Modified
American Robin	Per3 RIPARIAN - Control vs. State	4.19	26.76	0.051	State
Dark-eyed Junco	Per3 RIPARIAN - Control vs. State	8.83	27.30	0.006	State

Table 7-15 indicates the highest relative abundance of 22 songbird species for which data were available according to sampling interval, treatment (i.e., logged: state and modified), and habitat type (i.e., riparian and upland) Pre-treatment and 2-year post-treatment interval means were extracted from O'Connell et al. (2000, Table 4, p. 6-39). Because only summary statistics were available for the pre-harvest and first post-harvest sampling intervals, means were not tested for statistically significant differences.

Table 7-15. Sampling intervals with highest relative abundance (RA; shaded cells) for 22 species by habitat type (riparian, upland) at the two logged treatment types (modified, state) during three sampling intervals (SI 1–3). SI 2 is the first post-harvest sampling interval. Values for SI 1 and SI 2 are from O'Connell et al. (2000, p. 6-39). See Appendix 7-A for species codes.

Riparian							Upland					
State			Modified				Modified			State		
SI 3	SI 2	SI 1	SI 3	SI 2	SI 1	Species	SI 1	SI 2	SI 3	SI 1	SI 2	SI 3
		▨			▨	GCKI	▨			▨		
		▨			▨	BCCH	▨				▨	
	▨				▨	VATH	▨			▨		
		▨		▨		NAWA	▨			▨		
	▨				▨	BRCR		▨		▨		
	▨			▨		GRJA	▨			▨		
	▨					CHSP			▨		▨	
	▨				▨	HETH	▨					▨
	▨			▨		SOVI		▨		▨		
	▨			▨		DEJU		▨			▨	
	▨			▨		EVGR		▨			▨	
	▨			▨		RBNU		▨			▨	
	▨			▨		WIWR		▨			▨	
	▨			▨		YRWA			▨		▨	
		▨	▨			PISI			▨		▨	
▨					▨	MOCH		▨				▨
▨					▨	RNSA			▨			▨
▨			▨			AMRO	▨					▨
	▨		▨			TOWA	▨		▨	▨		
	▨		▨			SWTH			▨			▨
▨			▨			CBCH			▨			▨
▨			▨			HAFL			▨			▨

Bird–Habitat Relationships

The co-inertia analysis (COIA) combined information on songbird relative abundance and habitat variables measured at the sites. The biplots generated from the COIA depict the bird–habitat relationships along each transect. On each biplot, axis 1 represents the primary environmental gradient describing most of the variance; axis 2 corresponds to a secondary environmental gradient. The correlation of each environmental variable with axes 1 and 2 is indicated by its position relative to each axes, with variables closer to axis 1 or 2 more strongly correlated to the particular axis; variables occupying space between are correlated with both axes. Vector length is directly proportional to the degree of correlation: positive correlation to the right of the origin for axis 1 and above the origin for axis 2; negative correlation to the left of the origin along axis 1 and below the origin along axis 2. The relative position of a given species is indicative of its response to the environmental gradients described by the axes. Species nearer the origin are indicative of (a) species with too few detections to infer habitat associations, (b) habitat generalists, or (c) species that respond to intermediate conditions along the gradients described by the axes. Species farther from the origin are more abundant in environments that are dominated by particular variables. Beyond revealing the habitat features associated with higher levels of a species' relative abundance, biplots do not provide quantitative information on the relative abundance of a species (cf. Appendix 7-C), nor do they allow for comparisons of relative abundance among species.

The biplot for all riparian habitats (Figure 7-8) reveals an environmental gradient that was driven by forest openness (i.e., crown closure) and coniferous understory along axis 1. In addition, axis 2 shows a gradient from stands with few small snags (with bark) and low numbers of small conifers to stands with large snags (with and without bark) and large conifers. Notable habitat-feature associations of breeding birds along riparian transects are summarized in Table 7-16.

Table 7-16. Notable habitat associations of bird species detected on-transect along riparian transects during the third sampling interval (second post-harvest sampling interval, 2003/2004). Species presented in Figure 7-8 but not listed here are considered generalists with regard to the gradients depicted by the two axes of that figure.

Species	Notable Habitat Feature Associations
Dark-eyed Junco; Townsend's Warbler; Golden-crowned Kinglet	Abundant fir and spruce trees, small snags (with bark)
Chestnut-backed Chickadee; Swainson's Thrush	Low crown closure, abundant deciduous trees, seed-bearing shrubs
Yellow-rumped Warbler; Varied Thrush; Mountain Chickadee	Abundant conifers
Winter Wren; Warbling Vireo	Tall trees, abundant deciduous trees, large conifers, large snags

The biplot for all upland habitats (Figure 7-9) reveals an environmental gradient that was driven by the abundance of snags and conifers—particularly true firs. Axis 2 was driven largely by the gradient of stands with an abundance of large pine, Douglas-fir, larch, and with small red alder, and maple. Notable habitat-feature associations of breeding birds along upland transects are summarized in Table 7-17.

Although sample sizes (total number of individuals) varied greatly, bird species occurred along a continuum: from those that were abundant mostly in riparian areas to those that used upland areas almost exclusively (Figure 7-10). At least 75% of Song Sparrow, American Dipper, Ruffed Grouse, Winter Wren, Northern Waterthrush, Hammond's Flycatcher, Steller's Jay, and Wilson's Warbler were recorded along the riparian transects, whereas <25% of Dark-eyed Junco, Yellow-rumped Warbler, Olive-sided Flycatcher, Orange-crowned Warbler, Red Crossbill, Nashville Warbler, Gray Jay, Hairy Woodpecker, Pileated Woodpecker, Red-breasted Nuthatch, Brown-headed Cowbird, Chipping Sparrow, Townsend's Solitaire, Spotted Towhee, Northern Flicker, and Blue-headed Grosbeak detections occurred there. For species detected at least five times and with >50% of the total detections occurring along riparian transects, Figure 7-11 shows the percentage breakdown of observations by site type during the second post-harvest sampling interval. Four of the 12 species shown are considered to be riparian obligates or riparian-dependent by Bureau of Land Management (no date).

Table 7-17. Notable habitat associations of bird species detected on-transect along upland transects during the third sampling interval (second post-harvest sampling interval, 2003/2004). Species presented in Figure 7-9 but not listed here are considered generalists with regard to the gradients depicted by the two axes of that figure.

Species	Notable Habitat Feature Associations
Golden-crowned Kinglet; Swainson's Thrush; Chestnut-backed Chickadee	Abundant large pine, Douglas-fir, and larch; abundant small red alder and maple
Townsend's Warbler; Cassin's Vireo; Nashville Warbler; Hammond's Flycatcher; MacGillivray's Warbler Yellow-rumped Warbler; Western Tanager	Abundant seed and fruit-bearing shrubs; abundant small coarse woody debris; open canopy
Dark-eyed Junco; Red Crossbill; Orange-crowned Warbler; Chipping Sparrow	Low crown and understory crown closure (i.e., open areas)
Mountain Chickadee; Pine Siskin; Hermit Thrush	Abundant small pine, Douglas-fir, larch;

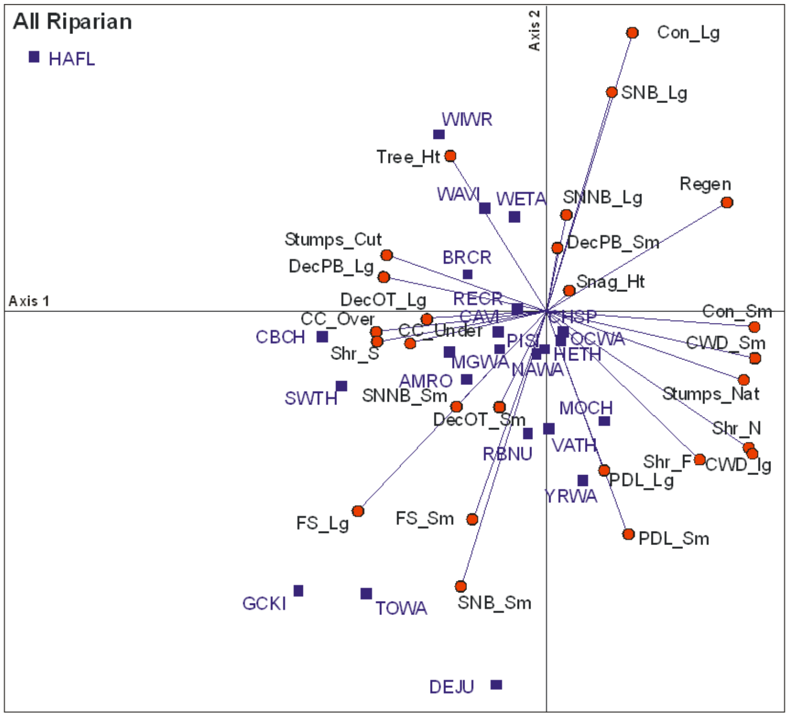


Figure 7-8. Biplot of bird–habitat relationships for all riparian transects in eastern Washington during the third sampling interval (second post-harvest sampling interval, 2003/2004). WD_Long = CWD >5m long; WD_Short = CWD <5m long; Stumps_Cut = Cut stumps; Stumps_Nat = Natural stumps; CC_Over = Overstory cover; CC_Under = Understory cover; Regen = Regenerating conifers <0.5m tall and <4 cm DBH; SNB_Sm = Snags with bark and <25 cm DBH; SNB_Lg = Snags with bark and >25 cm DBH; SNNB_Sm = Snags without bark, <25 cm DBH; SNNB_Lg = Snags without bark, >25 cm DBH; Con_Sm = Small-diameter conifers ; DecPB_Sm = Small-diameter birch and poplar ; DecOT_Sm = Small-diameter deciduous ; FS_Sm = Small-diameter fir and spruce ; PDL_Sm = Small-diameter pine, Douglas-fir, and larch; Con_Lg = Large-diameter conifers ; DecPB_Lg = Large-diameter birch and poplar ; DecOT_Lg = Large-diameter deciduous ; FS_Lg = Large-diameter fir and spruce ; PDL_Lg = Large-diameter pine. Douglas-fir, and larch ; Tree_Ht = Average tree height; Snag_Ht = Average snag height; Shr_S = Seed-bearing shrubs; Shr_F = Fruit-bearing shrubs; Shr_U = Non-food shrubs; CWD_Sm = Any piece <5m long; CWD_Lg = Any piece >5m long; AMRO = American Robin; BRCR = Brown Creeper; CAVI = Cassin's Vireo; CBCH = Chestnut-backed Chickadee; CHSP = Chipping Sparrow; DEJU = Dark-eyed Junco; GCKI = Golden-crowned Kinglet; HAFL = Hammond's Flycatcher; HETH = Hermit Thursh; MGWA = MacGillivray's Warbler; MOCH = Mountain Chickadee; NAWA = Nashville Warbler; OCWA = Orange-crowned Warbler; PISI = Pine Siskin; RBNU = Red-breasted Nuthatch; RECR = Red Crossbill; SWTH = Swainson's Thrush; TOWA = Townsend's Warbler; VATH = Varied Thrush; WAVI = Warbling Vireo; WETA = Western Tanager; WIWR = Winter Wren; YRWA = Yellow-rumped Warbler.

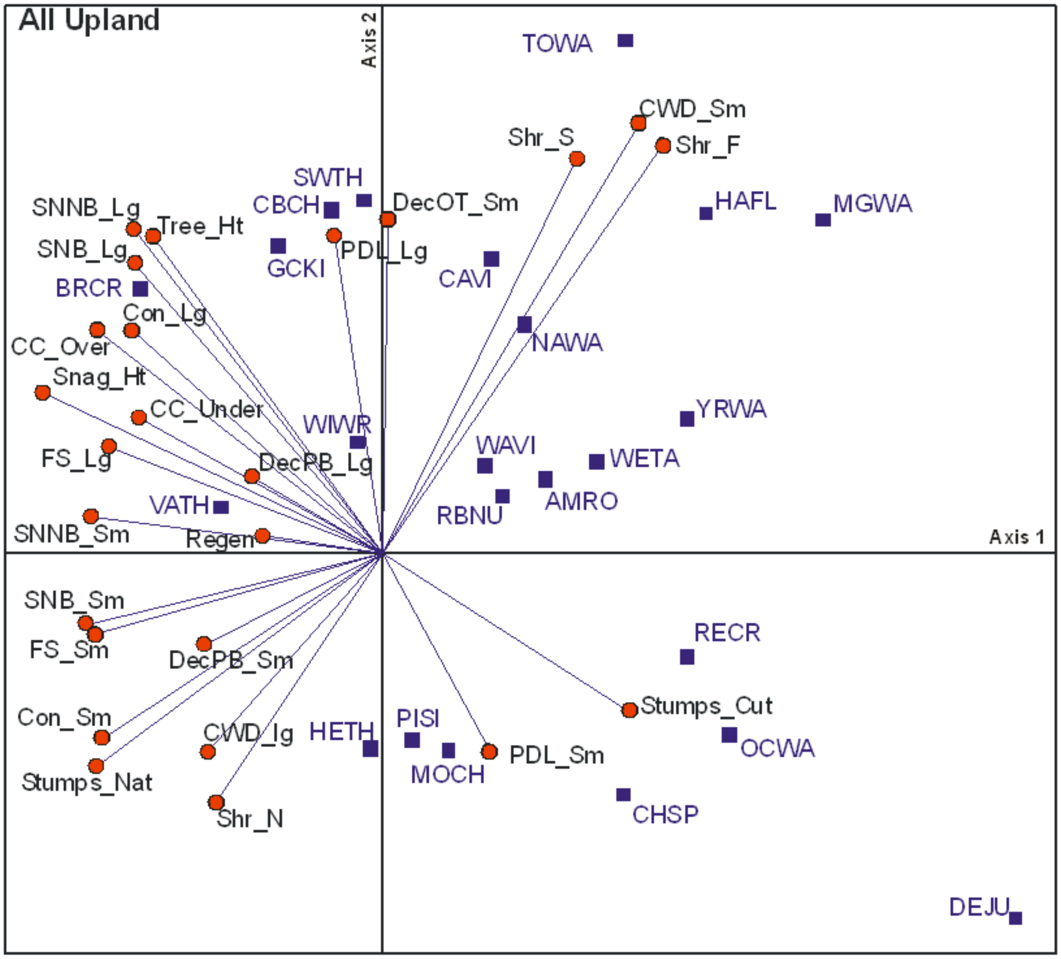


Figure 7-9. Biplot of bird-habitat relationships for all upland transects in eastern Washington during the third sampling interval (second post-harvest sampling interval, 2003/2004). WD_Long = CWD >5m long; WD_Short = CWD <5m long; Stumps_Cut = Cut stumps; Stumps_Nat = Natural stumps; CC_Over = Overstory cover; CC_Under = Understory cover; Regen = Regenerating conifers <0.5m tall and <4 cm DBH; SNB_Sm = Snags with bark and <25 cm DBH; SNB_Lg = Snags with bark and >25 cm DBH; SNNB_Sm = Snags without bark, <25 cm DBH; SNNB_Lg = Snags without bark, >25 cm DBH; Con_Sm = Small-diameter conifers ; DecPB_Sm = Small-diameter birch and poplar ; DecOT_Sm = Small-diameter deciduous ; FS_Sm = Small-diameter fir and spruce ; PDL_Sm = Small-diameter pine, Douglas-fir, and larch; Con_Lg = Large-diameter conifers ; DecPB_Lg = Large-diameter birch and poplar ; DecOT_Lg = Large-diameter deciduous ; FS_Lg = Large-diameter fir and spruce ; PDL_Lg = Large-diameter pine. Douglas-fir, and larch; Tree_Ht = Average tree height; Snag_Ht = Average snag height; Shr_S = Seed-bearing shrubs; Shr_F = Fruit-bearing shrubs; Shr_U = Non-food shrubs; CWD_Sm = Any piece <5m long; CWD_Lg = Any piece >5m long; AMRO = American Robin; BRCR = Brown Creeper; CAVI = Cassin's Vireo; CBCH = Chestnut-backed Chickadee; CHSP = Chipping Sparrow; DEJU = Dark-eyed Junco; GCKI = Golden-crowned Kinglet; HAFL = Hammond's Flycatcher; HETH = Hermit Thursh; MGWA = MacGillivray's Warbler; MOCH = Mountain Chickadee; NAWA = Nashville Warbler; OCWA = Orange-crowned Warbler; PISI = Pine Siskin; RBNU = Red-breasted Nuthatch; RECR = Red Crossbill; SWTH = Swainson's Thrush; TOWA = Townsend's Warbler; VATH = Varied Thrush; WAVI = Warbling Vireo; WETA = Western Tanager; WIWR = Winter Wren; YRWA = Yellow-rumped Warbler.

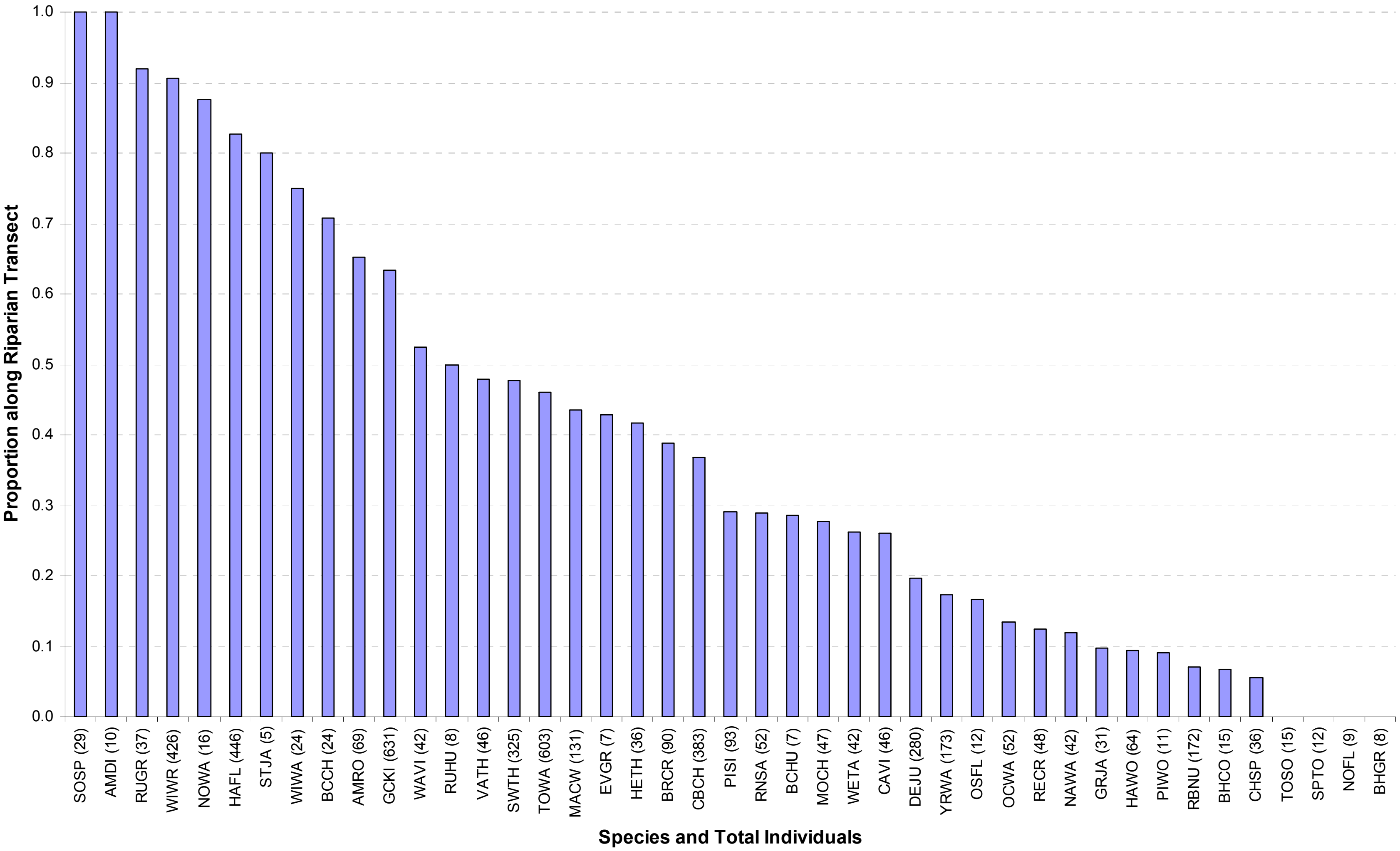


Figure 7-10. Proportion of bird species detected along riparian transects during the second post-harvest sampling interval (2003/2004) at the three (pooled) treatment types (control, modified, state) during breeding-bird surveys in eastern Washington. Data include detections on-transect but exclude flyovers. Total number of individuals detected follows species code (see Appendix 7-A). Only species detected ≥5 times are shown.

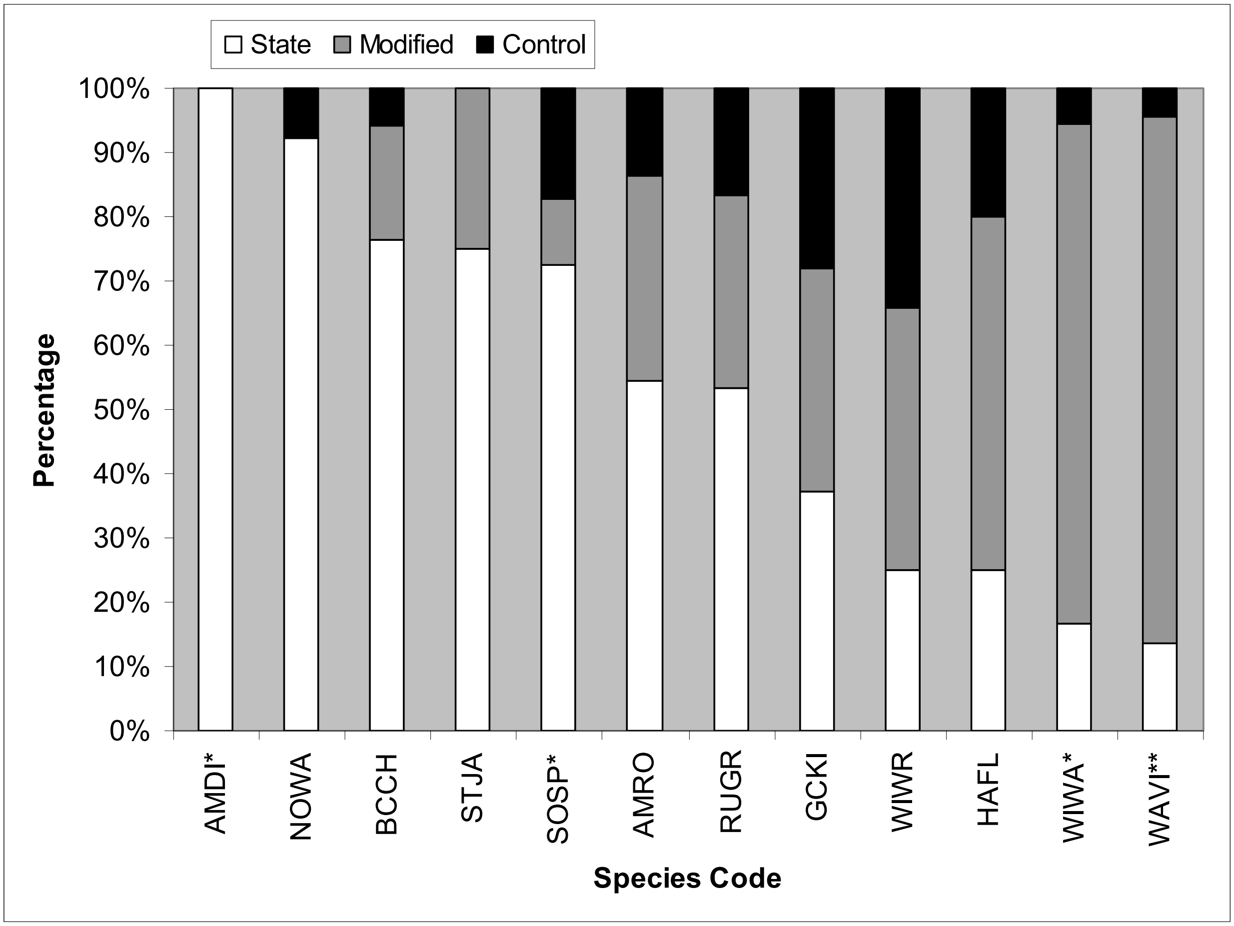


Figure 7-11. Percentage of bird observations along riparian transects partitioned by site type (control, modified, state) during the third sampling interval (second post-harvest sampling interval, 2003/2004). Data shown are for species with ≥5 detections and >50% of total detections along riparian transects. See Appendix 7-A for species codes. * denotes riparian-obligate and ** denotes riparian-dependent species according to Bureau of Land Management (no date).

Diversity and Community Comparisons

Sixty-two bird species were recorded on-transect in 2003/2004 on all sites combined. During that same period, control sites yielded 44 species, modified buffer treatment sites yielded 45, and state buffer treatment sites yielded 51. Figure 7-12 summarizes mean numbers of species detected along each transect during the second post-harvest sampling interval.

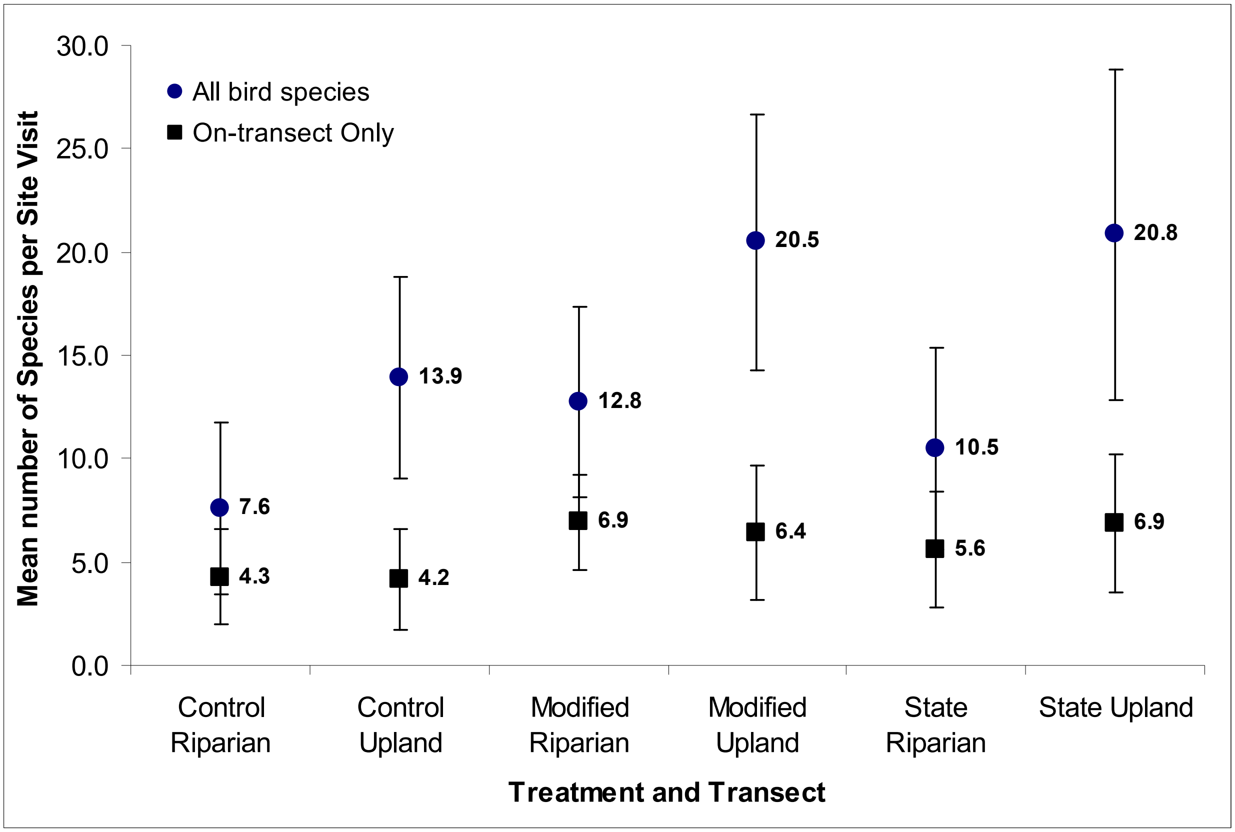


Figure 7-12. Mean number of bird species detected per survey along each transect and treatment in eastern Washington, 2003/2004. Means for all detections and only those on-transect are shown. Each mean was computed from the sum of the total number of species detected during each survey divided by the total number of surveys (*n* = 72 for control and state; *n* = 60 for modified). Error bars denote 1 SD.

Total numbers of species observed at each site during the second post-harvest sampling interval are summarized in Table 7-18. Means for treatments were significantly different ($F_{5,28} = 3.62$; $p = 0.012$). The subsequent Tukey–Kramer HSD test indicated that for pairwise comparisons there were few significant differences between the mean number of species per treatment–transect combination, with only the upland transects of modified and state buffer treatment sites having more species than the riparian transects of control sites (control riparian vs. state upland: $p = 0.03$; control riparian vs. modified upland: $p = 0.02$; Table 7-19).

Table 7-18. Total and mean numbers of bird species observed on-transect along the riparian and upland transects of study sites in eastern Washington during the third sampling interval (second post-harvest sampling interval, 2003/2004).

Treatment	Site	Riparian	Upland
Control	Bear	7	12
	Browns	16	24
	Chewelah	15	19
	Muddy Control	16	15
	Power	19	15
	Rocky Control	8	14
	mean	*13.5*	*16.5*
Modified	Buck East	16	24
	Butte	22	20
	Cee Cee Ah	18	21
	Mill	26	23
	Rocky Cut	22	26
	mean	*20.8*	*22.8*
State	Amazon	27	25
	Buck West	16	26
	Middle	14	29
	Muddy East	16	16
	Muddy West	19	15
	Sherry	19	21
	mean	*18.5*	*22.0*

Table 7-19. Mean numbers of bird species detected on-transect during bird surveys in eastern Washington for each combination of treatment (control, modified, state) and transect (R = riparian, U = upland) during the second post-harvest sampling interval (2003/2004). Means were calculated from the total number of bird species detected during the second post-harvest sampling interval by site by transect divided by total number of sites by treatment; excludes flyovers. Double-underscoring connects means that are not significantly different ($\alpha = 0.05$, Tukey–Kramer HSD). Sample sizes (i.e., number of sites) are indicated in parentheses.

Treatment and Transect					
Control-R	**Control-U**	**State-R**	**Modified-R**	**State-U**	**Modified-U**
(6)	(6)	(6)	(5)	(6)	(5)
13.5	16.5	18.5	20.8	22.0	22.8

Morisita's index of similarity (*C*) was calculated for comparisons of riparian and upland transects by treatment, using bird observations that were on-transect (Table 7-20). Similar values (range 0.64–0.67) were obtained for control, modified buffer, and state buffer treatment sites, indicating that the riparian and upland bird communities at each treatment type during the second post-harvest sampling interval differed to a similar degree. Morisita's *C* for bird communities along riparian transects at modified and state buffer sites were highly similar according to paired comparisons with the control sites (*C* = 0.94 in both cases). At 0.91, Morisita's *C* indicated that the bird communities along riparian transects of modified buffer and state buffer treatment sites were also similar, albeit by a lesser degree than for the other riparian comparisons.

Table 7-20. Morisita's index of similarity (*C*) for paired comparisons of bird communities along riparian and upland transects in eastern Washington during the second post-harvest sampling interval (2003/2004). Only bird observations that were on-transect are included.

Site or Transect Comparison	Morisita's *C*
Control: riparian vs. upland	0.67
Modified: riparian vs. upland	0.64
State: riparian vs. upland	0.65
Riparian: control vs. modified	0.94
Riparian: control vs. state	0.94
Riparian: modified vs. state	0.91

Discussion

Relative Abundance and Habitat Relationships

Significant differences in the mean abundance of all songbirds (on-transect) were observed among treatments. Forest harvesting at modified buffer and state buffer treatment sites resulted in habitat changes that improved habitat suitability for more species than the number of species for which habitat conditions declined (Table 7-12). Examination of habitat conditions for the upland transects of each treatment type shows that the three treatments existed along a vegetation gradient that went from higher amounts of trees, snags, and canopy cover associated with the control sites to open areas with an abundance of cut stumps (modified and state buffer treatment sites; Figure 7-13). The existence of a gradient in vegetation structure was less evident along riparian transects, indicating that riparian buffers were similar in terms of their vegetation communities (Figure 7-14).

Differences in the relative abundance of songbird species indicate that treatment and transect effects were evident during the second post-harvest sampling interval (Table 7-12). Overall, forest harvesting led to increased bird abundance along both transect types (though site effects cannot be discounted entirely). Both riparian and upland habitats contain important habitats for bird species. For 12 species, riparian habitats (transects) were used by >50% of the individuals within those species. Conversely, 30 species showed individuals that used riparian transects for <50% of the records (Figure 7-10).

Theoretically, the relationship between riparian buffer width and the upper limit of bird density within such buffers differs considerably among species. As such, there should be an upper limit to buffer width beyond which bird or species density (for those species requiring or preferring riparian habitats) does not increase appreciably. Modified riparian buffers tended to have higher numbers of birds and greater numbers of species compared with riparian buffers at the control or state buffer treatment sites. While those differences were not statistically significant, it is reasonable to expect that sites with a logged area adjacent to a wider riparian buffer would tend to have greater numbers of birds and species simply because of the juxtaposition of large (viable) patch sizes of different habitat types (i.e., early seral upland and mid-seral riparian). Such areas offer a range of habitat types (seral stages) for habitat specialists and generalists alike, within a smaller overall area. However, assuming that the present experimental design does not lack sufficient power, the absence of a significant difference in abundance or alpha diversity between the riparian transects at modified and state buffer treatment sites implies that the difference in habitat quality between modified and state buffers may not be appreciable for songbirds.

Beyond songbirds as a general group, when only those bird species commonly known for their strong affinity for riparian habitats were considered, the riparian buffer at state buffer treatment sites was used to a greater extent than the riparian areas at control or modified buffer treatment sites (Figure 7-11). In turn, riparian transects at modified buffer treatment sites were used more than at control sites. Of the 4 species considered to be riparian obligates or riparian-dependent by the Bureau of Land Management (no date), American Dipper and Song Sparrow were more abundant at state buffer treatment sites, while Wilson's Warbler and Warbling Vireo were more abundant at modified buffer treatment sites. All ten records of American Dipper were at the "Middle" state site. That species' association with that site was more likely a result of local stream conditions than of the habitat conditions in the riparian buffer. Further, because of repeated surveys at the same sites (i.e., 12 site visits during the second post-harvest sampling interval), it is highly probably that the same bird(s) were counted more than once. Song Sparrow is recognized as a riparian obligate, but as far as riparian habitat conditions, it prefers open areas in earlier seral stages. Hence, it is not surprising that most Song Sparrow occurred in state buffer treatment sites (Figure 7-14).

Wilson's Warbler is a neotropical migrant songbird whose breeding territories are usually located in deciduous forest habitats with extensive shrub thickets, where it usually nests at base of shrubs or small trees or under dense bunches of grass or sedge (Ammon and Gilbert 1999; Campbell et al. 2001). Increased abundance of Wilson's Warbler in the riparian buffers at modified buffer treatment sites likely reflected the juxtaposition of suitable conditions within the buffer (e.g., deciduous tree component) as well as conditions in the adjacent logged area (e.g., reduced overstory, presence of shrubs and regenerating trees) (Figure 7-13; Figure 7-14).

Warbling Vireo is a neotropical migrant songbird that prefers nesting in deciduous trees (Gardali and Ballard 2000). Warbling Vireo is strongly associated with mature mixed deciduous woodlands in riparian areas and occurs near forest edges and interiors (Gardali and Ballard 2000). In this study, the best explanation for the peak abundance of Warbling Vireo at the modified buffer treatment sites is increase in edge habitat that retained more large trees (Figure 7-14).

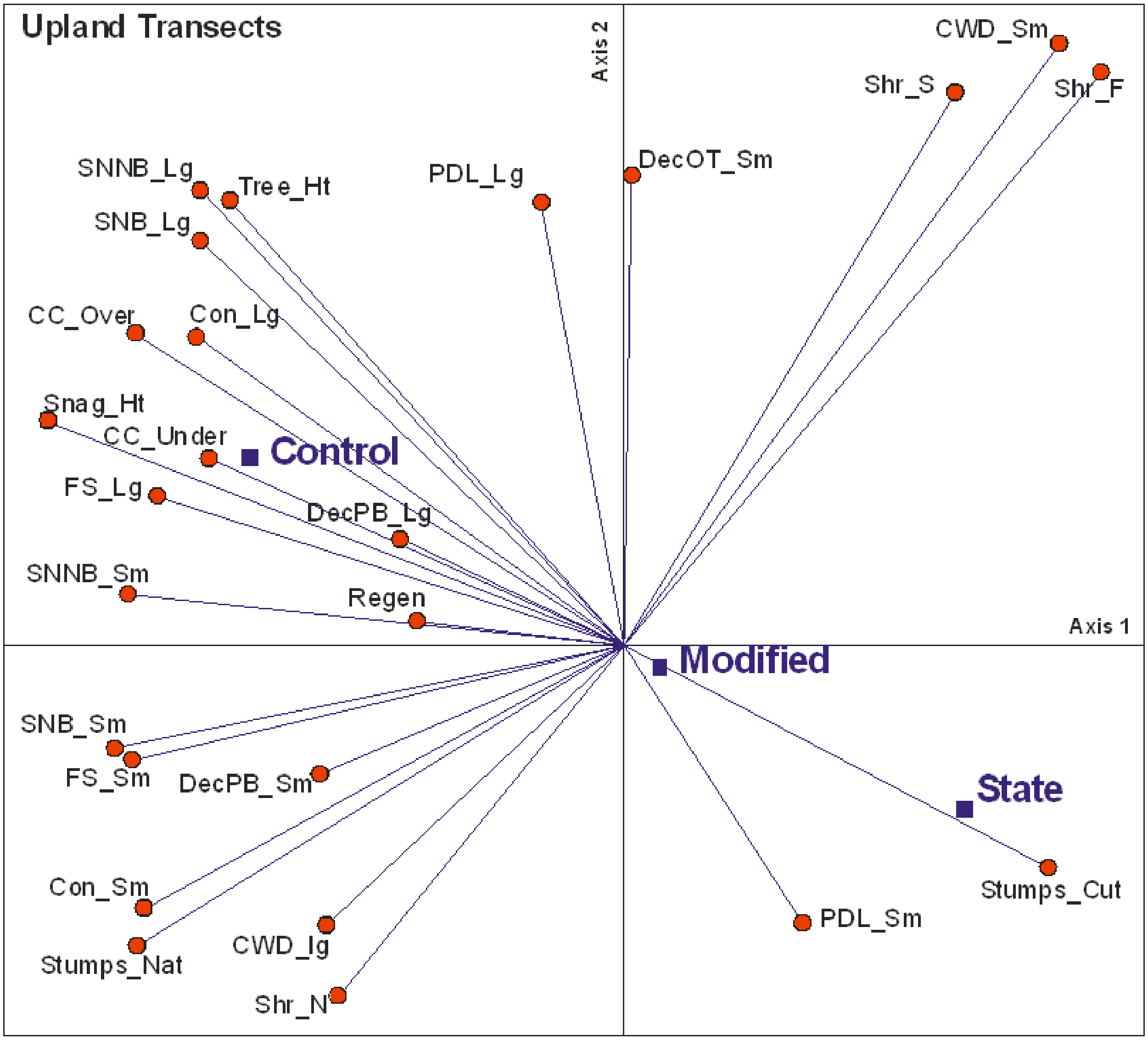


Figure 7-13. Biplot of upland transect habitat condition by treatment in eastern Washington during the third sampling interval (2003/2004). The data in this figure represent the mean condition as derived from the normalized score of the PCA on the habitat data matrix. WD_Long = CWD >5m long; WD_Short = CWD <5m long; Stumps_Cut = Cut stumps; Stumps_Nat = Natural stumps; CC_Over = Overstory cover; CC_Under = Understory cover; Regen = Regenerating Conifers < 0.5m tall and < 4 cm DBH; SNB_Sm = Snags with bark and <25 cm DBH; SNB_Lg = Snags with bark and >25 cm DBH; SNNB_Sm = Snags without bark, <25 cm DBH; SNNB_Lg = Snags without bark, >25 cm DBH; Con_Sm = Small-diameter conifers; DecPB_Sm = Small-diameter birch and poplar; DecOT_Sm = Small-diameter deciduous; FS_Sm = Small-diameter fir and spruce; PDL_Sm = Small diameter pine, Douglas-fir, and larch; Con_Lg = large-diameter conifers; DecPB_Lg = Large-diameter birch and poplar; DecOT_Lg = Large-diameter deciduous; FS_Lg = Large -diameter fir and spruce; PDL_Lg = Large-diameter pine, Douglas-fir, and larch; Tree_Ht = Average tree height; Snag_Ht = Average snag height; Shr_S = Seed-bearing shrubs; Shr_F = Fruit-bearing shrubs; Shr_U = Non-food shrubs; CWD_Sm = Any piece <5m long; CWD_Lg = Any piece >5m long.

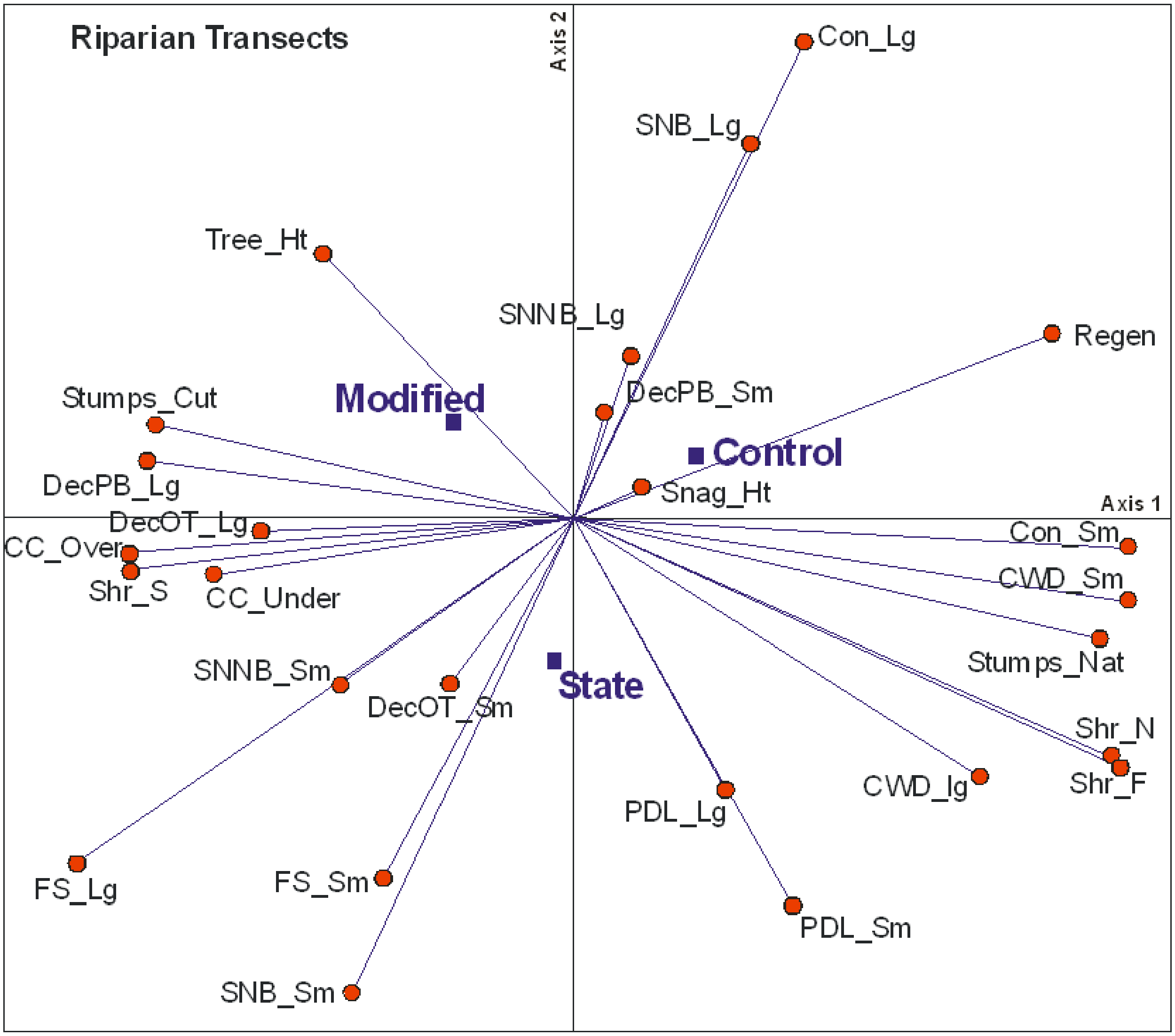


Figure 7-14. Biplot of riparian transect habitat condition by treatment in eastern Washington during the third sampling interval (2003/2004). The data in this figure represent the mean condition as derived from the normalized score of the PCA on the habitat data matrix. WD_Long = CWD >5m long; WD_Short = CWD <5m long; Stumps_Cut = Cut stumps; Stumps_Nat = Natural stumps; CC_Over = Overstory cover; CC_Under = Understory cover; Regen = Regenerating Conifers < 0.5m tall and < 4 cm DBH; SNB_Sm = Snags with bark and <25 cm DBH; SNB_Lg = Snags with bark and >25 cm DBH; SNNB_Sm = Snags without bark, <25 cm DBH; SNNB_Lg = Snags without bark, >25 cm DBH; Con_Sm = Small-diameter conifers; DecPB_Sm = Small-diameter birch and poplar; DecOT_Sm = Small-diameter deciduous; FS_Sm = Small-diameter fir and spruce; PDL_Sm = Small diameter pine, Douglas-fir, and larch; Con_Lg = large-diameter conifers; DecPB_Lg = Large-diameter birch and poplar; DecOT_Lg = Large-diameter deciduous; FS_Lg = Large -diameter fir and spruce; PDL_Lg = Large-diameter pine, Douglas-fir, and larch; Tree_Ht = Average tree height; Snag_Ht = Average snag height; Shr_S = Seed-bearing shrubs; Shr_F = Fruit-bearing shrubs; Shr_U = Non-food shrubs; CWD_Sm = Any piece <5m long; CWD_Lg = Any piece >5m long.

Community Diversity and Similarity

State buffer treatment sites had the highest alpha diversity (species richness) during the second post-harvest sampling interval, with 51 species on-transect compared with 44 and 45 species at the control and modified buffer treatment sites, respectively. Species richness at all three site types was driven by conditions along the upland transect. Although mean numbers of birds per site visit that were on-transect differed by only 0.1 to 1.3 species between the riparian and upland transects, the greater number of species off-transect along upland transects implies that there was a greater chance that different species would be sampled on-transect during subsequent visits. In essence, there was a greater "pool" of species nearby. An alternate explanation involves stream noise along the riparian transects. Several field technicians reported that while surveying riparian transects, stream flow noise often masked bird calls and songs to the point that upper detection distances were substantially shorter than along upland transects where stream noise did not aurally hinder bird detections. One observer noted that, at some sites, stream noise conditions resulted in birds in the riparian buffer being more audible from the upland transect than from the riparian transect. In such cases, the net result would be a reduced detection probability (McCallum 2005) along riparian transects, thereby biasing the estimates of species richness and abundance downward along those transects.

When the range of species sampled along each transect at each site was considered over the course of the second post-harvest sampling interval, the only statistically exclusive differences were the higher means of the upland transects of logged sites in contrast with the low mean for riparian transects of control sites (Table 7-18). Again, this supports the finding that mean species richness was highest along the upland transects at logged sites.

The relatively low values and small range in indices of community similarity (Morisita's *C*) for comparisons between the riparian and upland transects indicated two things. First, riparian and upland avian communities were appreciably different at each treatment type; second, the magnitude of the differences between the avian communities along riparian and upland transects were similar among treatment types, suggesting that treatment effects on community similarity were modest during the second post-harvest sampling interval. That is not to say that forest harvesting did not alter bird communities (it most likely did), but rather the extent of any differences between riparian and upland communities at a given treatment type did not appear to be driven by the type of riparian buffer retained at a site (the uplands at modified and state buffer treatments were logged following the same prescription).

Forest communities develop through a continuum that, for conceptual purposes, has been grouped into sequential stages (e.g., Thomas 1979; Bunnell et al. 1999). Figure 7-15 depicts 6 seral (structural) stages of forest succession[13] as they are conceptualized for a forest stand culminating in an old-growth climax. Stage 1 begins with a vegetation community dominated by non-woody plants such as rushes, grasses, and forbs. By stage 2, woody shrubs and shade-intolerant conifer seedlings are present. At stage 3, conifers begin to dominate the site as they compete with other plants for light, water, and nutrients. During stage 4, the conifer canopy closes, effectively shading out vegetation on a forest floor that is becoming increasingly barren. At stage 5, conifers are mature, and weaker trees begin to die. Those trees will stand as snags that will eventually fall to the forest floor, where they become coarse woody debris. Shade-tolerant species of conifers become established in the understory. During stage 6, shade-intolerant species of conifers are gradually outcompeted by the shade-tolerant ones. Those shade-tolerant species continue to germinate and grow up in the understory. The number of standing dead trees and downed logs increases. Dead and fallen trees create gaps in the canopy that, together with other stand-mediated changes to microclimatic conditions, create suitable conditions for shade-intolerant species. The increasing diversity of plant species—and their increasing density, size, age, and state of decay—promotes the structural heterogeneity of the forest during stage 6.

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[13] Other authors have used 4 stages (e.g., Franklin 1990; Klenner et al. 2000).

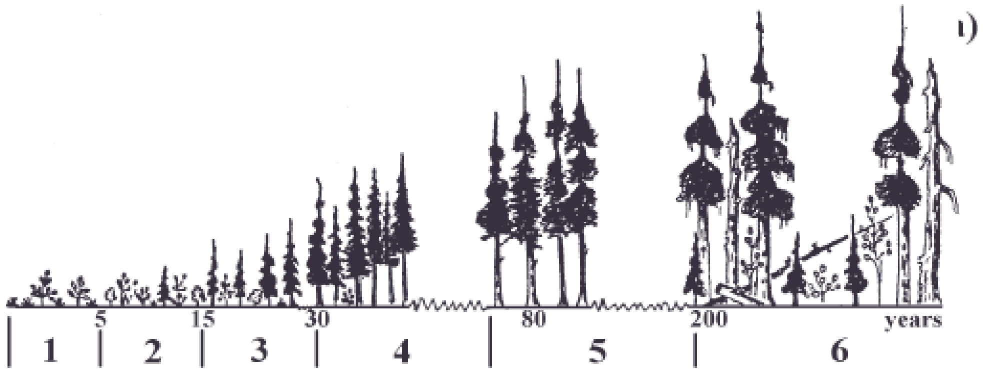

Figure 7-15. Conceptual model of the seral (structural) stages (1–6) and timeline (years) of natural coniferous-forest succession. The seral stages are: 1 = grass–forb; 2 = shrub–seedling; 3 = pole–sapling; 4 = young; 5 = mature; 6 = old growth. Source: Bunnell et al. (1999).

The increases in species richness recorded on upland transects at modified and state buffer treatment sites during the second post-harvest sampling interval fit well with standard conceptual models. Coniferous forests typically display the pattern of species richness depicted in Figure 7-16. In that pattern, species richness is relatively highest during the earliest and latest seral stages. As the tree canopy closes (around stage 4), the understory is shaded out, resulting in a decline in habitat suitability for many species. As succession progresses, the structural diversity and richness of plant species in the forest increases. This brings about an improvement in habitat suitability that in turn facilitates an increase in wildlife-species richness. Although wildlife-species richness in early and late successional stages can be similar and can have some species in common, the makeup of the wildlife communities is notably different. Thus, increases in relative abundance (at upland sites) and species diversity after forest harvesting of mid-seral second-growth stands in the study area are expected because, pre-harvest, stands in the present study were at a seral stage of ~4–5 (i.e., second-growth stands dominated by Douglas-fir ~55–65 years old), when natural levels of diversity and abundance are typically lowest. By harvesting sites and then allowing succession to proceed to stage ~2, an increase in species richness is expected. Similarly, over time the seral stage of the buffers will advance, resulting in further increases in species diversity.

Though not a novel finding by any means, this study clearly shows that the habitat suitability for a given bird species is tied to measures of the physical environment that can be altered both directly and indirectly by forest harvesting. The existence of this link means that forest managers have a degree of control over, and thus responsibility for, their effects on avian communities.

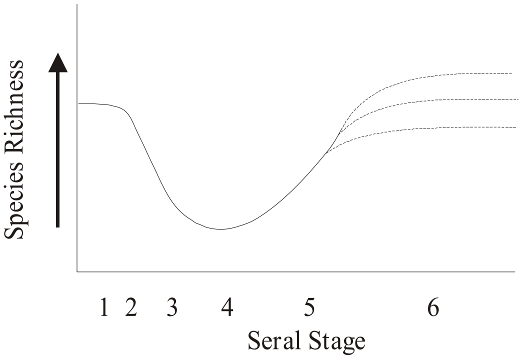

Figure 7-16. Conceptual model of the relative change in species richness (number of species) for wildlife communities as a function of forest seral stage, showing three possible outcomes (dashed lines) of lesser, similar, and greater species richness during the earliest and latest stages. Refer to Figure 7-15 for a description of seral stages. Source: adapted from Franklin (1990).

Conclusions and Management Implications

Forest harvesting alters the abundance, species richness, and community structure of songbirds in the Douglas-fir–dominated forests of northeastern Washington. The geographic scale of this study allows inference of study results across a large land base, particularly where similar habitats occur. Given that the sites selected for inclusion in this study represent a considerable proportion of the managed forest types in Washington State, the scale at which the conclusions of this study can be applied is substantial.

The purpose of this study was not to comment on specific objectives of songbird conservation but rather to assess songbird responses to two different stand-level harvesting prescriptions that differ primarily in the extent to which riparian buffers were retained after adjacent (upland) second-growth coniferous forests were clearcut. It is hoped that information gained by this study will assist managers and biologists in meeting the conservation objectives set by government and/or industry.

While two riparian-obligate or riparian-dependent bird species (i.e., Wilson’s Warbler, Warbling Vireo) might have responded to the enhanced riparian buffer conditions or modified buffer treatment sites (cf. state buffer treatment sites), there is no clear indication that the difference between modified and state buffer conditions have significant implications for habitat suitability for substantial numbers of breeding songbirds. It is not evident whether this lack of response was because of inconsequential differences in habitat suitability between the two buffer types or because even the modified buffer conditions were below those required by many riparian-obligate or riparian-dependent species. The fact that none of the more commonly encountered riparian-obligate or -dependent species were strongly associated with the riparian transects at control sites having Type 3 streams suggests that riparian buffer width in these forest types is not a critical habitat factor for these species. What is essential in terms of maximizing biodiversity, however, is that landscapes simultaneously support the full range of seral stages. Where individual species or guilds are of greater concern, species- and site-specific habitat management prescriptions are required within those landscapes. Modern forest management utilizes both approaches.

This study presents a short-term (~10-year) view of forestry–wildlife interactions at the stand level,[14] using mid-seral coniferous forests as a baseline, and not old-growth or climax seral forests at

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[14] "Stand level" describes the level of forest management at which a relatively homogeneous land unit can be managed under a single prescription or set of treatments to meet well-defined objectives. Stand-level management occurs on a land base ranging in size from a

a landscape level. This presents a few challenges. First, mid-seral forests are comparatively low in species richness. As a result, any forest harvesting is likely to cause an increase in alpha diversity within a few years of harvest. Second, it is unclear how unlogged buffers will progress through time on an industrial forest land base. For example, if the buffers are to be retained in perpetuity, they would conceivably reach old-growth status while the surrounding upland passed through multiple rotations. Yet, according to this study, they would continue to be greatly influenced by the adjacent harvesting. Third, and most importantly, the conservation of wildlife such as songbirds requires that management occur at landscape scales or larger. While site prescriptions are necessary to deal with stand-level conservation objectives, they are only part of a management hierarchy. Further, conserving ecological communities comprising other taxonomic groups (i.e., plants, invertebrates, amphibians, reptiles, and mammals) and non-songbirds will often require management actions very different from those that would be prescribed for songbirds alone.

A number of factors potentially influence the veracity of this study's results or limit the extent to which the conclusions can be used to aid in constructing management policy for managing and conserving songbirds in the coniferous forests of the Pacific Northwest. The purpose of presenting these limitations is to make the reader aware of (1) factors that could have influenced the results of the study and (2) factors that should be considered if this study is to be used to guide formal policy development.

Data Access—The lack of access to the raw data from the pre-harvest and first post-harvest sampling intervals precluded detailed comparisons of bird abundance on the riparian and upland transects at the study sites over time. It also precluded error-checking the summary results presented in chapter 6 of O'Connell et al. (2000). Assuming that the data in O'Connell et al. (2000) are free of substantial calculation errors, the absence of any significant differences in the abundance of songbirds during the course of the study (Table 7-11) suggests that, although changes in the abundance of individual species commonly occurred (Table 7-15), net abundance of the more common songbirds did not change appreciably either at control or treatment sites. For example, a decline in species such as Townsend's Warbler along the upland transects at state buffer treatment sites post-harvest was offset by a concomitant increase in species like Dark-eyed Junco along those same transects.

Data Accuracy and Study Design—Much has been written on the subject of observer bias and the effects of overcounting, undercounting, and misidentifying species (e.g., Bart 1985; McCallum 2005). We acknowledge that the database assimilated over the course of this study contains all three types of errors to some unknown extent. Although steps were taken to reduce this bias (see Methods section), its potential influence on the results must be considered nonetheless. This study only examined abundant, conspicuous songbirds that allow observers to be within 15 m. As such, rare, reclusive, nocturnal, quiet, and widely ranging species are likely to go underrepresented in the analyses. Notable taxa are the Accipitridae (diurnal raptors) and Strigidae (typical owls), both of which require survey techniques much different from those employed here.

Nesting Success—An underlying assumption untested by this study is that relative abundance of songbirds is indicative of habitat quality. To test this assumption, nesting success rates (a measure of fitness) would need to be monitored over time. Measuring nesting success is costly and logistically difficult. Until such time as there are compelling reasons to believe that monitoring trends in relative abundance is incapable of revealing wildlife responses to habitat, it seems reasonable to continue monitoring using the methods in this study.

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few to several thousand hectares. "Landscape level" describes planning areas delineated on the basis of topographic or geographic features. Typically they cover a watershed or series of watersheds and range in size from ~5000 to 100,000+ ha.

Habitat Assessment—Food is a critical habitat feature for breeding songbirds; however, this study did not measure food directly. While measures of the relative abundance of berry-producing shrubs likely provides a reasonable approximation of some food types, the influence of the abundance of insects, arthropods, and other invertebrates on the relative abundance of birds during the study is unknown. This study examined species–habitat relationships for only a portion (<20%) of the year. Spring is the most important time of year for neotropical and other migrants in the Pacific Northwest, but many species occur in the area year-round. Factors that influence habitat conditions during winter (e.g., roosting habitat, presence of and access to food) must also be considered.

Extensive forestry management in the Pacific Northwest has historically included clearcutting, prescribed fire, regeneration by seed trees, protection from forest fires, and salvage of timber killed by windthrow or disease (Carey and Harrington 2001). In the last 20 years, planting of Douglas-fir has increased, with most forest stands > 40 years old originating from seed. Current management practices include the retention of seed trees and coarse woody debris and the planting of seedlings. Implicit in the shift to ecosystem management has been the retention of riparian management zones (RMZs) around perennial fish- and non–fish-bearing streams, which have been required in some form since 1978. Diversity and structural complexity of vegetative communities have been shown to influence the number and diversity of niches for birds and other species (MacArthur and MacArthur 1961; MacArthur et al. 1966; Willson 1974; Roth 1976; Whelan 2001; Hayes et al. 2003).

Since this study’s inception in 1991, there have been changes to the Washington Forest Practices Rules and Regulations that have implications for our findings. The Forest Practices Rules of 1988 required a riparian buffer that met the state configuration (i.e., not less than 7.5 m from the ordinary high-water mark) for both fish-bearing (Type 3) and non–fish-bearing (Type 4) perennial streams. The development of the modified riparian prescription was an alterative to the state buffer that provided for greater structural complexity, heterogeneity, and ultimately greater diversity of both flora and fauna. Lee et al. (2004) identified the current trend in riparian management prescriptions as tending toward site-specific buffers that vary among broadly similar harvest areas to within a single harvest area. The Forest Practices Rules of 2001 seem to fit this model and, aside from changing how watercourses are defined, are site-specific with many options. Lee et al. (2004) suggest that the primary benefit of this type of prescription is the identification of well-defined criteria to delineate the riparian zone. These criteria are specifically defined by the modifying factors selected by jurisdictions, and buffers are applied in a predictable response to these criteria. The caveat to tailor-made buffers is the greater complexity in guidelines. For example, Washington now describes 3 zones of management for Type F streams (Type 3): (1) the core, (2) inner, and (3) outer zones, with various levels of tree removal, thinning, hardwood conversion, and other activities permitted within these zones depending on the site class of land, the management harvest option, and the bankfull width of the stream. In general, the site class will form the RMZ prescription. It appears that riparian management in eastern Washington has become (necessarily) site-specific. This implies that applying a one-size-fits-all approach to riparian management has been replaced by a more diverse, site-specific approach, which is consistent with the trend documented by Lee et al. (2004). For Type Np streams, or perennial non–fish-bearing streams (Type 4), the RMZ rules are slightly less complicated; however, there is ample room for site-specific management. Considering that the RMZ rules have evolved to embrace a site-specific approach, it seems only fitting that forest management in general adopt the same principals when considering biodiversity. Managing for biodiversity on a regional scale does not account for variation at the landscape or site level within the region, which could have dire consequences for species that are limited geographically, that occur at naturally low densities, or that are particularly sensitive to habitat change. As an alternative to the status quo, management that focuses on stand heterogeneity (as opposed to tree-species monocultures with reduced diversity) should produce landscapes with high biocomplexity and biodiversity at multiple scales, and therefore these landscapes should be more resilient to changes brought about by large-scale disturbances (Holling 2001).

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Appendices

Appendix 7-A. Species codes and common names of all birds encountered during breeding-bird surveys in eastern Washington, 2003/2004.

Species/Group	Common Name
ARDEIDAE	**Bitterns, Egrets, and Herons**
GBHE	Great Blue Heron
ACCIPITRIDAE	**Kites, Eagles, Hawks and Allies**
OSPR	Osprey
SSHA	Sharp-shinned Hawk
NOGO	Northern Goshawk
RTHA	Red-tailed Hawk
PHASIANIDAE	**Partridges, Pheasant, Grouse, Ptarmigan and Turkey**
BLGR	Blue Grouse
RUGR	Ruffed Grouse
SPGR	Spruce Grouse
WITU	Wild Turkey
COLUMBIDAE	**Pigeons and Doves**
MODO	Mourning Dove
STRIGIDAE	**Typical Owls**
NPOW	Northern Pygmy Owl
CAPRIMULGIDAE	**Goatsuckers**
CONI	Common Nighthawk
APODIDAE	**Swifts**
VASW	Vaux's Swift
TROCHILIDAE	**Hummingbirds**
BCHU	Black-chinned Hummingbird
CAHU	Calliope Hummingbird
RUHU	Rufous Hummingbird
ALCEDINIDAE	**Kingfishers**
BEKI	Belted Kingfisher
PICIDAE	**Woodpeckers**
RNSA	Red-naped Sapsucker
HAWO	Hairy Woodpecker
BBWO	Black-backed Woodpecker
NOFL	Northern Flicker
PIWO	Pileated Woodpecker
TYRANNIDAE	**Tyrant Flycatchers**
OSFL	Olive-sided Flycatcher
WWPE	Western Wood-peewee
HAFL	Hammond's Flycatcher
DUFL	Dusky Flycatcher
PSFL	Pacific-slope Flycatcher
VIREONIDAE	**Vireos**
CAVI	Cassin's Vireo
WAVI	Warbling Vireo
REVI	Red-eyed Vireo
SOVI	Solitary Vireo
CORVIDAE	**Jays, Magpies, Crows, and Ravens**
GRJA	Gray Jay
STJA	Steller's Jay
CLNU	Clark’s Nutcracker
CORA	Common Raven
HIRUNDINIDAE	**Swallows**
TRES	Tree Swallow
PARIDAE	**Chickadees**
BCCH	Black-capped Chickadee
MOCH	Mountain Chickadee
CBCH	Chestnut-backed Chickadee
SITTIDAE	**Nuthatches**
RBNU	Red-breasted Nuthatch
WBNU	White-breasted Nuthatch
CERTHIIDAE	**Creeper**
BRCR	Brown Creeper
TROGLODYTIDAE	**Wrens**
HOWR	House Wren
WIWR	Winter Wren
CINCLIDAE	**Dipper**
AMDI	American Dipper
REGULIDAE	**Kinglets**
GCKI	Golden-crowned Kinglet
RCKI	Ruby-crowned Kinglet
MUSCICAPIDAE	**Bluebirds, Thrushes and Allies**
MOBL	Mountain Bluebird
TOSO	Townsend's Solitaire
SWTH	Swainson's Thrush
HETH	Hermit Thrush
AMRO	American Robin
VATH	Varied Thrush
BOMBYCILLIDAE	**Waxwings**
CEDW	Cedar Waxwing
PARULIDAE	**Wood Warblers**
OCWA	Orange-crowned Warbler
NAWA	Nashville Warbler
YEWA	Yellow Warbler
YRWA	Yellow-rumped Warbler
TOWA	Townsend's Warbler
AMRE	American Redstart
NOWA	Northern Waterthrush
MACW	MacGillivray's Warbler

Species/Group	Common Name
COYE	Common Yellowthroat
WIWA	Wilson's Warbler
THRAUPIDAE	**Tanagers**
WETA	Western Tanager
CARDINALIDAE	**Cardinals, Grosbeaks and Allies**
BHGR	Black-headed Grosbeak
LAZB	Lazuili Bunting
EMBERIZIDAE	**Towhees, Sparrows, Longspurs, and Allies**
SPTO	Spotted Towhee
CHSP	Chipping Sparrow
FOSP	Fox Sparrow

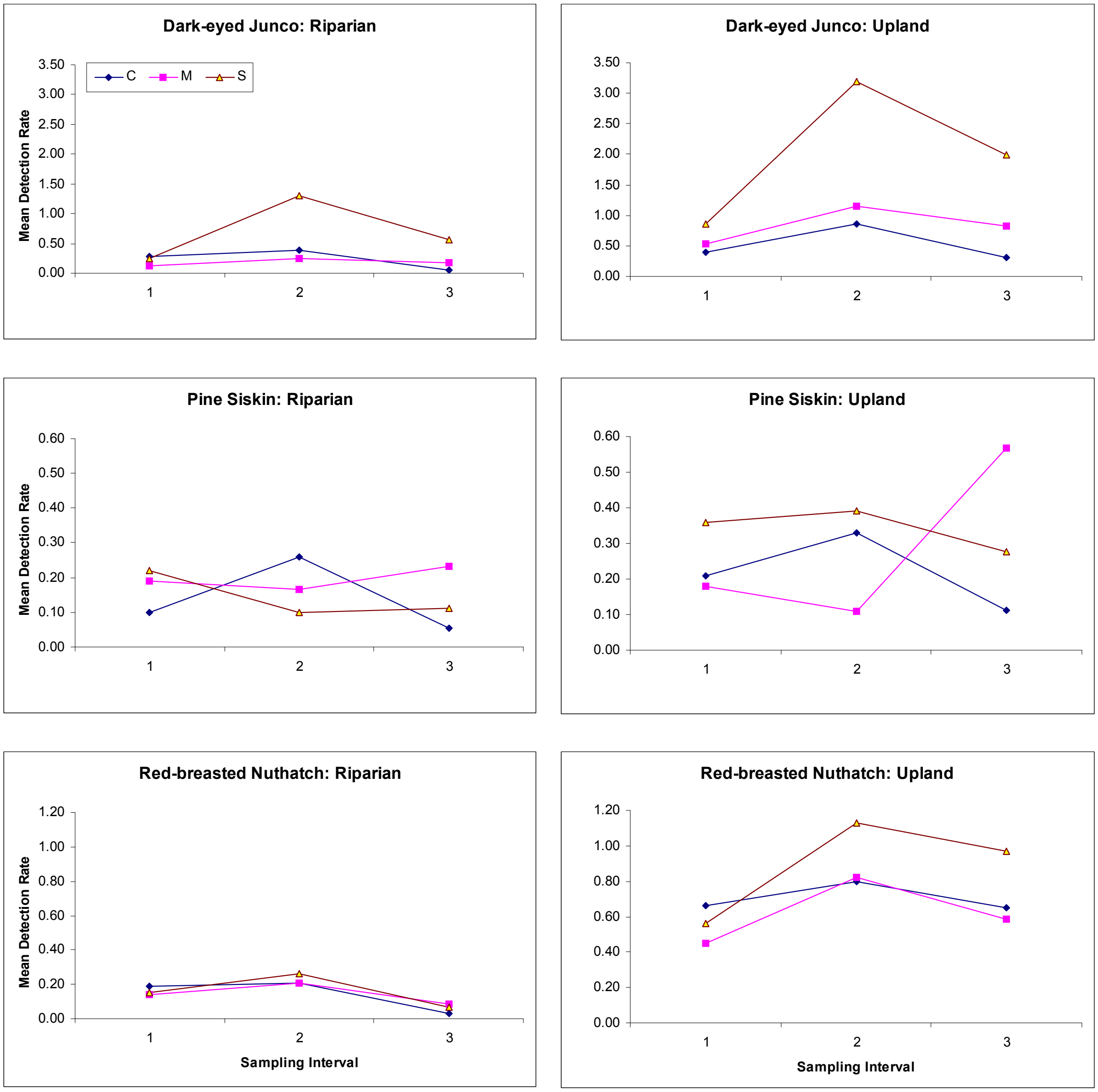
Species/Group	Common Name
SOSP	Song Sparrow
LISP	Lincoln's Sparrow
DEJU	Dark-eyed Junco
ICTERIDAE	**Blackbirds, Orioles and Allies**
BHCO	Brown-headed Cowbird
BUOR	Bullock's Oriole
FRINGILLIDAE	**Cardueline Finches and Allies**
CAFI	Cassin's Finch
RECR	Red Crossbill
PISI	Pine Siskin
EVGR	Evening Grosbeak

Appendix 7-B. Number of birds (excluding flyovers) detected on-transect (i.e., either within 0–8 m on the stream side or 0–22 m on the upland side of riparian transects; or 0–15 m from upland transects) along the riparian (R) and upland (U) transects of the three different sites (control, modified, state) during 6 surveys at each, 2003/2004 (i.e., 12 surveys per transect per treatment). See Appendix 7-A for species names.

	Transect and Treatment						
	Riparian			Upland			
Species	Control	Modified	State	Control	Modified	State	Total
AMDI			10				10
AMRE			2				2
AMRO	6	14	24	4	5	13	66
BBWO				1		1	2
BCCH	1	3	13			7	24
BCHU			2		4	1	7
BHCO		1				14	15
BHGR				2		6	8
BRCR	15	16	4	22	24	8	89
CAFI						1	1
CAHU					1	1	2
CAVI	3	6	3	13	8	11	44
CBCH	40	39	60	85	67	75	366
CEDW						4	4
CHSP			2	1	6	27	36
CONI					1		1
CORA	1						1
DEJU	4	11	40	22	49	143	269
EVGR	3				4		7
GBHE	1						1
GCKI	106	131	140	86	68	63	594
GRJA		1	2	13	3	11	30
HAFL	71	196	89	5	34	34	429
HAWO	1	2	3	1	12	45	64
HETH	7	6	1	5	1	15	35
LAZB					1	3	4
MACW	11	25	18	3	34	36	127
MOCH	1	3	9	10	3	20	46
MODO						1	1
NAWA		2	3	12	17	6	40
NOFL					3	6	9
NOGO		1		1			2
NOWA	1		12	1		1	15
OCWA		3	4		10	35	52
OSFL			2		3	7	12
PISI	4	14	8	8	34	20	88
PIWO		1		1	7	2	11
PSFL				1			1
RBNU	2	5	5	47	35	70	164
RCKI			1			1	2
RECR	1	5		2	1	39	48

	Transect and Treatment						
	Riparian			**Upland**			
Species	**Control**	**Modified**	**State**	**Control**	**Modified**	**State**	**Total**
REVI		1					1
RNSA	4	5	6	3	18	16	52
RTHA						1	1
RUGR	5	9	16	1	1	1	33
RUHU	1	2		1	2		6
SOSP	5	3	21				29
SPGR	2			1		1	4
SPTO						12	12
SSHA				1			1
STJA		1	3		1		5
SWTH	40	61	44	45	65	53	308
TOSO				1	8	6	15
TOWA	74	89	94	94	93	104	548
TRES				1	3		4
VATH	11	4	7	7	12	5	46
WAVI	1	18	3		8	12	42
WETA	2	9		3	10	16	40
WITU	2						2
WIWA	1	14	3		2	4	24
WIWR	125	150	92	10	18	8	403
YRWA	7	12	8	36	32	53	148
Grand Total	559	863	754	550	708	1019	4453

Appendix 7-C. Mean detection rate (birds per survey; relative abundance [RA]) for 23 selected bird species across treatments (C = control, M = modified, S = state), transects (riparian, upland), and sampling interval (1–3) during breeding-bird surveys in eastern Washington, 1992–2004. Plots are presented in descending order of the number of significant differences in F-probabilities on non-orthogonal contrasts for data from 2003/2004 (see Table 7-14). Data for sampling intervals 1 and 2 are from O'Connell et al. (2000, p. 6-39). Mean detection rates were not presented by O'Connell et al. (2000) for those species with no data points plotted (e.g., Orange-crowned Warbler).


Dark-eyed Junco: Riparian

Dark-eyed Junco: Upland

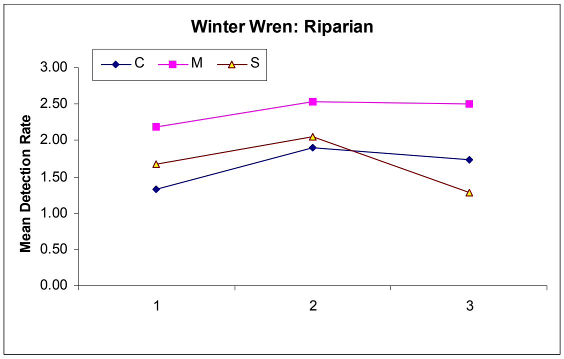
Pine Siskin: Riparian

Pine Siskin: Upland

Red-breasted Nuthatch: Riparian

Red-breasted Nuthatch: Upland

Appendix 7-C continued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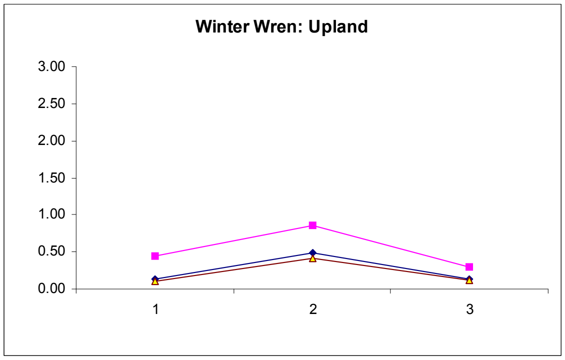
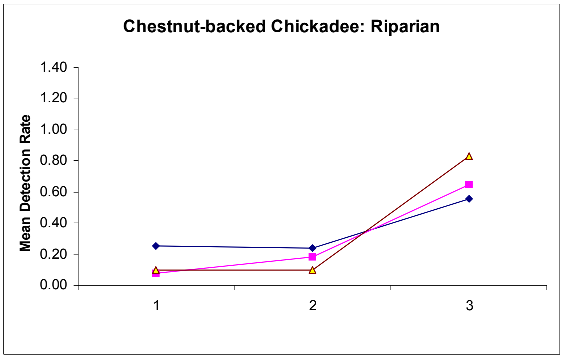
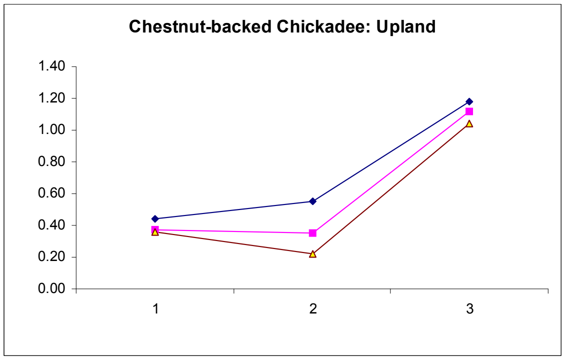
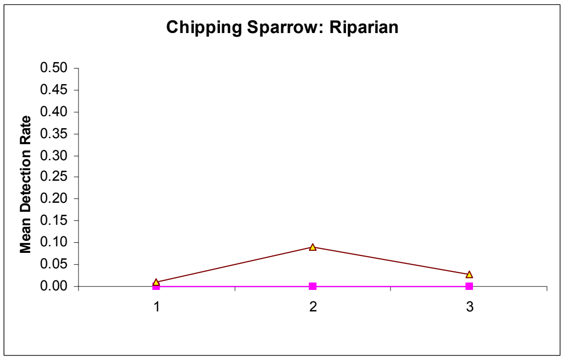
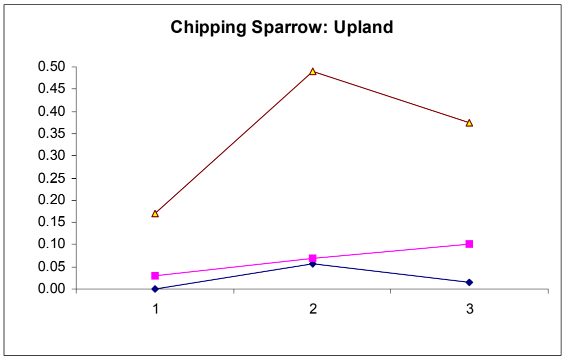
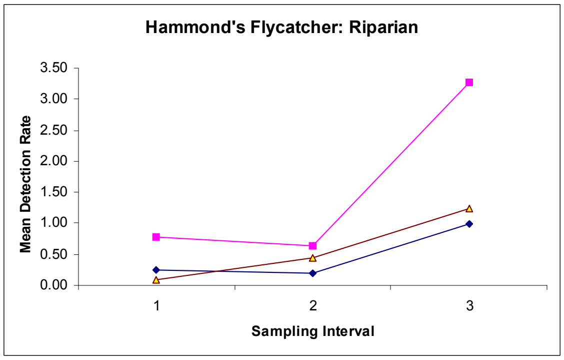

Winter Wren: Upland

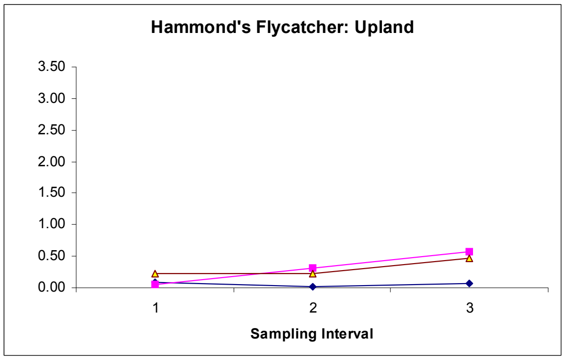

Chestnut-backed Chickadee: Riparian


Chestnut-backed Chickadee: Upland

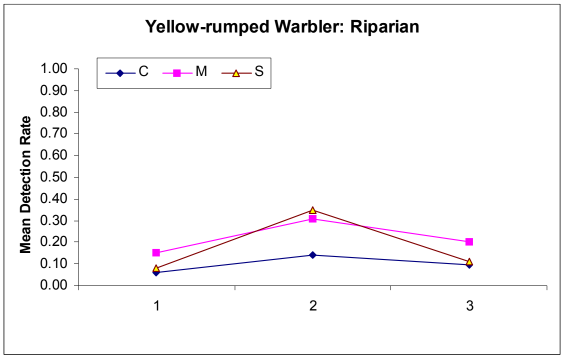

Chipping Sparrow: Riparian

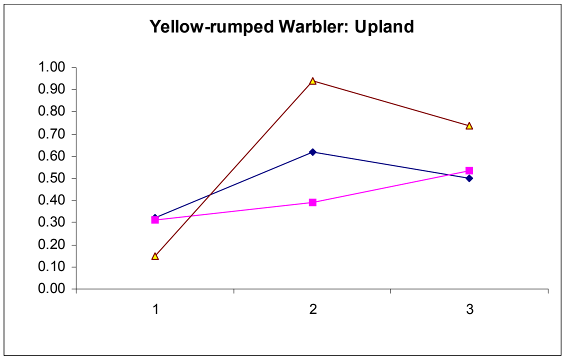

Chipping Sparrow: Upland

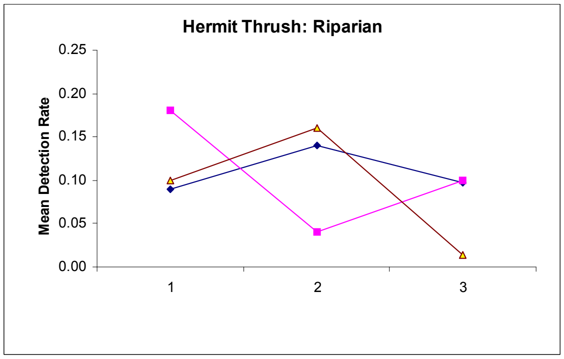

Hammond's Flycatcher: Riparian

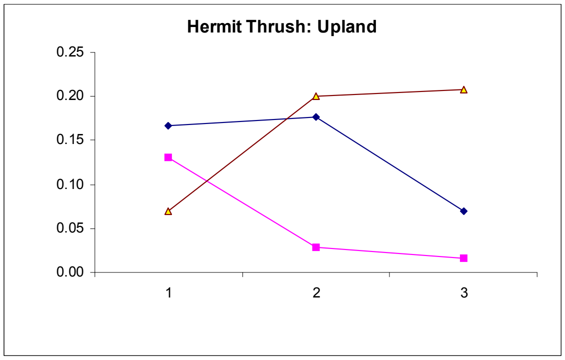

Hammond's Flycatcher: Upland

Appendix 7-C continued


Yellow-rumped Warbler: Riparian


Yellow-rumped Warbler: Upland


Hermit Thrush: Riparian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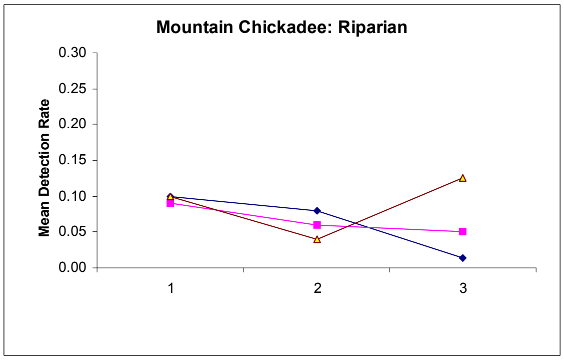
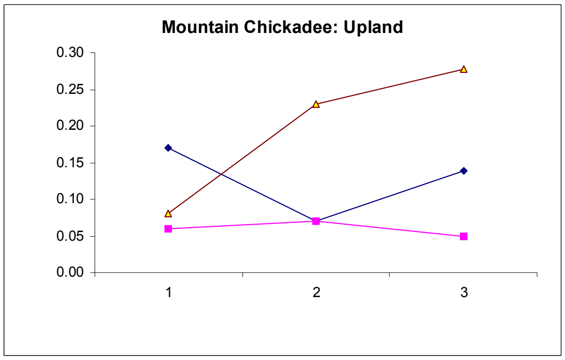
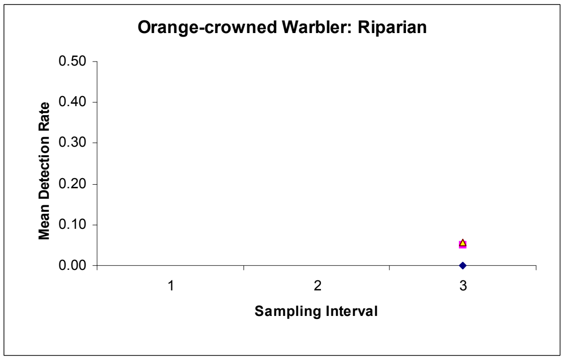
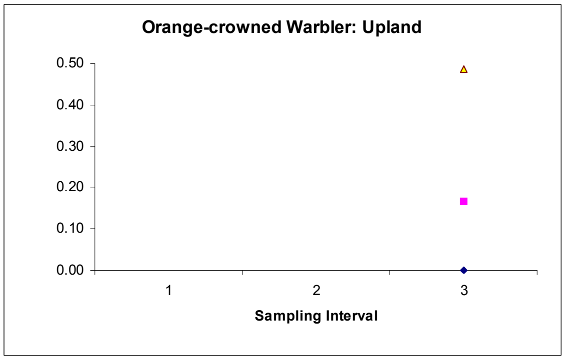
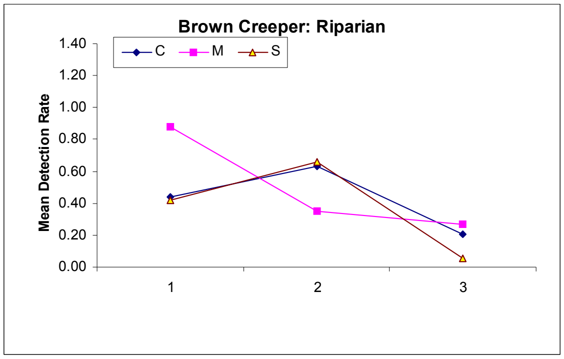

Mountain Chickadee: Riparian

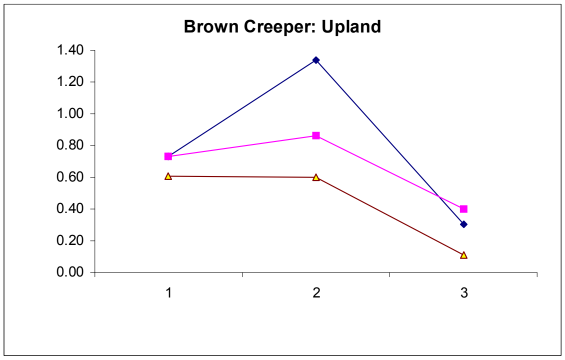

Mountain Chickadee: Upland

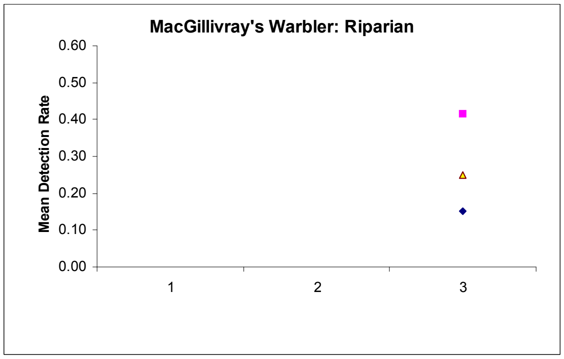

Orange-crowned Warbler: Riparian

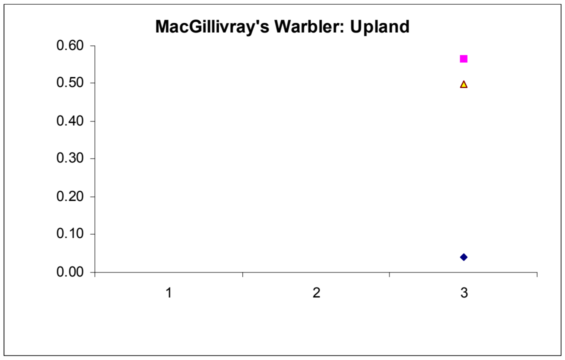

Orange-crowned Warbler: Upland

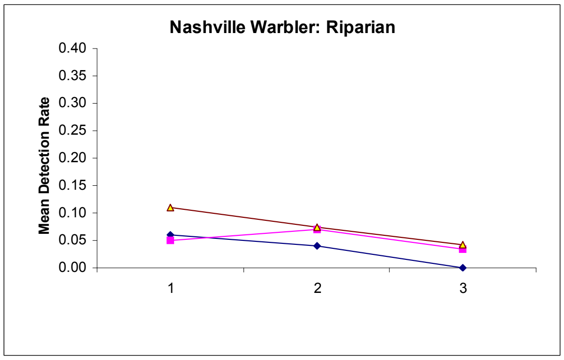
Appendix 7-C continued

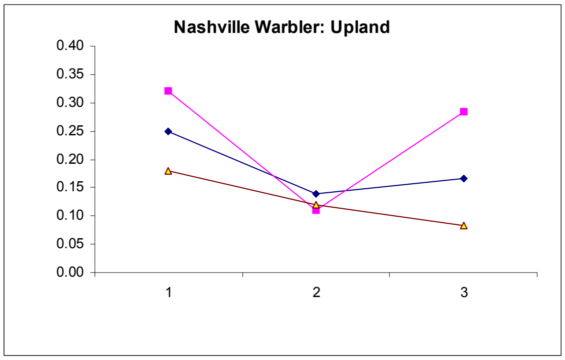


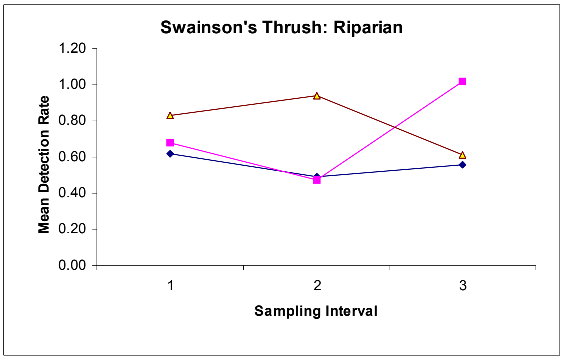


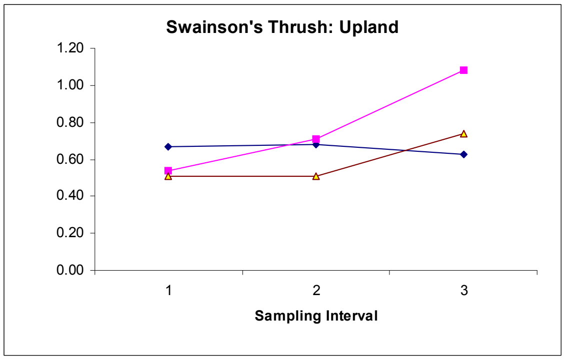












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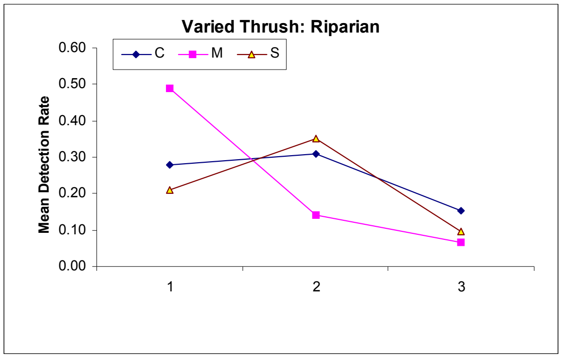

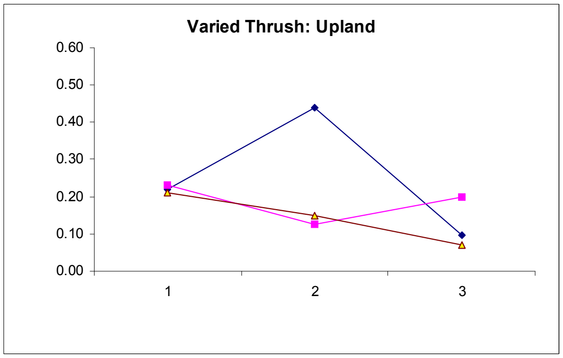

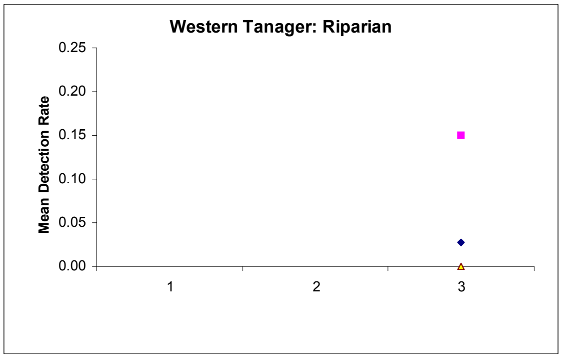

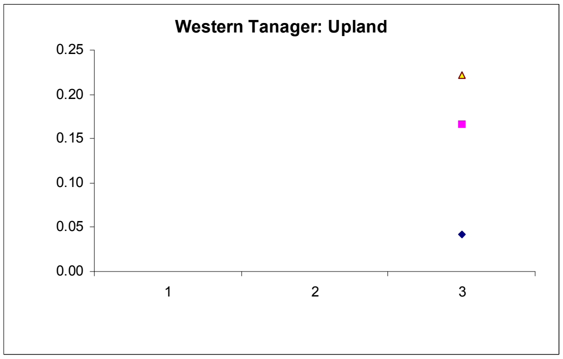

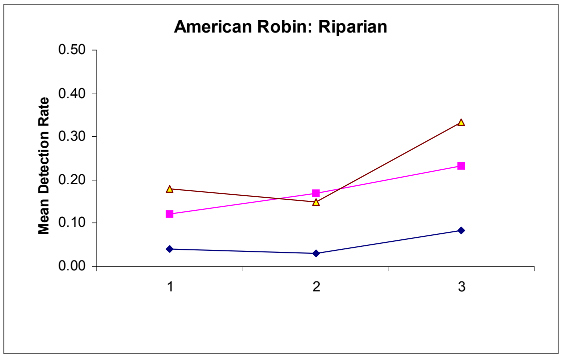

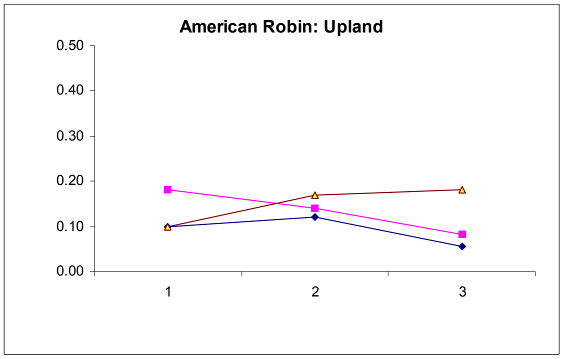

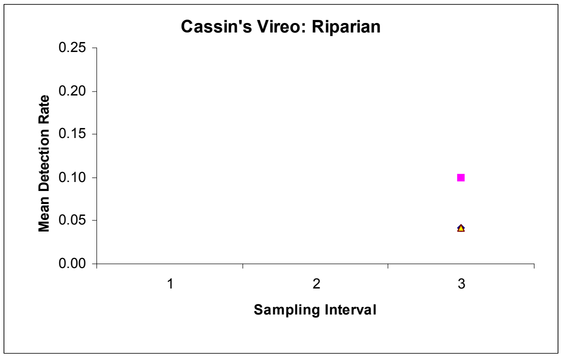

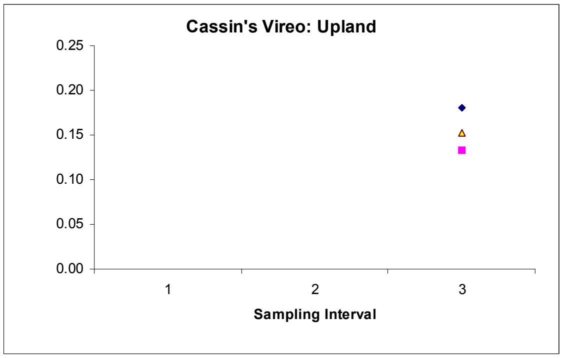

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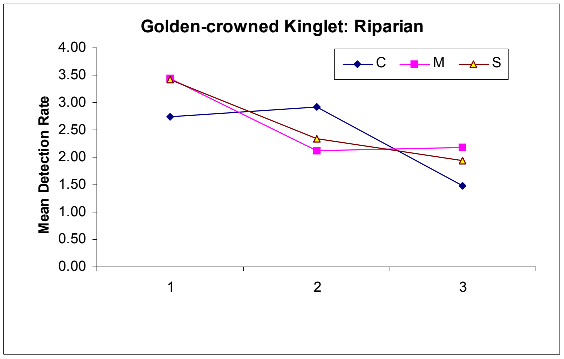

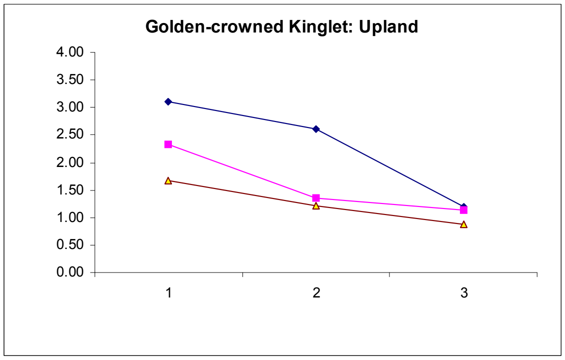

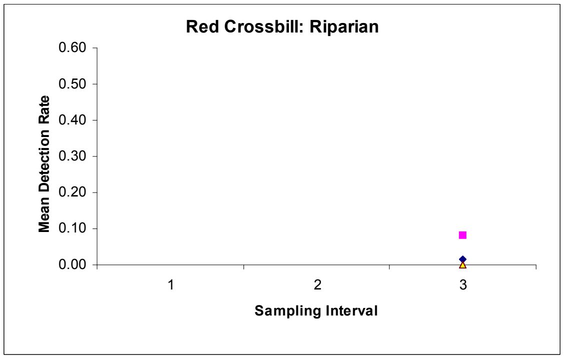

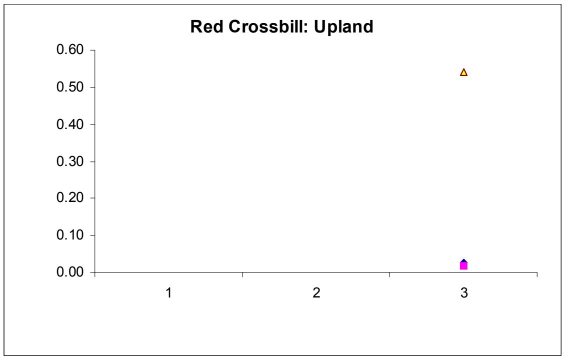

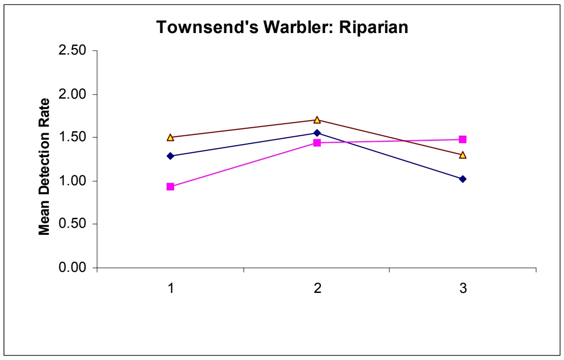

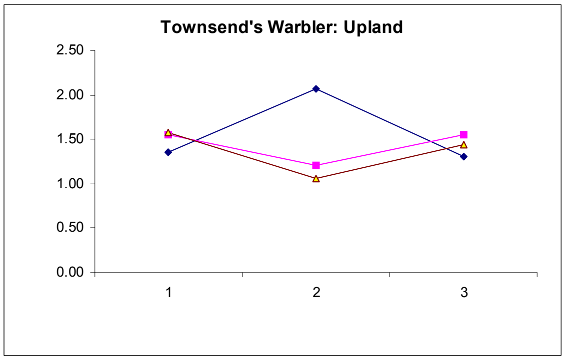

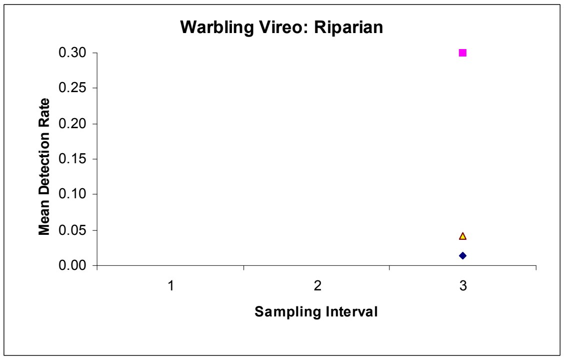

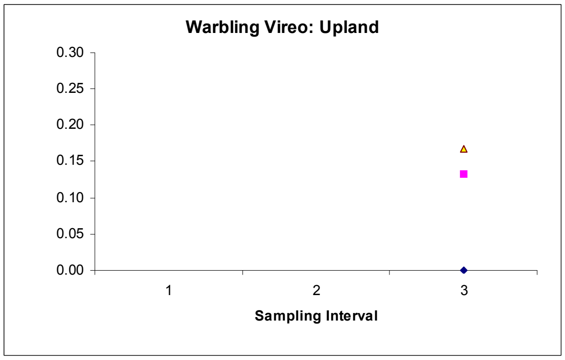

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Chapter 8 EASTERN WASHINGTON SMALL MAMMALS

James D. Fenneman and Virgil C. Hawkes

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Abstract

This study was designed to evaluate the small-mammal species richness and diversity in riparian zones by comparing two different buffer configurations to unlogged control sites. The buffer configurations were based on Washington's 1988 riparian management zone (RMZ) guidelines and on modifications to those guidelines. Small-mammal response to the retention of RMZs in eastern Washington was highly species-specific, with no global response shown by small mammals as a group. Confounding the interpretation of the results was the fact that the buffer treatments were insufficiently different from each other to allow for any meaningful comparison between the two. As a result, we were unable to draw any conclusions regarding the effectiveness of a narrower, uniform-width buffer compared to a wider, variable-width (modified) buffer in maintaining pre-harvest small-mammal populations on a managed landscape. Based on this, it appears that the state buffer prescription was adequate in maintaining the relative abundance and richness of small mammals in eastern Washington. It was possible, however, to detect some significant changes in populations of several species, specifically the chipmunks (*Tamias* spp.), within the upland habitats that had been selectively logged. A lack of significant changes in the relative abundance of small mammals as a whole within the RMZs suggests that either buffer configuration (state or modified) may be adequate for maintaining small-mammal populations at the levels present in typical second-growth montane coniferous forests of northeastern Washington. However, the mid-seral stands that were used as controls in this study may represent some of the lowest levels of diversity and abundance of species of any stage during the life of the forest.

Introduction

There has been a growing trend in forestry throughout North America toward an ecosystem-based approach to forest management, in which biological diversity has become a major objective for forested landscapes (Sullivan et al. 2000; Sullivan et al. 2001; Carey 2003). Traditional management techniques have degraded forests through the reduction of structural complexity, ecosystem simplification, exotic species introduction, imbalance of biotic communities, reduction of prey biomass, and the production of poorly functioning food webs (Carey 2003; Suzuki and Hayes 2003). However, new and innovative approaches to forest management can promote biocomplexity in these managed landscapes by providing a variety of forest successional stages, tree species, and stand structures in a mosaic of habitats across a landscape, as well as through the retention of legacies of individual trees, dead trees, and coarse woody debris (Sullivan et al. 2001; Carey 2003). These strategies attempt to re-create the ecological and structural characteristics of late-seral forests, such as crown-class differentiation, canopy stratification, decadence, understory development, and development of habitat breadth, which are likely more important to maintaining higher levels of biological diversity than is the age of the stand (Sullivan et al. 2001; Ransome et al. 2004). This would allow for the formerly competing concepts of forest harvest and conservation of biological diversity to exist concurrently on the same landscape.

Small mammals, including the shrews, moles, and small rodents, are an abundant yet often overlooked component of forested ecosystems in North America. Their small size and secretive habits have rendered many small-mammal species difficult to study in the wild, a fact compounded by the difficulties associated with species identification in several groups, such as the shrews and *Peromyscus* mice (Nagorsen 1996; Nagorsen 2005). Their inconspicuous nature, though, belies their abundance in many landscapes as well as their ecological importance within the environments they inhabit. Small mammals fulfill a broad range of roles within the ecosystem, including providing a prey source for a variety of predators, dispersing seeds and mycorrhizal spores, altering vegetation through herbivory and seed predation, preying on insects and other small invertebrates (as well as occasionally on other small mammals), aerating and fertilizing soils, and increasing the decomposition rates of organic matter (Sieg 1987; Pearson 2000; Laudenslayer and Fargo 2002; Marcks 2004). The health of the small-mammal fauna of forested ecosystems can provide a measure of the integrity of ecosystem function within temperate coniferous forests and can serve as indicators of change in forest structure (Sullivan et al. 2001).

Small mammals are widespread and often abundant in both riparian and upland forested habitats in northeastern Washington. Riparian habitats in this region contain high levels of floral and faunal diversity relative to other habitats and encompass a wide range of chemical, biological, and physical processes and features that differ from those represented within the larger forest matrix (Young 2000). These riparian forests are critically important to the ecological characteristics of forest ecosystems in managed landscapes because they provide unique microclimates, provide connectivity within increasingly fragmented landscapes, and serve as chemical, biological, and physical boundaries between streams and rivers and the terrestrial areas of the watersheds they drain (Young 2000). Until recently, relatively little was known about small-mammal use of riparian habitats in forested montane environments (Anthony et al. 1987; Doyle 1990) of the Pacific Northwest. Much of the earlier research on small-mammal use of riparian habitats in North America had focused on arid regions and floodplain ecosystems (e.g., Rosenzweig and Winakur 1969; Batzli 1977), where the distinction between the riparian and upland is much more pronounced than it is in montane regions (Doyle 1990). However, within the past 20 years, several studies have addressed these issues within the context of the Pacific Northwest (e.g., Anthony et al. 1987; Doyle 1990; Cockle and Richardson

2003). These studies have documented noticeable differences in species richness and diversity, adult mass, and reproductive rate between animals inhabiting riparian and upland habitats in managed forests of this region. For example, the species composition of the riparian small-mammal fauna in the Cascades of Oregon has been shown to be more diverse than that of adjacent upland habitats (Doyle 1990). This increased diversity is due to the presence of both riparian-obligate species (e.g., *Sorex palustris*) and species that are widespread across both the riparian and upland zones (e.g., *Peromyscus maniculatus*; Doyle 1990; Hanley and Barnard 1998). Attributes such as foliage height diversity (shrubs, herbs), vegetation density, and soil structure, all of which vary between the riparian and upland zones, significantly influence the relative distribution of small-mammal species across these habitats (Rosenzweig and Winakur 1969; Dueser and Shugart 1978; Wilson and Carey 2000). The riparian zone thus provides critical habitat conditions that are not replicated in the upland zone, allowing for a unique and diverse assemblage of small-mammal species to exist in the forested landscapes of northeastern Washington.

Coniferous forests of the Pacific Northwest, including those in northeastern Washington, are heavily managed for timber production, creating a landscape of forest stands that vary widely in age, structure, and composition (O’Connell et al. 2000). Although studies of small-mammal diversity in riparian habitats of managed forests have shown these habitats to be important to small-mammal populations (e.g., Doyle 1990; Anthony et al. 1987), the extent of the impacts of current management practices on the small-mammal fauna of both riparian and upland ecosystems in these landscapes has only recently been investigated (e.g., Sullivan et al. 2000; Sullivan et al. 2001; Cockle and Richardson 2003; Ransome et al. 2004; Sullivan et al. 2005). In particular, the use of riparian management zones (RMZs) as a means to conserve ecological integrity for terrestrial environments in heavily managed landscapes is receiving increased interest from researchers (e.g., Raphael 2002; Vesely and McComb 2002; Cockle and Richardson 2003). The retention of RMZs along waterways was originally utilized as a means to preserve the characteristics of the aquatic environment, particularly as it relates to fish production; however, RMZs have subsequently expanded to include preservation of terrestrial components of the environment (Doyle 1990; Cockle and Richardson 2003; Richardson 2004). These assumptions have not been thoroughly investigated, and some researchers have challenged their effectiveness and suggested that current riparian management techniques may not do a good job of meeting the needs of the terrestrial environment (e.g., Cockle and Richardson 2003; Richardson 2004). Riparian buffers may indeed benefit small-mammal populations by functioning as a travel corridor between forest stands, as a population source for regenerating stands following upland logging, and as a means to preserve the unique habitat characteristics of the riparian zone itself (Doyle 1990; O’Connell et al. 2000). Several other characteristics of riparian buffers, though, such as their linear shape and subsequent lack of forest-interior conditions, have led to doubt regarding their true effectiveness as a means to preserve pre-harvest small-mammal populations in heavily managed forests (Cockle and Richardson 2003). Since traditional RMZs are applied according to broad-based prescriptions and not the specifics of the individual sites, aspects such as width and configuration of the buffer may not provide adequate protection for terrestrial ecosystems in all situations (Richardson 2004).

In 1990, the Wildlife Steering Committee of the Cooperative Monitoring, Evaluation, and Research Committee (CMER) of the Timber, Fish, and Wildlife (TFW) Agreement initiated a study to evaluate the efficacy of different RMZ configurations in providing habitat for wildlife. The study was designed to evaluate the abundance and diversity of wildlife in RMZs by comparing populations of select vertebrate groups between two buffer configurations that differed in width and structure. Furthermore, these two buffer types were contrasted against unlogged control sites to determine the relative magnitude of changes in vertebrate populations in each buffer type compared to conditions in mature, unlogged landscapes. The buffer configurations were based on the RMZ guidelines mandated at the time (Washington State Forest Practices Board 1988) and on modifications to the 1988 guidelines. The wildlife groups selected for study were terrestrial and aquatic amphibians, riparian-associated songbirds, small mammals (rodents and insectivores), and bats. A precursor to this study (O'Connell et al. 2000) compared the diversity and abundance of these wildlife groups before and immediately after logging to identify the short-term temporal effects of logging both within and between treatments. In 2001, CMER initiated a follow-up study to evaluate the longer-term effects of RMZs on wildlife. This study is the only one in Washington to evaluate the longer-term post-treatment effects of riparian management strategies on wildlife based on data collected at the same sites.

Methods

The methods used in this study were carried forward from a study by O'Connell et al. (2000). A total of 17 previously established sites used in that study were resampled during this study (Figure 8-1). One of the sites sampled by O'Connell et al. (Calispell) could not be relocated on the ground and was excluded. The intent of this study was to evaluate the longer-term post-treatment efficacy of RMZs in providing habitat for small mammals by comparing data collected 10 years after logging with both the pre-harvest and 2-year post-harvest data. However, pre-treatment and 2-year post-treatment data were not available; therefore, our analyses are limited to the 10-year post-harvest data set.

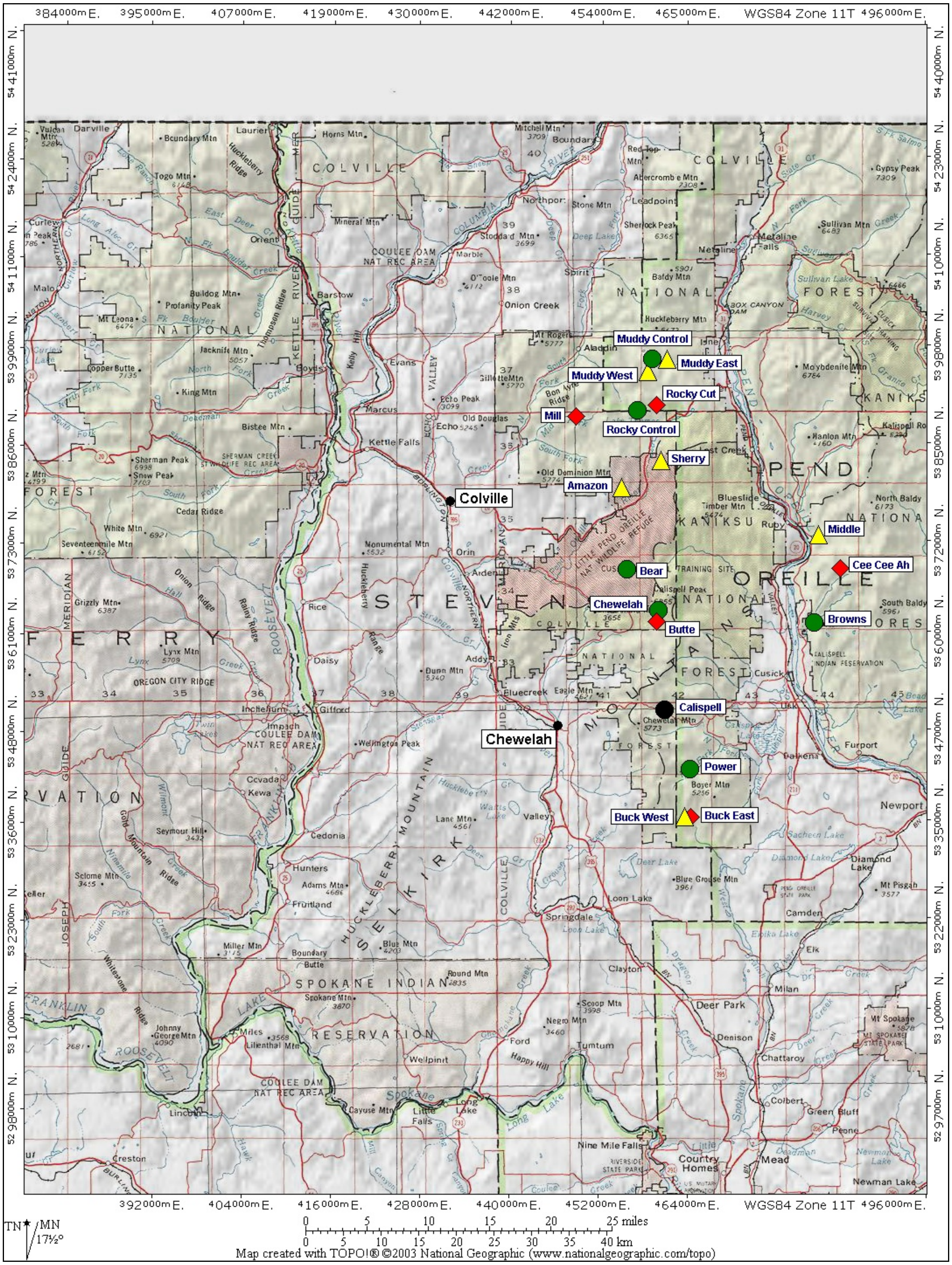
All sites were located in Stevens and Pend Oreille counties in the Okanagan Highlands physiographic province of northeastern Washington (Figure 8-1). Site selection in the original study (O'Connell et al. 2000) was based on five environmental criteria that minimized the variation between sites. The criteria used for site selection included:

1. Managed forests of harvestable age;
2. ≥800 m reach of Type 3 or permanent Type 4 streams, as delineated by Washington's 1988 Forest Practices Rules and Regulations (Table 8-1);
3. 16.2 ha previously logged stands on either side of the stream;
4. Mixed-coniferous forests; and
5. 600–1200 m elevation.

The selected sites were typical of mixed-coniferous forests in the Selkirk Mountains of northeastern Washington (O'Connell et al. 2000). The dominant tree species present at the sites were Douglas-fir (*Pseudotsuga menziesii* var. *glauca*), lodgepole pine (*Pinus contorta* var. *latifolia*), western redcedar (*Thuja plicata*), western hemlock (*Tsuga heterophylla*), western larch (*Larix occidentalis*), and grand fir (*Abies grandis*). Understory vegetation included gooseberry (*Ribes* spp.), devil' -club (*Oplopanax horridus*), Oregon grape (*Mahonia* spp.), falsebox (*Paxistima myrsinites*), red-osier dogwood (*Cornus stolonifera*), mallow ninebark (*Physcarpus malvaceus*), spirea (*Spiraea* spp.), Saskatoon (*Amelanchier alnifolia*), roses (*Rosa* spp.), and huckleberry (*Vaccinium* spp.). Despite attempts at consistency among sites, some random environmental events such as wildfire and extensive blowdown altered habitat conditions at some sites relative to others (e.g., Rocky Cut and Muddy East).

Table 8-1. Water-typing criteria (WAC 222-16-030) used in Washington in 1988. Source: Washington State Forest Practices Board (1988).

Parameter	Water Type				
	1	2	3	4	5
Channel Width	N/A	≥ 6 m between Ordinary High Water Mark (OHWM)	**Anadromous:** > 1.5m between OHWM **Resident Game:** > 3 m between OHWM	> 0.6 m between OHWM	> 0.6 m between OHWM
Gradient	N/A	< 4%	**Anadromous:** < 12% not upstream of a falls > 3m in height **Resident Game:** < 12%	N/A	N/A
Flow	N/A	N/A	**Anadromous:** N/A **Resident Game:** > 0.3 CFS at summer low flow	N/A	N/A
Impoundment	N/A	Water surface area of < 0.4 ha at seasonal low flow	**Anadromous:** Water surface area of < 0.4 ha at seasonal low flow **Resident Game:** Water surface area of < 0.2 ha at seasonal low flow	N/A	N/A
Fisheries	N/A	Used by substantial numbers of anadromous or resident game fish for spawning and rearing and migration	Used by significant numbers of anadromous or resident game fish for spawning and rearing and migration	Not used by significant numbers of fish	Not used by significant numbers of fish
Diversion	N/A	Domestic use for > 100 residences or campsites. Includes upstream reach of 450 m or until the drainage area is < 50%, whichever is less	Domestic use for > 10 residences or campsites. Includes upstream reach of 450 m or until the drainage area is < 50%, whichever is less	N/A	N/A
Other	All water within their OHWM inventoried as “Shoreline of the State” excluding related wetlands	Streams flowing through campgrounds available to public having ≥ 30 campsites	Contributes > 20% of flow to Type 1 or 2 water. Anadromous fish impoundments have outlet to stream with anadromous fish	N/A	All natural waters not classified as Type 1, 2, 3, or 4 or seepage areas, ponds and drainways having short run-off periods



Map created with TOPO!® ©2003 National Geographic (www.nationalgeographic.com/topo)

Figure 8-1. Distribution of eastern Washington study sites. ▲ state buffers; ♦ modified buffers; ● control sites; • excluded control site.

Study Design

The study design was based on a split-plot repeated-measures design, with 18 sites assigned to one of three treatments (Table 8-2). Each site was split into two distinct habitat types: (1) terrestrial riparian and (2) upland. Terrestrial riparian habitats occurred within 8 m of the ordinary high-water mark adjacent to a perennial stream and extended to the point where the vegetation visibly shifted to an upland type. Upland habitats extended upland from the zone of transition from riparian to upland vegetation. Upland sampling occurred 100 m upslope from terrestrial riparian habitat.

Table 8-2. Eastern Washington study sites sampled in 2003 and2004. T = Township; R = Range; S = Section.

Site Number	Site Name	Treatment	T	R	S
2	Bear	Control	T34N	R41E	12
3	Browns	Control	T34N	R44E	33
7	Calispell[a]	Control	N/A	N/A	N/A
9	Chewelah	Control	T34N	R42E	29
12	Muddy Control	Control	T37N	R42E	17 & 18
15	Power	Control	T32N	R43E	28 & 33
16	Rocky Control	Control	T27N	R41E	35
1	Amazon	State	T35N	R41E	2
5	Buck West	State	T31N	R42E	22
10	Middle	State	T35N	R44E	28
13	Muddy East	State	T37N	R42E	17
14	Muddy West	State	T37N	R42E	18
18	Sherry	State	T36N	R42E	28
4	Buck East	Modified	T31N	R42E	22 & 23
6	Butte	Modified	T34N	R42E	32
8	Cee Cee Ah	Modified	T34N	R44E	12
11	Mill	Modified	T36N	R41E	5 & 6
17	Rocky Cut	Modified	T37N	R41E	25

[a] Calispell was not sampled in 2003/2004.

The three treatments developed were based on their recent (or lack of) logging history and the RMZ prescription:

1. **Control sites**: (*n* = 7) These were tracts of unlogged second-growth coniferous forest between 65 and 75 years of age.

2. **Modified buffer treatment sites**: (*n* = 5) At these sites, the riparian zone was logged in a modified way in conjunction with the objectives of this study as outlined in O'Connell et al. (2000) ((Table 8-2).

3. **State buffer treatment sites**: (*n* = 6) At these sites, the riparian zone was logged according to RMZ guidelines in the Washington Forest Practices Rules of 1988 (Table 8-3).

The upland habitats at both the modified and state buffer sites were logged with partial retention (40–60%) during 1993–1994. The modified and state buffer sites differed in several ways (Table 8-3). Most notable were the configuration of the RMZ and the retention of wildlife reserve trees around the creek. Wildlife reserve trees are defined as follows:

- **Type 1**: Live trees with defective or deformed sound tops, trunks, and roots.
- **Type 2**: Dead tree with a sound top, trunk, and roots.
- **Type 3**: Live or dead tree with unstable tops or upper portions.

- **Type 4**: Live or dead tree with unstable trunk or roots, with or without bark. This includes "soft" snags as well as live trees with unstable roots caused by root rot or fire. Type 4 reserve trees are the most dangerous.

Table 8-3. Comparison of riparian harvest prescription for state and modified buffers.

Feature	State	Modified
Buffer	9–15 m buffer	30 m zone of consideration
Leave Trees	Leave Type 3 & 4 reserve trees; no protection	Leave Type 1, 2, 3 & 4 reserve trees; buffer one Type 3 or 4 tree per 2 acres by 1.5x the tree height
Seeps	Protect from machinery	Buffer by 9 m no-entry zone extending to the stream
Deciduous Trees	2 large or 3 smaller trees per acre	Leave all live trees
Shrubs	Avoid disturbance	Leave all within 9–15 m of the stream
Coniferous Trees	Leave all <30 cm DBH	Single tree selection
Delineation of Riparian Management Zone (RMZ)	Extend RMZ to maximum width of riparian zone	Extend zone of consideration to 1/3:2/3 riparian:upland. If no specific habitat features present within zone of consideration, follow State RMZ.

Each site had two 750 m–long transects on the same side of the stream, one in the riparian zone approximately 8 m from the edge of the stream and another in the upland zone approximately 100 m away from the riparian transect (Figure 8-2). Each of these two transects contained 18 pitfall traps at 15 m intervals (total of 36 per site) and 72 snap-trap stations at 10 m intervals (total of 144 per site).

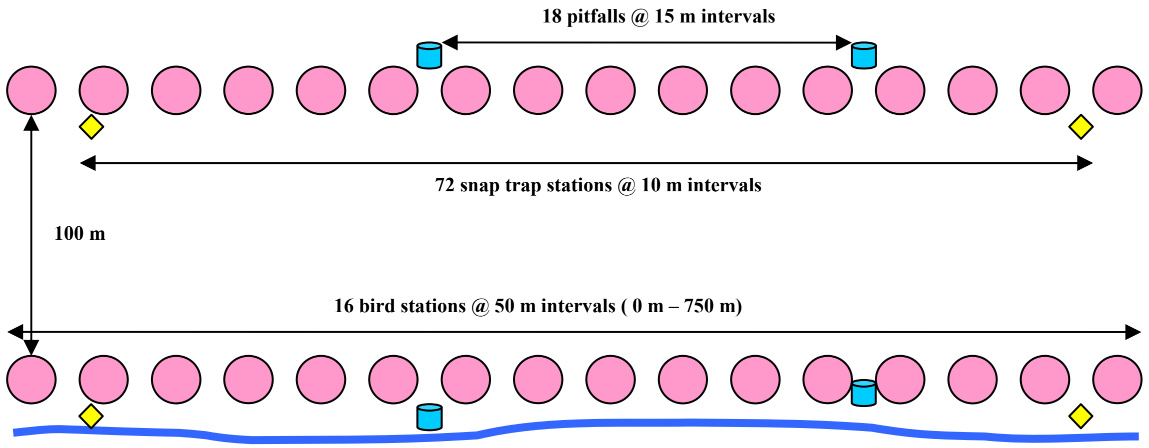


Figure 8-2. Eastern Washington study site layout. Pink circles = songbird point counts; Blue tubes = pitfall traps; Yellow diamonds = snap traps; Blue line = creek.

Pitfall trapping stations, which are more suitable for sampling shrews and other small insectivores, consisted of a single, double-deep #10 can (35 cm deep x 15 cm diameter) buried to a depth such that the open end of the can was flush with the surface of the soil. The ground around the pitfall trap, to a distance of approximately 30 cm, was swept clear of debris that could fall into the trap, and in some cases the ground around the can was built up to provide a smooth transition from the soil into the trap. Approximately 5 cm of water was placed in the bottom of the pitfall traps to facilitate the drowning of small mammal specimens. Drift fences were not utilized in this study. Snap-trap sampling locations, which are more suitable for sampling voles, mice, and other small rodents, were comprised of two snap-traps placed in suitable small-mammal microhabitats within 3 m of the station center and baited with a mixture of oats and peanut butter.

Field Sampling

Field sampling of small mammals occurred between 16 May–4 July 2003 and 16 May–28 June 2004. Sampling protocols were similar to those presented in O'Connell et al. (2000). Pitfall traps at each site were checked every 2–5 days for 2 weeks in 2003 and every second day for 2 weeks in 2004, remaining open throughout these sampling periods. Snap-trap stations at each site were set and checked for 4 consecutive nights during the sampling period. All small-mammal specimens obtained through both methods were weighed, measured (total length, tail length, and hind-foot length, time permitting), identified to species (whenever possible), labelled, and frozen. Labels provided information on species, site number, station number, trap number, date, and observer. All trap-check information was recorded in the field onto data forms. At the end of the day in the field camp, or at the end of the study in the office, all unidentified specimens were checked and identified to species (when possible).

Vegetation Sampling

Structural habitat characteristics were examined by establishing 16 x 20 m plots at 50 m intervals along both the riparian and upland transects. Fifteen plots were established on each transect for a total of 30 transects per site. Each plot was divided into four 8 x 10 m quadrats (Figure 8-3), and between each plot a 30 m point-intercept transect was established to evaluate floristic diversity. Fourteen point-intercept transects were established along each habitat transect (28 per site) (Figure 8-3). All vegetation field sampling (quadrats and point-intercept transects) was conducted during July and August 2004.

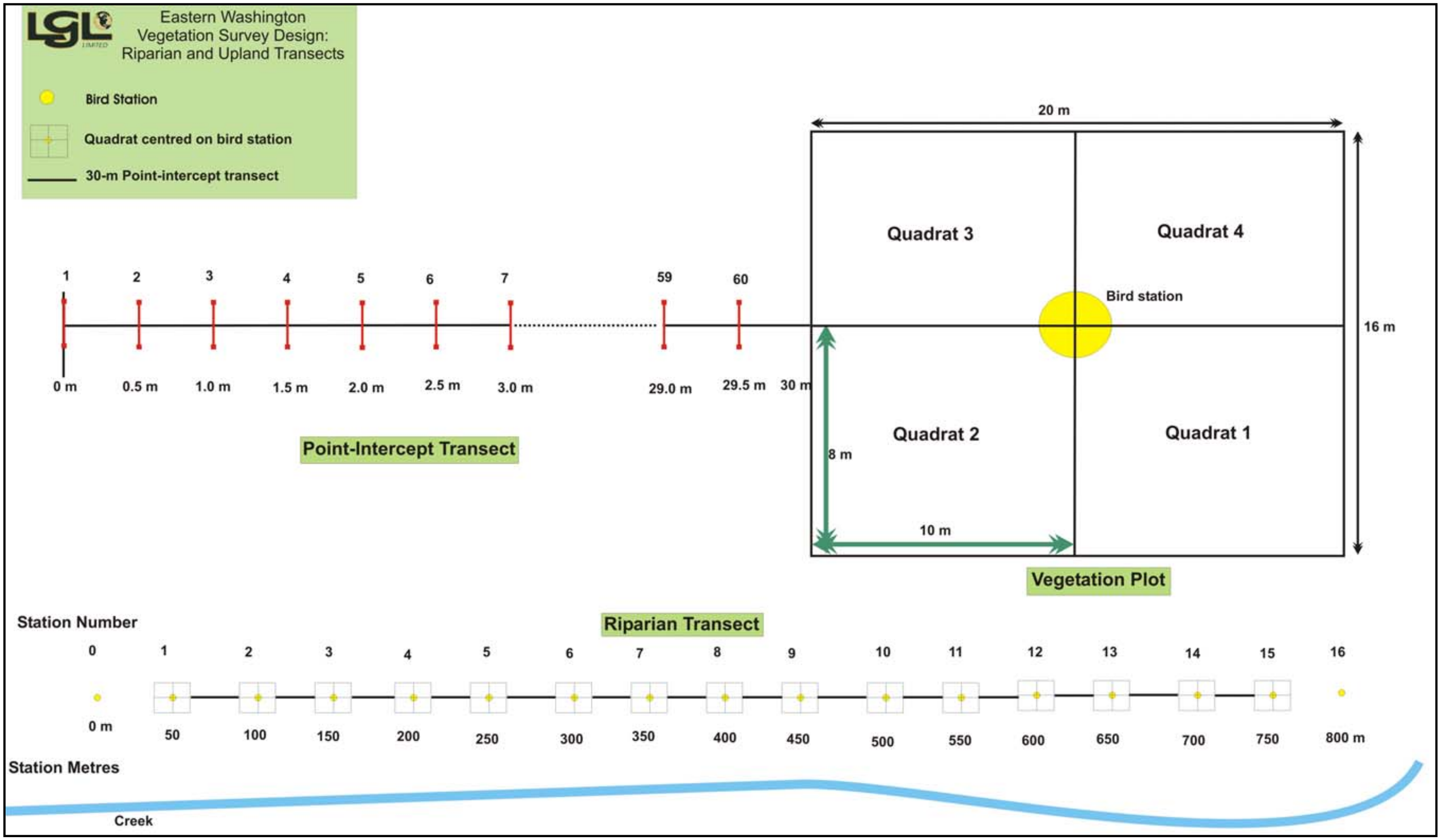


Figure 8-3. Schematic showing the distribution of 16 x 20 m vegetation plots in the riparian transect and the division of one plot into 4 quadrats. Point-intercept transects between each plot are also shown. The configuration was repeated in the upland transect.

In each 16 x 20 m plot, we documented the composition and distribution of shrubs >0.5 m high, down wood and stumps, trees and snags, canopy cover, and tree regeneration. The following sections describe the methods used to collect data on each of these components.

Shrubs

From the center point of each plot, we measured the distance (m) to the nearest shrub (>0.5 m high) in each quadrat, identified the species, and measured the area of the shrub (length x width).

Down Wood and Stumps

In quadrats 1 and 3 (Figure 8-3), we recorded the size and decay classes of woody debris and stumps. Logs were assigned to one of four size classes and decay classes (Table 8-4), and stumps were identified as either natural or cut and were categorized into the same decay classes as logs.

Table 8-4. Size and decay classes assigned to woody debris and stumps in eastern Washington. L = length; D = diameter. Source: after O'Connell et al. (2000).

Size Class	Definition
1	>5 m L; <15 cm D
2	>5 m L; 16–24 cm D
3	>5m L; >25 cm D
4	<5m L; >25 cm D
Decay Class	
1	Freshly fallen tree; bark essentially intact; wood solid; no decomposition
2	Bark beginning to slough or almost completely gone; start of decomposition; sapwood softened; log generally firm
3	Wood softened and breaks into chunks; each chunk still has integrity
4	Essentially no integrity to log; wood decomposed to soil like texture

Trees and Snags

Within each plot (all 4 quadrats), trees were identified to species and assigned to one of four diameter at breast height (DBH) categories: (1) 4–10 cm, (2) 11–25 cm, (3) 26–50 cm, (4) >50 cm. Snags were counted by condition class (condition 1: bark basically intact; condition 2: bark peeling off to absent) and assigned to the same DBH categories as trees.

In each plot, four average-sized trees and two average-sized snags (average size for the plot) were chosen at random and their crown-heights estimated using a clinometer (Figure 8-4).

For example:

- S1 = Sighting 1 = 60%
- S2 = Sighting 2 = -10%
- ∑S1S2 = 70% (sum of absolute values)
- Distance from observer to tree = 30 m
- Tree height = 70% X 30m = 21 m

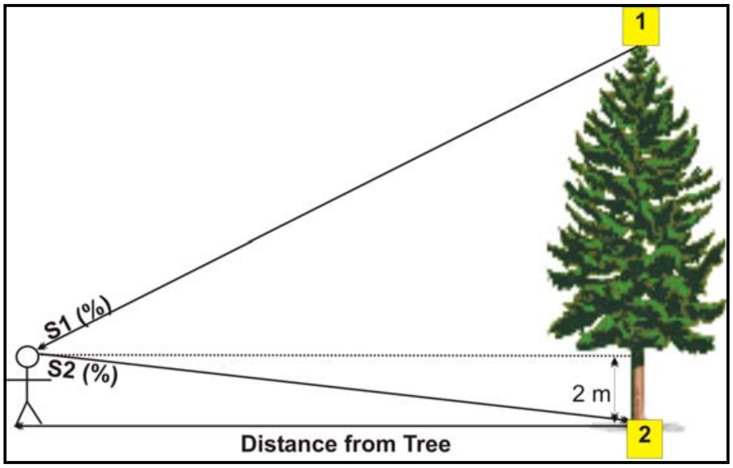


Figure 8-4. Estimating tree-crown height with a clinometer. S1 and S2 are sighting angles (in %).

Canopy Cover

We initially used a convex densiometer to measure overstory and understory canopy cover at the center of each plot and at the center of each quadrat. However, given the large amount of time required to obtain accurate measurements with the densiometer, and the very similar measurements obtained by ocular estimates, we elected to make most canopy-cover measurements by ocular estimation. The densiometer was used periodically to check the accuracy of ocular estimates.

Tree Regeneration

In quadrats 1 and 3 we recorded the number of regenerating coniferous trees (>0.5 m high and <4cm DBH).

Floristics

At 0.5 m increments along each 30 m transect between plots, a point-intercept rod was lowered perpendicular to the transect, and all vegetation, woody debris, and substrate that the rod contacted was recorded by height class (when relevant) (Table 8-5). Herbaceous plants, shrubs, ferns, and trees were recorded to species, while grasses were recorded as present or absent and not identified to species.

Table 8-5. Height classes used on point-intercept transects in eastern Washington.

Height Class	Definition
1	1.5 m
2	1.0 m
3	0.5 m
4	0.25 m
5	0 m

Logs were assigned to one of six size classes (Table 8-6) and one of four decay classes (Table 8-4). Stumps were classified as either natural or cut and assigned to one of the same decay classes as logs. Stumps were differentiated from snags by height; dead trees were considered to be stumps when they were <1.37 m high. Litter depth was measured every 5 m on each floristics transect.

Table 8-6. Size classes used for woody debris encountered on floristics point-intercept transects in eastern Washington. L = length; D = diameter.

Size Class	Definition
1	<5 m L; <15 cm D
2	<5 m L; 16–24 cm D
3	>5 m L; <15 cm D
4	>5 m L; 16–24 cm D
5	>5m L; >25 cm D
6	<5m L; >25 cm D

Data Management

The capture data for both the snap-trap and pitfall trapping methods are expressed as the number of individuals captured per 100 trap nights (catch per unit effort), as presented in Nelson and Clark (1973). The data for the snap-trapping method were corrected to account for all traps sprung during the night. This was done because, once sprung, these traps were removed from the overall trapping effort, decreasing the overall trapping efficiency of the site. These corrections applied both to traps that were successfully sprung by small mammals as well as to those accidentally sprung by such factors as rain, debris, slugs, amphibians, birds, and unsuccessfully by small mammals. No such corrections were necessary for the pitfall-trapping data due to the nature of the trapping method. The catch/effort values of each of the two trapping methods were then summed to obtain overall catch/effort values for use in the following data analyses.

Statistical Analyses

Statistical analyses were performed using SAS V9.1 (© 2002–2003), R V2.2.1 (© 2005), and Microsoft Excel 2002 (© 1985–2003). Prior to performing all analyses, data were assessed to see if they fit a normal distribution using the Analyst Application in SAS, which produces four tests of normality (Shapiro–Wilk, Kolomogerov–Smirnov, Cramer–von Mises, and Anderson–Darling). Comparisons of relative abundance were done using log-transformed means ($\log[x + 1]$), which approximated a normal distribution. Where appropriate, post hoc multiple range tests were used to account for experiment-wise error and thereby reduce Type I errors (Shaffer 1995). When pairwise comparisons of means were made, the Bonferroni method was applied to control the family-wise error (FWE) rate. The critical value of alpha was set at 0.1 and beta at 0.8 (*a priori*) for all analyses, which is appropriate for landscape-based studies and when analyzing data obtained from populations that are highly variable.

Species Richness and Community Similarity

We used the number of species found in each habitat type (i.e., riparian or upland) of each site during each sampling interval as a measure of species richness. Species richness data from riparian transects were used to test for differences relative to buffer width. To test whether logging influenced species richness or diversity, we combined data from the buffer treatment groups (i.e., state and modified riparian buffers), because both groups received similar upland treatments. Pooling the treatments provided a sense of whether or not the retention of riparian habitat, regardless of width, provided for the persistence of terrestrial species richness.

We used Morisita's coefficient of similarity (C) as a measure of species diversity because of its superior utility, relative to other measures, in pairwise comparisons of communities (Krebs 1999).

Morisita's index (Morisita 1959; Horn 1966; Brower et al. 1990) measures community similarity and is based on Simpson's index of dominance (Simpson 1949). Morisita’s index calculates the probability that specimens randomly drawn from two sites will be of the same species, relative to the probability that specimens randomly drawn from the same site will be of the same species. This index is desirable because sample size and diversities of the samples have little influence on its calculation (Morisita 1959; Wolda 1981). Morisita's index returns a value from 0.0 (no similarity) to 1.0 (identical) and was chosen because it is affected little by large variations in sample size.

Relative Abundance

We used the catch per 100 trap nights as a measure of relative abundance for each species, adjusted for missed nights or non-functional traps (Nelson and Clark 1973). Comparisons were made between mean relative abundance measures, transformed as $\log(x + 1)$ to approximate a normal distribution. We tested variation in relative abundance only for the most common species; sample size was too small for analysis of other species.

Using PROC MIXED, we ran repeated-measures ANOVAs to test hypotheses about variation in relative abundance, followed by orthogonal and non-orthogonal contrasts (Table 8-7) of main effects (treatment and transect) and their interactions. The main experimental units for which measurements were repeatedly collected were the individual transects in riparian and upland habitats, and the data from each transect were pooled to derive a treatment mean.

Table 8-7. Orthogonal and non-orthogonal contrasts used to identify differences in relative abundance (RA) within the ten-year post-harvest sampling interval (SI) relative to treatment (control, modified, state) and transect (riparian, upland).

Q	SI, Transect, and Treatment	Question
1	CONTROL - Riparian vs. Upland	Does RA in riparian habitats differ from the upland for the references?
2	MODIFIED - Riparian vs. Upland	Does RA in riparian habitats differ from the upland for the modified treatment?
3	STATE - Riparian vs. Upland	Does RA in riparian habitats differ from the upland for the state treatment?
4	RIPARIAN - Control vs. cut	Does RA of the control riparian differ from the cut riparian?
5	RIPARIAN - Modified vs. State	Does RA of the modified riparian habitats differ from state riparian?
6	RIPARIAN - Control vs. Modified	Does RA of control riparian habitats differ from modified riparian?
7	RIPARIAN - Control vs. State	Does RA of control riparian habitats differ from state riparian?
8	UPLAND - Control vs. cut	Does RA of the control upland habitats differ from the cut?
9	UPLAND - Modified vs. State	Does RA of the modified upland differ from the state upland?

All nine contrasts were used to answer questions about differences in mean relative abundance of the more common species detected. Based on the objective of the overall study, treatments, and transects were grouped to best answer the questions of differences between the state and modified buffer treatments relative to the control sites over time. The comparisons derived were based on the following assumptions:

1. All sites selected for this study were similar enough that results derived for one site could be extrapolated across all sites of a particular treatment; as such, pooling of sites into treatments was acceptable.
2. The upland habitats at the state and modified treatments represented the same treatment and thus could be pooled into a post-cut condition, defined by clearcut logging followed by replanting.
3. A comparison of the upland habitats of the control group with the pooled state and modified uplands (i.e., pre-cut vs. post-cut) would address logging effects (i.e., does logging uplands affect the relative abundance of small mammals?).

4. A comparison of control riparian habitats to the pooled state and modified condition would determine if logging affects relative abundance of small mammals irrespective of buffer width.
5. A comparison between the modified and state riparian transects would provide a measure of the effect of a wide buffer versus a narrow buffer and the effect that buffer width has on the relative abundance of small mammals.

Co-inertia Analysis

The program R V2.2.1 (© 2005) was used to investigate habitat relationships between abundant or common species and the habitat attributes measured at each site and treatment. For this study, a co-inertia analyses (COIA) was used and was based on the ade4 package (Chessel et al. 2005). COIA is a multivariate statistical technique that provides for the visualization of the structure of ecological data via ordination. COIA is very general and some existing methods appear as special cases of it (Dray et al. 2003). For example, interbattery analysis (Tucker 1958) is mathematically equivalent to a simultaneous principle components analysis (PCA) on two ecological tables followed by a COIA (i.e., PCA–PCA COIA). When table X contains qualitative variables and Y contains species numbers, it is usual to cross tables X and Y to obtain a matrix containing the distribution of species among the categories of environmental variables. A simple canonical analysis (CA) of this new table allows one to ordinate the species and the environmental classes (analysis of ecological profiles; Montana and Greig-Smith 1990). Binary discriminant analysis (Strahler 1978), which has been used in ecology (Del Moral 1982; Huang and Del Moral 1988), is mathematically equivalent to Romane’s CA (Dray et al. 2003). Although this approach allows plotting of species and environmental classes (Ben-Shahar 1987; Ben-Shahar and Skinner 1988), no information about ordination of sites is available.

COIA is similar to other ecological ordination techniques, such as a canonical correspondence analysis (CCA; ter Braak 1986) and redundancy analysis (RDA; Wollenberg 1977); however, COIA is a general coupling method that maximizes the covariance between the variables of two tables. Separate tables X and Y can be analyzed by various methods, leading to different coupling methods. This approach aims to find a site score that is a linear combination of environmental variables maximizing the variance of species centroids (i.e., separation of species niches). Unlike CCA, COIA does not have a regression step (Thiolouse et al. 2004). This difference has important implications for ecological data analyses: the regression step implies that in CCA and RDA, the number of samples must be high compared to the number of environmental variables; COIA does not have this constraint. CCA also has an additional constraint (the total variance must be equal to 1) and must be avoided in the case of numerous environmental variables, which is common in biological studies and is the case in this study.

COIA is a paired ecological table analysis that relies on the covariance matrix produced from (in this case) a simultaneous PCA of the environmental and species data (Figure 8-5). Each table is related through a common number of rows, which in this case represents the numbers of sites. The output of the PCA can be visualized as two different representations of the sites in space: one relating to the relative abundance of species; the other to the habitat attributes. The covariance matrices produced by the PCA are then joined via COIA so that the relationship between species relative abundance and habitat data can be viewed. The resulting biplot allows for visualization of how the two initially unrelated data sets covary at the same sites. From the biplot, a description of the habitat variables can be derived for species of interest. The ability to quickly assign habitat relationships based on the relative position of species to habitat variables provides an effective way to describe the habitat relationships for each species in each sampling interval and for each treatment.

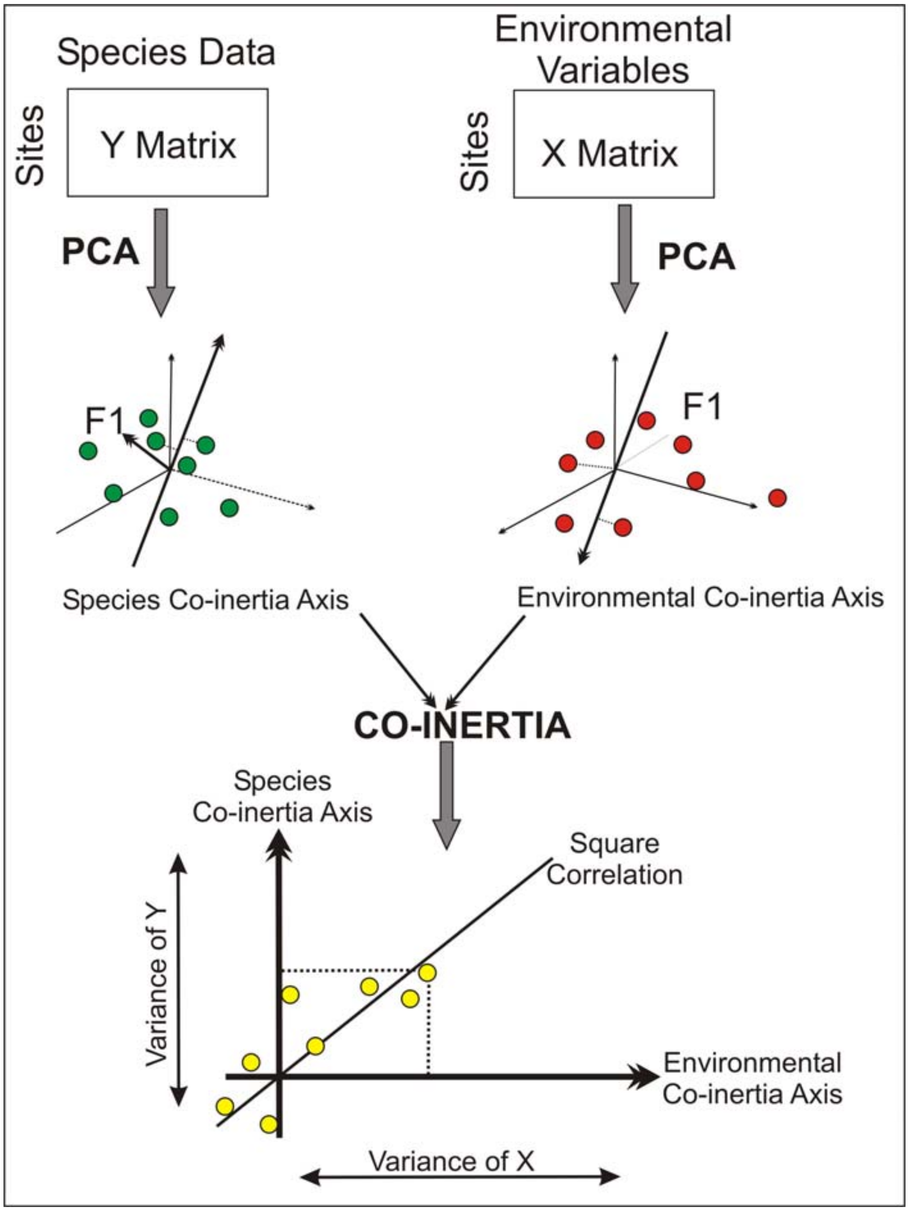


Figure 8-5. Schematic showing the principles of co-inertia analysis. The two ecological data tables X and Y produce two representations of the sites in two hyperspaces. Separate analyses find axes maximizing inertia (=variation) in space (F1 [first factorial axis]). Co-inertia analysis maximizes the square covariance between the projections of the sites on the co-inertia axes. Source: modified from Dray et al. (2003).

The output of a COIA can be viewed on a biplot. The biplots have vertical and horizontal axes, with their intersection being the origin. An environmental gradient can be described for each axis, with the horizontal axis (axis 1) representing the primary environmental gradient and the vertical axis (axis 2) describing a secondary gradient. Species that occur nearer the origin are indicative of (a) species with too few detections to infer habitat associations, (b) habitat generalists, or (c) species that respond to intermediate conditions along the gradients described by the axes. Species that plot farther away from the origin can be regarded as correlating with a particular habitat variable or suite of variables. The length of the vector from the origin to the location of the habitat variable in space is indicative of the strength of the correlation between habitat variable and the gradient described for each axis. Unlike the more commonly used multivariate analyses, such as CCA (ter Braak 1986), for which the number of environmental variables is limited to the number of sites minus one, COIA has no limitations on the number of environmental variables that can be used regardless of the number of sites sampled (Thioulouse et al. 2004). COIA has been evaluated extensively (e.g., Dray et al. 2003), and although it is not used frequently in ecological studies compared to other multivariate methods, it is a rigorous method producing easily interpretable results. The visual comparison of the relationship between a species and the environmental variables should be accompanied by a meaningful biological interpretation, which requires knowledge of the species habits and site characteristics. Caution, however, needs to be exercised when assessing species–habitat relationships, because sample size can influence the analysis.

Results

Effort

Similar levels of trapping effort were applied during 2003 and 2004. We sampled for a total of 49,726 trap nights, of which 25,303 trap nights were within the riparian zone and 24,423 trap nights were within upland habitats (Table 8-8). However, the trapping effort between the two trapping methods was more variable; only 15,552 trap nights were conducted with pitfall traps compared to 34,174 trap nights for the snap traps (

Table 8-9). These values were lower than the expected values for the number of trap nights per trapping method (17,136 trap nights for pitfall trapping; 39,168 trap nights for snap trapping) because of non-functioning traps that are removed from the overall trapping effort.

The number of trap nights associated with snap traps varied between sites due to the catch/effort calculations that were employed, which removed non-functioning traps from the overall effort; the number of trap nights associated with the pitfall-trapping program was generally similar among sites as well as between transects. There was no expectation that catch per effort would be similar between pitfall traps and snap traps. Snap traps are baited and thus attract selected small mammals, while pitfalls are not baited but are able to capture multiple animals each trap night. Because some species are not well sampled by one or the other trapping method, the imbalance of trapping effort by trap type must be remembered when interpreting results based on combined trap nights. We expect that differential susceptibility to trapping (snap and pitfall trapping) could bias estimates of difference in interspecific abundance, but is less likely to affect intraspecific abundance estimates among treatments. Therefore, results presented in this section emphasize treatment-related differences in abundance.

Table 8-8. Combined number of trap nights for pitfall and snap-trap studies in eastern Washington by site.

	2003		2004	
Site	**Riparian**	**Upland**	**Riparian**	**Upland**
Amazon	817	835	735	716.5
Bear	796	791.5	698.5	707.5
Browns	765.5	772	747.5	734
Buck East	787	784	780	749.5
Buck West	773	782.5	772.5	771.5
Butte	1102.5	552	683.5	720.5
Calispell[a]	0	0	0	0
Cee Cee Ah	738	723.5	730	727
Chewelah	651	655	696	685
Middle	779.5	747	762.5	583.5
Mill	728.5	732.5	715	700.5
Muddy Control	761.5	759.5	728.5	728
Muddy East	736.5	750	674.5	707.5
Muddy West	596.5	685	685	723
Power	760.5	762.5	760	618
Rocky Control	710	708.5	655.5	631
Rocky Cut	742.5	700.5	702.5	660.5
Sherry	808	828.5	723	690
Total	13053.5	12569.5	12249.5	11853.5

[a] This site was not sampled in 2003/2004

Table 8-9. Number of trap nights for pitfall and snap-trap studies in eastern Washington by treatment type and transect.

	2003				2004			
Treatment	**Riparian Pitfall**	**Upland Pitfall**	**Riparian Snap**	**Upland Snap**	**Riparian Pitfall**	**Upland Pitfall**	**Riparian Snap**	**Upland Snap**
Control	1422	1422	3023	3027	1278	1278	3008	2826
State	1530	1530	2981	3098	1296	1296	3057	2896
Modified	1206	1206	2893	2287	1044	1044	2587	2514
Total	4158	4158	8897	8412	3618	3618	8652	8236

Species Richness

During the 2003 and 2004 trapping sessions, 4534 individuals of at least 18 species[15] of small mammals were captured (Table 8-10). Of the 18 species recorded, 15 were detected on both the riparian and upland transects. Four species were detected at all 17 sites (*Sorex cinereus*, *S. vagrans*, *Clethrionomys gapperi*, *Peromyscus maniculatus*), while 3 of the species (*Glaucomys sabrinus*, *Microtus richardsoni*, *Neotoma cinerea*) were detected only once. *G. sabrinus* is primarily an arboreal species and was likely not properly sampled by our methods. Other species that occurred but were not adequately sampled due to the trapping techniques used were *Lepus americanus*, *Thomomys talpoides, Tamiasciurus hudsonicus*, and *Neotoma cinerea*.

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[15] This number is considered a minimum because some genera (*Sorex*, *Tamias*, *Microtus*) are extremely difficult to identify to species and, on a number of occasions, were simply identified to genus. The individuals that were identified only to genus may have contained additional species not recorded elsewhere in the study.

Table 8-10. Total number of small-mammal captures by treatment type and transect (2003 and 2004 pooled).

Species	**Control**		**State**		**Modified**	
	Riparian	**Upland**	**Riparian**	**Upland**	**Riparian**	**Upland**
Trap Nights	8730	8552.5	8863	8820	7709.5	7050.5
Sorex cinereus	182	164	145	148	168	229
Sorex hoyi	2	0	5	4	0	3
Sorex monticolus	32	11	12	12	25	15
Sorex palustris	4	0	0	5	10	1
Sorex vagrans	156	24	126	50	179	43
Sorex spp., unidentified	36	24	25	24	46	49
Lepus americanus	3	0	0	2	0	0
Glaucomys sabrinus	0	0	0	1	0	0
Tamias amoenus	0	10	12	67	7	31
Tamias rubricaudus	4	17	22	40	6	38
Tamias spp., unidentified	9	12	6	37	3	37
Tamiasciurus hudsonicus	0	1	1	0	1	0
Thomomys talpoides	2	7	2	7	4	2
Clethrionomys gapperi	221	122	176	93	197	145
Microtus longicaudus	1	0	1	1	0	1
Microtus pensylvanicus	1	0	1	1	1	0
Microtus richardsoni	0	0	0	0	1	0
Microtus spp., unidentified	3	1	3	11	5	2
Neotoma cinerea	1	0	0	0	0	0
Peromyscus maniculatus	233	155	191	150	134	107
Zapus princeps	42	5	63	51	50	18
Total	932	553	791	745	837	717

Total species captured per treatment (transects combined) ranged from 8 to 12 species for control sites (0 = 9.83 ± 1.33), 9 to 14 species for state buffer sites (0 = 10.17 ± 0.95), and 9 to 12 species for modified buffer sites (0 = 10.20 ± 0.58). With both transects combined, the logged sites (i.e., state and modified, pooled) were not found to differ significantly in the number of species captured at the control sites ($t = 2.13$; $p = 0.689$), nor were they found to differ between themselves ($t = 2.26$; $p = 0.978$). However, the mean number of species detected on riparian and upland transects was variable among each of the three treatment types (Figure 8-6). Although the number of species detected on riparian transects did not vary significantly between any of the three treatment types, species richness on upland transects was significantly lower at the control sites than at either of the logged sites ($t = 2.13$; $p = 0.014$). Conversely, species richness on upland transects at the state compared to the modified buffer sites was not significantly different ($t = 2.26$; $p = 0.421$), although the mean number of species recorded on the upland transects at state buffer sites ($n = 15$) was slightly higher than at modified buffer sites ($n = 13$).

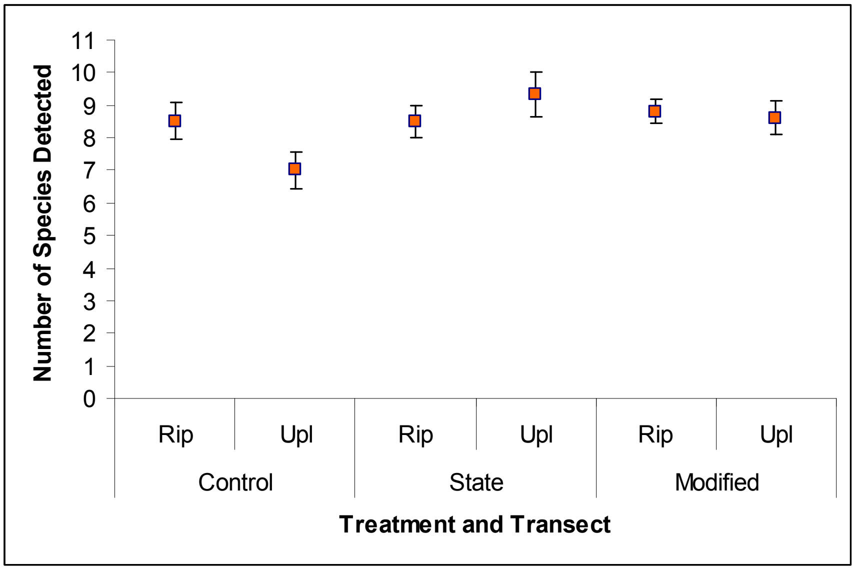

Figure 8-6. Number of species detected (mean ±SE) for riparian and upland transects in each of the three treatment types. Species identified only to genus (i.e., *Sorex*, *Microtus*, *Tamias*) are included here only if no other individuals from that genus were documented for the combined treatment and transect. Rip = Riparian; Upl = Upland.

Community and Diversity Relationships

Contrasting community similarity (Morisita's *C*) between treatments in the 2003 and 2004 trapping sessions reveals that all communities, both within and between treatments (control, state, modified), were extremely similar (Table 8-11). For example, during both sampling years, extremely high levels of community similarity were detected between the control sites and the logged sites on both the riparian (C = 0.985 in 2003; C = 0.992 in 2004) and upland (C = 0.926 in 2003; C = 0.963 in 2004) transects. The highest levels of community dissimilarity occurred on upland transects at state buffer sites between 2003 and 2004 (C = 0.802), with upland transects of the pooled logged sites (C = 0.876) and the control sites (C = 0.875) also showing higher levels of dissimilarity between sampling years. In general, riparian transects maintained higher levels of community similarity within and between treatments, as well as between sampling years, than upland transects.

Table 8-11. Morisita's index of similarity (*C*) for small-mammal communities at three treatments in eastern Washington.

	Comparison	Riparian	Upland
Treatment	Control 2003 vs. 2004	0.827	0.875
	State 2003 vs. 2004	0.923	0.802
	Modified 2003 vs. 2004	0.866	0.906
	cut 2003 vs. 2004	0.901	0.876
2003	Control vs. State	0.992	0.862
	Control vs. Modified	0.970	0.936
	State vs. Modified	0.988	0.890
	Control vs. cut	0.985	0.926
2004	Control vs. State	0.975	0.936
	Control vs. Modified	0.969	0.960
	State vs. Modified	0.926	0.953
	Control vs. cut	0.992	0.963

Relative Abundance

The total number of individuals captured between treatment types (transects pooled) was moderately variable and included 1485 captures at the control sites (8.59 per 100 trap nights), 1536 captures at state buffer sites (8.69 per 100 trap nights), and 1554 captures at modified buffer sites (10.51 per 100 trap nights). No significant differences in the number of captures were detected between transects within each treatment (Figure 8-7); however, control sites exhibited a marginally non-significant difference between transects (riparian > upland; $t = 2.23$; $p = 0.059$). This mirrors the pattern detected for species richness at control sites, in which fewer species were recorded on upland transects than riparian transects (Figure 8-6). No significant or notable differences in the relative abundance of small mammals between treatment types were detected between the state and modified buffer sites for either the riparian or upland transects.

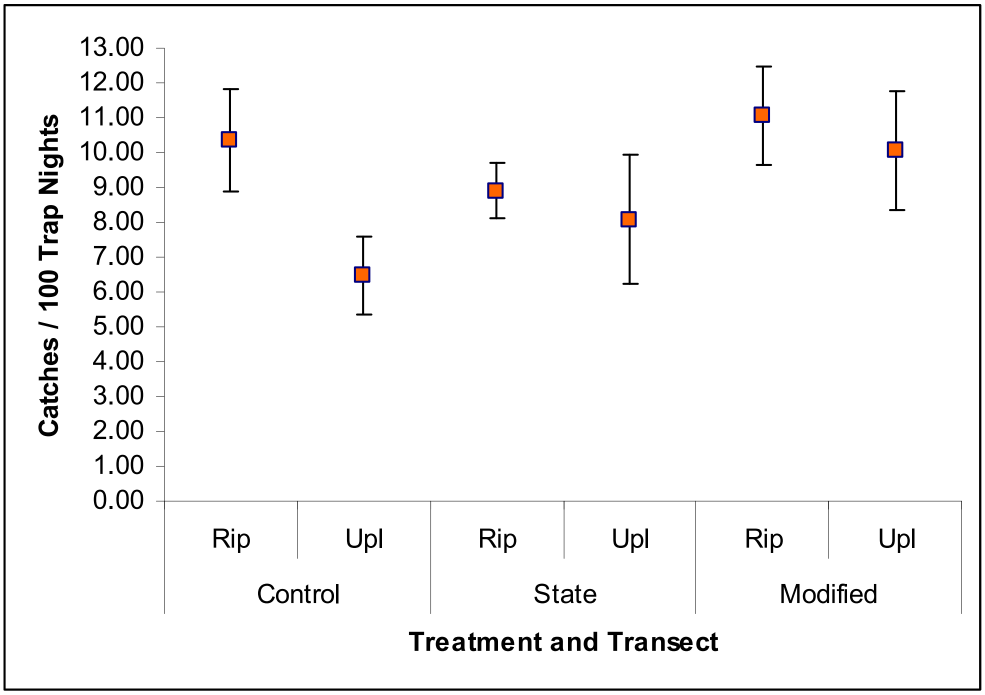

Figure 8-7. Number of small-mammal captures per 100 trap nights (mean ±SE) for riparian and upland transects at each of the three treatment types. Rip = Riparian; Upl = Upland.

Riparian and upland habitats of all three treatment types were dominated by the same 4 species: *Sorex cinereus*, *S. vagrans*, *Clethrionomys gapperi*, and *Peromyscus maniculatus*. These 4 species represented 2108 captures in the riparian zone (82.3% of all captures) and 1430 captures in the upland zone (72.4% of all captures) across all treatments. Other commonly encountered species included *S. monticolus*, *Tamias amoenus*, *T. rubricaudus*, and *Zapus princeps*, which together accounted for an additional 274 captures in the riparian zone (10.7%) and 315 captures in the upland zone (15.9%). The remaining 10 species, as well as all unidentified *Sorex*, *Tamias*, and *Microtus* species, accounted for only 137 captures (5.3%) in the riparian zone and 229 captures (11.6%) in the upland zone.

Certain species were more commonly captured in either the riparian or upland zones, while other species did not appear to be more abundant in one zone over the other. Species that were consistently more abundant in the riparian zone than in the upland zone were *S. vagrans*, *C. gapperi*, *P. maniculatus*, and *Z. princeps* (Figure 8-8). *S. vagrans*, in particular, was found to be significantly more abundant in the riparian zone of all three treatment types, being 6 times more abundant in the riparian zone of control sites ($F_{1,14} = 30.20$; $p < 0.001$), 2.5 times more abundant at state buffer sites ($F_{1,14} = 12.17$; $p = 0.004$), and 4 times more abundant at modified buffer sites ($F_{1,14} = 26.01$; $p < 0.001$). Other species that were significantly more abundant in riparian habitats compared to upland habitats were *C. gapperi*, which was 1.7 times more abundant in riparian habitats at control sites ($F_{1,14} = 11.07$; $p = 0.005$) and 1.9 times more abundant at state buffer sites ($F_{1,14} = 16.83$; $p = 0.001$), and *S. monticolus*, which was 3 times more abundant in riparian habitats of control sites only ($F_{1,14} = 7.73$; $p = 0.015$). No species exhibited a significant difference in the riparian zone between treatments, suggesting that harvesting activities and buffer width had limited or no impact on small-mammal populations in these habitats.

Species that were closely associated with upland habitats included *T. amoenus* and *T. rubricaudus* (as well as *Tamias* spp.; Figure 8-9). When pooled, the combined *Tamias* spp. were found to be significantly more abundant in upland habitats at state ($F_{1,14} = 16.31$; $p = 0.001$) and modified ($F_{1,14} = 14.00$; $p = 0.002$) sites only, suggesting that (partial retention) logging of upland areas was beneficial for this species (Figure 8-10). Between the logged sites, only *C. gapperi* was significantly more abundant in one treatment over another: they were significantly more abundant on upland transects at modified buffer sites than state buffer sites ($F_{1,22} = 6.90$; $p = 0.016$). The species with the most uniform distribution across both the riparian and upland transects was *S. cinereus* (Figure 8-11), which was found to be abundant on both transects at all three treatment types. Of the species with smaller sample sizes (ranging from 1 to 24 captures), *S. palustris* appeared to be more prevalent in the riparian zone ($n = 14$) than in the upland zone ($n = 6$), while *Thomomys talpoides* appeared to be more abundant the upland zone ($n = 16$) than the riparian zone ($n = 8$); however, the sample sizes for these two species were small and were not statistically analyzed. The remaining 8 species either showed no association with either habitat or had sample sizes that were very small.

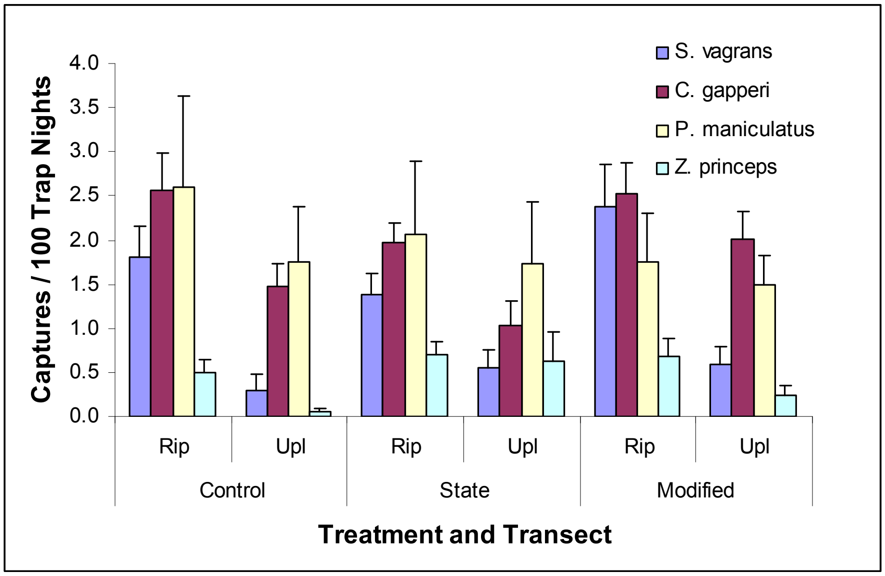

Figure 8-8. Captures per 100 trap nights (mean +SE) of *Sorex vagrans*, *Clethrionomys gapperi*, *Peromyscus maniculatus*, and *Zapus princeps* per transect and treatment type. Comparisons among species are not made. Rip = Riparian; Upl = Upland.

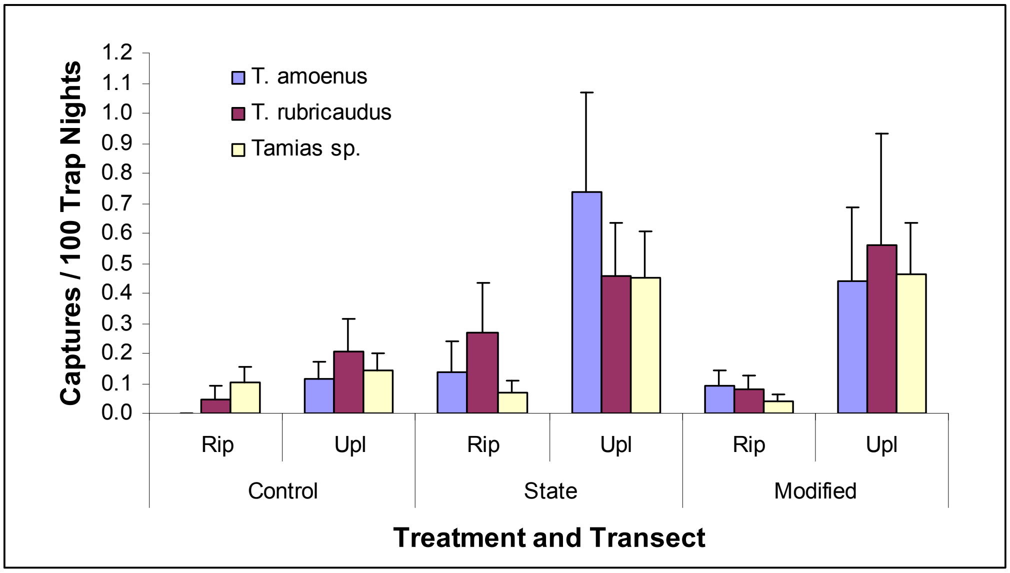

Figure 8-9. Captures per 100 trap nights (mean +SE) of *Tamias amoenus*, *Tamias rubricaudus*, and unidentified *Tamias* spp. per transect and treatment type. Comparisons among species are not made. Rip = Riparian; Upl = Upland.

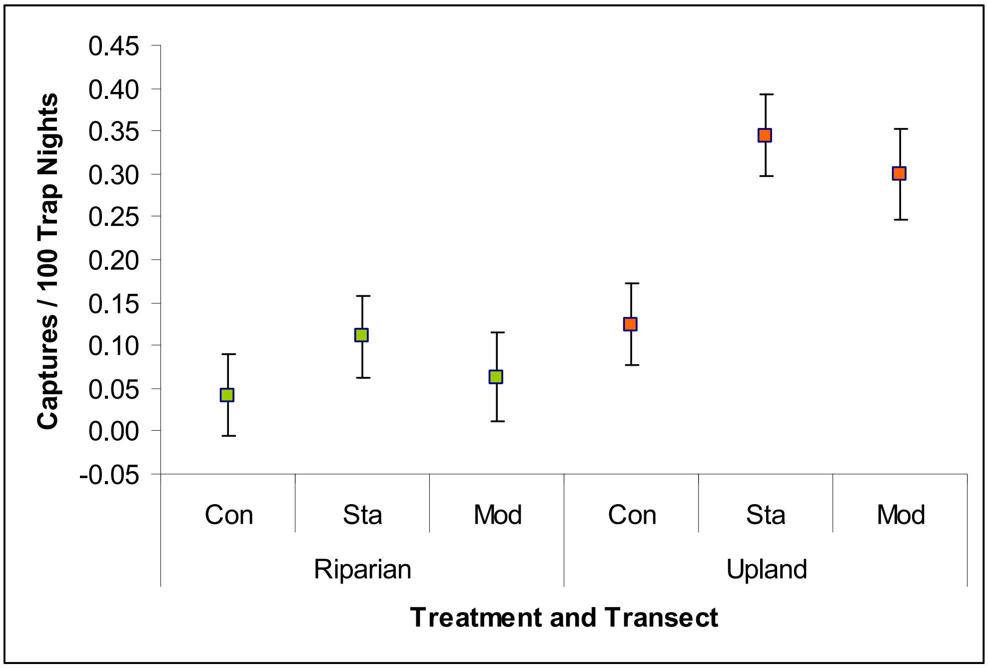

Figure 8-10. Capture rates (mean +SE) of pooled *Tamias* spp. per treatment and transect. Con = Control; Sta = State; Mod = Modified.

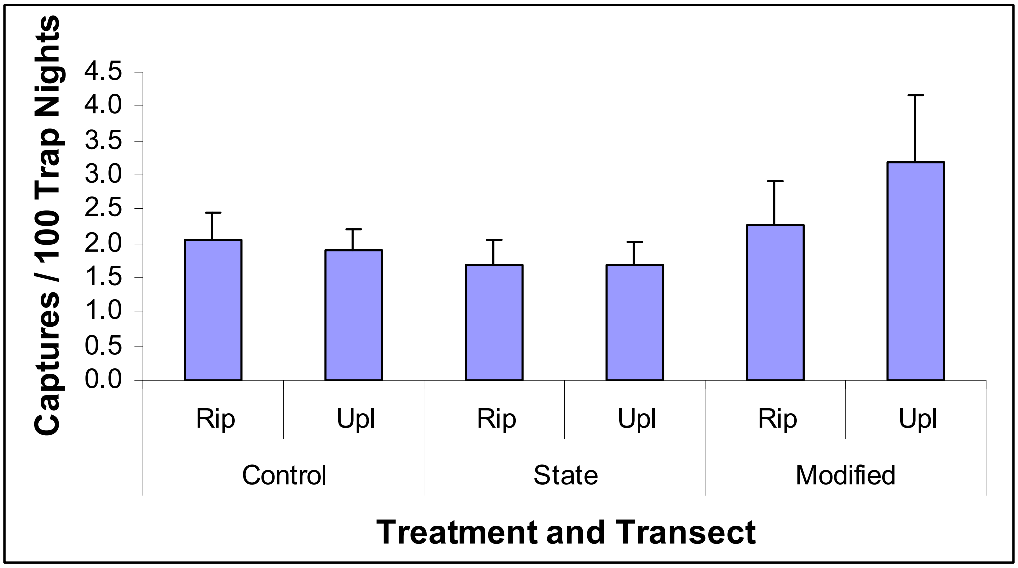

Figure 8-11. Capture rate (mean +SE) of *Sorex cinereus* per transect and treatment type. This species was a common and widespread habitat generalist. Rip = Riparian; Upl = Upland.

Habitat Relationships

Habitat relationships are described only for the 8 most commonly encountered species (*Sorex cinereus*, *S. vagrans*, *S. monticolus*, *Tamias amoenus*, *T. rubricaudus*, *Clethrionomys gapperi*, *Peromyscus maniculatus*, and *Zapus princeps*). Other species were encountered too infrequently to be analyzed effectively.

Riparian Habitat Associations

Variation in habitats and habitat relationships in the riparian zone between the three treatment sites was pronounced (Figure 8-12; Figure 8-13), despite the similarities in relative abundance and species richness that were detected (Figure 8-6; Figure 8-7). For the control sites (Figure 8-12), axis 1 describes 57% of the variation in the data and describes a habitat gradient that tended toward an increase in large pine, Douglas-fir, and larch trees and small pine, Douglas-fir, larch, fir, and spruce trees, as well as increasing small snags and natural stumps. Conversely, this habitat gradient was also characterized by decreasing canopy cover (both in the understory and in overstory) and decreasing large cedar and hemlock trees. Axis 2, which describes a further 30% of the variation in the data, describes a habitat gradient that was characterized by increasing tree height, snag height, and the number of cut stumps and decreasing large deciduous trees and large fir and spruce trees. The habitat conditions described by these two axes suggest a maturing coniferous forest in which increased stratification of the canopy and decreasing canopy cover, along with the resulting increased openness of the habitat, would create greater stand heterogeneity and complexity. Within this maturing forest, *S. cinereus* and *T. rubricaudus* were both positively associated with axis 1 (particularly *T. rubricaudus*), while *Z. princeps* was negatively associated with this axis. This suggests that, under these habitat conditions, the relative abundance of *S. cinereus* and *T. rubricaudus* would be relatively high, while that of *Z. princeps* would be lower. For axis 2, no species were either strongly or moderately associated in a positive direction with this habitat gradient, but both *S. monticolus* and *C. gapperi* were strongly negatively associated with it and would be expected to occur in relatively low numbers under those habitat conditions. Of the other commonly encountered species, *P. maniculatus* showed a weak negative association with axis 1 and a positive association with axis 2, while *S.*

vagrans was weakly associated in a negative direction with both axes. *T. amoenus* was not encountered on this transect at control sites.

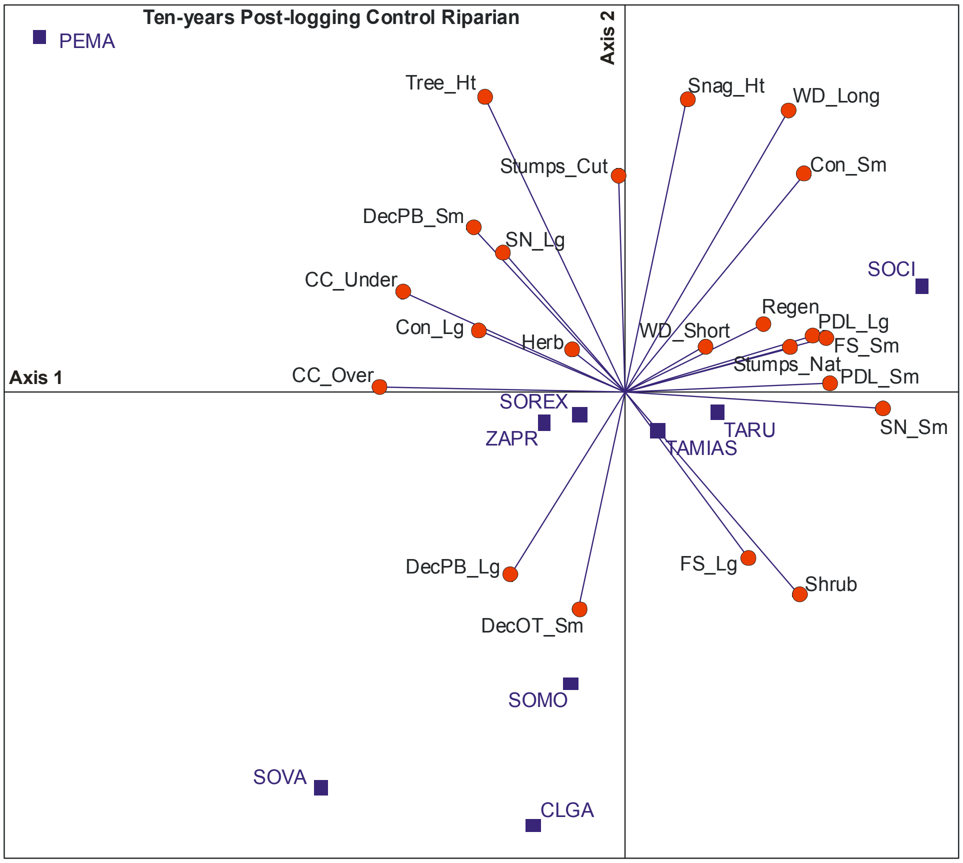

Figure 8-12. Biplot of small-mammal–habitat relationships for riparian habitats of the control treatment. WD_Long = CWD >5m long; WD_Short = CWD <5m long; Stumps_Cut = Cut stumps; Stumps_Nat = Natural stumps; CC_Over = Overstory cover; CC_Under = Understory cover; Regen = Regenerating conifers <0.5m tall and <4 cm DBH; SNB_Sm = Snags with bark and <25 cm DBH; SNB_Lg = Snags with bark and >25 cm DBH; SNNB_Sm = Snags without bark, <25 cm DBH; SNNB_Lg = Snags without bark, >25 cm DBH; Con_Sm = Small-diameter conifers; DecPB_Sm = Small-diameter birch and poplar; DecOT_Sm = Small-diameter deciduous; FS_Sm = Small-diameter fir and spruce; PDL_Sm = Small-diameter pine, Douglas-fir, and larch; Con_Lg = Large-diameter conifers; DecPB_Lg = Large-diameter birch and poplar; DecOT_Lg = Large-diameter deciduous; FS_Lg = Large-diameter fir and spruce; PDL_Lg = Large-diameter pine, Douglas-fir, and larch; Tree_Ht = Average tree height; Snag_Ht = Average snag height; Shr_S = Seed-bearing shrubs; Shr_F = Fruit-bearing shrubs; Shr_U = Non-food shrubs; CWD_Sm = Any piece <5m long; CWD_Lg = Any piece >5m long. CLGA = *Clethrionomys gapperi*; PEMA = *Peromyscus maniculatus*; SOCI = *Sorex cinereus*; SOMO = *Sorex monticolus*; SOREX = *Sorex* spp.; SOVA = *Sorex vagrans*; TAAM = *Tamias amoenus*; TAMIAS = *Tamias* spp.; TARU = *Tamias rubricaudus*; ZAPR = *Zapus princeps*.

At state buffer sites (Figure 8-13), the description of the habitat gradients, as well as the relationship of small mammal species to the axes, differs from that detected at control sites. At state buffer sites, axis 1 (which accounted for 82% of the variation in the data) describes a habitat gradient characterized by increasing herbaceous cover, tree height, cut stumps, and short-length coarse woody debris and decreasing pine, Douglas-fir, larch, cottonwood, and birch trees (both large and small), shrub cover, and natural stumps. However, axis 2 (which describes a further 16% of the variation in the data) refers to a habitat gradient of increasing habitat complexity, with habitats having a better-developed understory and greater tree-species diversity.

Specifically, the habitat gradient was characterized by increasing fir and spruce trees (both large and small), small snags, and canopy cover in the overstory and decreasing large snags, large cedar and hemlock trees, long-length coarse woody debris, and regenerating conifers. Riparian habitats at state buffer sites tended toward simplification and away from complexity, with reduced tree-species diversity, limited shrub and regenerating conifer cover in the open, herbaceous understory, and increased canopy cover in both the overstory and understory. The only species with a strong correlation to axis 1 was *C. gapperi*, which was negatively associated with this axis. *Z. princeps* and *S. vagrans* were both positively associated with axis 2, while *S. monticolus* was strongly associated with this axis in a negative direction. *T. amoenus*, *T. rubricaudus*, and *S. cinereus* were weakly associated in a negative direction with both axes, and *P. maniculatus* was weakly positively associated with axis 1 and weakly negatively associated with axis 2.

Figure 8-13. Biplot of small-mammal–habitat relationships in riparian habitats of the cut (state and modified) treatments. WD_Long = CWD >5m long; WD_Short = CWD <5m long; Stumps_Cut = Cut stumps; Stumps_Nat = Natural stumps; CC_Over = Overstory cover; CC_Under = Understory cover; Regen = Regenerating conifers <0.5m tall and <4 cm DBH; SNB_Sm = Snags with bark and <25 cm DBH; SNB_Lg = Snags with bark and >25 cm DBH; SNNB_Sm = Snags without bark, <25 cm DBH; SNNB_Lg = Snags without bark, >25 cm DBH; Con_Sm = Small-diameter conifers; DecPB_Sm = Small-diameter birch and poplar; DecOT_Sm = Small-diameter deciduous; FS_Sm = Small-diameter fir and spruce; PDL_Sm = Small-diameter pine, Douglas-fir, and larch; Con_Lg = Large-diameter conifers; DecPB_Lg = Large-diameter birch and poplar; DecOT_Lg = Large-diameter deciduous; FS_Lg = Large-diameter fir and spruce; PDL_Lg = Large-diameter pine, Douglas-fir, and larch; Tree_Ht = Average tree height; Snag_Ht = Average snag height; Shr_S = Seed-bearing shrubs; Shr_F = Fruit-bearing shrubs; Shr_U = Non-food shrubs; CWD_Sm = Any piece <5m long; CWD_Lg = Any piece >5m long. CLGA = *Clethrionomys gapperi*; PEMA = *Peromyscus maniculatus*; SOCI = *Sorex cinereus*; SOMO = *Sorex monticolus*; SOREX = *Sorex* spp.; SOVA = *Sorex vagrans*; TAAM = *Tamias amoenus*; TAMIAS = *Tamias* spp.; TARU = *Tamias rubricaudus*; ZAPR = *Zapus princeps*.

At modified buffer sites (Figure 8-13), axis 1 describes 53% of the variation in the data, while axis 2 describes a further 32% of the variation. Axis 1 describes a habitat gradient characterized by increasing tree height, canopy cover, short-length coarse woody debris, snag height, and large cedar, hemlock, maple, and alder trees and decreasing small fir, spruce, cedar, and hemlock trees, large fir, spruce, cottonwood, and birch trees, and long-length coarse woody debris. Axis 2 describes a habitat gradient characterized by increasing herbaceous cover, large snags, snag height, cut stumps, and large cedar and hemlock trees and decreasing shrubs, regenerating conifers, small alder and maple trees, and understory canopy. The habitat conditions described by these primary axes suggest a riparian habitat that resembles that of the state buffer sites, with increasing canopy cover, decreasing tree species diversity, and an increasingly open understory with high amounts of herbaceous vegetation and limited cover of shrubs and regenerating conifers. *S. vagrans* had negative association with axis 1, although both unidentified *Sorex* and *Tamias* species also showed weak and strong (respectively) positive associations with this axis. For axis 2, *T. amoenus* had a strong positive correlation and *P. maniculatus* had a moderate positive correlation to this axis, while *T. rubricaudus* was strongly negatively associated with the axis. *S. monticolus* was weakly negatively associated with both axes, *S. cinereus* was weakly associated with axis 1 in a negative direction and axis 2 in a positive direction, and both *C. gapperi* and *Z. princeps* were weakly associated with both axes in a positive direction.

Upland Habitat Associations

Habitat associations were more variable between the controls and cut treatments than among riparian zones due to selective harvest at the state and modified buffer sites. At control sites (Figure 8-14), where axis 1 represented 55% of the variation in the data, axis 1 describes a habitat gradient of increasing cut stumps, shrubs, and to a lesser extent, herbaceous cover and small alder and maple trees. Conversely, habitat characteristics such as small cedar and hemlock trees, small snags, canopy cover (overstory and understory), and short-length coarse woody debris decreased along this habitat gradient. In contrast, axis 2, which describes an additional 27% of the variation in the data, indicates a habitat gradient of increasing large snags and large cedar, hemlock, pine, Douglas-fir, and larch trees, with decreasing large fir and spruce trees, small fir, spruce, cottonwood, birch trees, natural stumps, and long-length coarse woody debris. The habitat described by these axes is one of maturing second-growth riparian coniferous forest, with a well-developed shrub layer (i.e., high shrub diversity and regenerating trees), large amounts of herbaceous vegetation in the understory, and higher levels of structural diversity resulting from an increase in the abundance of large snags. Species that were positively associated with axis 1 included *Z. princeps* and *P. maniculatus*; no species showed a particularly strong negative correlation with axis 1. Along axis 2, both *C. gapperi* and *S. monticolus* showed a strong positive association with this habitat gradient, while *T. amoenus* and *T. rubricaudus* both showed strong negative associations. *S. cinereus* was weakly positively correlated with axis 1 and negatively correlated with axis 2, and *S. vagrans* was weakly negatively associated with axis 1 and positively associated with axis 2.

Habitat conditions and species–habitat relationships on upland transects at the logged sites were different from what was found at the control sites (Figure 8-15). For state buffer sites, axis 1 describes 62% of the variation in the data. This axis describes a habitat gradient of increasing tree size, increasing cedar and hemlock, increasing large coarse woody debris, increasing stumps and snags, increasing understory canopy cover, and increasing regenerating conifers. Axis 2, which accounts for a further 25% of the variation in the data, describes a habitat gradient of increasing small trees (fir, spruce, pine, Douglas-fir, larch, cedar, hemlock, cottonwood, birch), increasing coarse woody debris (large and small), and increasing stumps, with decreasing overstory canopy cover, large trees (fir, spruce, pine, Douglas-fir, larch, cedar, hemlock), shrubs, herbaceous cover, large snags, and small deciduous trees (maple, alder). Together, the two axes describe a habitat of abundant small, dense trees (particularly regenerating conifers), with limited development of the shrub or herbaceous components of the understory and a decrease in the extent of the overstory canopy cover. No species was strongly associated with axis 1 in a positive direction, but *Z. princeps* showed a strong negative correlation with this axis. *T. rubricaudus* and *S. cinereus* were weakly positively associated with axis 1, while for axis 2, only *T. amoenus* (positive) and *S. vagrans* (negative) showed strong correlations to that axis. *S. monticolus* showed a moderate negative association with both axes, while *C. gapperi* was moderately positively associated with axis 1 and moderately negatively associated with axis 2.

For modified buffer sites (Figure 8-15), axis 1 represents 57% of the variation in the data, describes a habitat gradient tending toward increasing overstory canopy cover, shrub cover, large pine, Douglas-fir, larch, fir, and spruce trees, and small maple and alder trees, with decreasing regenerating conifers and cut stumps. Conversely, axis 2, which represents an additional 26% of the variation in the data, describes a habitat gradient that tends toward increasing small pine, Douglas-fir, larch, fir, and spruce trees, tree height, herbaceous cover, and short-length coarse woody debris, with decreasing understory canopy cover, large snags, snag height, small cottonwood, birch, cedar, and hemlock trees, large cedar and hemlock trees, and long-length coarse woody debris. These axes represent a habitat with a more diverse, well-developed shrub and herbaceous component in the more open understory, along with higher canopy cover and a greater component of large trees in the overstory than the habitat occurring in the upland habitat at state buffer sites. *C. gapperi* was strongly correlated with axis 1 in a positive direction, and *T. amoenus* was strongly correlated with axis 2 in a negative direction within the upland zone of this treatment. Three species (*T. rubricaudus*, *P.* maniculatus, and *Z. princeps*) showed moderate negative associations with both axes, while 2 species (*S. vagrans, S. monticolus*) exhibited moderate positive associations with both axes. *S. cinereus* was positively correlated with axis 1 and negatively correlated with axis 2, but neither association was particularly strong.

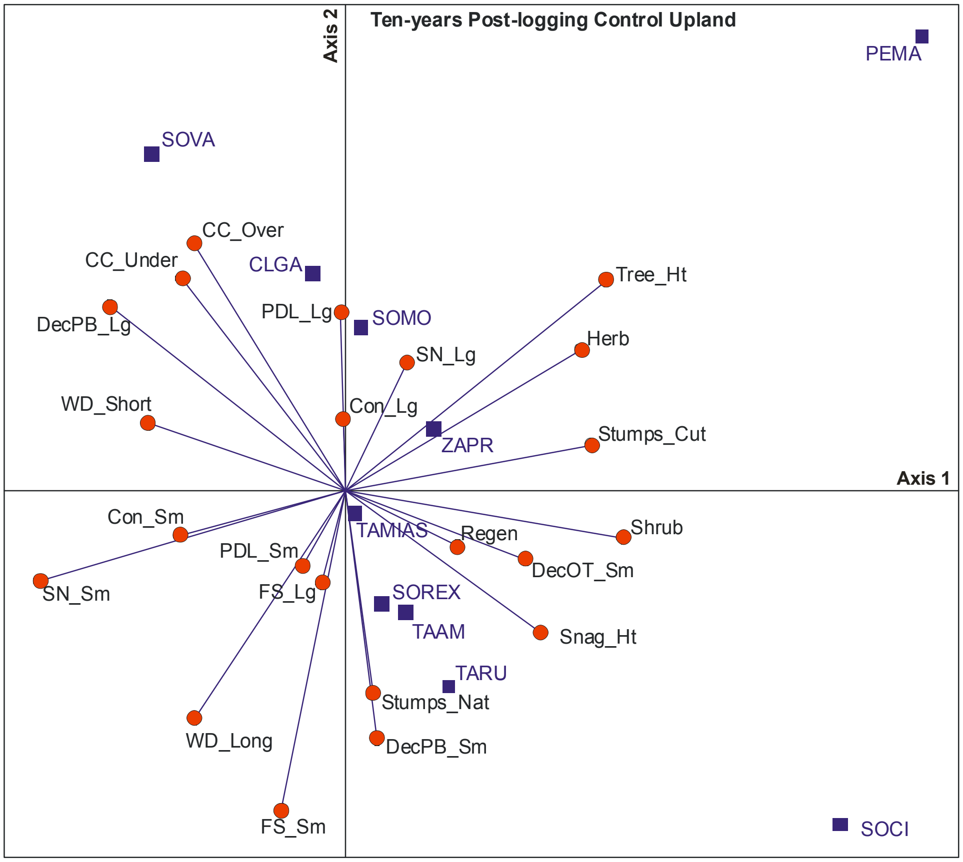

Figure 8-14. Biplot of small-mammal–habitat relationships in upland habitats of the control sites. WD_Long = CWD >5m long; WD_Short = CWD <5m long; Stumps_Cut = Cut stumps; Stumps_Nat = Natural stumps; CC_Over = Overstory cover; CC_Under = Understory cover; Regen = Regenerating conifers <0.5m tall and <4 cm DBH; SNB_Sm = Snags with bark and <25 cm DBH; SNB_Lg = Snags with bark and >25 cm DBH; SNNB_Sm = Snags without bark, <25 cm DBH; SNNB_Lg = Snags without bark, >25 cm DBH; Con_Sm = Small-diameter conifers; DecPB_Sm = Small-diameter birch and poplar; DecOT_Sm = Small-diameter deciduous; FS_Sm = Small-diameter fir and spruce; PDL_Sm = Small-diameter pine, Douglas-fir, and larch; Con_Lg = Large-diameter conifers; DecPB_Lg = Large-diameter birch and poplar; DecOT_Lg = Large-diameter deciduous; FS_Lg = Large-diameter fir and spruce; PDL_Lg = Large-diameter pine, Douglas-fir, and larch; Tree_Ht = Average tree height; Snag_Ht = Average snag height; Shr_S = Seed-bearing shrubs; Shr_F = Fruit-bearing shrubs; Shr_U = Non-food shrubs; CWD_Sm = Any piece <5m long; CWD_Lg = Any piece >5m long. CLGA = *Clethrionomys gapperi*; PEMA = *Peromyscus maniculatus*; SOCI = *Sorex cinereus*; SOMO = *Sorex monticolus*; SOREX = *Sorex* spp.; SOVA = *Sorex vagrans*; TAAM = *Tamias amoenus*; TAMIAS = *Tamias* spp.; TARU = *Tamias rubricaudus*; ZAPR = *Zapus princeps*.

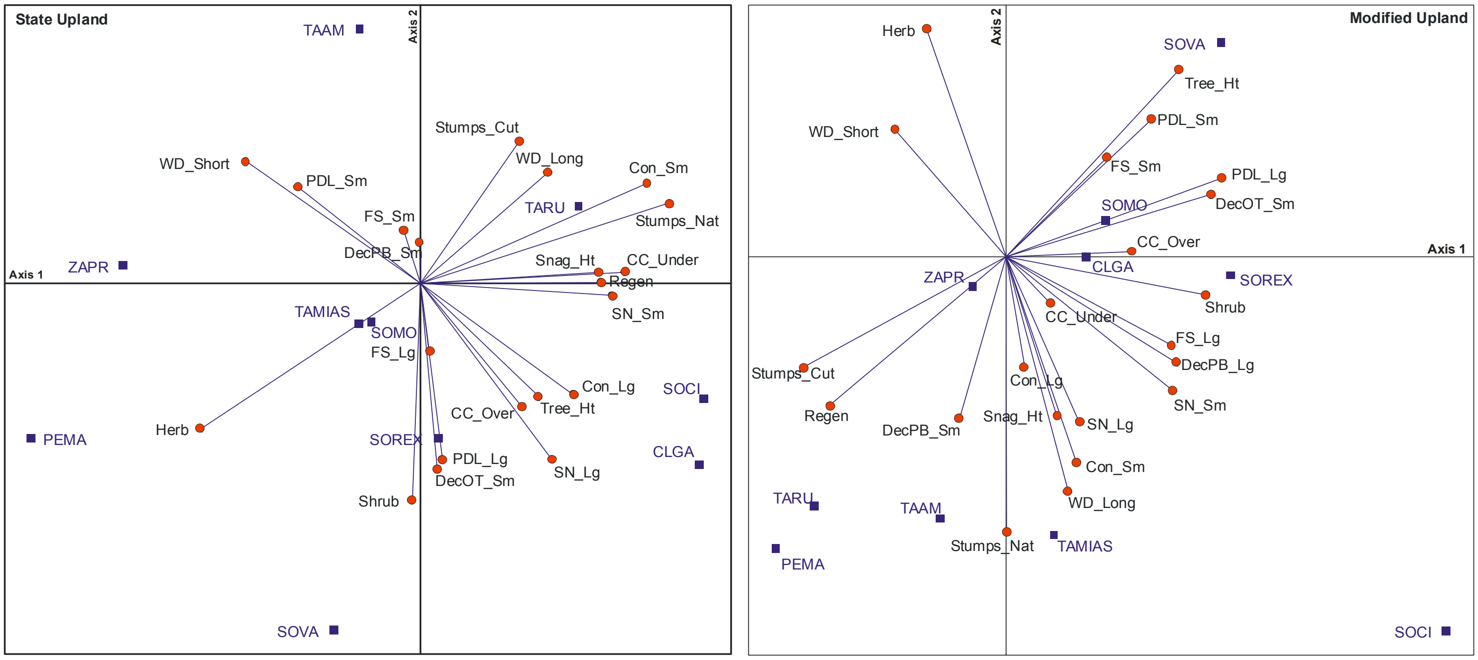

Figure 8-15. Biplot of small-mammal–habitat relationships in upland habitats of the cut (state and modified) sites. WD_Long = CWD >5m long; WD_Short = CWD <5m long; Stumps_Cut = Cut stumps; Stumps_Nat = Natural stumps; CC_Over = Overstory cover; CC_Under = Understory cover; Regen = Regenerating conifers <0.5m tall and <4 cm DBH; SNB_Sm = Snags with bark and <25 cm DBH; SNB_Lg = Snags with bark and >25 cm DBH; SNNB_Sm = Snags without bark, <25 cm DBH; SNNB_Lg = Snags without bark, >25 cm DBH; Con_Sm = Small-diameter conifers; DecPB_Sm = Small-diameter birch and poplar; DecOT_Sm = Small-diameter deciduous; FS_Sm = Small-diameter fir and spruce; PDL_Sm = Small-diameter pine, Douglas-fir, and larch; Con_Lg = Large-diameter conifers; DecPB_Lg = Large-diameter birch and poplar; DecOT_Lg = Large-diameter deciduous; FS_Lg = Large-diameter fir and spruce; PDL_Lg = Large-diameter pine, Douglas-fir, and larch; Tree_Ht = Average tree height; Snag_Ht = Average snag height; Shr_S = Seed-bearing shrubs; Shr_F = Fruit-bearing shrubs; Shr_U = Non-food shrubs; CWD_Sm = Any piece <5m long; CWD_Lg = Any piece >5m long. CLGA = *Clethrionomys gapperi*; PEMA = *Peromyscus maniculatus*; SOCI = *Sorex cinereus*; SOMO = *Sorex monticolus*; SOREX = *Sorex* spp.; SOVA = *Sorex vagrans*; TAAM = *Tamias amoenus*; TAMIAS = *Tamias* spp.; TARU = *Tamias rubricaudus*; ZAPR = *Zapus princeps*.

Treatment Considerations

The differences in species richness, relative abundance, and habitat relationships need to be considered in relation to the different treatments that were applied to the sites. The upland habitats at both the state and modified buffer sites had been selectively logged, with 40–60% retention and the preservation of a narrow, uniform-width (state buffer sites) or broader, variable-width (modified buffer sites) riparian buffer. The control sites, which were represented by stands of mid-seral, mature mixed-coniferous forest, were used as a reference point against which to compare the logged sites. Although comparisons of the diversity and relative abundance of small mammals between the cut treatments indicated minimal differences between the two (with the exception of the relative abundance of *C. gapperi* in upland habitats), when plotted in a co-inertia analysis the relative position of the treatments on a biplot, both for riparian and upland habitats, differs markedly in relation to the habitat attributes that were measured (Figure 8-16). Site effects are likely partially responsible for such differences; however, without knowledge of the pre-treatment condition of the treatment sites, it is impossible to gauge the actual magnitude of any site effects. Additionally, the significant differences in the relative abundance of *Clethrionomys* in upland habitats between the state and modified treatments suggests that certain habitat conditions may also be responsible for the differences that were detected between the logged sites.

Within riparian habitats, the modified buffer treatment is more closely aligned with the control treatment, while the state buffer treatment is widely divergent from both. This may indicate that, within riparian habitats, sites that had received the modified treatment were better able to rebound to pre-treatment conditions than sites that had received a state treatment. Upland habitats of the modified and state treatments are much more closely associated with each other, and both are widely divergent from the control sites. This is to be expected given the similar treatments that were applied to the uplands of the logged sites but not to the controls. However, even for upland habitats, the relative position of the two cut treatments is divergent enough to suggest that pronounced differences exist between the two.

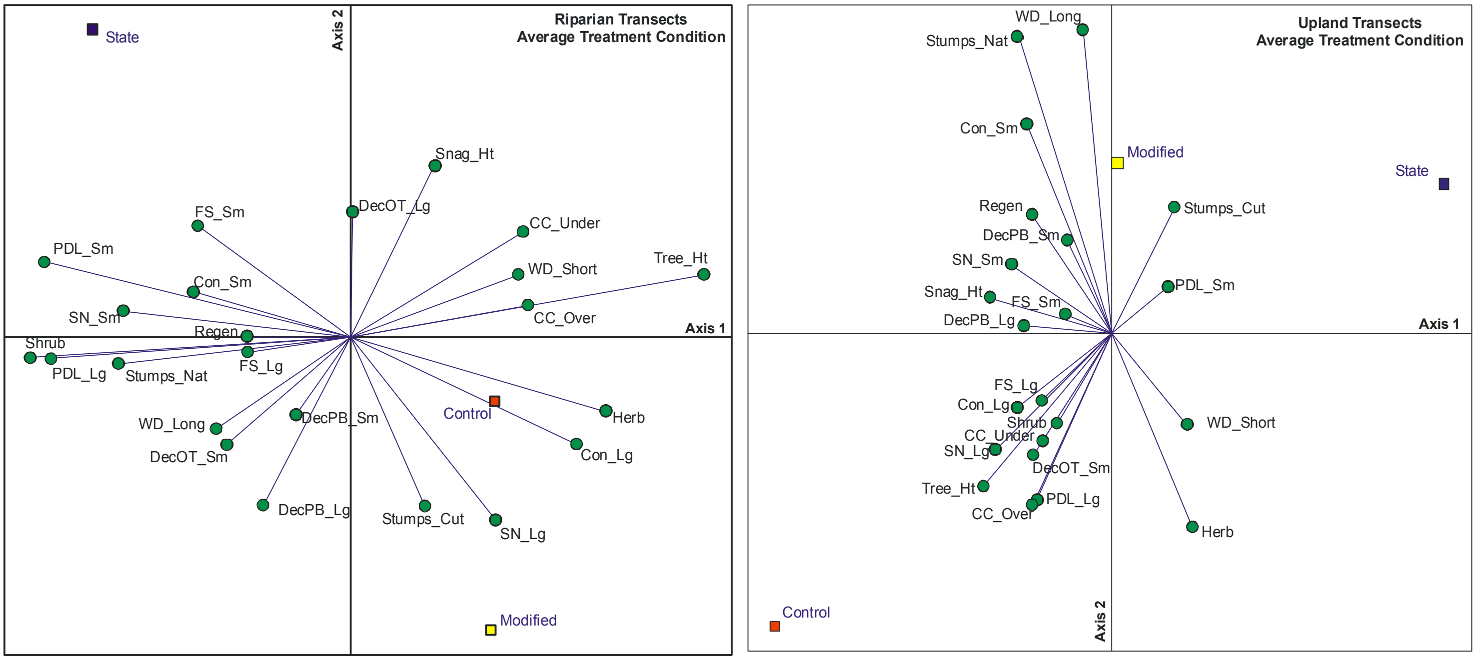

Figure 8-16. Biplots comparing the riparian and upland transect habitat condition by treatment. The data in this figure represent the mean condition as derived from the normalized score of the PCA on the habitat data matrix. WD_Long = CWD >5m long; WD_Short = CWD <5m long; Stumps_Cut = Cut stumps; Stumps_Nat = Natural stumps; CC_Over = Overstory Cover; CC_Under = Understory Cover; Regen = Regenerating conifers <0.5m tall and <4 cm DBH; SNB_Sm = Snags with bark and <25 cm DBH; SNB_Lg = Snags with bark and >25 cm DBH; SNNB_Sm = Snags without bark, < 25 cm DBH; SNNB_Lg = Snags without bark, >25 cm DBH; Con_Sm = Small-diameter conifers; DecPB_Sm = Small-diameter birch and poplar; DecOT_Sm = Small-diameter deciduous; FS_Sm = Small-diameter fir and spruce; PDL_Sm = Small-diameter pine, Douglas-fir, and larch; Con_Lg = large diameter conifers; DecPB_Lg = Large-diameter birch and poplar; DecOT_Lg = Large-diameter deciduous; FS_Lg = Large-diameter fir and spruce; PDL_Lg = Large-diameter pine, Douglas-fir, and larch; Tree_Ht = Average tree height; Snag_Ht = Average snag height; Shr_S = Seed-bearing shrubs; Shr_F = Fruit-bearing shrubs; Shr_U = Non-food shrubs; CWD_Sm = Any piece <5m long; CWD_Lg = Any piece >5m long.

Discussion

This study investigated the response of small-mammal populations to partial-retention logging in eastern Washington, with particular focus on the efficacy of riparian buffer widths in preserving components of the pre-harvest habitat conditions for the small-mammal faunas inhabiting those landscapes. Our findings indicate that the response of small mammals to the harvesting activities that occurred was highly species-specific and, with a few exceptions, minimal. Similarly, the size and configuration of the RMZ appeared to have very little effect on the diversity and relative abundance of small mammals in either the riparian or upland habitats, and most differences detected are likely a result of site effects rather than any broader treatment-level variation.

Because some species are not well sampled by one or the other trapping method employed, the imbalance of trapping effort by trap type must be taken into account when interpreting the results based on combined trap nights. Following Martin et al. (1991), we assumed that differences in mean capture rates of individual species among habitat types reflected intraspecific differences in abundance rather than habitat-related differences in trapping susceptibility. However, that study speculated that interspecific capture rate differences could be due to differential trapability as well as actual abundance differences. We expect that differential susceptibility to trapping (snap and pitfall trapping) could bias estimates of differences in interspecific abundance, but is less likely to affect intraspecific abundance estimates among treatments.

Comparisons of species richness among the three treatment types (control, state buffer, modified buffer) demonstrated that the number of species detected at the treatment level did not vary. At the transect level, no significant differences were detected in the number of species recorded in the riparian zone among any of the three treatment types. Therefore, it appears that the width and style of riparian buffer, as determined by treatment type, does not have a significant impact on the number of species inhabiting the buffer. Other research supports these findings and has demonstrated that the effect of forest harvest on small-mammal species richness within logged landscapes is reduced within buffer strips (Cockle and Richardson 2003).

Studies investigating the effects of forest thinning on small mammals in the Pacific Northwest have shown forest thinning to have a positive effect on the diversity of small mammal species inhabiting those stands (e.g., Sullivan et al. 2001; Sullivan et al. 2005; Suzuki and Hayes 2003). We also found that species diversity increase in the uplands of logged sites. Small mammal species that are associated with high levels of shrub and herb cover are particularly responsive to forest thinning (Suzuki and Hayes 2003). This increased primary productivity, combined with the higher levels of structural and microsite diversity, may allow for greater food availability (insects, other invertebrates, fungi) and increased habitat complexity following harvest, producing an increased number of niches available for a wider variety of small mammal species to occupy. As well, as suggested by Wilson and Carey (2000), changes in the type and texture of the humus layer of the soil can have implications for the small-mammal fauna inhabiting those sites. Changes to the humus layer would be expected following harvest, and this may account for some of the observed differences in species richness in upland habitats among the treatments. This factor is of particular importance, as it relates to changes in the fungal component of the soil, which forms a large part of the diet of several small-mammal species, such as *Clethrionomys gapperi* and *Glaucomys sabrinus* (Sullivan et al. 2001; Ransome et al. 2004; Nagorsen 2005).

Despite these findings, it should be noted that the species richness results of this study do not necessarily reflect the true diversity of species present at the sites. As demonstrated in Mackenzie et al. (2004), many ecological studies evaluating the spatio-temporal patterns of species assume equal detection of all species present. This assumption can lead to incorrect inferences about patterns of species richness, because some species may be inaccurately labelled as absent when they simply were not detected. For example, 2 species that were not adequately sampled during this study due to trapping techniques were *G. sabrinus* and *Tamiasciurus hudsonicus*. These species are both largely arboreal and spend relatively little time on the ground (Nagorsen 2005), and thus they would not have come in regular contact with the traps. As a result, a lack of these species from the capture totals at any of the treatments or transects cannot be taken as evidence of their true absence from the site, but rather is a result of the terrestrial-based trapping programs used and their ineffectiveness in targeting species that are largely arboreal in their habits. Other species that were not adequately sampled during this study were *Lepus americanus*, *Thomomys talpoides*, and *Neotoma cinerea*. Additionally, drift fences were not used in this study, and this may have resulted in fewer captures, therefore contributing to lower species richness measures within each treatment and transect for some of the smaller species (particularly *Sorex*).

The relative abundance of individual small-mammal species between treatments was minimal and insignificant for most species recorded. Of the 8 most frequently encountered species, none showed any significant changes in relative abundance in riparian habitats between treatments, and whatever minor variation was detected was generally attributable to site effects. Within upland habitats, however, one group of species (*Tamias*) significantly increased in relative abundance following upland logging at the state and modified treatments. This increase involved *T. amoenus* and *T. rubricaudus*, as well as unidentified *Tamias* individuals, and the detected increase is consistent with the habitat requirements of the species involved. Both of the *Tamias* species detected are characteristic of open coniferous forests and open or edge habitats, with the densest populations occurring in areas with abundant woody ground cover (Nagorsen 2005). Studies have shown *T. amoenus* to be 1.3–3.8 times more abundant in forests that had been commercially thinned than in forests that had not been thinned (Nagorsen 2005). As the increase in *Tamias* was similarly reflected in both the state buffer and modified buffer sites, but not at control sites, it appears that the type of riparian buffer was less responsible for increasing populations of *Tamias* than that the upland habitats had been selectively logged. Other species did not show any significant changes in relative abundance between treatment types.

Only *C. gapperi* exhibited a significant change in relative abundance between the two cut treatments. The relative abundance of *Clethrionomys* was significantly greater in upland habitats at modified buffer sites than upland habitats at state buffer sites. This suggests that modified buffer sites may provide better habitat conditions for this species. This species will tolerate a broad range of habitat conditions, but the highest populations are generally found in mesic, late-successional or old-growth forested habitats with high levels of structural diversity and microhabitat development (Sullivan et al. 2001; Sullivan et al. 2005). Furthermore, the presence of a well-developed understory of shrubs and young conifers appears to be correlated with higher populations of this species (Nagorsen 2005; Sullivan et al. 2005). Suzuki and Hayes (2003) found that *Clethrionomys* increased following thinning of the forest, suggesting that the presence of dense shrubs and vigorous understory growth may be more important to this species than the age of the forest. The discrepancies in the relative abundance of *Clethrionomys* between the modified and state buffer sites may be attributable to the abundance of shrubs in the upland regions of those treatments; although, as stated by Carey and Harrington (2001), rarely are wild vertebrates restricted by only one limiting factor. Nonetheless, comparison of the habitat conditions described by the biplots for those treatments (Figure 8-15) indicates that shrub cover was a much more prominent component in the upland habitats at modified buffer sites than at state buffer sites; and at modified buffer sites the amount of shrub cover was closely associated with the relative abundance of *C. gapperi*. The increased development of the shrub layer in upland habitats at the modified buffer sites is probably unrelated to the retention of the RMZ but more likely reflects site-level variation across the landscape.

Comparisons of the species richness, relative abundance, and habitat associations of small mammals between the state buffer and modified buffer sites suggests that the different prescriptions applied to the riparian zone of these two treatment types had minimal or no differential effect on the small mammals in either riparian or upland habitats. These minimal effects are not particularly surprising given the minor physical differences between the two treatments. Based on the results of this study, it appears that the presence of any form of riparian buffer may be important in preserving small-mammal populations, and the size, orientation, and configuration of the RMZ may be of little consequence to small mammals.

A number of factors potentially influence the veracity of this study's results or limit the extent to which the conclusions can be used to aid in managing and conserving small mammals in coniferous forests of eastern Washington. The purpose of presenting these limitations is to make the reader aware of (1) factors that could have influenced the results of the study and (2) factors that should be considered if this study is to be used to guide formal policy development. Factors to consider include the following.

Annual and Seasonal Population Fluctuations—Populations of many small-mammal species are well known for undergoing substantial population fluctuations between years, often with a peak in abundance followed by a sudden and sharp population crash. This pattern has been documented in a number of species that were detected in this study, including *S. cinereus*, *C. gapperi*, *Microtus longicaudus*, *M. pensylvanicus*, and *M. richardsoni* (Nagorsen 2005) and likely occurs in other species as well. Due to study limitations, particularly the inability to analyze a temporal data set, it was not possible to determine how the abundance of small mammals within the study was related to any short- or long-term variation in the natural population. So, for example, it was not possible to determine whether low capture rates of several *Microtus* species represented true scarcity of those species within the study area or whether the current survey period coincided with a population low for one or more of those species.

Furthermore, this study took place during the spring and early summer period of two consecutive years. During this period, following winter mortality and prior to juvenile dispersal, the abundance of most small-mammal species is at a seasonal low compared to other times of the year (Nagorsen 2005). This factor would be of particular importance for species that were encountered in lower numbers during this survey. Sampling in the late summer or fall, during the annual peak of small-mammal abundance, would be expected to yield greater capture rates for these less abundant species and thus a greater number of species for analysis. Since the less common species are often those that are more likely to be lost from the ecosystem following habitat disturbance, an ability to analyze population changes in those species may have provided more significant results.

Sampling Methods—Accurate comparisons of species richness values among treatments and transects requires that all species potentially present at a site be documented to avoid falsely labelling a species as not present. The restrictions of using only the snap-trap and pitfall-trap methods, the latter without any form of drift fences, reduces the effectiveness of catching a wider variety of species. Arboreal and fossorial species require special capture techniques to obtain accurate abundance results and, as such, were not properly sampled using the two trapping methods employed. Application of a wider variety of trapping methods would undoubtedly increase the effectiveness of the sampling for these other species and result in more comprehensive results on which to base conclusions.

The use of kill trapping techniques rather than live trapping techniques may have additional implications for the population structure and dynamics of small mammals, since each individual sampled is removed from the population. As a result, the study was continually sampling a declining population. This factor was not addressed in this protocol; therefore, it is unclear to what extent kill trapping altered the population structure of these species and how much influence this may have had on the results of the study.

Treatment Development—Although the two cut treatments applied in this study were intended to differ in a number of habitat features, such as tree retention, snag retention, and the amount of habitat preserved around the stream, the difference between the treatments was biologically insignificant with respect to small mammals. To better address the efficacy of RMZs in providing habitat for small mammals, the riparian buffers applied need to be measurably different with respect to habitat attributes. For example, Cockle and Richardson (2003) determined that a 30 m–wide buffer strip reduced the effects of logging on the riparian zone. Perhaps a wider buffer such as that may have resulted in a more measurable difference between the two cut treatments.

Conclusions

Small-mammal response to the retention of RMZs in this study was species-specific, with no global response shown by small mammals as a group. The treatments employed in this study, differing in the size and shape of the retained RMZ, were insufficiently different from each other to allow for any meaningful comparison between the two. As a result, we were unable to draw any conclusions regarding the effectiveness of a narrower, uniform-width (state) buffer compared to a wider, variable-width (modified) buffer in maintaining pre-harvest small-mammal populations on a managed landscape. It was possible, however, to detect some significant changes in populations of several species, specifically the chipmunks (*Tamias* spp.), within the upland habitats that had been selectively logged.

Additionally, we were unable to analyze a temporal data set (i.e., we did not have the pre-harvest or 2-year post-harvest data); as such, the dynamics of small-mammal populations over time following logging remain unclear. The sampling interval for this study was limited to 10-years post-harvest, and the immediate effect of logging (e.g., 1–2 years post-harvest) may differ dramatically from our findings. For example, O'Connell et al. (2000) found that populations of most species of small mammals in northeastern Washington increased sharply immediately following logging but declined significantly within 2 years after logging. The long-term post-harvest dynamics of small-mammal populations in northeastern Washington remain unknown, and it is therefore difficult to identify how the management prescription used in this study affected small-mammal populations 10 years after logging.

A lack of significant changes in the relative abundance of small mammals as a whole within the RMZs suggests that any such buffer may be adequate for maintaining small-mammal populations at the levels present in typical second-growth, montane, coniferous forests of northeastern Washington. However, as demonstrated by Carey (2003), the mid-seral stands that were used as controls in this study may represent some of the lowest levels of diversity and abundance of species of any stage during the life of the forest. This is exacerbated by some traditional management practices, such as removal of woody debris and maintenance of single-species, even-aged stands, which reduce structural and habitat complexity. This management-induced habitat homogenization and simplification negatively affects biodiversity (Carey and Harrington 2003) and has the ability to suppress the species richness and relative abundance of small mammals in those stands. Therefore, a similarity in the small-mammal populations inhabiting the riparian buffers and logged upland habitats may not indicate maintenance of populations at pre-harvest levels but, rather, maintenance of populations at the potentially suppressed levels typical of managed second-growth coniferous forests. Obtaining the pre-harvest and 2-year post-harvest data would certainly help elucidate any temporal changes in small-mammal populations and provide a better measure of the efficacy of RMZs in providing habitat for small-mammal populations in northeastern Washington.

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Appendices

Appendix 8-A. Small-mammal captures per 100 trap nights for riparian and upland transects of the control, state buffer, and modified buffer sites, sampled in eastern Washington, 2003/2004.

Riparian						
	2003			**2004**		
Species	**Control**	**State**	**Modified**	**Control**	**State**	**Modified**
SOCI	2.90	1.99	2.49	1.23	1.36	2.05
SOHO	0.00	0.00	0.00	0.05	0.12	0.00
SOMO	0.74	0.26	0.62	0.00	0.00	0.00
SOPA	0.07	0.00	0.16	0.02	0.00	0.08
SOVA	1.63	1.59	2.10	1.98	1.18	2.65
SOREX	0.16	0.16	0.24	0.69	0.39	0.99
LEAM	0.05	0.00	0.00	0.02	0.00	0.00
GLSA	0.00	0.00	0.00	0.00	0.00	0.00
TAAM	0.00	0.28	0.18	0.00	0.00	0.00
TARU	0.00	0.02	0.05	0.09	0.51	0.11
TAMIAS	0.04	0.00	0.00	0.17	0.14	0.08
TAHU	0.00	0.02	0.03	0.00	0.00	0.00
THTA	0.03	0.00	0.03	0.02	0.04	0.08
CLGA	2.72	2.12	2.91	2.42	1.84	2.13
MILO	0.02	0.02	0.00	0.00	0.00	0.00
MIPE	0.00	0.00	0.00	0.02	0.03	0.03
MIRI	0.00	0.00	0.03	0.00	0.00	0.00
MICRO	0.00	0.02	0.00	0.07	0.05	0.14
NECI	0.00	0.00	0.00	0.02	0.00	0.00
PEMA	4.18	3.08	2.90	1.03	1.05	0.59
ZAPR	0.61	0.88	0.89	0.37	0.50	0.46
Upland						
	2003			**2004**		
Species	**Control**	**State**	**Modified**	**Control**	**State**	**Modified**
SOCI	2.36	1.60	3.30	1.43	1.75	3.09
SOHO	0.00	0.00	0.00	0.00	0.10	0.08
SOMO	0.20	0.26	0.39	0.05	0.00	0.03
SOPA	0.00	0.11	0.03	0.00	0.00	0.00
SOVA	0.16	0.64	0.57	0.45	0.45	0.60
SOREX	0.18	0.05	0.10	0.41	0.50	1.25
LEAM	0.00	0.05	0.00	0.00	0.00	0.00
GLSA	0.00	0.02	0.00	0.00	0.00	0.00
TAAM	0.23	1.48	0.88	0.00	0.00	0.00
TARU	0.00	0.21	0.08	0.42	0.70	1.04
TAMIAS	0.00	0.11	0.26	0.29	0.79	0.67
TAHU	0.02	0.00	0.00	0.00	0.00	0.00
THTA	0.04	0.09	0.00	0.12	0.07	0.05
CLGA	1.24	0.97	1.95	1.70	1.10	2.08
MILO	0.00	0.02	0.03	0.00	0.00	0.00
MIPE	0.00	0.00	0.00	0.00	0.02	0.00
MIRI	0.00	0.00	0.00	0.00	0.00	0.00
MICRO	0.00	0.07	0.03	1 (0.02)	0.18	0.03
NECI	0.00	0.00	0.00	0.00	0.00	0.00
PEMA	2.45	2.16	1.85	1.06	1.30	1.14
ZAPR	0.07	0.54	0.35	0.05	0.73	0.14

Appendix 8-B. Description of habitat variables used in the co-inertia analysis.

Category	Measure	Description
WD_Long	Number of pieces / plot -- Quadrat	Total count for transect divided by 15 plots / transect / site
WD_Short	Number of pieces / plot -- Quadrat	Total count for transect divided by 15 plots / transect / site
Stumps_Cut	Number stumps / plot	Total count for transect divided by 15 plots / transect / site
Stumps_Nat	Number stumps / plot	Total count for transect divided by 15 plots / transect / site
CC_Over	Average CC for transect	Averaged across each transect at each site
CC_Under	Average CC for transect	Averaged across each transect at each site
Regen	Number regenerating stems per plot	Total count for transect divided by 15 plots / transect / site
SNB_Sm	Number snags with bark per plot -- small	Total count for transect divided by 15 plots / transect / site
SNB_Lg	Number snags bark per plot -- large	Total count for transect divided by 15 plots / transect / site
SNNB_Sm	Number snags no bark per plot -- small	Total count for transect divided by 15 plots / transect / site
SNNB_Lg	Number snags no bark per plot -- large	Total count for transect divided by 15 plots / transect / site
Con_Sm	Number per plot	Total count for transect divided by 15 plots / transect / site
DecPB_Sm	Number per plot	Total poplar, birch / 15 plots / transect / site
DecOT_Sm	Number per plot	Total alder, maple / 15 plots / transect / site
FS_Sm	Number per plot	Total true fir, spruce / 15 plots / transect / site
PDL_Sm	Number per plot	Total pine, Douglas-fir, larch / 15 plots / transect / site
Con_Lg	Number per plot	Total count for transect divided by 15 plots / transect / site
DecPB_Lg	Number per plot	Total poplar, birch / 15 plots / transect / site
DecOT_Lg	Number per plot	Total alder, maple / 15 plots / transect / site
FS_Lg	Number per plot	Total true fir, spruce / 15 plots / transect / site
PDL_Lg	Number per plot	Total pine, Douglas-fir, larch / 15 plots / transect / site
Tree_Ht	Average height for transect	Tree height averaged across transect for each site
Snag_Ht	Average height for transect	Snag height averaged across transect for each site
Shr_S	Frequency of occurrence for all seed-bearing shrubs	Number of occurrences divided by total intercepts for each site and transect
Shr_F	Frequency of occurrence for all fruit-bearing shrubs	Number of occurrences divided by total intercepts for each site and transect
Shr_U	Frequency of occurrence for all non-food shrubs	Number of occurrences divided by total intercepts for each site and transect
CWD_Sm	Frequency of occurrence for small CWD -- floristics	Number of occurrences divided by total intercepts for each site and transect
CWD_Lg	Frequency of occurrence for large CWD -- floristics	Number of occurrences divided by total intercepts for each site and transect

Appendix 8-C. Codes and scientific and common names of small mammals referred to in the text.

Code	Scientific Name	Common Name
SOCI	*Sorex cinereus*	Masked Shrew
SOHO	*Sorex hoyi*	Pygmy Shrew
SOMO	*Sorex monticolus*	Dusky Shrew
SOPA	*Sorex palustris*	Common Water Shrew
SOVA	*Sorex vagrans*	Vagrant Shrew
SOREX	*Sorex* spp., unidentified	
LEAM	*Lepus americanus*	Snowshoe Hare
GLSA	*Glaucomys sabrinus*	Northern Flying Squirrel
TAAM	*Tamias amoenus*	Yellow-pine Chipmunk
TARU	*Tamias rubricaudus*	Red-Tailed Chipmunk
TAMIAS	*Tamias* spp., unidentified	

Chapter 9 EASTERN WASHINGTON CONCLUSIONS AND MANAGEMENT IMPLICATIONS

Virgil C. Hawkes

The type of upland logging used in eastern Washington produced habitats with high structural complexity relative to the pre-harvest condition, when most stands were even-aged second-growth forests of Douglas-fir. In eastern Washington, it is apparent that the structural complexity and habitat heterogeneity created by partial logging benefited populations of songbirds and small mammals at the 10-year post-harvest sampling interval. In some cases, the relative abundance of certain species increased significantly relative to the control sites, further supporting the benefits of partial-retention logging in upland coniferous forest of northeastern Washington. As in western Washington, riparian management has become (necessarily) site-specific. The ability to manage riparian habitats based on the conditions of the site, coupled with the maintenance of partial logging of upland forests, will create forested habitats that not only maintain richness and relative abundance of non-game wildlife, but that are likely to promote species richness over time, leading to increased biodiversity.

It was unfortunate that we were unable to test the temporal and spatial aspects of riparian management zone (RMZ) retention adjacent to upland forests that were partially cut. Based on the findings of O'Connell et al. (2000), we can only surmise that species diversity and relative abundance values of small mammals and songbirds were not adversely affected. In all likelihood, the relative abundance of some species likely increased as a direct result of partial-retention logging. Moreover, the use of partial-retention logging in the upland made it very difficult to identify the zone of transition between riparian and upland habitats, except for at sites where stochastic environmental effects had changed the landscape (e.g., through forest fire and extensive windthrow). From a structural (i.e., plant) perspective, logged sites were more diverse than unlogged sites, in both riparian and upland habitats. The even-aged stands of the unlogged control sites typically had low vegetative diversity, which would inhibit their ability to support increasing biodiversity. As has been pointed out in previous chapters (e.g., chapter 5), the simplification of forested habitats can, over time, have adverse affects on wildlife and wildlife habitat.

For both small mammals and songbirds, the state buffer sites were as good as the modified buffer sites in maintaining richness and relative abundance values at levels equal to, or greater than, those obtained for unlogged control sites. The combination of riparian habitat retention and partial logging in the upland forest improved habitat suitability 10 years following logging and contributed to the increased richness of songbirds. In eastern Washington, riparian habitat retention in forested landscapes is extremely important. To maintain (or even increase) biodiversity on landscapes managed for timber, upland management strategies that contribute to the creation of forested habitats (riparian and upland) that have high structural complexity and heterogeneity (as opposed to simplified, even-aged, homogeneous stands) should produce landscapes with high biocomplexity and biodiversity at multiple scales, and therefore these landscapes should be resilient in the face of disturbance (Holling 2001).

Since this study's inception in 1991, there have been changes to the Washington Forest Practices Rules and Regulations that have implications for our findings. The Forest Practice Rules of 1988 required a riparian buffer that met the state configuration (i.e., not less than 7.5 m from the ordinary high-water mark) for both fish-bearing (Type 3) and non–fish-bearing (Type 4) perennial streams.

The development of the modified riparian prescription was an alterative to the state buffer that provided for greater structural complexity, heterogeneity, and ultimately greater diversity for both flora and fauna. Lee et al. (2004) identified the current trend in riparian management prescriptions as tending toward site-specific buffers that vary among broadly similar harvest areas to within a single harvest area. The Washington Forest Practice Rules of 2001 seem to fit this model and, aside from changing how watercourses are defined, are site-specific with many options. Lee et al. (2004) suggest that the primary benefit of this type of prescription is the identification of well-defined criteria to delineate the riparian zone. These criteria are specifically defined by the modifying factors selected by jurisdictions, and buffers are applied in a predictable response to these criteria. The caveat to tailor-made buffers is the greater complexity in guidelines. For example, Washington now describes 3 zones of management for Type F streams (Type 3): (1) the core, (2) inner, and (3) outer zones, with various levels of tree removal, thinning, hardwood conversion, and other activities permitted within these zones depending on the site class of land, the management harvest option, and the bankfull width of the stream. In general, the site class will form the RMZ prescription. It appears that riparian management in eastern Washington has become (necessarily) site-specific. This implies that applying a one-size-fits-all approach to riparian management has been set aside in favor of a more intensive approach that considers the specifics of the site, which is consistent with Lee et al. (2004). For Type Np streams, or perennial non–fish-bearing streams (Type 4), the RMZ rules are slightly less complicated; however, there is ample room for site-specific management. Considering that the RMZ rules have evolved to embrace a site-specific approach, it seems only fitting that forest management in general adopt the same principals when considering biodiversity. Managing for biodiversity on a regional scale does not account for variation at the landscape or site level within the region, which could have dire consequences for species that are limited geographically, that occur at naturally low densities, or that are particularly sensitive to habitat change.

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Appendix A

FINALIZATION SUMMARY FOR THE FINAL REPORT FOR THE STUDY: EFFECTIVENESS OF RIPARIAN MANAGEMENT ZONES IN PROVIDING HABITAT FOR WILDLIFE: RESAMPLING AT THE 10-YEAR POST-TREATMENT INTERVAL

FINALIZATION SUMMARY FOR THE FINAL REPORT FOR THE STUDY:

EFFECTIVENESS OF RIPARIAN MANAGEMENT ZONES IN PROVIDING HABITAT FOR WILDLIFE: RESAMPLING AT THE 10-YEAR POST-TREATMENT INTERVAL

BY

THE LANDSCAPE AND WILDLIFE ADVISORY GROUP (LWAG)

FOR

THE COOPERATIVE MONITORING, EVALUATION, & RESEARCH (CMER) COMMITTEE
WASHINGTON DEPARTMENT OF NATURAL RESOURCES (WDNR)

This summary is intended to provide background context, clarify finalization issues, and describe key results for the study Effectiveness Of Riparian Management Zones In Providing Habitat For Wildlife: Resampling At The 10-year Post-treatment Interval. This study investigated the efficacy of riparian management zone (RMZ) buffers for providing habitat to terrestrial vertebrate populations in managed forest stands in Washington. This study was completed in 2008, but the final report will have atypical finalization due to a variety of circumstances. Reading the information summarized in this memo is **absolutely necessary** to understanding these circumstances and reading this final report.

Background Context

- This study is **Part 2** of the RMZ buffer research begun in the early 1990s. This research was designed to experimentally test the efficacy of RMZ buffers for providing habitat to various terrestrial vertebrate groups. **Part 1** of this research is described in O'Connell et al. (2000. Effectiveness Of Riparian Management Zones In Providing Habitat For Wildlife TFW-LWAG1-00-001) and the bird segment of this research was published in Pearson and Manuwal (2001. Ecological Applications 11(3):840-853). One should refer to these documents for details of the research in **Part 1**. Both **Part 1** and **Part 2** of this RMZ research included the following general design elements in common:
    - Three treatments: two types of RMZ buffers left after harvest, and one no-harvest reference (= control).
    - Six replicates of each treatment (n = 18) for each geographic area.
    - Two geographic areas: western Washington and eastern Washington.
    - The forest harvest adjacent to the two RMZ buffer treatments followed standard regional methods: in western Washington, the harvest adjacent to the RMZ buffer treatments was a clearcut; in eastern Washington, the harvest adjacent to the RMZ buffer treatments was a selective harvest with 40-60% retention.
    - A before/after-treatment/control study (BACI) design where all 36 sites were sampled for two seasons immediately prior to forest harvest, two seasons after forest harvest (the combination of which comprised **Part 1**), and two seasons roughly 10 years after forest harvest, which comprises **Part 2**. **This final report describes the results of Part 2**.
    - Songbirds, small mammals, and amphibians (western Washington only) were sampled to assess changes in species and community (= assemblage) composition and relative abundance. Songbirds were sampled using point counts. Small mammals were sampled using snap traps and pitfall traps. Amphibians were sampled using pitfall traps.

  - Vertebrate sampling occurred along two transect types: a riparian transect parallel to the stream within riparian habitat, and an upland transect parallel to the stream in upland habitat 100 m upslope from the riparian transect.
  - Study sites in western Washington were located over a relatively broad geographic area on public and private commercial forest land, which was scattered from the Willapa Hills east to the west slope of the South Cascade Mountains and north to King County.
  - Study sites in eastern Washington were located on public and private commercial forest land and were clustered in a relatively small geographic area in northeast Washington north of Spokane.
- The design and methods of a long-term and geographically large-scale project such as this ultimately resulted in challenges that influenced the outcome of this research.
  - Not all treatments in western Washington were harvested in the same year in **Part 1** due to economic conditions surrounding commercial harvest. This resulted in temporal staggering of treatment application in **Part 1**. However, all sampling in **Part 2** occurred in the same years (2003 & 2004). Therefore, time elapsed since treatment application was roughly 1 year shorter for two western Washington study sites.
  - Not all reference (no-harvest) replicates had remained unharvested when **Part 2** was implemented, thus requiring surrogates (replacement sites).
  - Selected vertebrate sampling methods became out-dated by the time **Part 2** was implemented but were used regardless in order to maintain comparability between all sampling intervals of this research.
  - In **Part 1**, there were ambiguities and lack of statistical power for bat sampling using bat detectors, amphibian sampling using in-stream surveys, and all eastern Washington herpetofaunal sampling. As a result, LWAG decided to remove these sampling efforts from **Part 2**.
  - Two different contractors were used for **Parts 1** and **2**.
  - The raw data from **Part 1** was misplaced by WDNR and was not relocated until after the completion of **Part 2**. This resulted in cancellation of analysis using data from all sampling intervals. Instead, **Part 2** analysis for the eastern Washington sites included only data collected during **Part 2** sampling. **Part 2** analysis for the western Washington sites included data collected during **Part 2** sampling and a reconstructed version of **Part 1** data, which was developed before the original **Part 1** data was relocated by WDNR.

Final Report Completion

Steps leading to the finalization of this report were complicated for several reasons.

- Contractor funds were largely exhausted before the final report preparation stage and therefore resources to complete the report were limited. This resulted in a less responsive approach to addressing LWAG and CMER comments to the draft final report.
- LWAG asked the contractor to organize this report as a series of stand-alone chapters in anticipation that some chapters would be submitted for peer-reviewed publication. Unfortunately, this resulted in a lengthy report with uneven consistency across chapters which made all stages of the review process more difficult.
- LWAG participation (few people, busy schedules) hindered a thorough, timely, and effective review of early drafts of the final report.

- A large portion of review comments from LWAG and CMER related to format, grammar, and spelling. LWAG had an editor make these revisions prior to advancing the final report to the Independent Scientific Peer Review (ISPR) Committee.
- The ISPR Committee pointed out multiple issues including organization, consistency in application of selected aspects of statistics, approach in analysis, and interpretation. These comments are attached at the end of this document. These comments include important issues that ideally should be addressed prior to finalization of this report, especially for sections that might be considered for peer-reviewed publication.
- LWAG has recommended that these revisions should not be made at this time. The cost of addressing these changes in the report is a luxury not affordable in this fiscally constrained time. Instead this final report has been finalized with ISPR Committee comments and this summary appended. Additional revisions are unlikely to result in changes in interpretation in the key results of this study (see below), both in context of this report and from that of O'Connell et al. (2000).

Key Results Of This Study

- No marked differences between RMZ buffer treatments appear evident for any taxon group, which may partly reflect too great a similarity between the treatments. Differences between treatments and references (no harvest units) were limited and inconsistent over years, species, or geographic locations.
- Differences detected seemed attributable to predictable species-specific habitat associations (riparian/upland, successional stage, geographic location) or between-year variation.
- Western Washington songbirds seemed to show only one important difference among the three treatments. Similarity in riparian songbird assemblages was lowest in comparison to upland assemblages at harvested sites between and among sampling intervals irrespective of riparian treatment. Similarity in those assemblages was greater for the modified treatment than the state treatment, presumably reflecting greater species turnover in the latter, during both the post-harvest interval in this study and in the original study. Species richness and abundance seemed to increase with succession since harvest, but neither upland nor riparian transects seemed to differ significantly among treatments. Differences between upland and riparian transects seemed greatest for the harvest treatments.
- Eastern Washington songbirds showed no clear difference among the three treatments based on species richness, abundance, and species composition. As with western Washington songbirds, species richness and abundance seemed to increase with succession since harvest.
- Western Washington small mammals showed no clear response to the three treatments. Deer Mice (*Peromyscus* spp.), Shrews (*Sorex* spp.), and Voles (*Microtus* spp.) were the three species groups with the most captures. Species richness did not change significantly over time or between treatments. Abundance of individual species appeared to varied over time and between treatments but this could only partially be attributable to the treatments. Although there may have been some response to the harvest treatments (control vs. either of the two buffer treatments) this appeared to be confounded by natural population fluctuations over time.
- Eastern Washington small mammals showed marginal response between the three treatments. Captures were dominated by Southern Red-backed Vole (*Clethrionomys gapperi*), Deer Mouse (*Peromyscus maniculatus*), Masked Shrew (*Sorex cinereus*), and Wandering Shrew (*Sorex vagrans*). Response to the treatments was highly species-specific, and though some differences

were observed between the control and either of the two RMZ buffer treatments, site effects confounded comparisons.

- Amphibians (western Washington only) showed no uniform response to harvest treatments. Rather, species showed individual responses that varied over time and between treatments and transects. Two species, Ensatina (*Ensatina eschscholtzii*) and Western Red-backed Salamander (*Plethodon vehiculum*), had sufficient data upon which to withdraw meaningful analyses.

LWAG respectively reminds CMER that though we are confident that the results of the study summarized here represent the true final results, their peer-reviewed formalization will require significant special effort at the future time at which CMER may consider taking this step.

Appendix B

Independent Scientific Peer Review Comment Matrix

RMZ Re-sample Final Report ISPR
Are rigorous, transparent, and sound research and statistical methods followed?

Comment #	REVIEWER #	Comment location Page/ Paragraph/ Sentence	ISPR COMMENT
1	1		The researchers attempted to do a split-plot design and in most cases this was attained, but in several instances it was not. For example in western Washington two control areas (Hotel Creek and Taylor Creek) were close together without a state site nearby and with the modified site somewhat distant, leaving a second cluster of Blue Tick and Night Dance without a spatially linked control.
2	1		Vail, a control, was logged before 2003 and was replaced by a distant control, West Fork Falls Creek. Griffen Creek, a modified site, was thinned and was removed from the last sampling interval. These are all unfortunate, but I would have liked to see more information on how this was dealt with in the analysis, since presumably these deleted sites were included in the previous two sampling intervals.
3	1		What the effect of the surrounding landscape being harvested had on the control sites, but it was not clear how far away control sites were from any logging.
4	1		In eastern Washington, bird and small mammal data were not available pre and early post harvest for analysis which greatly impacts the conclusions that can be drawn from this study and obviates the use of a BACI analysis.
5	1		During song bird surveys there is no mention of how observers were trained, whether they were consistent between years and how bird detectability was ascertained.
6	1	Pg. 3-7 and Pg. 3-8	There is some confusion on pages 3-7 and 3-8 as to how observers were allocated across sites on the same day.

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
7	1		Why were 2 sites visited out of sequence (two in 1997 and 2 in 1998)? This would add to variability in the data since there is obviously a lot of annual variability in abundance of wildlife. I can't say with full certainty whether staggering two sites compromises study results as it was not clear how this was dealt with in the analysis, however, doing this increases annual variability and could make it less likely to detect a significant change on treatments relative to controls. Thus, any significant effects are likely to be real, whereas insignificant results could be spurious. The best way to determine this is to look at the magnitude of the change: if large changes are not statistically significant, then you might begin to worry that there are statistical power issues. It is difficult for me to assess this because I don't know how sample size and variance affect Morista's index and a repeated measures ANOVA was not done on single species abundances (at least I can't find where it was done). For me, the most relevant measure of an effect of buffer treatments is 1) community similarity from before to after the experiment relative to controls and 2) large changes in the abundances of some species known to be sensitive to forest loss (e.g. older forest species) in treatments relative to controls. As an example, for WW songbirds the Morisita's index of similarity was least similar from the pre-treatment condition for the modified upland and state upland sites and communities at both riparian treatments were slightly less similar after treatment than were control sites. The valid conclusion here is that logging in the uplands causes the biggest change in communities and logging in the buffers less so relative to controls. The next question is, given differences in the communities, which species are driving this? Here the relevant question is which species on treatments have declined from pre to post harvest more than on the controls.

Comment #	REVIEWER #	Comment location Page/ Paragraph/ Sentence	ISPR COMMENT
8			As far as I can see, this analysis was not done or if done, the overall significance not presented. On page 3-26, 3-27 are numerous post-hoc comparisons, none of which appears to compare changes in the treatments relative to the controls: a repeated measures ANOVA is required here. From Appendix 3-E, it appears that the following species have declined on one or both treatments in one or both post-treatment periods, relative to controls, but a repeated measures ANOVA should be done to confirm this (no error bars are shown on these graphs so it is difficult to assess variability): CBCH, GCKI, BRCR, WIWR, PSFL, HEWA, GRJA, HUVI.
9	1	Table 3-6	Why some sites were sampled again in 1997 and 1998. How was this discrepancy dealt with in the data analysis?
10	1		At Abernathy, the data were excluded from the analysis, so why present this as it confuses the reader? How was this discrepancy dealt with in the data?
11	1		At Ryderwood, the site was logged in 1996, but the data were excluded in 1995-why? The same site, was then monitored in 97 and 98- why if data were excluded in 95? How was this discrepancy dealt with in the data analysis.
12	1		EW songbirds were surveyed along transects at stations 50 m apart, thus surveys did not repeat the methods used in WW exactly. Why were survey methods changed?
13	1	Pg. 7-9	Why were filters for inclusion of data changed?
14	1	Table 7-10	Not clear why sampling effort shown for periods where data not available?
15	1	Ch. 5	Trapping methods were changed in the last period. While I understand this was done to reduce deaths and to prevent double counting, it could change the estimates of abundance relative to the earlier intervals. I am not an expert in amphibian sampling so am unclear as to how this could change the estimates, but this makes it doubly important to use repeated measures analyses appropriately so that any differences noticed in the last interval can be attributed to treatment effects and not sampling effects.

Comment #	REVIEWER #	Comment location Page/ Paragraph/ Sentence	ISPR COMMENT
16	1		Overall, I felt there were a number of problems with the data analysis. the main problems that should be addressed that could affect inferences drawn from the study. I left out comments on analyses that I don't think were useful (e.g. alpha diversity). • Data from modified and state treatments were pooled to test for effects of upland logging- this is not a good idea since species richness and abundance can be affected by the local treatment and combining them could wash out an effect of upland logging. This should be repeated with the treatments not pooled to ensure that conclusions are valid. • Results of the Morisita's index should be followed by describing how the communities have changed, using repeated measures ANOVA. From what I can see in the report this was supposed to have been done, but all that is presented is various post-hoc comparisons that do not appear to control for changes on the control plots. Controlling for changes on the control is essential otherwise conclusions may not be valid (e.g. a certain species may go down on a treatment pre to post harvest, but it may also decline on control plots). Perhaps I missed something here, but if so, there needs to be a much clearer results section showing the overall repeated analysis results. • In eastern Washington, any differences between controls and treatments attributed to treatment may be spurious if there were differences in the habitats of the three groups prior to harvest. If data are available, vegetation could be compared among pre-treatment groups to check for this. • Power analyses should be done to determine for which species there was significant power to detect biologically meaningful changes.
17	1	Pg 3-13	Data from different years and treatments were pooled. For example, for WW Songbirds data were pooled into sampling intervals of two years, but pre-harvest data consisted of one year only. Data from modified and state treatments were pooled to test for effects of upland logging- this is not a good idea since species richness and abundance can be affected by the local treatment and combining them could wash out an effect of upland logging.

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
18	1		Some sites were sampled in different years than others and there were data missing in some years. How was this dealt with in the analysis without introducing bias from annual variation?
19	1	Pg 3-13	For birds, the filtering of the data was appropriate, except I assume that —non-breeding migrant actually means transients that were passing through since breeding status was not evaluated. How was transient status determined?
20	1		For birds, in some analyses the strictly filtered data were used and in others more leniently filtered data were used. This added to the number of analyses and increased confusion for the reader. Why not stick to the basic filtered data throughout?
21	1	Ch. 7	For birds there did not appear to be any corrections made for differences in detectability. The authors note in chapter 7 that it was harder to hear birds along streams with a lot of noise, but this was not corrected for, leading to possible biases in diversity and abundance analyses. These days it would be hard to publish a paper that has not corrected for detectability.
22	1		In reports on alpha diversity it was not clear whether filtered data were used or the full data set. If the full data set, was rarefraction or some correction factor applied to deal with differences in detectability and the fact that some species are very rare? I saw no mention of this. For example, in eastern Washington small mammals there were three species that were detected only once, yet these were included in the analyses of species richness.

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
23	1		Measures of species richness (Alpha diversity) and diversity are not appropriate metrics to use in assessing the impacts of management actions. Managers do not manage for species richness (or at least they shouldn‘t) and the number tells you nothing about how community composition has changed. Morista‘s coefficient of similarity is appropriate and much more meaningful for managers than analyses of species richness. However, discussions of these coefficients should be immediately followed by describing how the communities have changed.I see the Morisita’s index as a first step to test whether there are differences in the communities from before to after the treatment. If the communities are highly similar, then presumably a manager would not need to be worried about the effects of the treatment. If the communities are dissimilar then the next step is to do a repeated measures ANOVA on individual species to determine which species have declined relative to the controls. Some of the differences in the index may be because some species have increased on the treatments or new species may have moved into the logged areas, so it is important to highlight which species have declined. I could not see any repeated measures ANOVA’s reported, only a bunch of post-hoc tests (although it says it was done in the methods) and this analysis should be done or if it was done, then the overall results of the analysis reported. Perhaps I have missed this, but the report is not clearly written so it may be embedded in there somewhere. A stripped down final report should include the similarity index, repeated measures ANOVA’s on single species, appendices with average abundance for each species on each treatment and year, power analysis (see below) and the co-inertia analyses if you want to know about habitat associations (but treatment or buffer width should be included as a factor).

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
24	1		Co-inertia analysis was used to examine relationships of species with habitat at sites/treatments. I am not familiar with this but it seems similar to CCA, which I have some familiarity with and the description was clear. It seems to me that these ordinations would be more useful if treatment was included as a variable (or at the very least actual buffer width), otherwise they are simply descriptions of species/habitat relationships. I guess this is useful to explain the changes in communities across treatments if vegetation variables used change across treatments.
25	1		Why was a technique such as Non-metric Multidimensional Scaling not used to compare bird communities across treatment types and controls- this can be followed by a Multiple Response and Permutation Procedure which allows you to test whether the communities are different at different treatments.
26	1		For a BACI design, one would analyze the data using some sort of repeated measures design. Although the authors state that this was done, what was presented was a very long series of post-hoc comparisons, most of which did not appear to control for the pre-harvest conditions (e.g. data from buffer treatments 2 yrs post-harvest were compared to controls 2-yr post-harvest without controlling for pre-harvest conditions (unless I am missing something here)).
27	1		For Eastern Washington it was not clear how a repeated measures ANOVA was used if pre-harvest data were not available. Thus, in eastern Washington, any differences between controls and treatments attributed to treatment may be spurious if there were differences in the habitats of the three groups prior to harvest. A regression analysis should have been used to look at whether abundance varied with vegetation parameters that are changed by harvest.
28	1	Table 4-12	Alpha was set at 0.10, but in most analyses 0.05 was used. Throughout I had some problems with conclusions based on whether a comparison was statistically significant at alpha 0.05. Significance is related to variance and sample size, hence more significant results could be found for common species than less common ones. Basically, it is the magnitude of change that is more important to interpret results.

Comment #	REVIEWER #	Comment location Page/ Paragraph/ Sentence	ISPR COMMENT
29	1	Table 4-14	In the small mammal chapter, an attempt to discuss changes after logging using an effect size was used, which is more appropriate. However, it is not clear why a 10% changes was chosen- how does this relate to the population dynamics of these species? Given the sample size and variance of the data, what is the power of the statistical tests to find a 10% change in abundance?
30	1		There should be much more discussion of the power of the statistical tests to detect ecologically meaningful change in wildlife abundance. Given the high level of variability in abundance of many species over time and space, any conclusions for less common or highly variable species based on treatment are probably spurious. It would be very difficult to attain the level of replication necessary to achieve suitable statistical power to detect real differences. Making a type II error can have disastrous implications for conserving wildlife populations. Yes, I would recommend deleting the less common or highly variable species from analyses on changes in abundance with treatment over time, but the means and 95% confidence intervals for those species should be presented in the report so that the reader can see the full range of species that were present and their variability.

Comment #	REVIEWER #	Comment location Page/ Paragraph/ Sentence	ISPR COMMENT
30a	1	This is a continuation of comment #30. The first sentence is a clarifying question from LWAG and the rest of the text is the response from the reviewer.	The reviewer needs to provide a fundamental reality check if the level of replication, which in this kind of ecological study, will never be large enough to detect “real” differences, it is worthwhile to conduct the study. Yes, this is a very real problem and one that we grapple with all the time. As a first step, ecologists should conduct a power analysis to determine whether or not the level of replication will allow them to detect statistical differences in abundance. Previously collected data can be used to get an idea of the expected abundance of each species on control areas and the variation between sites. Now comes the hard part: you have to decide how much change is biologically relevant. Will a 10% decline in a species be a problem, for example? For a very abundant species, a 10% change may not be that important, but for a rare species it could be very important. Deciding this is extremely difficult for many species because we don’t have the population data to decide what level of decline is important. One very large-scale program that is monitoring species over all of Alberta has gone through a number of exercises to come up with this %change figure and to ensure that they sample enough to be able to detect it. See http://www.abmi.ca/abmi/home/home.jsp for more information on their approach. We have to conduct studies to determine the effect of our actions, but I would argue it is probably not worth it to conduct a study with very low statistical power to detect differences in abundances. This is a weakness for many studies reported in the literature: they were powerful enough to detect changes in fairly common species, but not for rare ones. Unfortunately it is the rare ones we may have to be more concerned about. I would argue that we need to pool resources to conduct experiments which much higher levels of replication.

Comment #	REVIEWER #	Comment location Page/ Paragraph/ Sentence	ISPR COMMENT
30b	1	This is a continuation of comment #30 and 30a. The first sentence is a clarifying question from LWAG and the rest of the text is the response from the reviewer.	What is the reality of "real" differences? Are we stuck with a study where one cannot obtain confident conclusions with even the most common and least variable species? We need clarification on the substance of this comment. I think the level of replication in this study is reasonable for the common and less variable species, but not for the rare and more variable species. The point is to be clear about this in presentation of the results so that people know which species they can be confident in. The authors could, for example, use the data they have and conduct A post-hoc power analysis, NOT USING THE EFFECT SIZE (MAGNITUDE OF CHANGE IN ABUNDANCE) FOUND IN THE STUDY BUT USING A BIOLOGICALLY DERIVED EFFECT SIZE OR, IF THAT IS NOT POSSIBLE, ASKING WHAT LEVEL OF REPLICATION WOULD BE REQUIRED TO DETECT A 10% DECLINE, A 20% DECLINE, A 50% DECLINE ETC. If, for example, the level of replication would only detect an 80% decline in a rare species, then clearly there was not enough replication in the study.
31	1		The analysis of how the treatments differed in terms of vegetation should have been done up front and this should have guided the analyses that were done.

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
32	1		The other thing that should have guided the analyses and the whole structure of the report is what the management questions were or should be. For example, the effects of logging was assessed by comparing control uplands to logged uplands pre and post cut. This seems reasonable, but it is obvious that logging will affect the community- what managers want to know is how much logging will affect abundance of different species. In addition to knowing if narrow or wide buffers are better, managers want to know how wide a buffer should be. These are regression questions not ANOVA questions. What I am getting at here is the analyses should be driven by questions that managers need to know the answers for: what, where and how much should I leave in order to least impact wildlife? Streamlining these questions would reduce the number of analyses and make the report much easier to read and use. Instead of long lists of which treatments or controls had the highest species richness or abundance, the results should be structured by management questions based on predictions of what the effects should be. E.g. Does increasing tree removal decrease amphibian abundance? The prediction is controls should have higher abundance than modified than state sites, or if a regression approach is used, that amphibian abundance should increase with increasing forest cover. Do the data support this prediction?
33	2		Overall, I was satisfied with the research approach, methods, and statistics. Exceptions were detailed in the review of each chapter. The absence of drift fences associated with pitfall traps brings into question any amphibian results coming from Chapter 5.
34	2		I am unfamiliar with Co-inertia' analyses, I have never seen its use for this type of study, and question why it was used or why its better than multiple regression modeling with AIC model selection.
35	2	Pg. 3-25	HSD Alpha= 0.05. Why 0.05 when you used 0.10 for your other tests. Clarify.
36	2	Pg. 4-21	Need statistical values, dfs for each statement of a difference or equality.
37	2	Pg. 4-22	Why a t-test when your design was an ANOVA? Why alpha = 0.05?

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
38	2	Pg. 4-22	Need complete stats for each test.
39	2	Pg. 4-24. Table 4-9.	need to display dfs and t-values for each test for all species.
40	2	Pg. 5-9	How do you know going from 1 check per week to 2 per week doesn't affect captures?
41	2	Pg. 5-22	If you present 2 tests, you need 2 sets of statistical values. Also, the dfs don't add up to df=15. Four weather stations x 6 years = 24; unless you tested the interaction 3 dfs x 5dfs=15dfs. Clarify.
42	2	Pg. 5-23	Test of treatments, habitats, and time intervals requires 3 tests, you only report one. Clarify.
43	2	Ch. 5	I suspect that the lack of drift fences associated with the pitfall traps is a major design flaw in this amphibian study. I suggest that any analyses or conclusions are very tentative, at best. This chapter may have to be deleted.
44	2	Pg. 5-29	Need stats with p-values
45	2	Pg. 5-29	Test stats neede for all statement of difference or equality.
46	2	Pg. 5-31	Not sure what was tested here, years? Time interval 1 vs. 2 and 2 vs. 3? F-test or t-test? Why?
47	2	Pg. 5-61	Drift fences would have increased sample sizes and helped detect rarer species in all sample intervals.
48	2	Pg. 8-19	Add dfs to t-test statistics.
49	2	Pg. 8-21	Stats neede here; split-plot ANOVA?
50	3	Pg. 3-4	"Upland sampling occurred 100 m upslope from terrestrial riparian habitat". This is too close to the riparian area for many species. See previous work in Oregon.

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
51	3		"To estimate canopy cover we used a spherical, convex densiometer at the outer corners". Spherical densiometers can give biased estimates of cover in areas where the height of the overstory varies from one plot (e.g., alder) to another e.g., douglas-fir). See: Cook, J.G., Stutzman, T.W., Bowers, C.W., Brenner, K.A., and Irwin, L.L. 1995. Sherical densiometers produce biased estimates of forest canopy cover. Wildl. Soc. Bull. 23:711-717.
52	3	Pg. 4-7	"At each site, four transects (each 400m long) were established - two in the riparian and two in the adjacent upland area (one of each on either side of the stream) - with the upland transects located approximately 100m upslope of the riparian transects (Figure 4-2)". 100 m is too close to the stream to be called upslope for some species. See Oregon studies.
53	4		Although it would have been better to have two or more years of calibration data to assess natural variability in populations within the sampled stands, the data available are still useful. The authors incorporated the data provided to them (pre-treatment) with the data they collected (post-treatment) in a defensible manner.
54	4		The replicated BACI analysis are designed for detecting changes in means; they do not account for harvesting-induced change in variance and co-variation between treatments and control sites over time. Repeating BACI designs are insufficient to separate natural variation from treatment effect. In response to this problem, Underwood (1991) suggested "asymmetrical design" to assess natural variability over several control sites and treatments. While the authors did not employ "asymmetrical design", the random assignments of the selected sites to the three treatments mitigate the influence of random variation.

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
55	4		BACI ignores variation caused by differences between units in the trajectories through time as well as possible serial correlation of errors. If the spatial dependence and serial correlation of errors are not considered, the p-values might be depressed, leading to Type 1 errors. For example Maurthagh (2002) states that "correction for first-order serial autocorrelation in the time series of betwee-unit differences reduced these rejection frequencies to 15% and 28% respectively". Thus, the BACI design may not reveal the true impact associated RMZs retention and forest harvest activities.
56	4		The high variation within and among RMZs contributes to the difficulty of evaluating the impacts of forest harvest on wildlife habitats and on determining appropriate buffer prescription. It is not clear if the coefficients of variation (CV) of the relative abundance values obtained for the pre-treatment and post-treatment for the control sites were equal. If these CVs are not equal, the hypothesis may not adequately test the equality of percent changes among treatments. Recognizing this problem, examination of the "two-stage intervention analysis" developed for repeated measurements might discover important trends (Murtaugh 2000).
57	4		While the study failed to include examination or assessment of spatial and temporal variances, the methods employed to examine the efficacy of buffer retention strategies and forest harvesting impacts on abundacne and diversity of wildlife and their habitat are generally defensible.
58	4		Using "split-plot-repeated-measures design", the study addressed interesting, important and timely-questions. One of the distinguishing features of this design is that the levels of one factor are randomly assigned to main plots, and variation among split plots is less than among main plots. The text fails to clarify whether the 36 Westside and Eastside sites (18 for each) were randomly distributed among the three treatments (control, state, and modified state). Neither does the text document whether the variation among split plots was less than among main plots.

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
59	4		The RMZs data sets are highly variable and hierarchical. In light of this, the use of mixed effects model (PROX MIXED) that considers the hierarchical structure of data is appropriate, but the analysis would have been strengthened by using a generalized linear mixed model (GLIMIX) via the PROC GLIMIX or %GLIMMIX SAS macro. The GLIMIX approach accounts for the high variation of the RMZs data and considers the counts (measurements) made on each transect or plot as a unit with its own mean and variance, and provides a straight forward interpretation of the relative importance of differenct variables. Because of this, it is unclear why the MIXED procedure was used instead of the GLIMIX procedure. More details on the raionale for using the MiXED procedure and the error structures and assumptions made in using this procedure are warranted.Without examining the data or without conducting a re-sampling study, it would be difficult to suggest a specific error structure (error correlation structure) for the riparian management zone (RMZ) data set. For repeated measure analyses, commonly used error correlation structure include: compound symmetry, autoregressive, Huynh-Feldt, Toeplitz, variance components, and unstructured structures. Below are some references that provide guidance to select a specific error correlation structure for the RMZ data set. See references on original comments.

Comment #	REVIEWER #	Comment location Page/ Paragraph/ Sentence	ISPR COMMENT
60	4		The relative abundance values obtained from the different treatments were compared using log-transformed means (log[x+1]). Given that the logarithm of zero is undefined, it is customary to add a constant c to each data value before transforming. The constant c is chosen to be between 0 and 1. Given that the constant directly affects the computed significance level (McDonald 2000, pg. 273), the study failed to conduct uncertainty analysis using different constant values (e.g., 0.01, 0.167, .5, and 1) and examine the impact of adding one to the count data. On Page 39, Technical issue 1.5, the uncertainty analysis recommended is similar to those reported in Table 3 of McDonald et al. (2000). Given that the constant directly affects the computed significance level, it necessary to report the significance level obtained by changing the constant added to each data value. For details on such comparisons, please see McDonald et al. (2000). References: McDonald, T. L., Erickson, W. P., and McDonald, L.L. 2000. Analysis of count data from before-after control-impact studies. Journal of Agricultural, Biological, and Environmental Statistics. 5: 262-279.
61	4		It is possible that there are no differences in the abundance and diversity of wildlife associated with different treatments and buffer widths. Alternatively, the apparent similarities may reflect that the RMZs are internally highly variable. To verify this possibility, I suggest that the range in variation be assessed. If internal variation is not high, then the lack of a significant treatment effect may indicate that the data for the initial calibration were inadequate. This is particularly true for the Eastside data.
62	4		Lacking from the report are insights as to how some of the shortcomings in BACI designs are addressed or accommodated to evaluate the efficacy of treatments in RMZs. In addition, statistical models used for the analysis of the repeated measure RMZs data need to be specified. For the part of the study that involves count data, further analysis using recommended models is warranted.

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
63	5	Ch. 3	Where are the actual statistical analyses so we can see how d.f. were broken down? The only place degrees of freedom are mentioned is to suggest there are nearly 1000 d.f. which cannot be correct, and doesn't indicate how they broke down the data set. Unit of replication is a site (and maybe transect distance) - they appear to have used transects (both as seperate units), or maybe every date and point (pg. 3-24) given an F-value with 17,974 d.f.! i would expect that sampling date within pairs of years (two years were paired) would be a repeated measure.
64	5	Ch. 3	There were two pairs of transects at a site, but surely those would have been averaged. I would expect at mzximum a sample size of 108 (3 year pairs, 18 sites, upland and riparian-although those are split plots aren't they?).
65	5	Ch. 3, Pg. 3-32	The F-value on pg. 3-32 (d.f.=17,90) seems correct.
66	5	Ch. 3, Figure 3-10	Figure includes alpha at 0.05 after having stated alpha was going to be 0.1 (and again in Table 3-7, Figure 3-16).
67	5	Ch. 3	The captions on these tables and figures should give some indication of the degrees of freedom in their various contrasts and full models.
68	5	Ch. 3, Table 3-7	A lot of probabilities of zero (not possible) in this table - deserves a note in the table caption. Making clear how the numbers of samples are included is critical to making this analysis transparent.
69	5	Ch. 4	As in chapter 3 the data were checked for normality, but there is no statement that the data distributions actually conformed to that assumption.
70	5	Ch. 4	Set alpha = 0.1, beta = 0.8, with Bonferonni corrections.

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
71	5	Ch. 4	Same as chapter 3 - after saying that alpha was going to be 0.1, then the tables (e.g. 4-9) and text (e.g. Pg. 4-22, 4-28) indicate it is 0.05, and there is no indication of how the Bonferonni correction was applied. Of course, probabilities can't be zero as indicated in Table 4-12.
72	5	Ch. 5	Confusion about statistics again - why are there only 1,4 d.f. (pg. 5-23)? There are 3 factors listed on the line above and two of those would each have 2 d.f. in the numerator. Likewise on pg. 5-26 the tests described have 2,33 d.f. (despite apparently controlling for transect, which would mean an additional d.f. lost in the full model and a denominator of 32) and 2,31 (again controlling for transect, which might mean a denominator of 32, being 36-1 for total d.f., therefore 35-2 (treatment) -1 (transect) for 32 d.f.). As much as I dislike full ANOVA tables, it might make the results more transparent. These small inconsistencies diminish confidence in the work.
73	5	Ch. 5	Likewise, the denominator d.f. for all the tests of species by treatments seem to be inordinately large, again making one wish that there was more detail about what was being considered a unit of replication in each part of the results (e.g. Pg. 5-33).
74	5	Ch. 4 and Ch. 5	Chapter 4 and 5 appears to use Sattherwaite (name?) approximations for d.f. (e.g. 4-27, 28) but those were not used in other chapters and not mentioned in the methods. This is not an unusual thing to do with mixed models but it does deserve mention for its use, and for why it is used in this chapter but not for birds.
75	5	Ch. 7	Set alpha = 0.1, beta = 0.8, with Bonferonni corrections.
76	5	Ch. 7	Use of Welsh' ANOVA is fine (why wait until chapter 7 to mention it?). However, the unit of replication is still not obvious when there are over 182 d.f. in the denominator. Given 17 sites and 2 transects per site would at most give 34 total n (33 d.f.)

Comment #	REVIEWER #	Comment location Page/ Paragraph/ Sentence	ISPR COMMENT
77	Summary		The statistical analyses were conceptually appropriate, but there were many difficulties with their use, presentation and interepretation. The design should be a BACI design based on mixed model analysis, with repeated measures (multiple sampling periods). In general, that is what is used. However, one can have no confidence in the actual numbers produced by way of probabilities and significance if we can't trust the degrees of freedom and other reporting. Despite the study being of a common design across taxa, the degrees of freedom vary extremely for the same comparisons between chapters. The various contrasts might be appropriate but we never get enough detail on the partitioning of degrees of freedom to know what the effect and error terms are based on.

RMZ Re-sample Final Report ISPR
Is there sufficient detail in the document to reproduce the results?

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
1	1	Pg. 3-6	This was a weakness of the report. I found that the study site description was quite sparse and I could not visualize what the sites looked like, what the variability in the vegetation and width of riparian zones was, what the actual buffer treatments actually looked like, where the trees were left or how exactly the upland was logged. There is no way I could go out and try to lay out this design based on the description in the results.
2	1		The description of sites could be improved by describing the vegetation, including some pictures of the sites that capture the variability in the riparian zones and the treatments. This should be in the chapters and in the executive summary, since the summary may be all some people will read.
3	1		Control sites were continuous stands of maturing second growth conifer: what was the species composition of trees, were these sites next to streams (presumably, but this was not stated)?
4	1		For the State sites the treatment had a narrow (how narrow?) forested buffer retained in the riparian zone. How could this be when the riparian zone was described as unforested? I have no idea from this description as to how the trees were left- in a strip? In groups? Is each site within a single stand? This was a bit clearer in Chapter 7, but not in Chapter 3.
5	1		Modified sites- same questions as for state sites- how wide, how were trees left?
6	1		It is important also to clearly describe how the upland was harvested and what the variability was across treatments. An analysis should have presented up front to show that the treatments clustered out separately based on the vegetation in the riparian and upland areas.

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
7	2		Overall, the level of detail was adequate, however, a description of the actual logging and cut areas was absent.
8	4		More information is required concerning the statistical and log-linear models used in examining species abundance and diversity. It should be clarified whether a log-transformed multiplicative model or a log-transformed additive model was used in analyzing the data, and whether the model used to assess variation was proportional or additive.
9	4		In the analysis of repeated measurements, it is unclear which variables were random and which ones were fixed, and what assumptions were made with regard to the error distribution. For example, site is treated as a random variable, which in turn provides some insights on the influence of site in comparing the efficacy of different treatments on the abundance and diversity of wildlife, particularly on terrestrial amphibians (e.g., coastal salamander and Ensatina) at Westside. log γijk = $\mu+\tau_{i\,+}$ S_{ik} + α̇, + τα̇ij + εijk. Where γijk is relative abundance (e.g. number of time each bird species was detected, or number of small mammals caught/100 trrap nights, or mean number of amphibians caught/100 night) in the ith treatment group and counted during the jth survey at the kth site, μ is the overall mean, τ is the treatment effect, Sik is the random site error effect within treatment group,α̇j, is the effect of jth year effect, τα̇ij is the treatment-year interaction effect, and εijk is the random experimental error on repeated measurements through time. The Sik are assumed to be independently and identically distributed normal random variables with mean zero and variance σ^2 w . εijk is assumed to be independently and identically distributed normal random variables with mean zero and variance σ2e.
10	5		A phrase describing the sampling method for birds should be included.

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
11	Summary		In general the assumption in this report and elsewhere is that there is no significant difference from some alternative management action, in this case the controls. That also demands that a power analysis for the ability of the statistical design to detect a difference if one exists. The design here may have sufficient statistical power, but it is never indicated. The definition of efficacy (e.g. Pg. 3-4) needs to be more explicitly stated so the results can be compared against that definition. One cannot judge the trade-off between efficiency of conservation against the economic opportunity cost without such an explicit criterion.
12	Summary		The precise GPS co-ordinates of sites should be provided in some kind of permanent archive.

RMZ Re-sample Final Report
ISPR review

Where the data reasonably interpreted?

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
1	1		I found it a bit difficult to make my way through the results because there were so many different analyses, descriptions, tables and figures. Many of these could be deleted with no loss of information. I am assuming that the users of this report will be forest managers and regulators. As such, the report should be user-friendly so that it is easy to flip through and find the pertinent information. As mentioned above, the results should be structured by the pertinent management questions- by the end of the results I wasn‘t sure what the main messages were vis a vis the management questions.
2	1	Figure 3-7	Fig 3.7 is not very useful as data are lumped into groups by type of species, and flyovers, waterbirds etc. are included even though they may not be affected by treatments. No stats are done and data are pooled within treatments, so any apparent differences could be misleading.
3	1	Figure 3-8	Fig 3.8 not useful as there are too many categories of how data were pooled. Why are 3 different sets of data, obtained using different filters, presented? Which data set is useful for managers? There are biases due to unequal sampling effort among sites etc, so basically one shouldn‘t make inferences from this figure. What does all this information mean? Does it tell us anything we won‘t get from later analyses?
4	1	Figure 3-9	Fig 3.9- it would be easier for the reader to make comparisons if the data were presented grouped by sampling interval within the same treatment. This is done in some figures and not others.
5	1	Figure 3-24	On 3-24 it says that mean detection rates changed modestly over time on controls- was this significant or not and what was the pattern?

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
6	1	Figure 3-10	From figure 3.10, it seems that these changes are not significant, but it takes the reader a long time to figure this out. The author has to interpret the data better for the reader. Not clear to me what the * means on fig 3-10. The way the data are presented, it is unclear to me whether comparisons of change in the treatments over time are ―controlled‖ by the changes in the control plots over time. It seems from the post-hoc comparisons that they have not been. Abundance on all of the treatments, except the state riparian, appear to go down from pre to post1 and then rise in post2. The important point is whether the changes on the treatments are of larger magnitude than those on the control. Overall, what are the conclusions relative to management of this analysis?
7	1		One might argue that abundance of all species combined is not that informative and that it is the change in individual species that is important, especially since many of these did not follow the changes of the pooled species. Thus, what is the point of presenting the combined data?
8	1	Pg 3-26	The results on page 3-26 could be streamlined as outlined above. What are the relevant management questions and what does the analysis tell us about these questions? Structure the results that way instead of by the statistical comparisons, since the reader must sit and try and work out what the question is for each comparison. See my comments above about the relevance of the comparison when data are pooled and the control sites are not used to correct for changes.
9	1	Pg 3-28	first paragraph: not clear why five significant changes was chosen as a cut-off (see above for discussion of statistical power and effect sizes).
10	1	Figure 3-11 and 3-12	It doesn‘t seem useful to me to see how many changes were significant. It is the magnitude of the change relative to the controls and the species of concern (is a change in robin abundance as important as a change in brown creeper abundance?).
11	1	Figure 3-13	Are these changes important relative to changes on the controls? This is a very complicated diagram. The authors should make this easy for the reader.

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
12	1	Pg. 3-34	Morista's index: why was this level of the filter used as opposed to the most stringent filter? This analysis is useful but should be combined with the co-inertia ordination to explain how the communities have changed.
13	1	Pg. 3-35	A lot of the description of the co-inertia analysis was already said in the methods. What is the function of fig 3-18? If it is to show habitat relationships in natural riparian habitats, then why were all riparian habitats and all time periods combined? Ditto for fig 3-19. What is the relevance of this figure to treatment effects?
14	1	Figure 3-20	What is the relevance of fig 3-20 to the management questions? Why isn't buffer width included as an environmental variable?
15	1		Why are new data analyses presented in the discussion? These should be in the results, especially since these establish what differences were across sites before the harvest. Unfortunately mean habitat conditions within treatment categories were presented so we cannot assess what the variability across sites was.
16	1		There were a number of places where conclusions were made that were not supported by the data, due to lack of reference to pre-treatment controls sites, due to a lack of difference between the vegetation of the treatments and due to the high spatial and temporal variability of the wildlife. Some of this was acknowledged by the authors in some places, but in other places they appeared to ignore it. I outline some of this lack of support below.
17	2		Overall, the data were reasonably interpreted, however, Chapter 5 on amphibians is over-interpreted based on the extremely low captures.
18	2	Pg. 3-13	Is criteria for "common" or "abundant" species when the detection of species is > 15 times? Clarify.
19	2	Pg. 3-17	Abundant and common species not defined
20	2	3-26	When there is a statistical difference, just say its higher or lower, up front, not both.

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
21	2	Pg. 3-14	No criteria for "most common", clarify.
22	2	Pg 3-36-38	Because of the extensive data sets and many species, some meta-analyses may have been beneficial. It might be useful to consider using "relative effect size" as a method to sort responses to treatments relative to controls and yearly changes. One method is to calculate % effect by subtracting "state" and "modified" treatment responses from the control response and multiplying by 100 for each site and sampling interval. % effect would then be displayed with pluses or minuses. it may also be a way to summarize species differences, that is, early successional or edge species will likely respond positively to cutting whereas late successional/old growth species will respond negatively.
23	2	Pg.3-41	Spatial synchrony as discussed leads to dependence in data, which is why combining data from pointcount stations and/or transects may be necessary and clarified.
24	2	Pg. 4-24	need stats for all statements such as "differ", "higher", "more abundant", "equal".
25	2	Pg. 4-26	Stats needed for each statement.
26	2	Pg. 4-27, Table 4-11	Stats needed
27	2	Pg. 4-60	I suspect mammals can generally be grouped into "disturbance tolerant/early successional" species and "disturbance sensitive/late successional" species to help develop generalities.
28	2	Pg. 5-14	Please clarify how "common" was determined.
29	2	Pg. 5-33	Clarify the statistics in results. It appears that you are doing pairwise t-tests but somehow using an F-test (ANOVA). Please clarify how and why.
30	2	Pg. 5-47	These habitat analyses suggest the "state" and "modified" RMZs were not very different.
31	2	Pg. 7-15, #1	Define "common"; also how were data pooled, upland and riparian combined? Aren't they part of the split-plot?

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
32	2	Pg. 7-15,#1	Sites were not pooled, but used as replicates. Clarify.
33	3	Pg. 3-10	"All sites, regardless of treatment type, were surveyed 4-5 times during one year (1993) of pre-treatment sampling before any forest harvesting occurred at the modified and state buffer sites. Following forest harvesting, most sites were sampled 5-6 times during each of two years (1995/1996)". Since sites were sampled with inconsistend effort, any estimates of species richness or diversity are highly confounded with sampling effort. Sites with greater effort would be more likely to have more species detected than those with less effort. This could be easily standardized by randomly selecting 4 visits on which to base all estimates of richness.
34	3	Pg. 3-23	No error bars;why not? Unequal sampling effort- either remove species richness; or correct number of observations by sampling effort.
35	3	Pg 3-31	"Increase in species richness" - or was the increase in richness associated with an increase in sampling effort from 4-5 to 6 visits? Need to standardize all diversity estimates to a common number of visits per site.
36	3	Pg. 3-50	This seral stage progression is highly idealized - disturbances and stresses modify these stages dramatically. Successional pathways are a more informative approach to understanding stand dynamics. See: figure 2.9 in Spies, T. 1997. Forest stand structure, composition, and function. In Creating a forestry for the 21st Century. Edited by K.A. Kohm and J.F. Franklin. Island Press, Washington, D.C. pg. 11-30 and also Wimberly, M.C., Spies, T.A., Long, C.J., Whitlock, C., 2000. Simulating historical variability in the amount of old forest in the Oregon Coast Range. Conserv. Biol. 14, 167-180.
37	3	Pg. 4-17, Table 4-7	Unequal sampling intensity among times and treatments? Table 4-7 - This will confound and bias estimates of richness and diversity.
38	3	Pg. 4-24	Since sampling was unequal among sites these should at least be corrected for trap nights. Raw numbers of captures are not useful unless sampling intensity was consistent.

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
39	3	Pg. 4-37	Effect of unequal sampling effort on species richness estimates?
40	3	Pg. 4-41	Diversity indices are really meaningless - stick with community similarity. - See pg. 4-52. This is addressed in later chapters but two areas can have identical species diversity and entirely different communities.
41	3	Pg. 6-1	Chapter 6 seems to focus on amphibians - what about other species? Birds, mammals (including bats), plants, invertebrates? How can the observed patterns be used to conceptualize conservation of biodiversity within watersheds?
42	3	Pg. 7-9	"Particular attention was paid to detections within a 30m - wide strip along each transect, because those data formed the basis of most analyses: specifically, detections in the band of 0-8m streamside of the riparian transects (0-8R; Table 7-4); those in the band of 0-22m upland of the riparian transects (0-22R; Table 7-4); and those in the 0-15m band along either side of the upland transects (0-15U; Table 7-4). Birds within any of these three distances and not counted as flyovers are referred to as "on-transect". - Simply using distance based estimators of abundance is much more powerful and at least partially overcomes assumptions about detectability and biases associated with observers influencing bird behavior.
43	3	Pg. 7-15	"We used detection rate as a measure of relative abundance". Why wasn't distance estimated?
44	5	Ch. 3, Figure 3-22	Moriseta's analysis is a good visualization of the community results. The modified riparian changed less in composition than the state regulation sites, and the modified treatment was cery close to the controls. However, there is evidence for differences between riparian areas prior to harvest, e.g. Figure 3-22. Magnitude of the differences between treatment sites was greater than the differences before and after upslope harvesting in the cut sites, according to the bi-plot in Figure 3-22. It's hard to reconcile those differences given that they represent a predominant site effect overwhelming the treatment impacts.

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
45	5	Ch. 5, figure 5-22	There is an odd interpretation of this figure suggesting that the two riparian treatments were not really different from each other (Pg. 5-47) which throws the rest of the report and comparisons into a very different light. It seems that the two riparian treatments are as different from each other in the state space generated as they are from the controls (before and after harvesting), which could be site-specific differences which existed prior to harvesting. That these are artifacts of the riparian treatments has not been established, especially given that the abstract of this chapter seems to suggest 14 m buffers are not as good as 30 m buffers, by which one would expect differences in the treatments.
46	Summary		In general the data were reasonably interpreted. However, there are insufficient details for some aspects to properly judge the results and so the interpretation is a little harder to evaluate.The patterns in the data can be largely interpreted graphically, and don’t always depend on a probability value. In that sense the data seem appropriately interpreted. An over-reliance on statistical probabilities will not yield all the biological insights one should gain. Whether the statistics are correct or not (perhaps they are, but there are questions), the results are described in a narrative form that makes sense. One might expect that the degrees of freedom for the statistical model might be similar between taxonomic groups within a region (east versus west), but that is not the case (they vary from 17 to >1000 across chapters). For most statistical tests the actual error terms used are not evident from this document. Why do these models differ by chapter and by the particular contrast being made within a chapter? It is not obvious why there are so many ordinations done in some chapters, as all the data should be in a single ordination per chapter. Doing separate ordinations by subsets of the data results in different co-inertia matrices between analyses, even though there is one set of data.

RMZ Re-sample Final Report ISPR
Do the stated conclusions logically flow from the results?

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
1	1	Bird chapters	There is some speculation in the discussions that are not based on data presented, particularly in the bird chapter. For example the role of conditions on the study site (e.g. food) on variability in abundance of birds is not relevant nor was it tested and yet the authors conclude it was probably more important than larger scale factors.
2	1	Pg 3-43	I would like to see stronger and clearer conclusions stated. For example, page 3-43, second paragraph, the results are reiterated, but the conclusion is not clear. Is the conclusion that abundance over time was not related to treatment, but simply reflected regional changes in bird abundance? In the next paragraph, it is not clear whether these changes were above and beyond changes seen on the controls. The influence of regional changes were dealt with better in the discussion of individual species.
3	1	Pg 3-49, 7 lines from bottom	Conclusion that width of buffer did not influence community in the upland because communities at uplands at modified and state buffer sites were equally dissimilar does not follow. They could be dissimilar for different reasons.
4	1	Pg 3-50	Conclusion that modified riparian better than state riparian at maintaining pre-harvest community structure over the course of the study, based on fig 3-17 this appears to be true but how different ecologically is a C value of 0.85 from 0.7?
5	1	Pg. 7-33	Conclusion that forest harvesting lead to increased bird abundance not valid since pre-harvest abundances not taken into account and site differences could exist.

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
6	1	Pg. 5-50	The author correctly notes that annual and spatial variability in amphibian numbers makes it difficult to interpret both significant and non-significant results (5-50), however, he then goes on to suggest that Coastal Giant Salamanders were adversely affected on state riparian sites but went up on modified sites. He concludes that modified sites are better than state sites, however, CGS's also went down on control riparian sites (Fig. 5.15), so this conclusion is questionable.
7	1	Pg. 5-53	He concludes on page 5-53 that either 14 m or 30 m wide buffers function to preserve diversity and abundance. This is based on statistical non-significance of results for data with low sample size and high variance, thus power of statistical tests were probably quite low. Essentially, I would be very cautious about making interpretations that the buffers were preserving amphibians. Conclusions of no effect of buffer type for some groups are suspect and may be a type II error because of the small sample sizes and high variance of the data. Statistical power was low.
8	1		The sections in each chapter on succession and alpha diversity were interesting, but didn't really flow from the results.
9	1	Conclusion and management implications sections	These sections were disappointing. I would have liked to see a numbered list of the major conclusions of each section of the study and how each conclusion applied to management and policy decisions that have to be made in the Pacific northwest. This was not done and I think this report would leave most readers hanging over what the management implications are. If buffer management is becoming more site-specific, then what conclusions from this study will help managers manage for biodiversity at each site?

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
10	1	Conclusion and management implications sections	In particular I found the overall management conclusions sections (Chpts 6 and 9) to be very generic. Here was an opportunity to incorporate results from all of the taxonomic groups and come up with management scenarios that would be beneficial for all species that showed declines in logged buffers and uplands. There is a discussion of the ―best buffer widths found in other studies (e.g. page 6-1), and a conclusion that a 30m buffer is sufficient in western Washington for some taxonomic groups. I would argue that that is not a valid conclusion since buffers larger than this were not included and the vegetation on modified and state buffers was not appreciably different. If managers must manage site by site, how do they do this across a large landscape? Where can narrower buffers go? Where should larger buffers go? How much cutting in the buffer should be allowed? How much cutting in the upland should be allowed? Again, managers want to know what,where, and how much.
11	2	Pg. 3-31	If a decline is not statistically significant, it is not different. This comment pertains generally to whether you can say a response "declines" or "increases" without statistical support. So, when the report says (last 2 lines of P3-31) "a decline in alpha diversity (i.e. between the pre- and first post-harvest sampling intervals)." this needs to be supported by a significant test and not just a lower average value. I can't recall if the test was presented or not. If it was presented, I must have missed it.
12	2	Pg.3-49	Diversity and Similarity: Treatment effects are either significant or not, unclear writing in lines 3 and lines 9-11.
13	2	Pg. 4-37	I disagree about the variability, I don't see it.

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
14	2	Pg. 4-59	Be careful to extend a lack of RMZ treatment differences only to your choice of buffer widths (which I still don't know) and 40-60% retention cuts, not to all other buffer widths or types of logging.
15	2	Pg. 5-59	Very limited data on amphibians excludes any conclusions about timber harvest or RMZ treatments.
16	2	Pg. 8-21	Were there species missing from or uncommon in the data that may have been present in older growth control forests? Conclusions need to reflect this possibility and that they only apply to equivalent control forest stage/type. Also reslts only apply to timber harvests of a similar type, 40-60% partial retention, and adjacent RMZs. Results would probably have been quite different if forests adjacent to RMZs were clearcut.
17	3		Generally the stated conclusions logically flow from the results except for diversity metrics which are quite likely biased.
18	3		The power associated with the analyses to detect certain effect sizes should have been reported. I expect that at least in some cases the conslusions that there were no significant differences between certain treatments was as much due to lower power of detecting a difference as a true indication of the samples representing the same populations. Along those lines I was surprised that AIC information theoretic approaches were not used to identify the most parsimonious model structure. See: Anderson, D.R. et al (2000) Null hypothesis testing; problems, prevalence, and an alternative. J. Wildl. Manage. 64, 912-923.

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
19	3	Pg. 7-21, Table 7-11	Mean relative abundance (1 SD) and ANOVA results for selected species of songbirds detected on..." is this all species combined? Value? What does this tell the reader? The manager?
20	3	Pg. 8-34	No detectable effect...power? Understanding the ability to detect a certain effect size would be of value given the large number of cases were no significant difference was reported.
21	5		Comments about site-specific management and generating biocomplexity are reasonable, but outside the scope of this reassessment exercise. It's also a bit at odds with leaving riparian areas as reserves where structure and composition are not directly managed.
22	Summary		In general the conclusions seem to suggest that the modified treatment was marginally better than the State guidelines, but without better description of the treatments, this is hard to determine.

RMZ Re-sample Final Report
Do the literature citations include the latest applicable information and represent the current state of scientific understanding on this topic?

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
1	1		Generally I found the literature cited to be pertinent and up to date, but it did lean towards studies done in the Pacific Northwest with little reference to similar studies done in Alberta or eastern North America. However, I don't think this really limited the conclusions or scope of the report.
2	2		Very good coverage of the literature.
3	3		Throughout, there seemed to be literature from Oregon and California that was overlooked. I tried to point out some of it in the detailed comments below, but a search of google scholar may turn up some more. I also included an old literature review that was done in the 1990s that may still be of some use.
4	3	Pg. 4-64	Other references? Gomes, D.M. and R.G. Anthony, 1998. Small Mammal Abundacne in Riparian and Upland Areas of Five Seral Stages in Western Oregon. Northwest Science 72:293-302.
5	3	Pg. 5-1	Might want to check Martin, K.J., and B.C. McComb. 2003. Amphibian habitat associations at patch and landscape scales in the central Oregon Coast Range. Journal of Wildlife Management. 67:672-683.
6	3	Pg. 5-2	re: BACI see: Cole, E.C., Mccomb, W.C., Newton, M., Chambers, C.L., Leeming, J.P., 1997. Response of amphibians to clear-cutting, burning, and glyphosate application in the Oregon Coast Range. Journal of Wildlife Management 61, 656-664.

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
7	3	Pg. 5-50	Might want to compare these results to Cole et al., Gomez and Anthony, Martin and McComb, etc. Also "Vertebrates associated with reparian habitats on British Columbia's mainland coast Fred L. Bunnell, Glenn D. Sutherland, Tanya R. Wahbe Centre for Applied Conservation Biology University of British Columbia".There are a number of places on his page (and elsewhere) where the patterns of amphibians detected by Cole et al, Martin, and Gomez relate to watershed patterns including buffer strips. Are your results consistent with those of these others studies in Oregon? If so, then that is greater confirmation that the patterns that you describe are real. If not, then why not?
8	3	Pg. 8-32	See: McComb, W.C., R.G. Anthony,and K. McGarigal. 1991. Differential vulnerability of small mammals and amphibians to two trap types and two trap baits in Pacific Northwest forests. Northwest Science 65:109.
9	3	Pg. 8-32	"As demonstrated in Mackenzie et al. (2004), many ecological studies evaluating the spatio-temporal patterns of species assume equal detection of all species present". - applies to other taxa too.
10	3	Pg. 8-35	Perhaps refer to Martin, K.J., and W.C. McComb. 2002. Small mammal habitat associations at patch and landscape scales in Oregon. Forest Science 48 (2); 255-266.
11	4		Missing from the exposition are citations to recent advances on BACI designs, e.g., Underwood 1991, Underwood 1994, McDonald et al. 2000, Murtaugh 2000, 2002, 2003, and Grant et al. 2008. It would be useful to add some of these citations where appropriate. Otherwise, the report provides a good review of the literature, and cross-references the results of other studies.

RMZ Re-sample Final Report ISPR
Are uncertainties and limitations of the work stated and described adequately?

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
1	1	Ch. 5	There are sections on the limitations of the work, but this could have been enlarged to talk about overall ecological variability and difficulties of teasing out local and regional effects, effects of small sample size etc. For example, in the amphibian chapter, there are many non-significant results, but the numbers of amphibians captured per site is low and between-site variability is high, thus there could be issues with statistical power. There should be some discussion of this.
2	1	Pg. 5-57 and 5	The section on limitations of the study design for amphibian sampling is very thorough and should be used to guide future sampling if that is in the works.
3	2		Overall, I agree the uncertainties and limitations are adequately described. However, there was a lack of discussion about how the overall lack of differences between RMZ treatments may have been due to the lower impact logging (40-60% retention) adjacent to RMZs compared to, for example, clearcutting. This is an important point because just saying state and modified RMZs do the same job of protecting biodiversity opens up the possibility that these regs become universal for ALL logging. They are not. The 40-60% retention logging is a mild cut compared to clearcutting. So, my point is that ecological context of RMZ is likely very important and must be discussed.
4	3		Many of the uncertainties and limitations of the work are stated and described adequately , but they are inconsistently addressed among the chapters.

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
5	3	Pg. 3-5	"Accordingly, the results of the study are applicable to sites meeting the same criteria that were used to select the 18 study sites". Well, only if the 18 study sites were selected randomly and I looked but did not find information on random selection. Unless they were selected randomly then your statistical scope of inference are these 18 sites and only these 18 sites. If you want to assume that they are representative of other sites then you should provide data to support that assumption.
6	4		It is uncertain how treated and undisturbed buffers will change with time, and therefore, whether the observed trends will hold over two or three rotations. The authors do point out some limitations in the design of the study and in the inferences they made. I recommend that a bit more be added, however, to clarify theses uncertainties.
7	4		In using log-transformed means (log[x+1]) for the relative abundance values, it is customary to add a constant c to each data value before transforming. The constant c is chosen to be between 0 and 1. Given that the constant directly affects the computed significance level (McDonald 2000, pg. 273), the study failed to conduct uncertainty analysis using different constant values (e.g., 0.01, 0.167, 0.5, and 1) and to examine the impact of adding one to the count data.

RMZ Re-sample Final Report ISPR
Are assumptions stated and described adequately?

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
1	1	Pg. 3-14	The first assumption is that sites were similar enough that information from one site could be extrapolated across all sites. This is doubtful, given the variability across most landscapes. The pre-treatment vegetation of the sites should have been analyzed first to see whether they clustered in some way or whether each treatment encompassed the variability across the landscape in a similar way.
2	1		The second assumption is that the buffer treatments could be pooled to test for effects of upland logging. Again, I disagree. Birds in the buffer are more likely to be affected by local conditions (found in many studies of local vs landscape effects on community composition) than upland conditions. This also assumes that all logging and replanting (first we have heard about replanting! Replanted to what?) was the same in the uplands. This assumption could also be tested since vegetation was sampled in the uplands.
3	1		In the mammal and amphibian chapters the author states that replacement of sites (e.g. West Fork Falls Creek) is OK because all sites within a region are considered part of the available sampling population. I disagree, because in a repeated measures analysis the same sites must be measured over time. In any ecological study replicates can vary quite a bit which is why it is important to get as many replicates as possible and to try to choose sites to minimize some of that ecological variability.
4	2		Assumptions are stated and decribed very well.
5	3	Pg. 3-13	"We used the number of species found in each habitat type (i.e., riparian or upland) of each site…". With unequal sampling intensity among treatments and years? Assumptions? I do not understand the logic behind these unless you standardize sampling effort.

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
6	4		More details are needed on the assumptions made in using the log-linear and other statistical models to examine the impacts of forest harvest and RMZs retention strategies on the abundance and diversity of wildlife species. Repeated measures data require that the Huynh-Feldt condition (Huynh and Feldt 1970, 1976) holds to make inference using conventional repeated measures approaches. For the Westside with three re-measurements ,the study lacks an assessment whether the Hynh-Feldt condition (sphericity) was held. If the condition is not met, it necessitates conservative adjustments to the usual F-test. However, given that only two sets of measurements (pre-treatment and 10-year post harvest) were used in the analysis, a sphericity test was not critical.
7	5	Ch. 3	Bonferonni-corrected multiple comparisons - where would I find evidence of the corrections having been made?
8	5	Ch. 4	Assumptions spelled out (representativeness of replacement control; reasons for snap-traps to be sprung without a capture)

RMZ Re-sample Final Report

Is the information presented in an accurate, clear, complete, and unbiased manner and in a proper context?

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
1	1		I can't assess the accuracy of the information as I don't know what the controls were over data continuity between observers and between time periods, nor do I know how data were checked.
2	1		The report was unclear in many places, and while complete, was often —over-reported making it difficult to glean the main messages.
3	1		This report could be made much more "user friendly" and should be rewritten to exclude superfluous analyses, figures, and tables and reorganized so that there is not so much redundancy across chapters.
4	1		It would have been much better to have a common introduction and study area description and to join chapters on the same taxonomic groups.
5	1		List major conclusions and clearer, more specific management suggestions.
6	2	Pg. 7-14	How are data pooled; averaged, considered replicates? Clarify.
7	2	Figure 3-11 and 3-12	Unclear whether numbers are additions or subtractions.
8	2	Figure 3-13	Totally unclear.
9	2	Ch. 2	It would have been nice to have a separate chapter following Chapter 2 on the project design and treatment methods. It would also avoid redundancy in each of the bird, mammal, and amphibian chapters.

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
10	2	Pg. 4-20	"Natural variation" means changes in control plots over time, independent of treatments.
11	2	Ch. 4	I find the organization of results complex and difficult to follow in all the chapters I have read so far because of all the comparisons. I suggest partitioning the results into sections that reflect your primary hypotheses or biological effects of interest as major subsections, e.g.: 1. Natural variation - differences in controls among the 3 time periods, 2. Logging effects - Controls vs. state and modified RMZs combined, 3. Buffer effects - state vs. modified RMZs, 4. Habitat effects - upland vs. riparian. I don't believe that partitioning the results would affect the analysis.
12	2	Ch. 4	I think the discussion can be better organized by partitioning it into the disturbance tolerant vs. sensitive species. It is also not clear if the general goal is maximizing biodiversity or maintaining biodiversity at control levels and maintaining sensitive/rare species.
13	3		The context needs work-especially the conceptual framework within which to place the study and the results.
14	3	Pg. 2-2	"Riparian habitats; patchwork or habitats of different ages" - habitat is species specific. Do not confuse the term 'habitat' with forest type, or patch type. A riparian forest is a patch type; it may or may not be habitat for something.
15	3	Pg. 3-22, Figure 3-7	This figure is not particularly useful. I would recommend deleting it. On the previous page you state that over 90% of observations were passerines (a term that should be used instead of songbirds).
16	3	Pg. 4-27, Table 4-11	This table should have ranges or Cis on these estimates.
17	3	Pg. 4-53	Three species of shews - *S. monticolus, S. vagrans, and Sorex* sp. - there are only 2 species here... if one was unidentifiable then it should be *Sorex spp*.

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
18	4		Missing from the description of the experimental units, harvest treatments, stand description, and the difference among treatments and controls is some measure of variation (e.g., coefficients of variation, CV) for different stand attributes including stand structure recorded before and after treatments.
19	4		The report, in places is somewhat tedious, but is generally unambiguous and informative. Although the report could be condensed, it is coherent and logically arranged.
20	4		The title of the report does not precisely reflect its contents. The term "effectiveness" implies RMZs are capable of providing habitat for wildlife regardless of the circumstances. The term "Resampling at the 10-year" implies sub-sampling or a re-sampling method such as Monte Carlo Simulation was conducted. Instead, the experimental units wer re-measured over time. I suggest modifying the title of the report to reflect these facts. One suggestion of a title is "Comparison of Riparian Management Zones Retention Strategies to Maintain favourable Habitat for Wildlife: Re-measurements at the 10-year Post-treatment".
21	5		The report is repetitive. The general goal is repeated 5+ times, design repeated 6 times, etc.
22	5	Ch. 1	Summary seems to have been patched together by different authors and not integrated together - in fact the abstracts were cut and pasted into the summary from each of the chapters.
23	5	Ch. 3	What was the purpose of upland (100m) transects (in clearing, but with different distances to standing trees)?
24	5	Ch. 4	For species richness, how were specimens that were not identified to species dealt with? In Figure 4-4 and 4-18 (These are the same figures shown twice) Peromyscus sp. Was included as a separate taxon, while other genera with unidentified species were not included. It would be useful to know if specimens unidentified to species were left out of richness estimates, etc. as it is not clear how they were dealt with through most of the analysis. For species richness were the estimates rarified?

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
25	5	Ch. 5, Figure	This figure (and others) indicates these are PCA axes. Since PCA did not seem to be the ordination of choice I wonder if these are perhaps not PCA axes. Needs to be confirmed.
26	5	Ch. 7, Table 7-3	In Table 7-3 is the State guidelines widths supposed to be 9 to 15, or is '5' correct? The caption for this table should indicate that these are for some particular class of sites - all of eastern Washington (these are not for the entire state, are they?). Were the widths of the transects really different? It seems that for the upland it was a 30m wide transect, but the riparian one had only 8m to one side, and 22 to the other. Is there any accounting for the differences in detection rates with different band widths?
27	5	Ch. 7	Why would anyone show arcsin-transformed data in a figure? This is completely appropriate for analysis of proportion data, but unneccessary for a figure. A figure is intended to protray the data, and if it needs some transformation to avoid disprportionately showing some values, then sometimes transformation (log or ln) can be useful. Arcsin-transformed data are not helpful.
28	5	Ch. 7, Table 7-14	Table also includes probabilities of zero. Lists alpha as 0.05.

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
29	5		Confusing discussion of “rules”. As in my first point under the summary heading, it is not clear exactly what was applied to these sites, nor is it made clear in the rest of the report. The rules are only very vaguely stated in the summary and suggest that the original 1988 rules were for buffers of 14 m width on average and that the modified buffers were wider. This is inadequate description of the rules or their modification. The “rules” are confusing throughout. There is no single statement of what the actual sites were like after treatment. This particular comment was for the Executive Summary, but applies throughout. The Summary says the comparison was between the 1988 state regulations and a modified version of those, but the modification is never clearly stated, except that there were “… (comparatively wider riparian buffers)…” [page 1.2]. The western WA bird section leads one to think it is simply a change of width, although that width is never specified. The small mammals of western WA section states that “… RMZ rules have evolved to embrace a site-specific approach, …” which leads one to think that there may be some variation among the way sites were treated. The amphibians section categorically specifies widths as 14 m versus 30 m for the state regulations and modified regulations, respectively, but this is the first time this is stated. However, the birds of eastern WA section makes me question whether rules were similar in east and west, which is never stated. The eastern songbirds section mentions “… greater timber removal, the state buffer …” (is this from within the buffer?) and “… logging in the riparian areas…” [p.1.4]. Finally, under the mammals of eastern WA (lines 6-8) it states that “… the buffer treatments were insufficiently different from each other t allow for any meaningful comparison between the two.” That section also talks about variable width buffers. I think you can see why it is confusing, with no single statement of what the prescriptions were meant to be, and statements all the way from specifically 14 m versus 30 m to being no real difference between the riparian treatments. The first full paragraph on page 1.6 talks a bit about the 1988 rules, stating buffers must be ≥7.5 m for both type 3 and 4 streams. The modification is intended to “… provided for greater structural complexity…”. Does that mean increased widths or some additional modifications? However, it also states that the modifications use 3 zones for type 3 streams, and the rules are less complex for type 4 streams – where are we ever told which are type 3 or 4 streams? How are those 3 zones applied to the RMZ and how does that specifically differ by stream type? Is there partial harvest allowed in these 3 zones, and to what levels? Are the rules different in eastern Washington [page 1.7]? The section also states that the modifications allow for “… ability to manage riparian areas based on the conditions of the site, …”, by which one wonders if the lack of difference mentioned for mammals in eastern Washington might have been convergence of treatments because of this. Somewhere there has to be a clear statement of the way the rules were applied, and how they varied between eastern and western Washington, if at all. Nowhere in the document is there an actual description of the site treatments, or a comparison. Hence we are left with one view that the two buffer treatments did not differ through to fixed widths of 14 m or 30 m for the two treatments, respectively. And this is just from the executive summary. The rest of the document does not help to elucidate the differences between treatments across sites.

RMZ Re-sample Final Report
Given the limitations of the study design, are the statistical analyses appropriate?

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
1	1		See comments for CMER question #1.
2	2		I am unfamiliar with "Co-inertia" analyses, I have never seen its use for this type of study, and question why it was used or why its better than multiple regression modeling with AIC model selection.
3	2	Ch.3	A fairly common approach for habitat analyses is multiple regression modeling with AIC model selection criteria. Was this considered? I am totally unfamiliar with the co-inertia analysis used here. This may lead to apparent subjective decisions about what habitat variables are important or an inclusion cutoff.
4	2	Ch. 3	The analyses seemed to focus more on comparing the "state" treatment with the "modified" treatment, or control with "cut" treatment (combined "state" and "modified"), but less on comparing the control vs "state" and control vs "modified" treatments. The 3 pairwise comparisons should be treated equally in each tested data set.
5	3		What I did not see, and expected to, was a conceptual framework that provided the basis for a species-specific response to buffer width. I think that it would be much more useful to relate the patterns observed to a concept that includes the spatial requirements of species - for instance the habitat relationships observed for salamanders likely reflect a situation in which multiple generations can exist within a buffer in multiple home ranges. Alternatively, some bird species associated with reparian areas are either riparian obligates (e.g., dippers) or may have small territories that can be elongated sufficiently to fit in a buffer (e.g., warbling vireo), but there conceivably would be a limit to how narrow a buffer can be before the species is less likely to occur there. What the species is doind in the buffer is also important. Occurrence is one thing; nesting, reproducing, dispersing are more important measures of use.
6	3	Pg. 3-17	Why weren't AIC tests used?

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
7	3	Pg. 3-18	Sampling effort varied from 4-6 visits per site. This confounds estimates of richness and diversity.
8	3		For all analyses - why wasn't an analysis done of the combined stand level effects (riparian and upland data for each site combined) to represent beta diversity? It is after all the watershed biodiversity that will be important to conserve. Not just what is in streamsides.
9	3		The approaches to statistical analyses (testing of null hypotheses) were traditional and the exploratory multivariate approach was novel and interesting. The patterns that were observed are not particularly surprising, but the power associated with the null hypothesis tests to detect certain effect sizes should have been reported. I expect that at least in some cases the conclusions that there were no significant differences between certain treatments was as much due to lower power of detecting a difference as a true indication of the samples representing the same populations. Along those lines I was surprised that AIC information theorectic approaches were not used to identify the most parsimonious model structure. See; Anderson, D.R. et al. (2000) Null hypothesis testing; problems, prevalence, and an alternative. J. Wildl. Manage. 64, 912-923.
10	4		The methods used are straightforward, and the statistical analyses employed were generally appropriate. The descriptions of the different statistical methods, particularly the Co-inertia Analysis (COIA) and Morisita's index of similarity, are well presented. There is a general deficiency, however, in specifying the statistical models used in the analyses of data from repeated measures.

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
11	4		While the study failed to include examination or assessment of spatial and temporal variances, the statistical analysis employed to examine the efficacy of RMZs retention strategies and forest harvesting impacts are appropriate and generally defensible. The authors have used sound statistical analysis to examine the overarching questions of habitat-environment relationhips.
12	Summary		The analyses are generally appropriate, although the reporting of the actual results leaves a bit to be desired. In general the actual statistical models were never made entirely clear in terms of the model degrees of freedom and their partitioning. This needs to be clearer for a reader to be confident in the outcome.

RMZ Re-sample Final Report ISPR
Are the results as presented supported by the analyses?

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
1	1		See comments for CMER question #3.
2	2	Pg. 3-15	Raising alpha here does not solve the problem of to many contrasts, in fact, Bonferroni correction usually lowers alpha.
3	2	Pg. 3-24	Test results: F=15.3, df=17, 974, p<0.001 - I am not sure what df=17 is derived from, all treatments or the 3-way interaction?; I am not sure how df=974 was derived. This is where a clear design statement would be useful. That is, from your methods, were data from each pointcount station used in the analysis or were pointcounts along each transect combined or were dates combined? The answer to these questions ultimately determined your dfs and whether data in your test are independent or possibly represent pseudoreplication. Please clarify.
4	2	Pg. 3-32	Test results: F=8.9, df=17,90 - I am not sure where df=17 or df=90 came from and precisely what test this represents. Be explicit.
5	2	Pg. 4-10 #1	You say "pooling of sites into treatments was acceptable". This is not what I think you want to do. What I think you mean is that sites were "acceptable" replicates of treatments. Yes? No?
6	3		I suspect that the failure to detect differences could have been for reasons other than the presumption that there was no effect. The authors need to be very careful about issues of pooling data and assuming that a P value greater than -0.05 or 0.10 means that the samples are from the same populations. It could simply be that they did not have the power to detect an effect.
7	5	Ch. 3	Used alpha = 0.1 and beta = 0.8 ("...for landscape-based studies...") but all comparisons seem to be at alpha of 0.05 (e.g. pg. 5-36 "close to significant" at probability of 0.09)?

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
8	5	Ch. 7	It doesn't seem that the species richness values were rarified to account for unequal numbers per site.
9	Summary	Fig. 5-22 and others	Fig. 5-22 and others suggest that the sites were as different before treatments were applied as they were afterwards. This is a bit of a problem, and in various places the text states that the two riparian treatments were not really all that different once implemented, which makes their contrast somewhat suspect. This comes back to how well the pre-treatment data were used to control for site-specific differences before treatments were applied. The authors give the impression that there is a difference between "treatments" but what exactly that difference might be is confusing, and may not exist (see additional comments below, page 7 and 8). The authors several times state that the two buffer treatments were not different from each other in the end. Throughout it is not possible to tell because the specific differences are never shown. When they do show multivariate data for the analysis it suggests the three categories of sites were as different before any cutting as they were afterwards. This is potentially undermining of the study. I have assumed that one need not read the original report from the first post-harvest measurement, but perhaps the details are there.

RMZ Re-sample Final Report ISPR
Are the discussion and conclusions consistent with the results?

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
1	1		See comments for CMER question #4
2	2		Need to discuss RMZ's in the context of the type of logging in adjacent areas.
3	2	Pg. 3-13	There is an error in the statement of Type I error, lines 6-7 from the bottom state Type II error not Type I.
4	2	Pg. 3-24	Each statement of a difference or lack of difference needs statistical support. If p>0.10 then there is no change or difference. It's currently unclear what statistic is used to support each statement. Please clarify.
5	2	Pg 3-36-38, Figures 3-18 and 3-19	It is unclear how variables are selected for inclusion in the results as "important" or not. Some examples like "CC" are obvious but where is the cutoff or how was the decision made?
6	2	Pg. 4-9	It is not at all clear how data were pooled. Each site had 4 transects, 2 reparian and 2 upland. Traps were checked numerous times. Were the 2 replicate transects combined? Were all sampling dates/times within a year combined? This is critical to the analyses and conclusions drawn.
7	2	Pg. 5-49	Not enough capture data for meaningful analyses or conclusions.

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
8	3	Pg. 1-6	"Although we did not assess the efficacy of RMZs from a community ecology perspective, management strategies that are implemented at regional scales should be premised on the preservation of biodiversity and not predicated on single-species or single-wildlife group preservation." This needs expansion - species are the common denominator with regards to biodiversity conservation. And you will not ever 'preserve' biodiversity. We can conserve it, but extinction and speciation are a given. Paterns of biodiversity willshift over space and time. Provide a conceptual framework for biodiversity conservation that places the riparian information in a larger context. To not have a conceptual framework at the outset of the work for how the information will be useful to resolving current or future problems is a serious oversight. Assuming that either one was not developed or it no longer is available, then recreating one so that the information can be placed within such a framework would seem an important first step to interpreting the results. Buffer strips do not occur in isolation and the various environmental gradients affecting the patterns in the results extend far beyond the 100 m sampled. Providing a clear understanding of the states and processes that would influence both the interpretation of the results and the utility of the information would seem critical. See: Gregory SV, Swanson FJ, McKee WA, Cummins KW. 1991. An ecosystem perspective of riparian zones. BioScience 41:540–51 for an early example of such thinking.
9	3		I believe that the discussion reflected their interpretation of the results quite well. There was a bit of overstatement of results that dealt with landscape-level planning necessary to conserve biodiversity and this should have been clearly stated as conjecture, or based on other studies, and it should not be implied that it was based on their data.

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
10	5		Alternative mangement for conservation - these are good suggestions, although perhaps outside the mandate of this study.
11	5	Pg. 1-7	Odd that author says that their ability to make conclusions was confounded, etc., then still make a conclusion.
12	5	Ch. 7, Figure 7-16	Figure 7-16 (similar to others in the report) shows three different species richness trajectories once forest stage develops. However, nowhere does it say anything about why one might expect three different outcomes.
13	5	Ch. 7	The landscape-level discussion (p. 7-39 and elsewhere) about a full range of seral stages on the landscape being essential for biodiversity is likely so. However, it is not something that is tested or examined in this report and not a conclusion from this study.
14	5		Seems that the whole thing is over-analyzed to the point that it is an incredibly tedious read. All the comparisons of upland versus riparian, etc., seem a bit needless.
15	5		Extrapolation to all forest types from two study areas? Most of these sites are predominantly Douglas-fir and few are dominated by western (or mountain) hemlock.

RMZ Re-sample Final Report ISPR

Are all the pertinent literature citations captured and are they incorporated adequately into the report?

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
1	1		See comments for CMER question #5
2	3	Pg. 4-2	McComb 1993 is an inappropriate citation for this sentence. That study did not investigate harvested stands.
3	3		I believe that they should have reviewed the literature from Oregon and BS a bit more than they did. There are a number of papers in national and regional journals that would seem pertinent, but those citations did not turn up in the literature cited. There are a number of newer papers that came out since this was done.
4	4		Missing from the exposition are citations to recent advances on BACI designs. Otherwise, the report provides a good review of the literature, and incorporates most relevant literature in comprehensible manner.
5	5	Ch. 2	Cockle and Richardson is 2003 not 2002.

RMZ Re-sample Final Report

Are the harvest treatments and the stand conditions between treatments well-defined, providing an adequate basis for understanding and making inferences from the results?

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
1	1		See comments for CMER question #2
2	2		Descriptions of the RMZ treatments at each site are lacking. Regulations are given but not data on what was done or left at each site. A table showing how the RMZs varied in structure and size is needed. A description of the actual logging of the sites is absent.
3	2	Ch.1, summary introductions	RMZ buffers were an obvious focus of this research yet it was hard to figure out what size or distance was manipulated in treatments. The average size and structure of RMZ buffers actually left in the cuts should be more prominently indicated. I had no idea how big they were and what the difference was, large or small.
4	2	pg. 3-5	The regulations were clear but it's unclear how much buffer area was actually left.
5	2	Table 3-1	This table is not very clear. This needs to be quantified somewhere and the numbers/distance used when referring to the "state" and "modified" buffer treatments.
6	2	Ch. 4, pg. 4-3	Please provide the physical mean width of the modified buffer.
7	2	Ch. 4, pg. 4-4	Not clear what the actual "state" buffer width was here, seems like it could vary from 7.5 - 15 m.
8	2	Ch. 4, pg 4-4	I have read the "modified" treatment description many times now and still do not know how wide it is. Please give mean, standard error, and range on your sites.

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
9	2	Pg. 5-4	Were RMZs measured at the sites? If so, provide those data.
10	2	Ch. 5	My poor understanding of the modified RMZ regs is that they provide some increase in riparian habitat "quality" but add little habitat "quantity". The biology of riparian species like some salamanders indicates quality and quantity are both important for species persistence. I suspect a lack of RMZ treatment effects is due largely to such subtle difference in "state" and "modified" RMZ specs.
11	2	Ch. 5	Not accurate to say riparian buffer widths were different if regs are based on quality and not quantity of habitat. Were treatment buffer widths ever measured and tested for differences? I have not seen those data or stats.
12	2	Pg 5-53	Need to clarify RMZ widths used in your study.
13	2	Pg. 6-1	I am still wondering how wide your modified treatment buffers were?
14	2	Pg. 7-7	Are there roads or skidder trails thorough control sites? I am not sure if the size and shape of the logged areas has ever been described. This is important for understanding the results and management implications.
16	2	Pg. 8-1	RMZs varied in structure (quality) and width (I assume), so hard to tell which factor is most important. As pointed out before, it is important to document and test how treatments (structure and size) varied across sites. I assume you have such data.
17	2	Pg. 8-8	Unclear what was actually left, need to show data.
18	3	Pg. 3-52	As, such buffers would need to be wider to be immune from such influences. Again, a need for a conceptual framework that sets a context for buffers within a watershed/region.

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
19	3		The upslope transects were quite close (too close for some species) to the riparian area to be considered independent. A transect 100m from the stream is probably too close to the stream to represent an independent sample of upslope conditions, at least in western Washington. See: McComb, W.C., K, McGarigal, and R.G. Anthony. 1993. Small mammal and amphibian abundance in streamside and upslope habitats of mature Douglas-fir stands, western Oregon. Northwest Sci. 67: 7-15. - the transriparian data indicate that for many species inflection points are seen 100-200m from the stream. I think that this potential weakness should be recognized.
20	5		State the treatments - it finally appears under amphibians that the buffer widths appear to be 14m and 30m (pg 1-3), which should be stated at the first mention of the guidelines and design. However, it turns out that widths were 7.5 to 23m for the narrower width sites (Pg. 3-5). This gets further confused when on Pg. 1-6 widths of 7.5m are mentioned as the standard.
21	5		State how far away the "upland" and "riparian" bird transects were from the stream in the summary.
22	5	Ch. 3	With riparian treatments (without actually knowing specific widths at each site) some or most of the 15m radius plots could be outside of the protected zone.
23	5		I'd like to see a more complete description of the treatments, but the writing also makes one wish there was a simple description for the whole study. It becomes less and less clear with each chapter as there are different details about the widths and additional management (removal of merchantable timber). Perhaps a table (like 3-A) might include what the actual widths left behind were and how much timber volume was removed. I could not find such details in the report. Pg. 4-49 seems to be the first place that it says treatments weren't that different between the state and modified methods.

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
24	Summary		The details of the actual treatments are more than a little critical to the evaluation of this work. If indeed the riparian treatments did not end up very differently, then it would be no wonder that the State and Modified riparian treatments were not exceptionally different for some measures, but of course some did differ, and significantly so. If there were site-specific modifications these need to be explicitly listed. This omission makes the entire report difficult to assess since most of the questions presume we can distinguish the actual difference between the two riparian management treatments (State and Modified).

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
25	Summary		The treatments aren't well defined at all. The state guidelines for how the two management regimes should be implemented are described in general, but the specific actions at each site were not given. As mentioned there are several places in the document where the discussion states that the two riparian managment treatments were not all that different due to the large amount of site-specific variation or differences in harvesting. A specific description of the 1988 State guidelines (widths, retention levels, etc.) and the modifications would be helpful. For the modification the descriptions are very vague. They mention that in general they are wider, but only in the amphibian section do they state widths. It is never clear if these are reserves or buffers. The authors also imply there are differences in implementation of the modifications based on type 3 versus 4 streams, and that there are 3 zones within the modified guidelines for type 3 streams. I would suggest a table outlining the actual rules and modifications would be helpful. It would also be useful if somewhere in the document they actually describe the treatments applied to the sites, and if the sites were all type 3 or all type 4, or a mix. As mentioned several times in this response, the description of treatments varies from 14 versus 30 m buffers (amphibian section), to there being no differences in the buffer treatments because of site-specific variations (section on eastern WA mammals).

RMZ Re-sample Final Report
General comments by reviewers.

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
1	2	Pg. 4-7	Were live animals bagged and frozen orjust dead animals? Were live animals released?
2	2	Pg. 5-9	At least 28 days, Table 5-3 says 28-46 days.
3	2	Pg. 5-14	"data from each transect were pooled" - How exactly, by site, by site and date, etc?
4	2	Pg. 5-29	The actual capture data is extremely low for 18 sites over 3 time periods. Only 23-48 individuals per site pooling all years, or 8-16 individuals per site per year!
5	2	Pg 6-2	What level of biodiversity? Regional maximum, old-growth late successional, or pre-cut levels? Define what the expected level is based on.
6	2	Pg. 7-21, Tabl	Clarify whether 1 or 2-way ANOVA.
7	2	Pg. 7-39	It is important for managers to know that "maximizing" bird diversity may not always be the site objective and therefore not best or possibly detrimental for species of concern e.g. late successional species.
8	2	Pg. 8-1	Here it says uplands were "selectively" logged, however "clearcut" was mentioned on Pg 7-39. Typo?

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
9	3		The study design was likely flawed from the beginning but there is little that can be done about that at this point. A transect 100 m from the stream is probably too close to the stream to represent an independent sample of upslope conditions, at least in western Washington. See: McComb, W.C., K, McGarigal, and R.G. Anthony. 1993. Small mammal and amphibian abundance in streamside and upslope habitats of mature Douglas-fir stands, western Oregon. Northwest Sci. 67:7-15. - the transriparian data indicate that for many species inflection points are seen 100-200 m from the stream. I think that this potential weakness should be recognized.
10	3		I did not see, but may have missed it, the IACUC protocols used - given that kill trapping was used, it will be important to document approval of techniques by the IACUC.
11	3	Pg. 1-1	"Pre-treatment or first post-treatment data (i.e., 1995/1996) were collected but are no longer available for the eastern Washington study sites except for those data included in the original report (O'Connell et al. 2000)". This is a serious weakness and needs further explanation.

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
12	3	Pg. 1-5	"The problem faced by land managers is that far too often management plans become specific to one group of organisms and do not use biodiversity preservation as the basis for management." Absolutely! So what is the conceptual framework that should be used to address biodiversity concerns? How will upslope associated be considered? What guidelines can managers use? All biodiversity issues and solutions are species-based.To not have a conceptual framework at the outset of the work for how the information will be useful to resolving current or future problems is a serious oversight. Assuming that either one was not developed or it no longer is available, then recreating one so that the information can be placed within such a framework would seem an important first step to interpreting the results. Buffer strips do not occur in isolation and the various environmental gradients affecting the patterns in the results extend far beyond the 100 m sampled. Providing a clear understanding of the states and processes that would influence both the interpretation of the results and the utility of the information would seem critical. See: Gregory SV, Swanson FJ, McKee WA, Cummins KW. 1991. An ecosystem perspective of riparian zones. BioScience 41:540–51 for an early example of such thinking.
13	3	Pg. 1-7	Weren't bats included in these analyses? They were mentioned in several places as one of the focal groups, but there is no indication of why those data are not included here.

Comment #	REVIEWER #	Comment location Page / Paragraph / Sentence	ISPR COMMENT
14	3	Pg. 2-4	"Riparian areas often support the highest diversity and abundance of wildlife in temperate zones because of their diversity of natural features, because they can support species uniquely adapted to riparian and early successional habitats," True in some places - not in all. There clearly are data that document that in some forest types and successional stages, upslope forests support higher alpha diversity than riparian areas. It is the beta diversity in the watershed that is what is important to consider however.
15	3	Pg. 2-5	"Preservation of high-quality habitat is paramount for preserving the integrity of the entire ecological system of the riparian zone." Habitat for which species? This sentence really does not mean anything as written.
16	3	Pg. 2-6	"Today, however, as the forested landscape is logged and regenerated, these same zones, in the form of buffers, act as refugia for species more typically associated with mature and old growth forests (O'connell et al. 2000)" They MAY act as refugia - we do not know that… And they are less likely to function as refugia for upslope associates.
17	3	Pg. 3-8	"Birds were surveyed using 15m fixed-radius point counts (Verner 1985). Birds beyond this" - distance recorded? Why wasn't the distance to the bird recorded? This gives so much more utility to the data to estimate abundance in a less biased manner. Further, counting birds SO close to the observer can bias results if the observer is having any effect on bird behavior.
18	3	Pg. 3-10	"Snags were classified as short (<1.5m), medium (1.5-15m) and tall (>15m) in three diameter classes: Class 1 - all limbs attached and structurally sounds; Class 2 = losing limbs and showing reduces". I think you are confusing diameter and decay class here.
19	3	Pg. 4-7	kill trapping? - this issue is raised in the discussion of the eastern Oregon analyses, but kill trapping introduces biases associated with depleted populations and alteration of population age structures.

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20	3	Pg. 5-7	Old pitfall traps could not be found... they were left there? How were they inactivated?
21	3	Pg. 5-12	re: "Snags wer counted in all 4 large quadrats and gouped according to quadrat location. Snags were classified as short (<1.5m), medium (1.5-15m), and tall (>15m) in three diameter classes: Class 1 = all limbs attached and structurally sound; Class 2 = losing limbs and showing reduced structural integrity; and Class 3 = about to fall down due to minimal structural integrity". do you mean decay classes?
22	3	Pg. 7-3	"However, pre-treatment and 2-year post-treatment data were not available;" why not?
23	5		There are no amphibians in Eastern WA?
24	5	Ch. 2	Evidence for majority of fish food organisms coming from overhanging trees - best evidence to date is still that of Wipfli (1997) and that indicates much less than half of number of food organisms for a couple of species. Most are dependent on instream production.
25	5	Ch. 3	Mentions vegetation data - were these also analyzed?
26	5	Ch. 3	Strong habitat structure-bird relative abundance relations evident from COIA. These are useful for understanding whycertain patterns occur in the different sites and transects, but link to original objectives not very obvious.
27	5	Ch. 3	Agreed that nesting success might be useful but that there is no compelling reason to pursue that over estimates of relative abundance through time.

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28	5	Ch. 3	The chapter ends with mention of site-specific management, but the work here does not seem to inform in any way how that might be done to meet the conservation objectives for wildlife. I wonder why this is here in the document, or if there is an intention of providing additional guidance.
29	5	Ch. 4	Not obvious why small mammals had to be killed with snap traps, as opposed to various live trap options.
30	5	Ch. 4	In various places the red-backed vole is considered the western red-backed vole, the southern red-backed vole, the western redbacked vole. A consistent spelling would be nice. Pg. 409 Kolmogorov (spelling).
31	5	Ch. 6	This chapter seems like mostly a repeat (verbatim) of what is in earlier chapters. I'm not certain if there is any utility to this chapter if it is simply repeating things.
32	5		The report is highly repetitive between chapters, especially in the methods of each chapter.
33	Summary		Although the design includes riparian transects and upslope transects at each site, the reason for including the upslope transects is never obvious.