129 TFW-LWAG1-00-001

EFFECTIVENESS OF RIPARIAN MANAGEMENT ZONES IN PROVIDING HABITAT FOR WILDLIFE

Final Report

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TIMBER/FISH
&WILD-LIFE

May 2000

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Final Report

Submitted to the LWAG, Timber Fish and Wildlife Program
May 2000

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INTRODUCTION

The managed forests of Washington State encompass approximately 15.9 million acres of which about 63% are owned by the State, various tribes, and private landowners (Washington-Department of Natural Resources 1992). The Timber Fish and Wildlife (TFW) Agreement of 1987 introduced both a framework for management of Washington's state and private lands to protect natural and cultural resources within the context of the managed forest, and a mechanism to evaluate and modify management practices. The Agreement incorporated recommendations and guidelines for the protection of water, fish, wildlife, and archaeological resources. The representatives of state resource agencies, Native American tribal organizations, timber companies, and conservation organizations who forged this Agreement recognized both the immediate need for new forest management policies to protect these resources and the long-term need for these policies to be flexible and responsive to new information. Thus, a central feature of the TFW Agreement was the introduction of adaptive management to Washington's natural resources. Adaptive management involves the continual evolution of management practices in response to scientific knowledge gained through careful monitoring of natural resources and well-designed experimental studies to evaluate how resources are impacted by management practices (Walters 1986).

A set of management goals for the different resources provided the starting point for participants to develop the TFW Agreement. For wildlife, the goal "... is to provide the greatest diversity of habitats (particularly riparian, wetlands, and old growth), and to assure the greatest diversity of species within those habitats for the survival and reproduction of enough individuals to maintain the native wildlife of Washington forest lands" (TFW

Agreement 1987, p.2). Inherent in this statement was the recognition of the importance of maintaining habitat diversity to ensure wildlife species diversity and of the disproportionate importance of certain habitats, including riparian habitats. Given the importance of riparian habitats for wildlife (e.g., O'Connell et al. 1993), it is critical that we understand wildlife response to habitat conditions created by management practices in riparian habitats. In an attempt to balance wildlife and economic goals, the TFW Agreement established Riparian Management Zones (RMZs) for the protection of riparian areas and recommended appropriate sizes, tree densities, and management practices for RMZs associated with several defined water types. These guidelines were incorporated into the Forest Practices Board Rules and Regulations (Washington State Forest Practices Board 1988). The goal of this project was to examine the effectiveness of RMZs in providing habitat for wildlife. The specific objectives were 1) to determine whether current Riparian Management Zone (RMZ) habitat specifications provide adequate habitat to maintain wildlife as specified in the TFW wildlife goal, and if they do not, 2) to identify those habitat conditions created by current RMZ management practices that adversely affect species assemblages, and 3) to provide recommendations for improving RMZ guidelines. These objectives approached on both the east and west side of the state: in an experimental fashion by monitoring the population responses of selected wildlife species and species groups within riparian zones and adjacent upland habitats on 18 sites of harvestable age. The initial study design was for six sites to be harvested according to RMZ guidelines current at the time, six according to a modified RMZ harvest prescription that the research team would design in cooperation with the Wildlife Steering Committee, and six to remain unharvested as controls. Wildlife monitoring would be for 2 years prior and 2 years immediately after harvest. This strategy would establish the baseline conditions from which to compare future changes. In addition, this approach would allow comparison of wildlife use of riparian and upland habitats in the forests of western and northeastern Washington and to examine the habitat correlates that might provide insight into the observed patterns of species richness, diversity, and abundance.

This report is organized into five main sections. First, we provide background information reviewing the importance of riparian habitat for wildlife. Second, we describe our technical approach including experimental design, general sampling strategies, selection and general description of study sites, the design of Riparian Management Zones under the TFW Agreement, and the rationale and design of our Modified RMZs. The third and fourth sections present our studies of the habitat and wildlife, respectively. Each of these sections provides information on the results of the West-side portion and then the East-side portion of the research project. Each section is organized around a comparison of riparian versus upland conditions followed by consideration of the treatment effects. The final section provides a summary and management recommendations.

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ACKNOWLEDGMENTS

We thank the Wildlife Steering Committee of Washington's Timber Fish and Wildlife Program for their evaluation of the initial study designs, assistance in the long task of identifying and securing study sites, their critique of the first draft of-this report, and their general support for this project. Special thanks are due to Lenny Young who was instrumental in the initiation of the project. Peter Haug, Tom Hoppensteadt, Lisa Lombardi, and Georg Ziegeltrum visited our study sites and attended meetings to inform the public of the project. Thanks to Bob Anderson, Eric Hansen, Tim Quinn, Carl Hruska, Derek Stinson, Dave Brock, C. Cushman, Dave Volson, Tim Cullinan, and Neal Wilkins for their support and suggestions at quarterly review meetings. Cheryl Quade and Jim McCracken organized the RMZ Project Workshop and took the lead in organizing review of the report. Our association with members of the Wildlife Steering Committee has been most rewarding.

We thank Linda Chiles and Nancy Sturhan of the Washington Department of Natural Resources for their support and fiscal oversight of the project.

West Side

Given the complications of the experimental approach, the use of state and private lands, and the challenges of funding, work on the west-side was critically dependent on the contributions of many people. We thank Craig Beals, Jessica Eskow, Brian Gilbert, and Jack Ward of Champion International; Dwayne Paige of the Cedar River Watershed (City of Seattle); Stu Blocker, Bob Brown, Wes Culp, Chuck Johnson, Bob Larson, Mark Mauren, Clay Sprague, Chuck Turley, and Lenny Young of the Washington Department of Natural Resources; John Links and Don Wallace of Hampton Tree Farms; John Gross of

International Paper; Lorin Hicks, Hoit Laney, Doug Myers, and Henning Stabins of Plum Creek Timber; Neal Wilkins of Port Blakely Tree Farms; Georg Ziegeltrum of the Washington Forest Protection Association; and Bob Anderson, Kymm Boire, Tim Larkoski, Ken Lenz, Doug Runde, and Don Schuh of the Weyerhaeuser Company. At the University a legion of hourly field assistants collected the field data and assisted with laboratory work. We thank graduate students Robin Christy, Janet Erickson, Matthias Leu, Ellen Martin-Yanney, Scott Pearson, Cheryl Quade, and Angela Stringer for their expertise and long hours. In the budget shortfall year of 1993, Plum Creek Timber, the Weyerhaeuser Company, the Washington Forest Protection Association, the Washington Department of Natural Resources, and the Northwest Hardwoods Commission stepped in and saved the west-side work with financial contributions.

For field assistance we thank Grant Canterbury, Peter Gibert, John Frisch, Neil Furgeson, Paul Hendig, Dave Meade, Mark Moon, and Jennifer Pretare. For comments on the avian chapter we thank Kathryn Kelsey, John Kilgo, Matthias Leu, and Cheryl Quade. We thank Champion Pacific Timberlands, City of Seattle, International Paper and Hampton Tree Farms, Plum Creek Timber, The Weyerhaeuser Company, and Washington State Department of Natural Resources for providing study sites and logistical support. We thank Washington Forest Protection Association Washington Hardwoods Commission, The Weyerhaeuser Company, and Plum Creek Timber for financial support in 1993 and 1995.

East Side

The identification and selection of study sites for the east-side study was possible only through the support and coordination of many people from federal and state agencies, private timber companies, and private landowners. We owe a debt of gratitude to all the people with whom we worked during the initial establishment of the project. Personnel from the Washington Department of Natural Resources (DNR) Northeast Office assisted in site identification, access to orthographic photographs, and many aspects of project logistics. Bob Anderson instructed us in Riparian Management Zone protocol. We thank District Managers, Dick Dunton and Jim Newton, for their support. Walt Obermeyer and Al Lang assisted in identification of potential sites. Pat Ryan and Guy Gifford established and managed the sales of the DNR sites. The Colville National Forest was a critical player in this project. District Rangers, Meredith Webster and Peggy Kane, are thanked for their support. During the site selection process, Diane Fors, of the Colville Ranger District, was indispensable. Her knowledge of the area and of harvest practices, and her support of the project were critical elements in the successful identification of sites. Wildlife Biologists, Tom Burke, Jim McGowan, Sandy Mosconi, Chris Loggers, and Michael Herrin, provided valuable input in both site selection and knowledge of local faunas. Dan Len of the Colville Ranger District assisted in management of the timber sales. Three of our sites were located on private timber company 'land. We thank Boise Cascade and Plum Creek Timber for access to their lands, support of the project, and logistic support. Thanks to Dave Whitwill and Steve Tweit of Boise Cascade and to Dwight Opp, Mike Rueffert, and Lorin Hicks of Plum Creek Timber. We thank the Washington Department of Fish and Wildlife and the US Fish and Wildlife Service for permission to establish one Control site on the Little Pend Oreille Wildlife Refuge. Former manager, Chuck McComb, and current manager, Lisa Langelier, were both most supportive. During the site selection process we also met with personnel from the Sullivan Lake and Kettle Falls Ranger Districts of the Colville National Forest, Colville

Federated Tribe, Arden Tree Farm, Vaagins Timber, Stevens and Pend Oreille County Conservation Districts, and the Okanogan National Forest.

We gratefully acknowledge the hard work and dedication of the many students who assisted in the fieldwork, data entry, specimen preparation, and laboratory work. Several graduate students assumed leadership roles in different aspects of the project. We thank Robert Griffith for maintaining quality control of specimen preparation and small-mammal identification, Doug Call and Steve Mech for their coordination of field crews, Lori Campbell for her work with the bat crew, Ray Entz and K. J. Hackworthy for the nest predation work, Velma Matthew for the plant collection, Peter Fonken, Kevin Pullen, Anthony Floyd, and Deborah Beutler for their work with the avian crews, John Hall for his design of amphibian and reptile protocols, and Lia Wetzstein for her help with small-mammal surveys. Many students and hourly employees returned for several consecutive years and this continuity was a tremendous asset. Scott Barnes was instrumental in the site selection and layout of harvest units. Thanks are due to Kate Healy for her help with many tasks, including assistance in the analysis of bat tapes. Dan Friesz, John DeLong, Jessie Jewell, Heather Carlisle, and Cindy McCormick were perennial members of the avian crew. Joey McCanna, Val Koehler, Neil Lockwood, and Ed Fochtman assisted in the small-mammal surveys and vegetation sampling. Jeff Boyd, Jennifer Ballard, and Sierra Hayden helped with the bat surveys.

We owe a special debt of gratitude to Lauren Hallett for her cheerful cooperation in all aspects of this project. Her companionship and assistance in the field are much appreciated.

Nancy Miller, Ruth Galm, Hiroko Hanes, and Jean Cavanaugh of Eastern Washington University assisted in logistic support and fiscal oversight. At Washington State University, Bev Meyer and Tim Mildren provided much support. We thank both universities for their support of work-study students associated with the project. The nest predation research was supported, in part, by a Northwest Institute Faculty Research Grant to M. A. O'Connell.

Chapter 1

RIPARIAN HABITATS AND RIPARIAN BUFFERS

Riparian zones are found adjacent to watercourses such as streams, rivers, springs, ponds, lakes, or tidewaters and represent the interface between terrestrial and aquatic environments. The riparian zone can be variously defined in terms of vegetation, topography, hydrology, or ecosystem function (e.g., Swanson et al. 1982, Kovalchik and Chitwood 1990). The latter approach integrates the former factors and defines the riparian zone as the zone of interaction between the aquatic and terrestrial habitats (Swanson et al. 1982, Bilby 1988). This definition encompasses the concept that the terrestrial system influences the aquatic system and, in turn, is influenced by the aquatic system. The zone of interaction can be identified as the water's edge or on a broader scale, as a zone extending from the water through the canopy of the vegetation associated with the zone (Swanson et al. 1982). On the latter scale, riparian zones include the relatively mesic vegetative communities and associated faunas occurring between aquatic and more xeric upland sites (Knopf et al. 1988).

Watercourses associated with riparian zones have been variously classified. A widely adopted system to describe drainages classifies small, headwater channels as first-order streams with each union of first-order streams forming a larger second-order stream, each union of second-order streams forming a still larger third-order stream, and so forth (e.g., Strahler 1957, Everest et al. 1985). For regulatory purposes, Washington State Forest Practices Board (1988) recognized five water types on the basis of size and presence of game fish, with Type 1 corresponding to large rivers and shorelines and Type 5 to small headwaters that do not support fish. From a wildlife perspective, a key element of the riparian zone is the amount of open water, but as Hall (1988) points out, the amount of open water necessary to qualify an area as riparian will depend upon individual requirements of species. Wildlife use of riparian areas does not necessarily correspond to the above classifications and it might be preferable to define "operational habitat units" relevant to specific taxa as Bury (1988) does for reptiles and amphibians. The function of the riparian zone is closely related to the size of the watercourse. In the Pacific Northwest, most riparian zones are found adjacent to streams (Oakley et al. 1985) and this is especially true for the forestlands of the region (Swanson et al. 1982, Bury 1988).

The structure and function of riparian zones are determined by several key elements (Cummins 1980, Brinson et al. 1981, Swanson et al. 1982, Oakley et al. 1985, Bilby 1988, Brosofske et al. 1997). These elements are topography, surface water, soils, microclimate, and vegetation. The interaction between terrestrial and aquatic environments that occurs in the riparian zone is mediated by these elements. On the one hand, they combine to create common features that distinguish riparian zones from upland areas. For example, riparian zones are characterized by increased primary productivity, higher levels of energy transport, and often, more frequent natural disturbance than upland areas. On the other hand, differences between these key elements result in differences observed among riparian habitats.

RIPARIAN VEGETATION

The hydrological, topographic, substrate, and microclimatic features of riparian zones result in distinctive physiological, compositional, and structural features of riparian vegetation (e.g., Campbell and Franklin 1979, Franklin et al. 1981, Swanson et al. 1982, Oakley et al. 1985). The hydrology of the riparian zone affects the metabolism and growth of vegetation through three primary factors: increased moisture, nutrient transport, and ventilation of soil by flowing water (Brinson et al. 1981). These three factors contribute to faster growth rates and increased primary productivity of riparian plant communities relative to upland communities.

Composition considers both the number of plant species and the abundance of each species. Riparian areas typically have greater species diversity than upland sites. Variation in the diversity of vegetation between riparian sites is related to a site's size, aspect, soil moisture, amount of woody debris, and time since disturbance (e.g., Gawler 1988, Malanson and Butler 1990). The riparian vegetation is composed of generalized species that inhabit both riparian and upslope sites, but are often more abundant in riparian areas because of favorable conditions, as well as specialized species that are found only in the moist riparian habitat. The latter can include species adapted to conditions created by patterns of natural disturbance characteristic of riparian areas (Gawler 1988). Riparian plant species have evolved a variety of strategies in response to flooding and alluvial deposition. Rowe (1983) defined five categories of plants -- invaders, endurers, resisters, evaders, and avoiders -- based on their mode of adaptation to disturbance and Agee (1988) developed these categories in the context of riparian vegetation of Pacific Northwest forests.

The structure of the vegetation refers to the horizontal and vertical stratification of the plant community. Riparian areas typically have greater structural diversity than upland sites and broader riparian zones have greater structural diversity than narrow, steep-sided riparian areas.

INFLUENCE OF VEGETATION ON STREAM STRUCTURE AND FUNCTION

Many characteristics of riparian plant species and communities are shaped by the presence and flow of water; however, riparian vegetation, in turn, has a direct effect on stream structure and function. First, roots of riparian vegetation stabilize streambanks and streambeds that help to define stream morphology and reduce erosion (Brinson et al. 1981, Swanson et al. 1982).

Second, tiparian vegetation is an important source of large organic debris (LOD, e.g., tree boles, root masses, large branches) in Pacific Northwest streams. Although large organic debris was once considered detrimental to stream quality (Triska and Cromack 1980), it is now recognized as an integral link between terrestrial and aquatic components of forest ecosystems. Indeed, Swanson et al. (1982) suggest that LOD might be the primary influence on lower order mountain streams in forests of the Pacific Northwest. LOD can help define stream structure by retaining gravel and sediment, forming pools, and creating waterfalls (Swanson et al. 1976, Triska and Cromack 1980, Bilby 1981, Swanson et al. 1982, Bilby 1984, Bilby 1988). LOD facilitates deposition of sediments in the stream and consequently affects the morphology and energy transport in lower order streams (Keller and Swanson 1979, Swanson, et al. 1982, Bilby 1988). For example, Megahan (1982) found LOD to retain 49% of the sediments in Idaho streams. This retention of sediment can lead to the formation of sediment terraces that form broad, level areas adjacent to the channel, increasing the size of the riparian area (Bilby 1988). With the input of LOD, a stream becomes characterized by long, level portions, in which the gradient is less than the overall gradient of the valley, separated by short, steep falls in which much of the potential energy of the water flow is dissipated (Swanson et al. 1982). Removal of LOD in smaller streams results in a decrease in the percent area of pools and number ofwaterfalls (Bilby 1981, Bilby 1984) and an increase in particle export from a watershed (Bilby 1988). As a result of this pattern of pools and falls, streams with LOD typically have less erosion, slower loss of organic detritus, and greater habitat diversity than straight, even-gradient streams (Swanson et al. 1982). LOD plays a more important role in creating habitat in smaller streams than in larger streams. The woody debris is large relative to stream width and the smaller streams generally do not have strong enough water flow to redistribute LOD. Wood-created habitat is formed by individual pieces of debris or small accumulations. Periodic debris torrents in smaller streams can remove LOD.

Third, standing riparian vegetation has an important effect on stream function, Riparian vegetation influences the chemistry of the stream through nutrient assimilation and transformation. The absence of vegetation in the riparian zone can result in greater export of dissolved materials (Brinson et al. 1981, Bilby 1988).

Fourth, the shading of streams by riparian vegetation can affect water temperature, and the magnitude of this effect is directly related to stream size. In smaller streams, riparian vegetation can completely shade the water from sunlight and these streams typically exhibit stable, cool temperatures year-round. Larger streams are too wide to be completely shaded so that riparian vegetation has minimal effect on water temperature. Stream size and the degree to which streams are shaded by riparian vegetation also influences whether the energy source supporting the biotic community of streams is primary production in the stream or detritus from surrounding vegetation. In smaller streams, shading by riparian vegetation prevents sunlight from reaching the water, thereby reducing primary production by algae. Organic material from the surrounding vegetation represents the main source of energy in these streams. For example, Swanson et al. (1982) reported that 95% of the organic matter in lower order streams in Pacific Northwest forests is detritus, derived from terrestrial sources. This detritus represents the main food source for many aquatic invertebrates, which in turn, provide food sources for other aquatic and terrestrial species (Bilby 1988). In contrast, primary production by algae and diatoms in larger streams represents the primary energy source for the aquatic community (Cummins 1980, Swanson et al. 1982).

The interaction between the terrestrial and aquatic environment that occurs in the riparian zone changes with stream size. On the one hand, stream size is one of the main factors determining the size of the riparian zone. Small streams produce smaller riparian zones than larger streams. On the other hand, the effect of the terrestrial system on the aquatic system is inversely related to stream size. The forest dominates in small streams, controlling the physical structure and energy base. As Bilby (1988) stresses, understanding this relationship between stream size and interaction between aquatic and terrestrial systems is important when we examine the effects of disturbance in the riparian zone.

DISTURBANCE IN RIPARIAN ZONES

Riparian zones are a product of disturbance (Agee 1988) and an understanding of how natural disturbance affects riparian zone structure and function is necessary to assess how human activities alter riparian zones. In Pacific Northwest forests, natural disturbances, such as flooding, fire, and wind, vary in frequency, magnitude, and relative importance in upland versus riparian areas. Within riparian areas, the effects of disturbance are related to stream size. Agee (1988) modeled the probabilities of fluvial, wind, and fire disturbance relative to position in the riparian zone for small, medium, and large streams in Pacific Northwest forests. In small streams there is a high probability of fluvial disturbance in the center of the riparian zone and the probability of fire or wind disturbance at the center of the zone is equal to, and under some conditions, greater than, that in the surrounding forest. Consequently, the combined probabilities of disturbance are greater in the center of the riparian zone rather than on the edges. Frequent disturbances result in a mixture of patches of invader species with upslope vegetation. The probability of water-based disturbance in riparian zones associated with medium-sized streams is also greatest at the center and decreases towards the edges of the riparian zone. However, the probabilities of fire or wind disturbance are decreased because of higher moisture conditions and more protected topography, respectively. Therefore, the combined disturbance probabilities tend to be reduced at the edges of medium-sized streams. The probability of water-based disturbance relative to position in the riparian zone is similar in large streams to that discussed above for smaller streams. The probability of wind disturbance is relatively great in larger riparian zones because valleys can be corridors of wind movement and saturated soils make trees susceptible to blowdown. High moisture conditions reduce the probability of tire. Combined disturbance probabilities indicate that in larger streams water-based disturbances are the primary disturbance, leading to establishment of invader species.

Agee's (1988) model of disturbance probabilities relative to stream size and position in the riparian zone has implications for assessing impacts of human disturbances in riparian zones and in the design of riparian buffer zones to protect against these disturbances. Although riparian habitats are the products of disturbance, they can also be especially susceptible to human disturbance because 1) humans are attracted to and therefore concentrate many activities in riparian habitats, 2) riparian habitats constitute a relatively smaller amount of area than upland areas, 3) the long, thin shape of riparian areas creates extensive interface with upland areas and makes riparian areas vulnerable to upland disturbances, and 4) riparian habitats support a unique flora that is often sensitive to disturbance (Oakley et al. 1985). Human impacts on riparian habitats are varied and include timber harvesting, livestock grazing, road building, impoundments, housing, channelization, introduction of toxic compounds, hunting and fishing, and non-consumptive recreation (e.g., Brinson et al. 1981, Hall 1988). Given the goals of our research, we focus on the effects of timber harvest.

The impact of timber harvesting in riparian and adjacent upland habitats varies with the type of harvest and characteristics of the watershed. Clear-cutting, for example, might have a greater negative impact on riparian habitats than single tree selection (e.g., Oakley et al. 1985). Research concerning the effects of logging in watersheds has suggested varying levels of impact on riparian zones from little or no impact to substantial impact. Much of the variation reflects the initial definition of the riparian zone, the variables measured, and the design of the studies. For example, a comparative study of logged versus undisturbed sites in northeastern Oregon (Carlson et al. 1990) suggested little differences in LOD and pool volume between sites. In contrast, other studies have identified several major stream-habitat changes associated with logging (e.g, Harr 1976, Harr et al. 1979, Swanson 1980). Water temperatures increase after tree harvesting due to the reduction of shading. Increased sedimentation often results from logging because 1) logging activities (i.e., timber felling, yarding, roading) increase input of soil and detritus into streams, 2) sediments trapped by LOD prior to logging can be released if LOD is removed from the channel, and 3) a reduction in ground cover adjacent to streams increases erosion of soils. Stream flow, especially in smaller streams, can significantly increase following timber harvests. Microclimatic variables such as air temperature and soil temperature in forested buffers along streams resemble conditions in the clearcut areas rather than forest interior (Brosofske et al. 1997). Timber harvest in riparian areas can alter the composition and structure of both the overstory and understory plant communities. Finally, removal of vegetation from small streams can alter the dynamics of the food chain because, as discussed above, terrestrial vegetation represents the primary source of organic input in these streams. Maintenance of vegetative buffer zones adjacent to streams and retention of LOD in stream channels can decrease many of these negative impacts (e.g., Franklin et al. 1981).

Anthropogenic modifications potentially reduce the value of riparian habitat for native wildlife. In the remainder of this background section we discuss the characteristics of riparian habitats which make them of high wildlife value, wildlife use of these areas, and how buffer zones designed to mitigate the effects of human disturbances in managed forests might affect wildlife.

WILDLIFE USE OF RIPARIAN HABITAT

Naturalists have long recognized the high value of riparian habitats to wildlife. Quantitative studies conducted during the past several decades have supported observations and have identified biological and physical attributes of riparian habitats which enhance their value to wildlife. Brinson et al. (1981), Oakley et al. (1985), and O'Connell et al. (1993) provide summaries of these biological and physical features.

First, the presence of surface water provides a critical habitat component for wildlife and the abundance of soil moisture creates habitat conditions favorable to many wildlife species. Second, the increased humidity, higher rates of transpiration, and greater air movement often found in riparian zones create microclimate conditions that differ from surrounding uplands and are preferred by some wildlife species during hot weather. Third, riparian habitats tend to be complex wildlife habitats because of the interspersion of many biological and physical features. Plant communities in riparian habitats are more diverse in their composition and structure than in uplands. Associated with this complexity is an increase in internal edges at the interface between stream channel and riparian vegetation and in the transition between riparian and upland vegetation. A developed deciduous component in riparian plant communities creates additional habitat complexity because of changes in habitat conditions at different times of the year (Thomas et al. 1979). Fourth, the linear shape typical of riparian habitats creates maximum edge effect with adjacent upland forests which is beneficial for some wildlife species. Finally, the shape and habitat conditions of riparian zones make them natural migration routes and travel corridors for many wildlife species (e.g., Thomas et al. 1979, Brinson et al. 1981, Oakley et al. 1985) and therefore might represent routes of gene flow (West 1988).

Brinson et al. (1981) and Johnson (1977) provide extensive reviews ofwildlife resources in various regions of the US and Thomas et al. (1979), Oakley et al. (1985), and Raedeke (1988) review wildlife use of Pacific Northwest forests. Most surveys indicate that wildlife species use riparian habitats disproportionately more than other types of habitat, Although especially true in the more arid regions of the US (Johnson and Jones 1977, Brinson et al. 1981), this pattern is generally found in the forests of the Pacific Northwest. Thomas et al. (1979) report that 278 of the 285 terrestrial wildlife species in the Blue Mountains are found exclusively or more commonly in riparian areas and Oakley et al. (1985) report similar patterns for 359 of the 414 wildlife species of western Washington and Oregon forests. In contrast, McGarigal and McComb (1992) report little difference in avian species diversity between riparian and upland habitats along lower order streams in the coastal mountains of Oregon.

Although there are common environmental attributes of riparian ecosystems that enhance the wildlife value of these areas, other ecological characteristics vary between riparian areas and further determine the value of these habitats to wildlife. These ecological variables have been reviewed by Brinson et al. (1981) and include vegetation type, size and shape of riparian area, stream type and hydrologic pattern, adjacent land use, and elevation.

In sum, riparian areas, provide habitat for many wildlife species, but assessing the relative value of a particular ~riparian area for wildlife must take into account a variety of ecological characteristics. Therefore, habitat management of riparian areas becomes a critical element of wildlife management. To mitigate the effects of timber harvesting in managed forests many states have adopted the use of buffer zones along streams. In Washington, for example, the Forest Practices Board (1988) prescribed the creation of Riparian Management Zones (RMZs) for managed forests on state and private lands. These RMZs vary in width and number of trees left in the buffer depending upon water type and region of the state. The primary intent ofmandating buffer zones along streams has often been the preservation of water quality and fisheries habitat. The maintenance of buffer zones can also benefit terrestrial wildlife species, but the effectiveness of these buffers must take into account a variety of factors.

WILDLIFE USE OF RIPARIAN BUFFERS

In the managed forests of the Pacific Northwest buffer zones can serve two distinct roles. Historically, when the prevailing successional stage in the PNW was old forest, riparian zones provided refugia for species characteristic of early successional stages. Aside from the presence of water, the unique features of riparian zones center on the admixing of early successional characteristics within old forests. The presence of such areas was especially important for the continued existence of species with limited powers of dispersal, For example, herbivorous small mammals, which survived at low population densities in such areas, and could rapidly colonize large areas after forest disturbance, needed the small strips of open ground supporting grasses and herbs. With the maintenance of riparian buffer zones in managed forests, a second function envisioned for riparian zones is in providing elements of old forest in a predominantly young forest landscape. Forest harvest, which creates riparian buffer zones in managed forests, however, results in the fragmentation of the previously continuous forest habitat. This leads to the creation of a mosaic of forest patches of various age and structure, that are scattered over the landscape and which vary spatially and temporally. McIntyre and Hobbs (1999) proposed a framework for conceptualizing such landscape patterns that recognizes a continuum of habitat alteration states from intact to relictual and different levels of modification for the surrounding matrix habitat. In addition, forest patches created by the retention of riparian buffers are unique in their linear shape and because of the special features inherent to riparian zones. Examination of the effectiveness of riparian buffer zones in the two above-mentioned functions must therefore take into consideration the effects of forest fragmentation on wildlife.

The positive relationship between area and species richness has long been recognized for island situations (e.g., MacArthur and Wilson 1967) and has been applied to forest landscapes (e.g., Harris 1984). Larger areas support greater species richness because of greater habitat diversity and likelihood of colonization from surrounding areas. The maintenance of buffer zones along streams creates forest patches of potentially different sizes. Studies of terrestrial vertebrates (Stauffer and Best 1980, Dobkin and Wilcox 1986, Rudolf and Dickson 1990, Kinley and Newhouse 1997) indicate that wider buffer zones (i.e., larger area) often support greater species richness. Although maintenance of species diversity is a primary goal of current conservation strategies, maximizing species richness without regard to differences between species is not always a desirable management goal (e.g., Van Home 1983, Murphy 1989, Lehmkuhl and Ruggiero 1991). Considering the potential dual function of riparian buffer zones in providing habitat for both early and late successional species, managing for species diversity becomes a complex issue. For example, in pine plantations of eastern Texas Dickson and Williamson (1988) found that narrow (< 25 m) streamside management zones supported more small mammals than medium (30-40 m) or wide (50-90 m) zones, but that only the wider zones provided habitat for species associated with mature forest stands. Considering birds, Darveau et al. (1995) found that narrower (20-40 m buffers) had higher initial densities than wider (60 m) buffers, but that the narrower buffers had the highest decreases in abundance after several years.

As background it might be helpful to distinguish three categories of wildlife species that might inhabit a riparian zone. The first group, riparian obligates, are those species that require free water for some aspect of their natural history and must inhabit the riparian zone. They will reach maximum abundance within the riparian zone, and decline in abundance with distance from it. The second, and larger group of species is those that are characteristic of the older successional stages. Numbers of these species will increase as the area of old forest available to them in the riparian zone increases, resulting in relatively few of these species in small forest blocks and generally a full complement of species in large blocks. These species might not require the resources of the riparian zone to survive, but will inhabit it and might even have more productive populations within the zone than in the adjacent uplands. The third group of species consists of those species characteristic of early successional stages. They have an interesting relationship to riparian zones in that, as previously mentioned, riparian zones almost always provide some level of resources to support these species. This is the result of the periodic disturbance regimes characteristic of riparian zones. They will inhabit riparian zones embedded within old forest in small but persistent numbers. Should the adjacent upland forest be harvested, the forest successional sequence will be initiated, and these species will rapidly colonize these areas. Given this scenario, they might exert considerable pressure on the resources available to species characteristic of old forest, which might be trying to exist within the riparian management zone. How much pressure they exert will be related to the width of the zone.

Riparian habitats are characterized by high levels of inherent (natural edge) and maximum edge effect. The creation of riparian buffer zones in managed forests results in equally high levels of induced (disturbance created) edge. Wildlife biologists have long recognized that the abundance and richness of some species is greater along edges because of the presence of species adapted to the two adjacent habitat types as well as those specifically adapted to edge conditions. Wildlife habitat management has traditionally sought to maximize edge effect in managed forests. This has benefited species such as white-tailed deer, elk, and ruffed grouse. Fragmentation of habitat and increased edge, however, might be detrimental to other wildlife species. Lehmkuhl and Ruggiero (1991) summarized seven detrimental edge effects: 1) competition between forest interior and edge species might occur which could reduce the viability of interior species populations; 2) generalized species found in forest patches at time of fragmentation might benefit from the altered environmental conditions outside the patches (a “cross boundary subsidy”) and increase in population size or viability to the potential detriment of interior species (e.g., Raedeke and Lehmkuhl 1986); 3) nest predation and nest parasitism can increase in forest patches with substantial edge (Wilcove 1985, Temple and Cary 1988); 4) the forest edge might be a “unidirectional filter” that animals will pass out of but cannot return, for example some species are more vulnerable to predation outside of forest patches; 5) elimination of interior species as a result of forest fragmentation might lead to secondary extinctions because of altered community interactions; 6) extrinsic processes such as blowdown or ground fire, can reduce forest patch size or quality through “edge creep”; and 7) forest patch edges are subject to microclimatic changes which alter conditions for interior plant and animal species - in the Pacific Northwest, for example, these microclimatic changes are thought to extend up to two tree lengths (160 m) inside a forest patch (Harris 1984, Franklin and Forman 1987).

The potential negative impacts of forest fragmentation on wildlife, the unique features of the riparian habitat, and the dual function envisioned for riparian zones in providing wildlife habitat, require that careful attention be given to the design of buffer zones if they are to be effective in providing that habitat. Although there is general consensus for the need to provide riparian ‘buffers in managed forests, there is much less agreement as to the size and desired characteristics of these buffers. In part this is because riparian buffers have been designed for a variety of purposes. At one end of the spectrum, if the function of the riparian buffer strip is to protect water quality, a narrow buffer of 8 m (e.g., Trimble 1959, Washington State Forest Practices Board 1988) might suffice, but at the other end, wider buffers are recommended if these strips are designed to maintain wild or scenic values of river corridors (e.g., 400 m; Wild and Scenic Rivers Act, P.L. 90.542).

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Chapter 2

STUDY DESIGN

INTRODUCTION

The objectives of the: statewide study were threefold: 1) to determine whether current Riparian Management Zone (RMZ) habitat specifications provide adequate habitat to maintain wildlife as specified in the TFW wildlife goal (TFW Agreement 1987, Wildlife Action Plan 1990), and if they do not, 2) to identify those habitat conditions created by current RMZ management practices that adversely affect species assemblages, and 3) to provide recommendations for improving RMZ guidelines. These objectives were addressed on each side of the state in an experimental fashion by monitoring the population responses of selected wildlife species and species groups within riparian zones and nearby upland habitats on 18 sites of harvest age. The original study design designated that six sites would be harvested according to RMZ guidelines current at the time, six according to modifications of the guidelines that the research team would design in cooperation with the Wildlife Steering Committee, and six control sites would not be harvested. These sites will be classified as State, Modified, and Control, respectively, throughout this report. Following the original study design, wildlife monitoring would be for 2 yr prior to and 2 yr immediately after harvest. This strategy would establish the baseline conditions from which to compare future changes in the State and Modified treatments.

WEST SIDE

Research was conducted in coniferous forests of the coastal and Cascade Mountains of western Washington.

Site selection

Site selection began in spring 1991, but, due to a variety of factors, was not completed until late 1992. Site selection criteria were chosen to make the study broadly applicable to forest lands in western Washington. At the same time, adequate replication required that the scope of the study be limited with respect to the number of varying environmental factors. Within an area of study, sampling sites were sought to minimize variation in forest age and composition, elevation, moisture condition, and water type. In consultation with the TFW Wildlife Steering Committee, we selected sites that had the following characteristics: 1) low elevation (<620 m); 2) second-growth forest (55-65 yr old), dominated by Douglas-fir: 3) Type 3 water by forest regulations, Type 4 could be chosen if streams differed only in the presence of salmonids; 4) predominately coniferous riparian canopy with deciduous tree component; 5) at least 500 m in stream length; 6) road access within 0.5 km; 6) could be harvested according to the project's specifications and time lines, The selection process resulted in the 18 sites listed in Table 1. The sites were distributed widely in western Washington (Fig. 1).

Table 1. West side study sites by treatment type, harvest completion date, and ownership

Stream	Treatment	Schedule	Ownership
Abernathy	Control	No harvest	Washington State Department of Natural Resources (DNR)
Blue Tick	Modified	Mar 1994	Washington State DNR
Elbe Hills	Control	No harvest	Washington State DNR
Eleven Creek 3 1	Modified	Sep 1994	The Weyerhaeuser Company
Eleven Creek 32	state	Mar 1994	The Weyerhaeuser Company
Griffen Creek	Modified	Mar 1994	The Weyerhaeuser Company
Hotel Creek	Control	No harvest	City of Seattle Cedar River Watershed
Kapowsin	state	Mar 1995	Champion Pacific Timberlands
Ms. Black	Modified	Jan 1994	Washington State DNR
Night Dancer	state	Mar 1995	Washington State DNR
Porter Creek	Control	No harvest	Washington State DNR
Pot Pourri	State	Mar 1994	Washington State DNR
Ryderwood 860	Modified	Mar 1994	International Paper and Hampton Tree Farms
Ryderwood 1557	State	Jun 1994	International Paper and Hampton Tree Farms
Side Rod	Modified	Mar 1994	Washington State DNR
Simmons Creek	State	Mar 1994	Plum Creek Timber
Taylor Creek	Control	No harvest	Cedar River Watershed, Seattle
Vail Control	Control	No harvest	The Weyerhaeuser Company

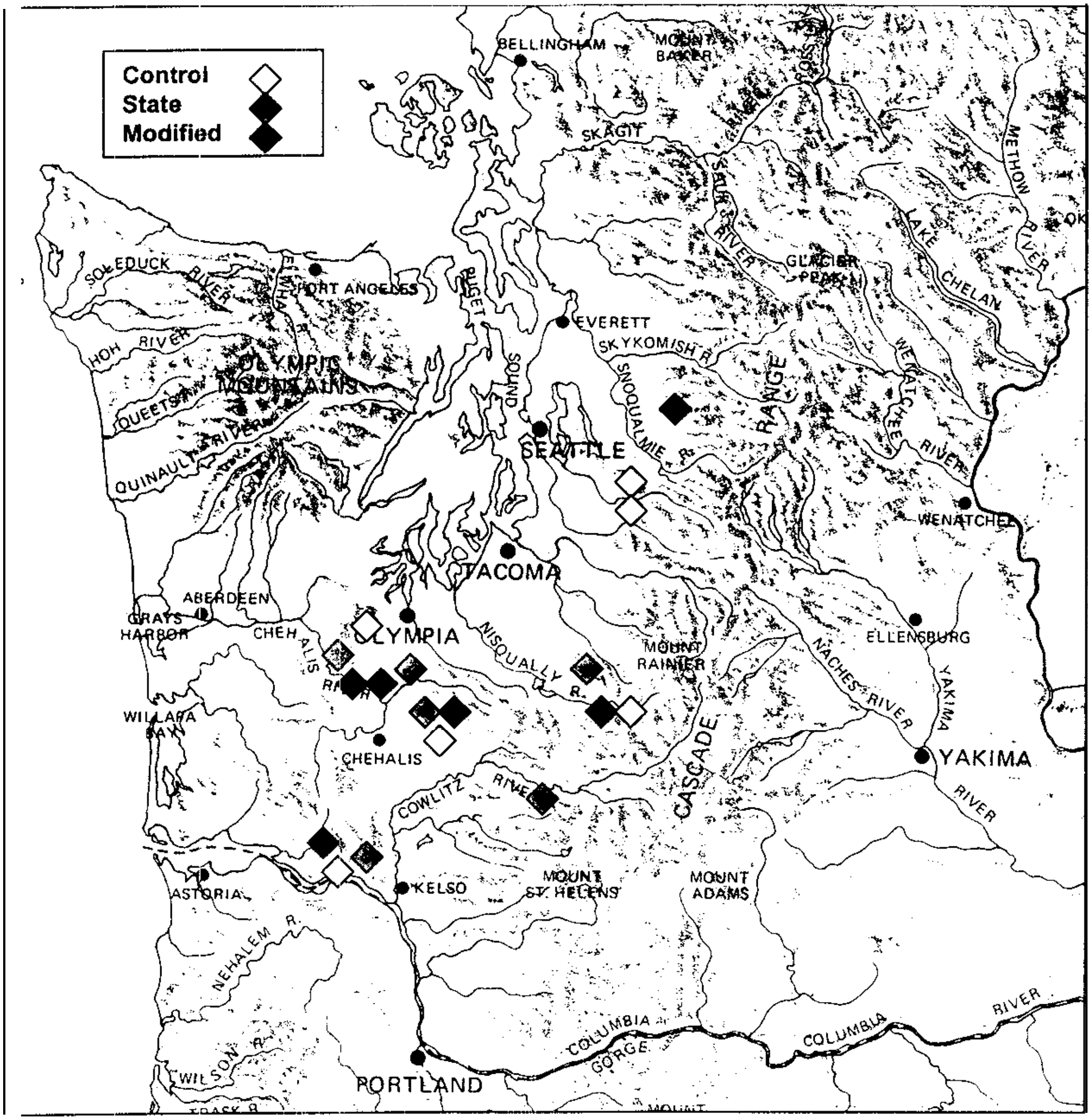

Figure 1. Distribution of study sites in western Washington

Field sampling

Due to the broad range of wildlife taxa we sampled, the field season extended from April until November. Details of sampling methodologies are given in the pm-harvest section for each taxon (below). Vertebrate sampling occurred on the following schedule each year:

Mid-April Early July	Breeding Bird Surveys: variable circular plots
Mid-June - End August	Bat Surveys: echolocation detectors
Mid-July - Mid-September	Stream Amphibian Surveys: stream searches
Early October - Early November	Small Mammal and Terrestrial Amphibian Surveys: pitfall trapping

Vegetation sampling occurred during mid-July-August in 1993 (pre-harvest), 1996 (second post-harvest year), and in 1998 (second post-harvest year for the two late sites)

The delays in site selection and interruptions in funding resulted in some asynchronies in sampling between sites. In 1992, avian sampling was not done during spring and early summer because of the lack of sites. Thirteen sites were available for wildlife censusing during the fall sampling period. Trapping for small mammals and terrestrial amphibians was completed for all 13 sites; stream surveys for aquatic amphibians were conducted on 12 sites; and bat echolocation surveys conducted on 10 sites. During winter 1992, we acquired the full complement of 18 sites (Table 1). Avian censusing was completed the following spring/early summer on all sites.

In early summer 1993, state funding for the project ended. At that time, the project had one year of sampling stratified by riparian and upland habitat. Fortunately, cooperators (Washington Department of Natural Resources, The Washington Forest Protection Association, The Weyerhaeuser Company, and Plum Creek Timber) funded the fall sampling period. This had several important consequences. In terms of the usefulness of the data set, the second sample provided: 1) a 2-year average for the baseline condition which allowed for statistical comparisons for all wildlife groups between sites and over time (except birds), 2) the opportunity to make comparisons within sites, i.e., riparian vs. upland comparisons, 3) all of the vegetation/habitat measurements which were scheduled for collection in the second year, and 4) information for small mammals, stream-dwelling amphibians, terrestrial amphibians, and bats for all 18 sites.

Given delays in harvesting on four sites and funding shortfalls, we did not sample in 1994. The Washington Forest Protection Association and the Washington Hardwoods Commission provided fimding for 1995. Eighteen sites were sampled in 1995 and 1996. One of the State harvest sites had to be reestablished due to a siting error. We decided to sample this site simultaneously with a control during 1997 and 1998 to realize the full complement of six sites per treatment type:. Funding for 1996 was provided by the state, as was funding to complete the post-harvest sampling during 1997 and 1998. At present, we have full data (vertebrates and vegetation) for both post-harvest years.

Timber harvest

All cut sites were harvested between March 1994 and March 1995.

Upland Harvest

The harvest prescription of the upland harvest on West Side cut sites was a clearcut.

Riparian Harvest

The riparian buffer zones were harvested according to the Washington State Forest Practices current at the time or a Modified buffer prescription designed for this research project.

State.— ‘The riparian zones of the 6 RMZ sites were harvested in accordance with the Washington State Forest Guidelines for buffer width and number of leave trees.

Modified.— In designing the harvest prescription for the Modified buffers, we sought to contrast the habitat features of buffer zones generated from the State guidelines and Modified guidelines with unharvested riparian zones. Our strategy in designing the guidelines for the Modified buffers was to create a buffer of intermediate structure between the State RMZ buffers and unharvested riparian zones. We accomplished this relative to the State RMZ buffers by increasing the amount of undisturbed ground near the stream by specifying a no-entry zone. We retained wildlife reserve trees within the buffers, and increased the variation in buffer width and tree density within the buffers by buffering wildlife reserve trees according to Labor and Industry safety regulations when their structure so required. The no-entry zone and wildlife tree buffering also increased the overall width of the buffers.

Described below is the harvest prescription for the block of six Modified harvest sites in western Washington.

The following three recommendations have been formulated to assist the persistence of species highly associated with forested riparian habitats. Forest practices not addressed below are assumed to follow current guidelines[1]. These recommendations are written for type 3 streams, For such streams, maximum buffer widths vary between 25 and 50 feet as specified in current guidelines.

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[1] As adopted by the Forest Practices Board June 26, 1992 and effective August 1, 1992.

Recommendation 1: Observe a no-entry zone within the riparian zone as defined by the break between riparian and upland vegetation. When the riparian zone extends beyond 25 feet from the ordinary high water mark, observe a minimum no-entry zone of 25 feet measured horizontally from the ordinary high water mark.

Justification: species identified as obligate (aquatic amphibians, water shrew) or highly associated with riparian zones require strong shade for low water temperature and buffered temperatures at ground level. They also benefit from uncompacted soils, minimal sediment input, and deciduous vegetation. Such characteristics are best accomplished by avoiding extensive ground disturbance and retaining an intact canopy with its associated understory vegetation immediately adjacent to the stream. A no-entry zone will also provide an enhanced supply of large organic debris to the stream.

Departure from current Washington State Guidelines: Current RMZ guidelines specify 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. The proposed harvest prescription differs from the guidelines in prohibiting harvest within the no-entry zone.

Recommendation 2: Within a strip ofvariable width (25 to 50 feet) adjacent to the no-entry zone, conduct harvest practices according to current RMZ guidelines with the exception of recommendation 3 below.

The width of the variable strip is a function of riparian zone width (defined by vegetation), and as described below, the distribution of wildlife reserve trees. When riparian vegetation does not extend further than 25 feet from the ordinary high water mark, the variable strip will average at least 25 feet wide. When riparian vegetation extends beyond 25 feet from the ordinary high water mark, the variable strip will widen from the no-entry zone to maintain at least a 25-foot buffer along the riparian/upland boundary.

Justification: This recommendation provides a minimum buffer for the no-entry zone and helps retain its interior forest characteristics. The recommendation also increases the probability of persistence for species highly associated with the riparian zone by providing a wider area of potential inhabitation. Selective cutting within this strip may help reduce blowdown rates within the no-entry zone, and will open the canopy allowing for increased growth of understory vegetation. This enhanced productivity and cover should benefit many ground-active species.

Departure from current Washington State Guidelines: The proposed guideline applies current guidelines for selective harvest in Slate RMZ's to a variable-width zone located 25-75 feet from the ordinary high water mark.

Recommendation 3: Retain all type 1, 2, 3, and 4 wildlife reserve trees within the zones defined above unless they violate Labor and Industry safety regulations. Type 3 and 4 wildlife reserve trees buffered according to Guidelines for Selecting Reserve Trees'.

Justification: This recommendation increases present and future opportunities for species requiring reserve tree characteristics and it provides additional material for LOD recruitment. The clumping of trees generated by buffering type 3 and 4 reserve trees will increase the spatial complexity of the riparian management zone and intermittently extend an intact canopy to the uplands.

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[2] Guidelines for selecting reserve trees. 1992. P 417-092-000. Available from the Washington Department of Natural Resources, Olympia, WA

Type 3 and 4 wildlife reserve trees felled in accordance with Labor and Industry safety regulations shall be left on the ground. Trees used in buffers for type 3 and 4 wildlife reserve trees will not be counted in the minimum RMZ leave tree requirements.

Departure from current Washington State Guidelines: Current guidelines call for three wildlife reserve trees, two green recruitment trees, and two downed logs to be left for each acre harvested. The proposed guideline requires that all wildlife reserve trees be left within the RMZ.

EAST SIDE

Research was conducted in mixed-coniferous forests in the Selkirk Mountains of northeastern Washington (Stevens and Pend Oreille counties). Forest composition in this region is variable and is affected by slope, aspect, edaphic factors, fire history, and timber management practices. Dominant tree species include Douglas-fir (*Pseudotsuga menziesii*), lodgepole pine (*Pinus contorta*), western redcedar *(Thujaplicata),* western hemlock (*Tsuga heterophylla*), western larch (*Larix occidentalis),* grand fir (*Abies grandis),* and alders (*Alnus incana* and *Alnus sinuata).* Shrubs included gooseberry *(Ribes* spp.), devil's club (*Oplopanax horridum*), Oregon grape (Berberis spp.), mountain boxwood (*Pachistima myrsinites*), red-osier dogwood (*Cornus stolonifera*), ninebark (*Physocarpus malvaceus*), spireae (*Spireae* spp.), serviceberry (*Amelanchier alnifolia*), rose *(Rosa* spp.), and huckleberry (*Vaccinium* spp.).

Site selection

Selection of the 18 study sites was based on six primary criteria: 1) managed forests of harvestable age, 2) ≥800 m reach of Type 3 or permanent Type 4 stream, 3) ≥16.2 ha previously harvested stands on either side of stream, 4) mixed coniferous forests,. 5) >600 and <1200 m elevation, 6) landowners agreed to either leave sites unharvested for 10 yr (controls) or to harvest sites within timeframe and specifications of study design. Initially we had planned to have six sites in each of the three treatments, but harvest schedules on one of the Modified sites could not be accommodated, resulting in 7 Control, 6 State, and 5 Modified. The 18 sites are listed in Table 2. Sites were located in northeastern Washington (Fig. 2) on lands managed by the Washington Department of Natural Resources (DNR), the United States Forest Service, Colville National Forest (USFS), Boise Cascade Corporation, Plum Creek Timber Company, and the Little Pend Oreille National Wildlife Refuge (USFWS).

Table 2. East side study site:; by location, elevation, ownership, and treatment type,

Stream	Legal description	Elevation (m)	Ownership	Treatment
Amazon	T35NR41 ES2	1133	DNR	State
Bear	T34NR41ES12	1200	USFWS	Control
Browns	T31NR44ES33	800	USFS	Control
Buck East	T3 1NR42ES22,23	1000	Boise Cascade	Modified
Buck West	T3 1NR42ES22	1000	Boise Cascade	State
Butte	T34NR42ES32	1200	USFS	Modified
Calispell	T32NR42ES14	1067	USFS	Control
Cee Cee Ah	T34NR44ES 12	1233	Plum Creek	Modified
Chewelah	T34NR42ES29	1233	USFS	Control
Middle	T35NR44ES28	800	DNR	State
Mill	T36NR41ES5,6	1133	USFS	Modified
Muddy Control	T37NR42ES17,18	1233	DNR	Control
Muddy East	T37NR42ES 17	1200	DNR	State
Muddy West	T37NR42ES18	1233	DNR	State
Power	T32NR43ES28,33	933	USFS	Control
Rocky Control	T27NR41ES35	1167	USFS	Control
Rocky Cut	T37NR41ES25	1167	USFS	Modified
Sherry	T36NR42ES28	1167	DNR	State

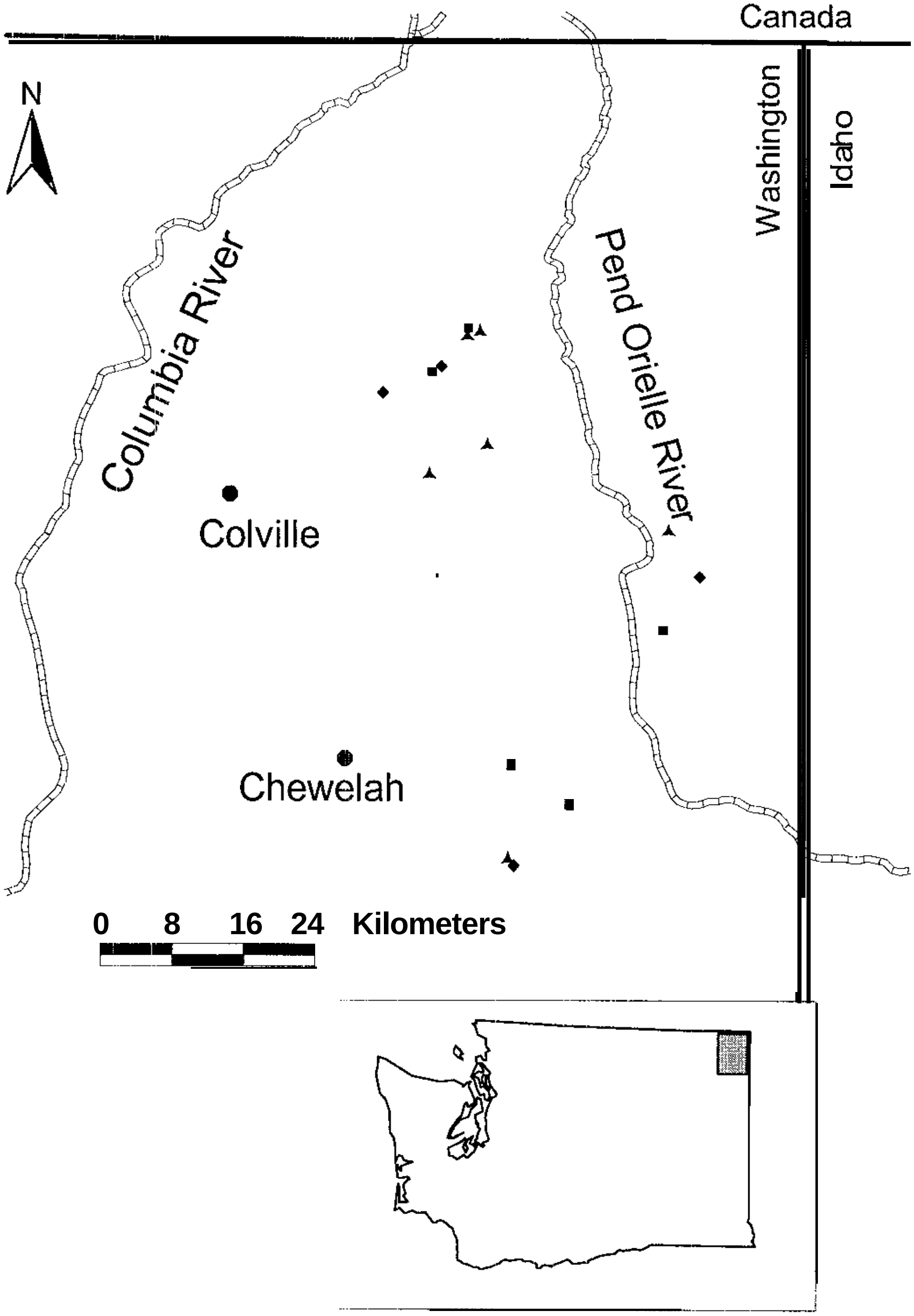

Figure 2. Distribution of the seven Control (■), six State (▲), and five Modified (◆) study sites in northeastern Washington.

Transect design

At each of the 18 study sites we established an 800-m riparian transect at 8-m distance from the stream (about half way from the stream edge to the boundary of a state-mandated RMZ) and another 800-m upland transect 100-m upslope from the riparian transect. 'We marked each of these transects with flags at 50-m increments to serve as reference points during the bird surveys and vegetation studies.

Field sampling

The east side field seasons took place in the spring and summer of 1992-1996. Breeding bird surveys were conducted from May through mid-June. Small mammal, amphibian, and reptile surveys took place from mid-May through June. Bat echolocation surveys were conducted monthly from June through late August-early September. Vegetation sampling was conducted in July and August.

Timber harvest

All cut sites were harvested beginning in fall 1993 (after sampling) and extending through summer 1994.

Upland harvest

The harvest prescription of the upland harvest on the East Side cut sites was a partial cut yielding a 6-12-m spacing.

Riparian Harvest

The riparian buffer zones were harvested according to the Washington State Forest Practice RMZ guidelines or a Modified buffer prescription designed for this research project, The two harvest prescriptions are compared in Table 2.

State Buffer.-- The riparian zones of the 6 State sites were harvested in strict accordance with the Washington State Forest Guidelines for RMZ buffer width and number of leave trees.

Modified Buffer.-- The riparian zones of the 5 Modified sites were harvested according to a harvest prescription that we designed after examination of the initial years' data and in consultation with the TFW Wildlife Steering Committee and landowners. The intent of this harvest was to incorporate a site-specific approach to riparian management. The following is the description of the East Side harvest prescription as presented to and approved by the participants and TFW Steering Committee.

There was be a 100-ft zone of special consideration on both sides of the stream. If habitat features and forest conditions identified below were present in this zone, harvest practices followed the Modified prescription. If identified habitat features and forest conditions were not present, harvest practices followed the State RMZ prescription. Forest practices not addressed in the Modified prescription followed State RMZ guidelines (e.g., no use of heavy machinery in the riparian zone).

1. Goal: To provide for the habitat needs of species identified as riparian specialists.

Recommendation 1: Within the 100-ft zone of special consideration, all type 1, 2, 3, and 4 reserve trees were left unless they were in violation of Labor and Industry regulations. Within the 100-ft zone of special consideration, an average of one tree every 2 acres of type 3 or 4 reserve trees > 12" DBH was buffered by 1.5 times the tree height as suggested for type 3 and 4 reserve trees in "Guidelines for selecting reserve trees" (1992).

If buffered areas extended into the 30-50 ft standard RMZ width of the Washington State Forest Practices, non-buffered areas of the standard RMZ maintained the specified stem value of a riparian buffer. If both uplands and riparian habitats are managed with structural diversity and attention to habitat features of importance to wildlife in mind, protection of the region's terrestrial vertebrate diversity can be enhanced.

By design, our results focus on the years immediately following harvest. We have provided a baseline from which future changes within the buffers and adjacent uplands can be compared. Studies of wildlife response to different buffer harvests in other regions have indicated changes in composition and abundance between the immediate post-harvest years and later years. From some trends in this study and our experience with the habitat patterns shown by vertebrates in the TFW Landscape Study, we expect several such changes in the next few years.

To document these changes these sites must be resurveyed at regular intervals. We suggest returning about five years post-harvest and again at about 10 yr post-harvest. The first decade should encompass the most active period for decline in species associated with riparian and closed canopy forest. Without additional sampling the effectiveness of these RMZ designs cannot be assessed.

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O'Connell, M. A., J. G. Hallett, and S. D. West. 1993. Wildlife use of riparian habitats: a literature review. Washington Department of Natural Resources **TFW-WLl-93-001.**

maintained the specified stem count of the Washington State Forest Practices. Non-buffered areas of the standard 30-50 ft RMZ width were aggregated for stem count determination, In other words, leave trees in buffered areas could not subsidize non-buffered areas.

Justification: Seeps and marshes provide habitat for many amphibian species (e.g., spotted frog) and some small mammal species identified as riparian species (e.g., western jumping mouse, northern bog lemming). Recommendation 2 provides shading for these areas and potential travel corridors between these areas and the stream.

Departure from Current Washington State Guidelines: Current guidelines provide protection for seeps and marshes within the 30-50 ft standard RMZ. Recommendation 2 extended the area within which these features are protected and provided for travel corridors linking seeps or marshes to the stream.

Recommendation 3: Within the 100-ft zone of special consideration, all live deciduous trees were left and within 30-50-ft of the stream all deciduous shrubs were left. Mechanical operations that would result in the inadvertent removal of these trees were avoided.

Justification: Deciduous trees and shrubs are a critical habitat component for many birds that breed in the riparian areas (e.g., MacGillvray's warbler). Recommendation 3 helped ensure the retention of the deciduous overstory and understory.

Departure from Current Washington State Guidelines: Guidelines current at the time of harvest, specified that two live deciduous trees > 16" DBH and three live deciduous trees 12-16" DBH be left within the 30-50 ft standard RMZ. Recommendation 3 extended the area within which deciduous trees were protected and extended protection to all live deciduous trees. Current guidelines did not specifically protect shrubs. Recommendation 3 provided protection of shrubs within the standard RMZ width.

Recommendation 4: If past forest practices or tire history, had resulted in even-aged stands of $\leq$ 12” DBH conifers with an average of $\leq$ 5 ft spacing, single tree selection was used within the 100 ft zone of consideration to yield a post harvest spacing averaging 15-30 ft and representative of the surrounding forest.

Justification: The diversity of breeding bird species in riparian areas is associated with the structural diversity of the riparian habitat. Opening dense, even-aged stands promotes structural diversity by allowing development of the deciduous component and uneven aged forest stands.

Departure from Current Washington State Guidelines: Guidelines current at the time of harvest specified that all trees < 12” DBH be left within the 30-50 ft State RMZ. This recommendation provided for greater site-specific flexibility.

Goal: To provide for native wildlife as specified in the TFW’s wildlife goal.

Recommendation 5: If the riparian vegetation extended for > 1/3 the width of the 100 ft zone of special consideration (i.e., > 30 ft) then the zone of special consideration would be extended to encompass 1/3 riparian habitat and 2/3 upland habitat. In other words, in areas where the riparian zone was (extensive and riparian vegetation extended 40 ft from the stream, for example, the zone of consideration would extend to 120 ft from the stream.

Justification: This recommendation helped provide a balance of both riparian and upland habitat and reduced potential negative edge effects on the riparian habitat.

Departure from Current Washington State Guidelines: Guidelines current at the time of harvest resulted in RMZs that were variable in width depending upon width of riparian zone. This recommendation ensured that the zone of special consideration was also reflective of riparian zone width.

Table 3. Comparisons of riparian harvest prescription for State and Modified Buffers,

Feature	State	Modified
Buffer	30-50 ft buffer, with entry	100-ft 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 trees per 2 acres by 1.5 tree height
Seeps	Protect from machinery	Buffer by 30-B no entry zone extending to stream
Deciduous Trees	2 large or 3 smaller trees/acre	Leave all live trees
Shrubs	Avoid disturbance	Leave all within 30-50 ft of stream
Coniferous trees	Leave all < 12" DBH	Single tree selection
Delineation of RMZ	Extend RMZ to maximum width ofriparian zone	Extend zone of consideration to 1/3 riparian:2/3 upland If no specific habitat features present within zone of consideration, follow State RMZ

Chapter 3

‘WEST-SIDE VEGETATION

Abstract. We sampled vegetation along streams in managed, industrial forests to examine the relationship between habitat changes and wildlife responses to clearcut timber harvest when buffer strips are left adjacent to the stream. Vegetation sampling occurred once prior to timber harvest and once following timber harvest at 18 study sites in western Washington. We estimated the percentage cover of understory and midstory vegetation and down logs and counted trees and snags to quantify and compare wildlife habitat. Sampling transects ran parallel to the stream at 16 m and at 100 m from the stream. Forested riparian habitat contained significantly greater cover of rocks, bare soil, herbs, ferns, shrubs, and red alder trees (*Alnus rubra*) than forested upland habitat. Upland habitat had significantly greater cover of leaf litter and numbers of short and tall snags than riparian habitat. Following timber harvest, canopy cover was significantly greater at forested control sites than treatment sites with riparian buffer strips both within the buffers and in upland clearcuts. Within the riparian buffer, we found significantly greater cover of ferns, bare soil, regenerating vine maple (*Acer cercinatum*), and numbers of mid-sized alder trees on forested control sites following timber harvest than within buffer strips on harvested sites. Sites with either 15-m or 30-m buffer strips had significantly greater cover of berry producing shrubs and numbers of short snags (stumps). Upland sampling results showed greater numbers of medium and tall snags in forested controls compared to upland clearcuts. Although extreme habitat changes occurred along upland transects, few changes were measured within the buffer strips.

INTRODUCTION

A description of riparian and upland habitats provides the setting for examining vertebrate use of these areas. We adapted standard vegetation sampling methods to measure riparian and upland habitat characters at all sites both before and after timber harvest. We designed our methods to characterize the streamside habitat, upland habitat, and the transitional area between the two. Analysis of the pre-harvest vegetation data collected from all sites provides a comparison between riparian, upland and transitional riparian-upland habitats. Following timber harvest, comparisons of habitat measures between the Control, Modified, and State sites help explain vertebrate responses to the treatments. We included only stand-level measurements in this analysis and therefore limited our interpretations to the stand. All sites except two were located within private or state-owned timberlands that actively manage forests for timber harvest on 40-60 year rotations. The City of Seattle owns the Cedar River Watershed where we established two forested control sites. The watershed was managed for timber as well as water resources until just recently. The two control sites share a similar stand management history with the other sites.

METHODS

Sampling design

Habitat sampling at all 18 sites occurred in 1993, one year prior to timber harvest, and then once following timber harvest. Treatment sites were resampled in 1995; one yr following timber harvest and control sites were resampled in 1996. One treatment site, Ryderwood 1557, was not harvested until 1996, 2 yr after the others. We conducted post-treatment habitat sampling al: this site in 1998.

At all sites and in all sampling years, vegetation and habitat sampling quadrats were placed along riparian and upland transects relative to bird point count stations on both sides of the stream (Fig. 1). At each survey area four 8 × 10 m quadrats were delineated. Quadrats paralleled the stream for 10 m and ran 8 m perpendicular to the stream.

Along riparian transects, six habitat surveys were centered at 50-m intervals from the midpoint of bird point count stations 1 and 2 to point count station 4 on one side of the stream, and from the midpoint between point count stations 6 and 7 to point count station 9 on the other side. Thus, 12 areas were surveyed per site within the riparian zone describing both sides of the stream. Two quadrats (Q1, Q2) extended from the ordinary high water mark away from the stream 8 m. Quadrats 3 and 4 (Q3, Q4) extended from 8 m to 16 m away from the ordinary high water mark. Determination of the high water line was not always easy because surveys were conducted during periods of low stream flows. When evident, the point at which the slope of the bank increased dramatically and vegetation replaced rocky substrate was taken to be the high water mark. Small side channels, even if dry at the time, were considered part of the streambed. When laying out plots, quadrats 1 and 4 sometimes had to be offset from 2 and 3 because of a bend in the stream or a very steep slope. Priority was given to placing as much of tbe baseline at the high water mark as possible rather than keeping the quadrats contiguous with each other. The total sampling area was kept as consistent as possible.

Along upland transects, vegetation surveys were centered at 50-m intervals from bird point count station 2 to point count station 4 on one side of the stream and from point count station 7 to point count station 9 on the other. Five surveys were done along each upland transect for 10 upland surveys at each site. Quadrats were bounded on the upland side by the bird transects and extended 16 m perpendicular from the transect toward the stream. A list of acronyms, scientific, and common names of plant species mentioned in this chapter is provided in the Appendix.

Ground cover measurements

At 1, 4, 7, and 10 m from the streamside edge of vegetation quadrats, we placed 2 × 2-m and 1 × l-m plots and estimated percentage cover of herbaceous and woody vegetation and rock, litter and bare soil (Fig. 1). We grouped shrub species into three categories: berry-producing, evergreen, and other deciduous shrubs. Berryproducing shrubs included *Lonicera involucrata* (black twinberry), *Oemleria cerasiformis* (Indian-plum), *Ribes* spp. (currant and gooseberry), *Rosa* spp. (wild rose), *Rubus* spp. (blackberry, thimbleberry, salmonberry, and raspberry), *Sambucus* spp. (elderberry), *Sorbus sitchensis* (Sitka mountain-ash), *Vaccinium* spp. (blueberry and huckleberry), and *Viburnum edule* (highbush-cranberry). Other deciduous shrubs included *Ceanothus* spp., *Crataegus douglasii* (black hawthorn), *Holodiscus discolor* (oceanspray), *Linnaea borealis* (twinflower, although it is an evergreen shrub), *Oplopanax horridus* (Devil's club, although it is a berry producer), *Physocarpus capitatus* (Pacific ninebark), and *Spiraea douglasii* (hardhack). Evergreen shrubs included *Berberis* spp. (Oregon Grape), *Gaultheria shallon* (salal), *Juniperus communis* (common juniper), *Pachistima myrsinites* (mountain boxwood), and *Rhododendron macrophyllum* (Pacific rhododendron). In each category, we estimated percentage cover of shrubs <1 m high. Percentage cover of taller shrubs was measured in larger quadrats. Within the 1 × l-m plots we measured percentage cover of herbs, ferns, moss, grass, *Lobaria* lichen, seedlings <1 m tall, course woody debris >10 cm in diameter, litter, rock, and bare soil. Litter depth was measured in mm at two points on each 1 × 1-m plot.

Tall shrubs

Percentage cover of tall shrubs (l-3 m) was estimated in quadrats 2 and 3. Shrubs *were* identified in same three classes: berry-producing, evergreen, and other deciduous shrubs.

Down wood

Percentage cover of down wood (logs) was estimated in quadrats 2 and 3. Wood was considered down if its angle of incidence with the ground was less than 45". Each piece was categorized by diameter and decay class (DC). Two diameter classes, 10-30 cm and >30 cm diameter, and three decay classes were used. Decay class 1 describes structurally sound wood with intact limbs. Decay class 2 describes wood with reduced structural integrity and some limb loss. Decay class 3 describes wood with minimal structural integrity and presence of epiphytes.

Snags

Snags were counted within all four quadrats and grouped according to quadrat location. We counted short (<1.5 m), medium (1.5-15 m) and tall (>15 m) snags in three diameter classes (<10 cm diameter at breast height [DBH], 10-30 cm DBH, >30 cm DBH) and three decay classes (DC 1, all limbs attached and structurally sound; DC2, losing limbs and showing reduced structural integrity; DC3, about to fall down due to minimal structural integrity). The only small diameter snags counted were in the shortest height class.

Tree regeneration

Percentage cover of sapling tree species was estimated in quadrats 2 and 3. Saplings 1-3 m in height were included regardless of whether they grew from the ground, stumps, or down wood. Each individual species was identified and percentage cover estimated for that particular species, Trees <1 m in height were not included.

Tree counts

Trees >3 m tall were 'counted within all four quadrats and grouped according to quadrat (Q1 and Q2 or Q3 and Q4), diameter size (10 cm, 10-50 cm, 50-100 cm, >100 cm DBH), and species. All trees with split boles, except vine maple *(Acer cercinatum),* were counted as more than one tree if the split occurred below breast height. Vine maple clusters were counted as one tree because multiple stems is the typical morphology for vine maple. Trees with more than half of the base outside of the plot were not counted.

Buffer width

Vegetation sampling following timber harvest included measuring the width of the buffer strip from the stream edge perpendicular to the stream to the outermost tree. We did this at five different locations on both sides of the stream. Slope distance was measured rather than the distance of a horizontal plane from the last tree to the stream.

Canopy cover

Canopy cover readings were taken with a spherical, convex densiometer at outer corners of each quadrat and at the center point where the four quadrats met. At each of the five points, four readings were taken while facing the stream, away from the stream, upstream, and downstream and then averaged. In 1993, only one reading was taken at each corner while facing towards the center point and the center reading was taken facing the stream.

Statistical analyses

Comparing riparian and upland habitat

For all pm-harvest measures, we calculated overall means, standard deviation, minimum and maximum values of riparian and upland transect sampling. Means from riparian and upland transects were compared using paired t-tests with data from 17 sites to test the null hypothesis of no difference between riparian and upland habitat. Parameters for which we believed there to be an abrupt change from riparian to upland vegetation, riparian quadrats 1 and 2 (O-8 m from the stream) were compared to riparian quadrats 3 and 4 (8-16 m from the stream),

Comparing riparian habitat and upland habitat among treatment types

We used one-way ANOVA to test the null hypothesis of no difference among treatment types for each vegetation parameter measured in both riparian and upland transects. We calculated the difference between pre- and post-treatment means by subtracting the pre-treatment mean from the post-treatment mean and used the difference as the test statistic. Count data were log transformed before subtracting pre-treatment means from post-treatment means.

RESULTS

Riparian and upland habitat comparisons

We found significantly more red alder trees (*Alnus rubra*) in riparian than upland habitats ($P = 0.001$; Table 1). Alders 10-50 cm in diameter dominated the streamside area within 8 m of the stream. Outside this area, their numbers decreased significantly (P = 0.002; Table 1). Within the transitional zone, 8-16 m from the stream, numbers of Douglas-fir (*Pseudotsuga menziesii*) SO-1 00 cm in diameter and western hemlock (*Tsuga heterophylla*) 10-50 cm in diameter increase significantly (I' = 0.002, $P = 0.001$, respectively) when compared to streamside habitat (Table 1). The deciduous riparian canopy allows greater penetration of sunlight to the forest floor. Canopy cover was significantly greater in upland habitat than in riparian habitat (P = 0.016).

The percentage cover of herbaceous plants is influenced by tree composition and amount of sunlight that reaches the forest floor. Herbs and ferns cover more ground area in riparian habitat than upland habitat ($P < 0.001$, $P = 0.056$, respectively). Moss, grasses and lichen were present in similar and rather low densities in both habitat types. The coverage and depth of tree litter was significantly greater in upland habitat ($P < 0.001$, $P = 0.064$, respectively). Consequently, riparian habitat had a greater percentage of bare soil and rock cover than upland habitat ($P = 0.031$, $P = 0.009$, respectively).

Woody deciduous shrub cover was significantly greater in riparian than upland habitat ($P < 0.05$, Table 1). Both berry-producing and other deciduous shrubs followed this trend. Evergreen shrubs appear to be similarly abundant between riparian and upland habitats.

No significant differences in amounts of down wood in riparian and upland habitats were found. However, results show significantly more short and tall snags in upland than riparian habitats ($P = 0.005$, $P < 0.001$, respectively). Differences between medium height snags in riparian and upland areas were marginally significant ($P = 0.084$).

Treatment type comparisons

Riparian treatment comparisons were less obvious and more interesting than upland comparisons of forests and clearcut areas (Table 2). As our goal was to evaluate vertebrate use and habitat conditions in two riparian buffer strip configurations, we spent more time examining the riparian results than those from upland habitats.

After harvest, riparian buffer ships remained dominated by alder trees. Differences in tree counts in riparian habitat following timber harvest occurred on the outer riparian edge of sampling quadrats 3 and 4 where trees were cut. Buffer strip widths were significantly different between the two configurations ($P = 0.025$, Table 2). Modified buffer strips averaged (±1 SD) 30.5 ± 10.3 m and state regulation buffers averaged 15.4 ± 6.9 m. Numbers of red alder trees decreased at State and Modified sites with when compared to Control sites ($P = 0.062$). The numbers of Douglas-fir and western hemlock trees within both buffer strips were similar in number to those of forested control sites following timber harvest ($P > 0.10$). Photographs best summarize the changes reflected in upland tree counts (Fig. 2). Upland habitats were clearcut in both treatment types, leaving standing trees as required by law, approximately 2 trees per acre.

Following timber harvest, riparian canopy cover was significantly different in each of the treatment types ($P < 0.001$, Table 2). Control sites provided the densest riparian canopy cover, 90-100%, while buffers on State sites provided 20-75% coverage. Riparian canopy cover on Modified sites ranged from 40-90%. As expected, upland canopy cover of Control sites differed significantly from clearcuts of both State and Modified sites ($P < 0.001$).

Understory riparian vegetation changed slightly after timber harvest. Within riparian buffer strips, percentage cover of ferns and moss was significantly less than forested control sites ($P = 0.003$, $P = 0.069$, respectively). Streamside berry-producing shrubs were significantly greater in riparian areas with a clearcut edge than in forested control sites ($P = 0.02$). The percentage cover of bare soil within buffer strips decreased significantly ($P = 0.016$) when compared to forested control sites.

Percentage cover of regenerating saplings changed little following timber harvest. Vine maples in the outer riparian quadrats (3 and 4) decreased in state regulation buffer strips when compared to forested control sites ($P = 0.038$). We found no significant differences among treatment types of other regenerating tree species in riparian and upland habitats (Table 2).

There were no clear trends in changes in the cover of down wood. Within riparian buffers and upland clearcuts, amounts of new down wood, decay class 1, tended to increase but results were not statistically significant (Table 2). Small pieces of down wood, 10-30 cm diameter and decay class 3, in upland clearcut sites decreased significantly when compared to forested control sites ($P = 0.001$).

Timber harvest affected snag densities in riparian quadrats 8-16 m from the stream more than along the stream at sites with state regulation buffers. Snags <1.5 m in height (stumps) increased significantly following timber harvest in the outer riparian habitat ($P = 0.047$). Numbers of medium height snags (1.5-15 m) decreased significantly in both the streamside (Q1 and Q2), outer (Q3 and Q4) riparian, and upland quadrats of clearcut sites (P

= 0.012, $P = 0.01$, $P < 0.01$, respectively). The number of tall snags >15 m decreased in upland clearcuts when compared to forested controls (P = 0.069).

DISCUSSION

Riparian and upland habitat comparisons

For many of the parameters measured, we found no difference between riparian and upland habitat before timber harvest (Table 1). Results help establish a framework from which to interpret vertebrate species distributions. Vertebrates able to move great distances utilize many habitat types within a forest while others with limited movement capabilities must meet their needs in smaller areas or forgo reproduction, reduce activity, emigrate, or die. Extensive changes in the vegetation community, for example following timber harvest, alter the vertebrate community because they dramatically change the resources and microhabitat conditions upon which animals depend (Aubry et al. 1998).

Riparian areas often provide habitat that is denser in the under- and mid-storms than upland habitats. 'This occurs when hydrophilic deciduous trees dominate streamside areas and allow more light to reach the understory. Streamside areas provide habitat for vertebrate species needing resources provided by deciduous and understory vegetation. Our results support this generalization. Alder, devil's club, *Rubus* spp. and *Sambucus* spp. dominated riparian habitats, providing fruits, seeds and shrub materials for various vertebrate species, The more abundant herbs and ferns on the riparian floor provide resources for herbivorous vertebrates. Coniferous trees of upland habitats of managed forests create dense canopies that prevent sunlight from reaching the forest floor, limiting the amount of understory vegetation, Snag densities were greater in upland than riparian habitats. Densities of animals that nest or roost in cavities may be higher in upland areas due to the significantly fewer snags found in riparian habitats, Down wood appeared to be available in similar quantities in riparian and upland habitats. Rock cover, deposited by the stream during flood events and from rockfall down valley slopes, was greater in riparian areas whereas conifer tree litter dominates upland ground cover. The dynamic nature of stream courses and adjacent habitat may restrict vertebrate use to those animals adapted to very moist and sometimes inundated areas.

Treatment type comparisons

The most striking differences between vegetation in the two buffer strip configurations and the forested control were in canopy cover and numbers of snags. Buffers on the Modified sites contained more trees than those on the State sites. Consequently, they provided significantly more canopy cover. The decreased canopy cover in riparian buffer strips when compared to Control sites had only slightly altered riparian understory vegetation 2 yr following timber harvest.

Fern cover and amount of bare soil decreased in both riparian buffer strips configurations. Although the increase in litter cover was only marginally significant in buffer strips, this increase can explain the decrease in bare soil spots. Increases in insolation due to clearcutting appears to have stimulated growth of berry-producing shrubs within the buffer strips. Brosofske et al. (1997) report decreases in relative humidity and increases in solar radiation and daytime surface temperatures at buffer strip edges and within riparian buffer strips. Vegetation may not respond immediately to these microclimatic changes following timber harvest. Brosofske et al. (1997) recommend leaving a buffer strip of at least 45 m to maintain pre-harvest riparian microclimate. Buffers on the Modified sites ranged from 17-58 m, with an average of 30.52 ± 10.27 m). Microclimate changes during the first 2 yr after harvest did not affect overall percentage cover of herbs, grasses and most shrubs, The actual species composition of these vegetation classes, however, was not measured and may change as a result of timber harvest.

Numbers of short snags, or stumps, increased within buffers on the State sites when compared to Modified and Control sites. Some cutting occurred within the sampling areas of state regulation buffers. Clearcutting occurred as close as 5.4 m to the stream at one State site. In addition to the increase in stumps, a significant number of mid-sized snags was removed from the buffers on the State sites during timber harvest. Although the decrease of mid-sized red alder trees within buffer strips is most likely due to effects of timber harvest, the difference was only marginal ($P = 0.062$).

For the most part, buffers on the State sites were sampled from the stream edge to the buffer edge and buffers on the Modified sites from the stream edge to approximately half their width. This sampling area can be thought of as the core area of the buffer, surrounded by an additional 15 m of forest to the clearcut edge. Whether or not this additional buffer width will have a significant effect on near-stream vegetation remains to be seen.

Vegetation differences among treatments in upland transects are the straightforward results of habitat changes following clearcut harvesting. Cover of ferns and moss was significantly reduced in clearcuts compared to forested sites while grass and lichen cover showed marginally significant increases in clearcut sites. Low levels of light due to dense canopy cover appeared to limit grass and lichen growth in forests, while moisture may limit fern and moss growth in clearcuts. The amount of down wood on upland transects remained similar among treatment types or increased slightly in clearcuts.

Decreases in snag counts in clearcuts are of greatest concern. Snags >1.5 m in height, of varying diameters and decay class were significantly less abundant on clearcut sites than forested sites. This result supports the findings of a previous TFW upland landscape study

(Aubry et al. 1998). Furthermore, we found significantly more snags in upland forest than in riparian forest habitat. Protecting streams with limited riparian buffers does not mitigate for snag loss from clearcutting upland forests. The collective decrease of this important wildlife resource from harvested sites merits attention. Snags provide essential foraging, roosting, nesting and perching habitat for many species of vertebrates, including woodpeckers, raptors and bats. Bark slough and down wood from old trees that become snags provide habitat for terrestrial salamanders and small mammals. Forest managers need to consider protecting existing snags and creating new snags within riparian and upland habitat during harvest operations to mitigate the apparent loss of snags in managed forests.

Overall, habitat within buffers on the Modified sites appears very similar to riparian habitat on Control sites. Post-harvest sampling occurred within 2 yr of actual clear-cutting operations. Thus, subsequent changes or their lack has yet to be documented. The modified buffer strip configuration designed for this project may functionally maintain forest riparian conditions through a full harvest rotation. The ultimate test of the suitability of riparian buffer strips in providing habitat for wildlife rests in documenting changes in the vertebrate community.

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Table 1. Comparisons of riparian and upland habitat prior to timber harvest (paired t-test).

Habitat variables	No significant difference ($P > 0.10$)	Marginally significant difference ($0.10 \geq P \geq 0.05$)	Significant difference ($P < 0.05$)
canopy cover (%)			R<U
Ground cover (%)	moss grass lichen down wood	litter depth (R<U) ferns (R>U)	herbs (R>U) litter (R<U) soil (R>U) rock (R>U)
Cover of shrubs >1 m (%)	evergreen shrubs, Q2 evergreen shrubs Q3		berry-producing, Q2 (R>U) berry-producing, Q3 (R>U) other deciduous, Q2 (R>U) other deciduous, Q3 (R>U)
Cover down wood (%)	LO-30 cm diam., DC1, Q2 IO-30 cm diam., DC2, Q2 10-30 cm diam., DC3, Q2 to-30 cm diam., DC1, Q3 10-30 cm diam., DC2, Q3 10-30 cm diam., DC3, Q3 >30 cm diam., DC1, Q2 >30 cm diam., DC2, Q2 >30 cm dim., DC3, Q2 >30 cm dim., DC1, Q3 >30 cm dim., DC2, Q3 >30 cm diam., DC3, Q3		
N". of short snags, <1.5 m	<10 cm dim., DC3, Q1Q2 10-50 cmdiam., DC1, Q1Q2 >50 cm dim,., DC2, Q1Q2 <10 cm diam., DC3, Q3Q4 IO-50 cm diam., DC1, Q3Q4 10-50 cm diam., DC2, Q3Q4 IO-50 cm dim., DC3, Q3Q4 >50 cm diam., DC2, Q3Q4	<10 cm diam., dc1, Q1Q2 (R<U) IO-50 cm diam., dc2, Q1Q2 (R<U) <10 cmdiam., dc2, Q3Q4 (R<U) <10 cm diam., dc1, Q3Q4 (R<U)	<10 cmdiam., dc2, Q1Q2 (R<U) 10-50 cm diam., dc3, Q1Q2 (R<U) >50 cm diam., dc3, Q1Q2 (R<U) >50 cm diam., dc3, Q3Q4 (R<U)
Med. snags, 1.5-15 m	10-50 cmdiam., DC1, Q1Q2 LO-50 cm diam., DC2, Q1Q2 10-50 cm dim., DC3, Q1Q2 >50 cm diam., DC1, Q1Q2 >50 cm diam., DC2, Q1Q2 IO-50 cm diam., DC2, Q3Q4 10-50 cm diam., DC3, Q3Q4 >50 cm diam., DC 1, Q3Q4 >50 cm diam., DC2, Q3Q4 >50 cm diam., "C3, Q3Q4		10-50 cm diam., dc1, Q3Q4 (R<U)
Tall snags, >15 m	IO-50 cmdiam., DC3, Q1Q2 >50 cmdiam., DC1, Q1Q2 >50 cm diam., DC2, Q1Q2 >50 cm diam., DC3, Q1Q2 IO-50 cm diam., DC3, Q3Q4 >50 cm diam., DC1, Q3Q4 >50 cm diam., DC2, Q3Q4		10-50 cmdiam., dc1, Q1Q2 (R<U) IO-50 cm diam., dc2, Q1Q2 (R<U) 10-50 cm diam., dc1, Q3Q4 (R<U) 10-50 cm diam., dc2, Q3Q4 (R<U)
Tree count, Riparian Q2 vs. Q3	ACCI <10 cm diam. ALRU, <10 cm diam ALRU, 50-100 cm diam. PSME, <10 cm diam. THPL, <10 cm diam. THPL, 10-50 cm diam. THPL, 50-100 cm diam.	PSME, 10-50 cm diam. (R1,2<R3,4) TSHE, <10 cm diam. (R1,2<R3,4) TSHE, 50-100 cm diam. (R1,2<R3,4)	ALRU, IO-50 cm diam. (R1,2>R3,4) PSME, 50-100 cm diam. (R1,2<R3,4) TSHE, 10-50 cm diam. (R1,2<R3,4)

Table 2. Comparisons of hahitat measures among treatments (ANOVA). For post-hoc comparisons (Tukey) C = Control sites, M = Modified sites, and S = State sites. Probability values for statistically significant comparisons listed in bold font.

Habitat variable	Riparian *P*	Tukey	Upland *P*	Tukey
canopy cover (%)	**<0.001**	C≠M, C≠S, M≠S	**<0.001**	C≠M, C≠S
Buffer strip width	0.025	M>S	NA	
Ground cover				
Herbs	0.371		0.883	
Ferns	0.003	C≠M, C≠S	0.017	C≠M
Moss	0.069		0.013	C≠S
Grass	0.43 1		0.060	
Lichen	0.388		0.062	
Litter	0.094	C≠M	0.374	
CWD	0.232		0.457	
Soil	0.016	C≠M, C≠S	0.568	
Rock	0.544		0.066	
Tall shrub cover				
Q2 berry-prod.	**0.020**	C≠M, C≠S	0.765	
Q3 berry-prod.	0.499		0.510	
Q2 other decid.	0.847		0.770	
Q3 other decid.	0.145		0.905	
Q2 evergreen	0.224		0.399	
Q3 evergreen	0.366		0.582	
Tree regeneration				
ACCI, Q2	0.622		0.549	
ACCI, Q3	0.038	C≠S	0.830	
TSHE, Q2	0.097		0.187	
TSHE, Q3	0.198		0.065	
ALRU, Q2	0.526		0.217	
ALRU, Q3	0.965		0.223	
PSME, Q2	0.111		0.774	
PSME, Q3	0.315		0.154	
THPL, Q2	0.727		NA	
THPL, Q3	0.577		NA	

Table 2. Continued.

Habitat variable	Riparian *P*	Tukey	Upland *P*	Tukey
Down wood				
10-30 cm, Decay Class 1, Q2	0.753		0.290	
10-30 cm, Decay Class 2, Q2	0.842		0.389	
10-30 cm, Decay Class 3, Q2	0.281		0.227	
>30 cm, Decay Class 1, Q2	0.354		0.199	
>30 cm, Decay Class 2, Q2	0.539		0.326	
>30 cm, Decay Class 3, Q2	0.230		0.369	
10-30 cm, Decay Class 1, Q3	0.173		0.265	
10-30 cm, Decay Class 2, Q3	0.889		0.413	
10-30 cm, Decay Class 3, Q3	0.054		**0.001**	C≠M, C≠S
>30 cm, Decay Class 1, Q3	0.094		0.936	
>30 cm, Decay Class 2, Q3	0.802		0.526	
>30 cm, Decay Class 3, Q3	0.929		0.641	
10-30 cm, All Decay Classes, Q2	0.661		0.081	
>30 cm, All Decay Classes, Q2	0.567		0.586	
10-30 cm, All Decay Classes, Q3	0.580		0.977	
>30 cm, All Decay Classes, Q3	0.680		0.892	
Snags, <1.5 m, Q1Q2				
<10 cm diameter, DC1	0.517		0.359	
<10 cm diameter, DC2	0.773		0.888	
<10 cm diameter, DC3	0.515		0.440	
10-30 cm diameter, DC1	0.315		**<0.001**	C≠M, C≠S, M≠S
10-30 cm diameter, DC2	**0.010**	C≠S	0.030	C≠M, C≠S
10-30 cm diameter, DC3	0.435		0.761	
>30 cm diameter, DC1	0.270		**<0.001**	C≠M, C≠S
>30 cm diameter, DC2	0.039	M≠S	0.039	C≠M, C≠S
>30 cm diameter, DC3	0.030	C≠M	0.133	
ALL diameter, ALL DC	0.197		**<0.001**	C≠M, C≠S
Snags, 1.5-15 m, Q1Q2				
10-30 cm diameter, DC1	0.564		0.423	
10-30 cm diameter, DC2	0.020	C≠S, M≠S	0.208	
10-30 cm diameter, DC3	0.370		0.017	C≠M, C≠S
>30 cm diameter, DC1	0.391		0.087	
>30 cm diameter, DC2	0.810		0.682	
>30 cm diameter, DC3	0.004	C≠S, M≠S	0.036	C≠S
All diam, all DC	**0.012**	C≠S, M≠S	0.007	C≠M, C≠S

Table 2. Continued

Habitat variable	Riparian P	Tukey	Upland P	Tukey
Snags, >15 m, Q1Q2				
10-30 cm diameter, DC1	0.649		0.207	
10-30 cm diameter, DC2	0.892		0.282	
10-30 cm diameter, DC3	NA		0.152	
>30 cm diameter, DC1	NA		NA	
>30 cm diameter, DC2	NA		NA	
>30 cm diameter, DC3	NA		NA	
ALL diameter, ALL DC	0.588		0.069	
Snags, <1.5 m, Q3Q4				
<10 cm diameter, DC1	0.670		0.276	
<10 cm diameter, DC2	0.424		0.460	
<10 cm diameter, DC3	0.336		0.387	
10-30 cm diameter, DC1	0.015	C≠S, M≠S	**0.001**	C≠S, M≠S
10-30 cm diameter, DC2	0.124		**0.002**	C≠M, M≠S
10-30 cm diameter, DC3	0.637		0.363	
>30 cm diameter, DC1	0.066		**<0.001**	C≠M, C≠S
>30 cm diameter, DC2	0.071		0.006	C≠M, C≠S
>30 cm diameter, DC3	0.396		0.068	
ALL diameter, ALL DC	0.047	C≠S	0.002	C≠M, C≠S
Snags, 1.5-15 m, Q3Q4				
10-30 cm diameter, DC1	0.349		0.835	
IO-30 cm diameter, DC2	0.019	M≠S	0.003	C≠M, C≠S
IO-30 cm diameter, DC3	0.011	M≠S	0.012	C≠M, C≠S
>30 cm diameter, DC1	0.136		NA	
>30 cm diameter, DC2	NA		NA	
>30 cm diameter, DC3	0.018	C≠S	0.003	C≠M, C≠S
ALL diameter, ALL DC	0.010	C≠S, M≠S	**<0.001**	C≠M, C≠S
Snags, >15 m, Q3Q4				
10-30 cm diameter, DC1	0.296		0.423	
10-30 cm diameter, DC2	0.853		0.334	
10-30 cm diameter, DC3	0.278		0.010	C≠M, M#S
>30 cm diameter, DC1	NA		NA	
>30 cm diameter, DC2	NA		NA	
>30 cm diameter, DC3	NA		NA	
ALL diameter. all DC	0.873		0.385	

Table 2. Continued.

Habitat variable	Riparian P	Tukey	Upland P	Tukey
Tree counts				
PSME, <10 cm, Q1Q2	0.705			
PSME, 10-50 cm, Q1Q2	0.396			
PSME, 50-100 cm, Q1Q2	0.191			
PSME, >100 cm, Q1Q2	NA			
PSME, <10 cm, Q3Q4	0.289			
PSME, 10-50 cm, Q3Q4	0.138			
PSME, 50-100 cm, Q3Q4	0.128			
PSME, >100 cm, Q3Q4	NA			
TSHE, <10 cm, Q1Q2	0.368			
TSHE, 10-50 cm, Q1Q2	0.08			
TSHE, 50-100 cm, Q1Q2	NA			
TSHE, >100 cm, Q1Q2	NA			
TSHE, <10 cm, Q3Q4	0.306			
TSHE, 10-50 cm, Q3Q4	0.174			
TSHE, 50-100 cm, Q3Q4	0.218			
TSHE, >100 cm, Q3Q4	NA			
THPL, <10 cm, Q1Q2	0.185			
THPL, 10-50 cm, Q1Q2	0.341			
THPL, 50-100 cm, Q1Q2	0.695			
THPL, >100 cm, Q1Q2	NA			
THPL, <10 cm, Q3Q4	0.491			
THPL, 10-50 cm, Q3Q4	0.471			
THPL, 50-100 cm, Q3Q4	NA			
THPL, >100 cm, Q3Q4	NA			
ALRU, <10 cm, Q1Q2	NA			
ALRU, 10-50 cm, Q1Q2	0.748			
ALRU, 50-100 cm, Q1Q2	0.456			
ALRU, >100 cm, Q1Q2	NA			
ALRU, <10 cm, Q3Q4	0.119			
ALRU, 10-50 cm, Q3Q4	0.062	C≠M, C≠S		
ALRU, 50-100 cm, Q3Q4	0.814			
ALRU, >100 cm, Q3Q4	NA			
ACCI, <10 cm, Q1Q2	0.425			
ACCI, 10-50 cm, Q1Q2	0.393			
ACCI, <10 cm, Q3Q4	0.629			
ACCI, 10-50 cm, Q3Q4	0.431			

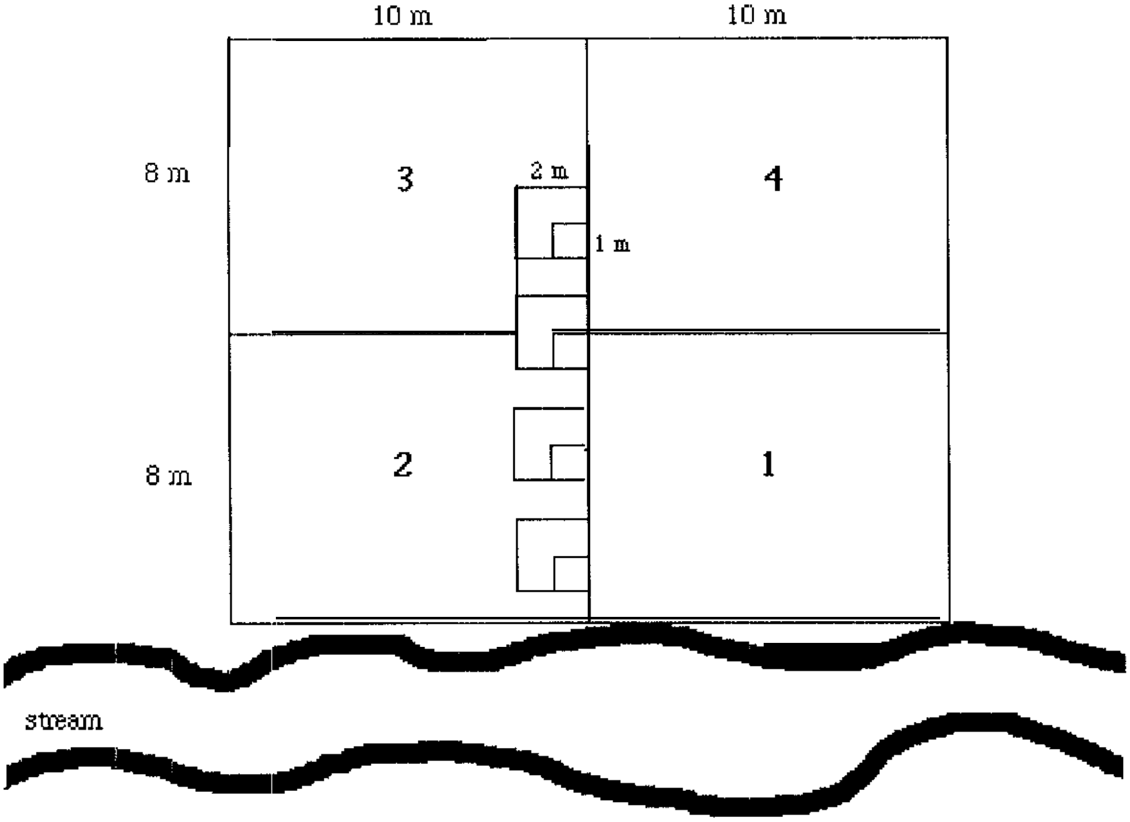

Figure 1: Vegetation sampling design. Understory vegetation cover was measured in 1 x I-m and 2 x 2-m plots. Down wood, shrubs, snags, and trees were measured in 8 x 1 O-m quadrats. Riparian sampling ran along the stream on both sides. Upland sampling occurred 100 m from the stream. Sampling quadrats were coincided with bird point count stations at 100-m intervals and at 50-m intervals in between the point count stations.

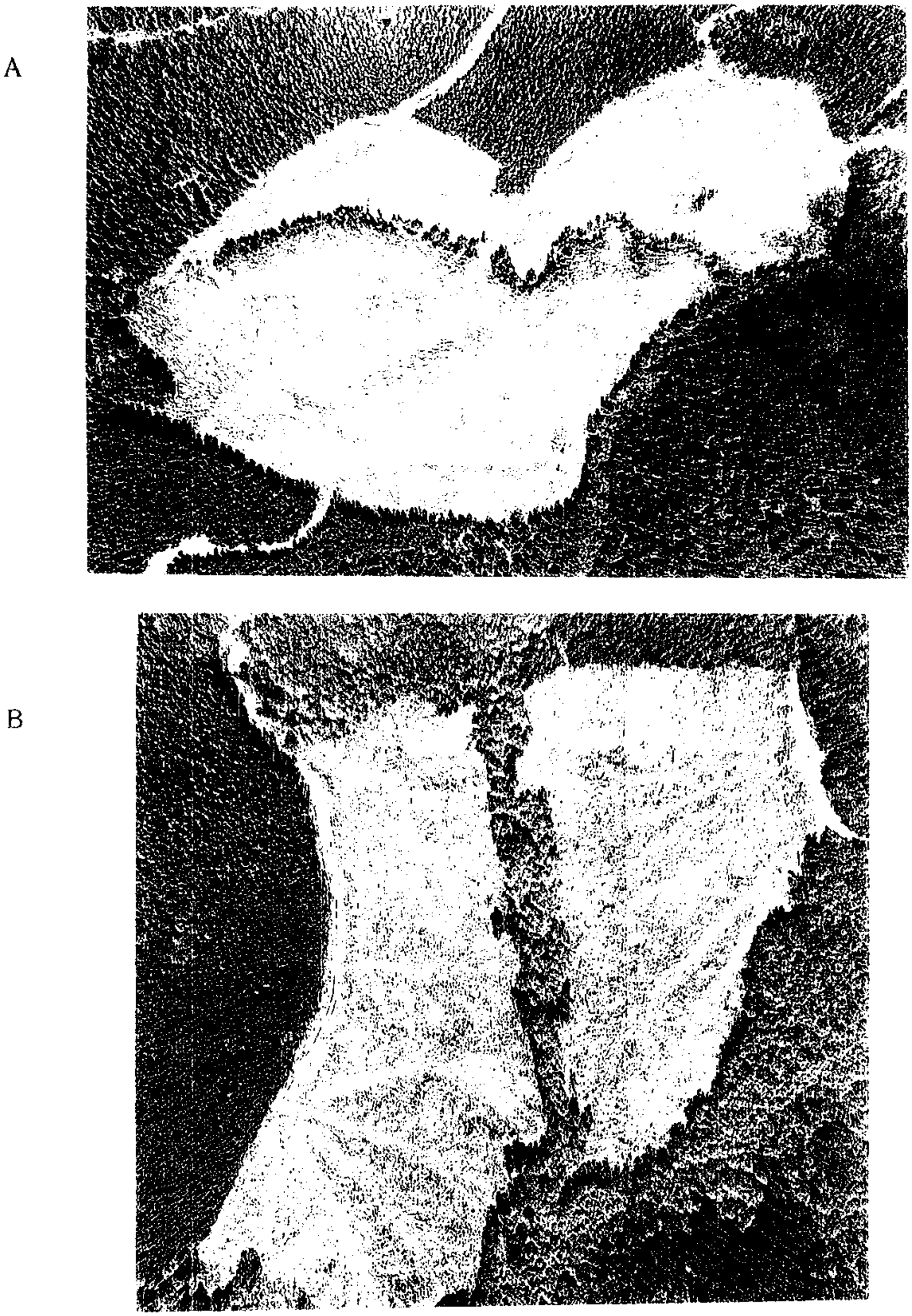

Figure 2: Aerial photograph:; illustrating post-harvest buffet-s of A) a State site and B) a Modified site.

APPENDIX

Acronyms, scientific, and common names of plant species mentioned in the text:

TREES

ABGR - *Abies grandis*, Grand Fir
ACCI - *Acer cercinatum,* Vine Maple
ACMA - *Acer macrophyllum,* Bigleaf Maple
ALRU - *Alnus rubra,* Red Alder
PIEN - *Picea engelmannii,* Englemann Spruce
POTR - *Populus trichocarpa,* Black Cottonwood
PRSP - *Prunus species,* Cherry
PSME *Pseudotsuga menziesii,* Douglas-fir
QUCA - *Quercus garryana,* Oregon White Oak
RHPU - *Rhamnus purshiana,* Cascara Buckthom
SASP - *Salix species,* Willow
TABR - *Taxus brevifolia,* Pacific Yew
THPL - *Thuja plicata.* Western Red Cedar
TSHE *Tsuga heterophylla,* Western Hemlock
UNK - Unknown

SHRUBS

Berry Producing

LOIN *Lonicera involucrata,* black twinberry
OECE - *Oemleria cerasiformis,* Indian-plum
RISP - *Ribes* spp., currant and gooseberry
ROSP - *Rosa* spp., wild rose
RUSP *Rubus* spp., blackberry, thimbleberry, salmonberry, raspberry
SASP *Sambucus* spp., elderberry
SOSI *Sorbus sitchensis,* Sitka mountain-ash
VASP *Vaccinium* spp., blueberry and huckleberry
VIED - *Viburnum edule,* highbush-cranberry

Other Deciduous Shrubs

CESP - *Ceanothus* spp., Ceanothus
CRDO *Crataegus douglasii,* black hawthorn
HOD1 *Holodiscus discolor,* oceanspray
LIBO - *Linnaea borealis,* twinflower (although it is an evergreen shrub)
OPHO - *Oplopanax horridus,* Devil's club (although it is a berry producer)
PHCA - *Physocarpus capitatus,* Pacific ninebark
SPDO - *Spiraea douglasii,* hardhack

Evergreen Shrubs

BESP *Berberis* spp., Oregon Grape
GASH - *Gaultheria shallon,* salal
JUCO - *Juniperus communis,* common juniper
PAMY - *Pachistima myrsinites,* mountain boxwood
RHMA - *Rhododendron macrophyllum,* Pacific rhododendron

Chapter 4

EAST-SIDE VEGETATION

Abstract. The structural and floristic components of 18 riparian and adjacent upland managed forest sites in northeast Washington were characterized before and after logging to compare the two habitats, examine structural and floristic changes following different riparian timber harvest prescriptions, and to provide information to analyze habitat associations of terrestrial vertebrates. Seven of the sites served as unharvested controls and the uplands of 11 were logged following a partial harvest prescription. The riparian habitat of six sites were harvested according to the Washington State guidelines for Riparian Management Zones (State) and those of five sites were harvested according to a modified riparian harvest that identified and protected habitat features such as seeps and snags (Modified). Structural habitat conditions varied in only a few respects between riparian and upland habitats prior to harvest. These differences included greater dispersion of shrubs in riparian that upland habitats, larger and more decayed woody debris and more natural stumps in the riparian habitat, and more deciduous trees in the riparian habitat. Few species of shrubs and herbs were unique to either riparian or upland habitats. Most species of shrubs were more abundant in the upland habitat, but the pattern was reversed for herbaceous vegetation. Riparian buffers on the Modified sites were wider, but considerably more variable that those on the State sites. Logging reduced overstory and understory canopy, reduced the mean height of snags and trees, and decreased the number of shrubs, trees, and snags on the cut as compared to Control sites. There were few differences in habitat structure between the State and Modified sites. Floristically, there were greater changes on the State than Modified sites following logging. The abundance of shrubs and most herbaceous plants was decreased in the riparian habitat of the State sites as compared to either the Modified or Control sites.

INTRODUCTION

The structure and function of riparian habitats are determined by several key elements including topography, surface water, soils, microclimate, and vegetation. Important among these for understanding vertebrate use of these habitats, especially if altered by timber harvest, is the structure and composition of the vegetation. The structure of the vegetation refers to the horizontal and vertical stratification of the plant community. Composition refers to the species richness and abundance. Riparian areas typically have greater structural and species diversity than upland areas, but differences between these habitats are influenced by natural factors such as slope, aspect, size of stream, proximity of drainages to one another, disturbance patterns, and by anthropogenic factors such as timber harvest. Timber harvest affects both the structure and composition of forest vegetation and assessing the response of vertebrates to habitat changes in the riparian and adjacent uplands requires description of habitat conditions. In the following we first describe the riparian and upland habitats prior to timber harvest and then compare habitat conditions between the Control, Modified, and State sites after harvest.

Northeastern Washington is part of the upper Columbia drainage system and the numerous creeks in the area drain primarily into the Pend Oreille or Colville Rivers, which merge with the Columbia. The forested lands of the region are managed primarily by the U.S. Forest Service, U.S. Fish and Wildlife Service, Washington Department of Natural Resources, private timber companies, and private landowners. Timber management practices during the past 100 yr have included high-grading, clearcutting, post harvest burning with replanting, fire suppression, selective cutting, and uneven-aged management. These management practices, coupled with natural variation in forest structure due to slope, elevation, aspect, edaphic characteristics, and fire, have resulted in a mosaic of forest stands of varying structure. Recognizing that variation is inherent to these forests, we selected 18 riparian forest stands that met the following criteria: mature, mixed coniferous forest that had been previously harvested, similar elevation, and similar stream type. Because of the importance of both forest structure and composition to terrestrial vertebrates, we adapted standard vegetation sampling methods to examine structural as well as floristic components of the riparian and adjacent upland habitats on these 18 sites before and after timber harvest. Seven of these sites served as unharvested controls and the uplands of 11 were logged following a partial harvest prescription. The riparian habitat of 6 sites were harvested according to the Washington State guidelines for Riparian Management Zones (State) and those of 5 sites were harvested according to a modified riparian harvest that identified and protected habitat features such as seeps and snags (Modified).

METHODS

Habitat structure

To examine structural habitat characteristics, we established a 16 × 20-m plot at each 50-m interval along the riparian and upland transects for a total of 15 riparian plots and 15 upland plots per site. Each pllot was divided into four quadrants (8 × 10 m; Fig. 1). Pre-harvest sampling of plots was conducted in August 1992. Post-harvest sampling of all State and two Modified sites was conducted in July 1995 and of the three remaining Modified sites in July 1996.

Shrubs

We examined the composition and dispersion of taller shrubs (>0.5 m high) on the 15 riparian and 15 upland plots. From the center point of each plot, the distance to the nearest shrub in each of the 4 quadrants was measured, and we recorded area (length × width) and species of each shrub.

Down wood and stumps

In two opposite quadrants, the number and decay class of woody debris and stumps were recorded. Logs were assigned to one of four size classes and to one of four decay classes. Size classes were: 1) >5m long and <15 cm diameter, 2) >5 m long and 16-24 cm diameter, 3) >5 m long and >25 cm diameter, and 4) <5 m long and >25 cm diameter. Decay classes were defined as: 1) freshly fallen tree with bark essentially intact, wood solid, no decomposition; 2) bark beginning to slough or almost completely gone, decomposition begun with sapwood partially softened by log generally still firm; 3) decomposition progressed to a point that wood is generally soft and breaks into chunks, each chunk still has integrity; 4) essentially no integrity to log, wood has decomposed to point of soil-like texture. Stumps were assigned as either "natural" or "cut" and to one of the four above decay classes. Stumps were differentiated from snags by height; stumps were <1.37 m.

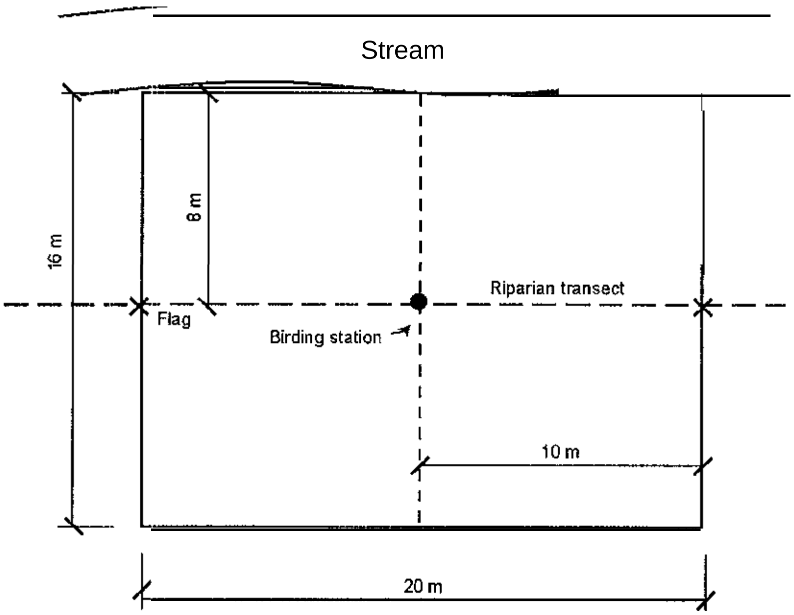

Figure 1. Sampling scheme for habitat measurements at plot stations.

Trees and snags

Within each plot all trees were recorded by species and assigned to one of four DBH size classes: 1) 4-10 cm, 2) 1 l-25 cm, 3) 26-50 cm, 4) >50 cm. All snags within each plot were counted and designated as either Condition 1 (bark basically intact) or Condition 2 (bark peeling off to absent). Four average live trees and two snags were chosen at random and their heights were estimated using a clinometer.

Canopy cover

Using a convex densiometer, percentage of overstory and understory cover were measured at the center of each plot and at the center of each quadrant. We averaged the five measurements per plot for each variable.

Tree regeneration

In two opposite quadrants, the number of regenerating coniferous trees (>0.5 m high, <4 cm DBH) was recorded.

Floristics

We evaluated floristic diversity by establishing 30-m point-intercept transects between each of the above plots (Fig. 1) for a total of 14 riparian and 14 upland transects per site. Pre-harvest sampling of the point-intercept transects was conducted in August 1992. All cut sites were resampled 2 yr post harvest in either July 1995 or July 1996. Control sites were resampled in July 1995.

Ground cover

We measured the composition of the ground vegetation and characterized the litter using 14 30-m point-intercept on each riparian transect and the upland transect. Vegetation was measured at 50-cm increments along the 30-m transect. 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 (1.5 m, 1 .O m, 0.5 m, 0.25 m, and 0 m). Herbaceous plants, shrubs, ferns and trees were recorded to species. Grasses were recorded as present and were not identified to species.

Logs were assigned to one of six size classes and to one of four decay classes. Size classes were: 1) <5 m long and <15 cm diameter, 2) <5 m long and 16-24 cm diameter, 3) >5m long and <15 cm diameter, 4) >5 m long and 16-24 cm diameter, 5) >5 m long and >25 cm diameter, 6) <5 m long and >25 cm diameter. Decay classes were defined as: 1) freshly fallen tree with bark essentially intact, wood solid, no decomposition; 2) bark beginning to slough or almost completely gone, decomposition begun with sapwood partially softened by log generally still firm; 3) decomposition progressed to point that wood is generally soft and breaks into chunks, each chunk still has integrity; 4) essentially no integrity to log, wood has decomposed to point of soil-like texture.

Stumps were assigned as either "natural" or "cut" and to one of the above decay classes. Stumps were differentiated from snags by height; stumps when il.37 m high (standard breast height). Woody debris was assigned to the same size and decay classes listed above, with the exception that two additional size classes (<5 m long and <15 cm diameter; <5 m long and 16-24 cm diameter) were recorded.

Litter depth was measured every 5 m on each transect for a total of seven depth measurements per 30-m point-intercept transect. This yielded a total of 98 measurements per riparian or upland transect for a total of 196 measurements per site.

Physical features and buffer width

Slope was determined using a clinometer and aspect was measured with a compass at each plot. After harvest on the Modified and State sites we measured the buffer width as the perpendicular distance from the stream to the edge of the riparian harvest unit (boundaries had been marked prior to harvest). We measured the buffer width at 17 points spaced 50 m apart along the riparian transect.

Statistical analyses

Habitat structure

For all variables with, count data (e.g., logs, stumps, trees), we obtained total counts per plot. For all other variables, we calculated the mean for each plot. To characterize differences between riparian and upland habitats, we conducted analysis of variance (ANOVA) on the overall means of each variable for each site and habitat type.

We examined changes in habitat variables between pre- and postharvest, modified and state harvest, and riparian and upland habitats using factorial ANOVA. Tests were conducted on the means for each site and habitat type. We excluded control sites for these analyses.

Floristics

We examined species; richness of herbaceous and shrub vegetation on the point-intercept transects by two measures: mean number of species per 30-m point intercept transect per site and mean number of species per site. We used one-way ANOVA to compare species richness between riparian and upland habitats and repeated measures ANOVA to examine changes in species richness due to harvest treatment. To examine the relative abundance of herbaceous plant and shrub species, we summed the number of intercepts per species per 30-m transect. Each species was counted only once per point-intercept (i.e., a taller plant might be recorded at two different heights per point intercept, but we only counted it: once for analysis). We used one-way ANOVA and a Tukey's mean separation test to compare the mean number of intercepts for each species between riparian and upland habitats and between the pre- and post-harvest sampling.

To examine the ground surface we compared mean litter depth and mean number of litter and bare soil intercepts per 30-m point-intercept transect per site between riparian and upland habitats and between pre and post harvest samples in both habitats. We averaged the seven litter depths per 30-m point-intercept transect. To compare litter depths between riparian and upland habitats before harvest, we used one-way ANOVA and a Tukey's mean separation test. To examine the effects of the different harvest treatments, we subtracted the mean litter depth for each 30-m point-intercept transect sampled in 1992 from the mean for the same transect sampled in 1995. We then compared the differences per site using one-way ANOVA and a Tukey's mean separation test. We summed the number of litter and bare soil intercepts at ground level (i.e., height = 0) per 30-m transect and used one-way ANOVA and a Tukey's mean separation test to compare the mean number of intercepts for each species between riparian and upland habitats. To examine the effects of the different harvest treatments, we subtracted the mean litter and bare soil intercepts for each 30-m point-intercept transect sampled in 1992 from the mean for the same transect sampled in 1995. We then compared the differences per site using one-way ANOVA and a Tukey's mean separation test (SAS Institute 1989).

RESULTS

Riparian and upland habitats

Habitat structure

Prior to harvest, mean distance to the nearest shrub was significantly greater in riparian than in upland habitats (Table 1). Shrub area and shrub height did not differ between riparian and upland habitats. Only minor habitat differences were observed for logs or stumps. Small diameter logs in decay class 2 were more common in the upland, whereas two classes of the largest diameter logs were more abundant in the riparian (Table 1). Natural stumps in advanced decay were more common in riparian areas.

Overstory canopy cover was significantly greater in the riparian, but the magnitude of the difference was small. Deciduous trees <25 cm DBH were more abundant in riparian than upland habitats, as were large conifers (Table 1). Numbers of snags did not differ between upland and riparian except for the greater number of large (>50 cm DBH) condition 2 trees in the riparian.

Of the six taxa of deciduous trees, only alder and willow were broadly distributed across the 18 sites (Table 2A). Most of the 10 species of coniferous trees also occurred across most sites. The principal exceptions were ponderosa pine and western white pine (Table 2A), which had limited distributions.

Floristics

Shrubs. -We observed 26 taxa of shrubs along riparian and upland point-intercept transects (Table 3). Four shrub species (devil's club, mock orange, common chokecherry, and alder buckthom) were found only in the riparian habitat and three species (creeping Oregon grape, red-stemmed ceanothus, and mountain balm) were found only in the upland. Most species were found in both riparian and upland habitats (Table 3). Prior to harvest, the mean number of shrub species counted on each of the 30-m transects per site was greater in upland (4.2 ± 0.2) than in riparian habitats (2.7 ± 0.1) across all sites ($F = 49.3$, df = 1,403, $P < 0.001$), and similarly for each of the three treatments (Table 4). However, the mean number of shrub species observed per site did not differ between riparian (8.8 ± 0.7) and upland (8.9 ± 1.1) habitats either overall or for any of three treatments (Table 4).

We examined the relative abundance of the more common shrubs (measured as the mean number of point-intercepts per 30-m transect) in riparian and upland habitats (Table 5). Of 17 species, three (17.6%) were equally distributed between habitats and only three were more abundant in the riparian (17.6%; e.g., red-stemmed dogwood), whereas 11 (64.7%; e.g., mountain boxwood, huckleberry, rose) were more abundant in the upland.

Herbaceous vegetation. – We observed ca. 115 species of herbaceous plants along riparian and upland point.-intercept transects (Table 3). Few species were found exclusively in the riparian (e.g., slender bogorchid, cow-parsnip, sharptooth angelica, licorice root) or exclusively in upland habitats (Table 3). Most species were found in both riparian and upland habitats (Table 3). There was considerable variation in plant species richness within sites; the minimum number of species per point-intercept transect ranged from 0 to 8 and the maximum from I to 24. For all treatments before harvest, there were more species of herbaceous vegetation per 30-m point-intercept transect in the riparian (7.2 ± 0.3) than in the upland habitats (5.1 ± 0.2; $F = 38.2$, df = 1,473, $P < 0.001$). Overall, the mean number of herbaceous species per site was greater in the riparian (23.4 ± 1.6) than upland habitats (17.3 ± 1.7; $F = 6.7$, df- 1,34, $P < 0.01$). However, comparisons of the mean number of species per site on each of the three treatments revealed no differences between total species richness between riparian and upland habitats (Table 4).

Although few species, of herbaceous plants were found exclusively in the riparian habitat, 48.6% of all taxa were more abundant in the riparian (Table 5). Only 18.9% of the taxa were more abundant in the upland habitat.

Litter and ground surface characteristics. -Before harvest, there were few differences in characteristics of the ground surface between riparian and upland habitats. Litter depth was

similar between riparian and upland habitats when compared across all sites (Table 6). Within treatments, litter depth was similar between riparian and upland habitats for Control and Modified sites (Table 6), but greater in the riparian habitat for the State sites (Table 6). The mean number of point intercepts of litter was greater in the upland than riparian habitat when compared across all sites, but only for the Control sites when compared within treatments (Table 6). The mean number of bare soil intercepts was similar between the riparian and upland habitats
(Table 6).

Effects of harvest

Habitat structure

The distance between shrubs increased after harvest (Table 7), but shrub area and height did not change significantly. Harvest resulted in reductions in woody debris in older decay classes, but increased the amount of fresh logs in the <24 cm diameter size classes (Table 7). Similarly, naturally created stumps in decay classes 2-4 were lost, whereas, as expected, the numbers of recent cut stumps increased significantly. These changes were reflected in differences between riparian and upland habitats. The upland transects had greater numbers of small diameter fresh logs and recent cut stumps (Table 7). Logs in the older decay classes were retained to a greater extent in riparian than in upland habitats.

Harvest also reduced number of regenerating stems, canopy cover, mean height of trees and snags, and number of deciduous and coniferous trees <25 cm DBH (Table 7). Reductions in overstory canopy cover were limited to the upland. Mean height of live trees remained greater in riparian than in upland habitats, as did the numbers of deciduous trees <25 cm DBH. Coniferous trees >1 1 cm DBH were more abundant in the riparian than in upland following harvest (Table 7). The numbers of snags also were lower on postharvest sites, primarily in the <25 cm DBH classes for both decay classes (Table 7). These reductions were greater in upland habitats for most size classes of snags (Table 7).

As expected, forest harvest altered the abundance and distribution of most tree species (Table 2B). These reductions were of course more pronounced on the upland transects, and were reflected in reductions :in the number of sites, the numbers of plots, and numbers of trees per plot.

Floristics

Shrubs. -Species richness of shrubs per site did not differ between Control, State, or Modified sites before harvest for riparian ($F = 0.07$; df = 2,15; $P = 0.93$) or upland $F = 0.40$; df = 2,15; $P = 0.68$) habitats, nor were there any differences in species richness of shrubs between treatments after harvest for riparian ($F = 0.50$; df = 2,9; $P > 0.60$) or upland ($F = 0.53$; df = 2,9; $P > 0.60$) habitats (Table 4). There were also no differences in species richness of shrubs when measured per 30-m point-intercept transect/site between the Control, Modified, or State sites before harvest in riparian ($F = 0.6\ 1$; df = 2,240; $P > 0.54$) or upland ($F = 0.49$; df = 2,229; $P > 0.61$) habitats or after harvest of the two cut treatments (riparian: $F = 1.782$; df = 2,246; $P > 0.17$; upland: $F = 0.66$, df = 2,227, $P > 0.52$).

The number of shrub species per 30-m point-intercept transect/site and per site in the riparian was greater in 1995 than 1992 (Table 4; point-intercept transect: $F = 5.9$; df = 1; $P < 0.015$; site: $F = 10.0$; df = 1:; $P < 0.005$). These differences were not attributable to the effects of the harvest treatment (point-intercept transect: $F = 0.5$; df = 2; $P > 0.59$; site: $F = 0.2$; df = 2; $P > 0.81$). The mean number of shrubs per 30-m point-intercept transect/site and per site in the upland also was greater in 1995 than in 1992 (Table 4; point-intercept transect:

$F = 5.0$; df = 1, $P < 0.02$; site: $F = 9.0$; df = 1, $P < 0.008$), but there were no differences among harvest types (point-intercept transect: $F = 0.7$ df = 2, $P > 0.51$; site: $F = 0.5$; df = 2; $P > 0.60$).

Counts of shrub species on both riparian and upland point-intercept transects on Control sites did not differ in abundance between 1992 and 1995 (Table 5). No shrub species was more abundant before than after harvest on the Modified cut sites, whereas 7 of 16 (44%) shrub species in the upland and 3 of 17 (18%:) species in the riparian were more abundant before harvest on State sites. Three species were more common after harvest on the Modified sites (Table 5).

Herbaceous vegetation. -Species richness of herbaceous plants per site among Control, State, or Modified sites did not differ before harvest in riparian ($F = 0.48$; df = 2,15; $P > 0.63$) or upland ($F = 0.86$; df = 2,25, $P > 0.44$) habitats (Table 4). Similarly, no differences in species richness of herbaceous plants between treatments were observed after the State and Modified sites 'were harvested for riparian ($F = 1.23$; df = 2,15; $P > 0.32$) or upland ($F = 0.71$; df = 2,15; $P > 0.51$) habitats (Table 4). There were also no differences in species richness of herbaceous plants when measured per 30-m point-intercept transect/site between the Control, Modified, or State sites before harvest in riparian ($F = 0.61$; df = 2,240; $P > 0.54$) or upland ($F = 0.49$; df = 2,229; $P > 0.61$) habitats, or after harvest (riparian: $F = 1.78$, df = 2,246, $P > 0.17$; upland: $F = 0.66$; df = 2,227; $P > 0.52$).

There were more herbaceous plant species per 30-m transect/site in the riparian habitats on the Control, Modified, and State sites in 1995 (post harvest) than in 1992 (pre harvest) ($F = 38.9$, df = 1, $P < 0.0001$), but these differences were not attributed to the effects of harvest treatment ($F = 0.1$, df = 2, $P > 0.94$.; Table 4). The number of herbaceous plant species per site in the riparian also averaged higher on all sites in 1995 than in 1992 ($F =$ 13.5, df = 1, $P < 0.001$; Table 4), but again the effects of harvest treatment did not contribute to these differences (F = 1 .0; df = 1; $P > 0.39$).

There were more species ofherbaceous plants per 30-m point-intercept transect in the upland in 1995 than in 1992 ($F = 8.6$; df = 1; $P < 0.003$), but there were no differences in species richness per site between sampling periods ($F = 2.29$; df = 1; $P > 0.1$; Table 4). The effect of harvest treatment had no effect on the richness of upland herbaceous plant species by either measure (per point-intercept transect, $F = 0.2$; df = 2; $P > 0.84$; per site: $F = 2.3$; df = 2; $P > 0.1$).

Abundance of herbaceous species did not differ between the two sampling periods for most species in riparian or upland habitats on the Control and Modified sites or for species in the upland on the State sites (Table 5). In contrast, most species in the riparian on State sites were more abundant before harvest. The abundance of about 20% of the riparian herbaceous species and 10% of the upland herbaceous species was greater in 1995 than in 1992 on Control and Modified sites (Table 5). On the State sites, the abundance of more upland herbaceous species was greater post-harvest than that of riparian species. The increase in abundance of herbaceous vegetation in 1995 was most likely the result of two factors: point-intercept 'transects were sampled earlier in the season of 1995 than of 1992 and the abundance of several weedy species (e.g., bull thistle) was greater after harvest on the cut sites.

Litter and ground surface characteristics. -After harvest, litter depth was significantly greater in riparian habitat (Table 6). Comparison of the differences in litter depth at each station in the 1992 and 1995 samples revealed significant effects of harvest for both riparian ($F = 5.61$; df = 2,249; $P < 0.004$) and upland habitats ($F = 3.44$; df = 2,249; $P <$

0.03). The effects of harvest on litter depth in riparian habitat were similar between the Modified sites (mean decrease in litter depth = 1.2 ± 0.8 cm) and the Control sites (mean increase in litter depth = 0.3 ± 0.6 cm). The change in litter depth on the State sites (mean increase of litter depth = 2.5 ± 0.9 cm) differed significantly from the Control and Modified sites. In the upland habitat, the effects of harvest on litter depth were similar between the Modified (mean decrease of litter depth = 4.8 ± 1.0) and State (mean decrease of litter depth = 2.8 ± 0.8 cm) sites and differed from the Control sites (mean decrease of litter depth = 2.3 ± 0.6 cm).

The mean number of intercepts of litter was greater in riparian than upland habitats after harvest of the Modified and State sites (Table 6). Comparison of the differences in litter intercepts at each station in the 1992 and 1995 sample revealed significant effects of harvest in both the riparian ($F = 9.5$; df = 2,249; $P < 0.0001$) and upland habitats ($F = 24.0$; df = 2,249; $P < 0.0001$). The effects of harvest on number of litter intercepts in both the riparian and upland habitat were similar between, the Modified and State sites as compared to the Control sites. Litter increased in the riparian habitats of the cut sites and decreased in the upland habitats.

After harvest of the Modified and State sites, the number of intercepts of bare soil became greater in the upland than in the riparian habitat (Table 6). Although the number of intercepts of bare soil remained comparable between the three treatments in the riparian habitat ($F = 1.7$, df = 2,501; $P > 0.18$), both Modified and State sites had significantly more intercepts of bare soil in the upland habitats than Control sites ($F = 12.2$; df = 2,502; $P < 0.001$). Comparison of the differences in bare soil intercepts at each station in the 1992 and 1995 sample revealed significant effects of harvest in the upland ($F = 8.6$; df = 2,249; $P <$

0.0002), but not the riparian habitats ($F = 0.74$; df = 2,249; $P > 0.48$). The number of bare soil intercepts increased in the upland habitats of the cut sites and remained the same in the riparian habitats.

Buffer width

The mean width of the State buffers was 14.1 ± 3.0 m with a range of 8 to 22.6 m. The mean width of the Modified buffers was 29.7 ± 17.4 m with a range of 12 to 144 m. Figure 2 contrasts the uniform width of a State buffer with the more variable widths of a Modified buffers.

DISCUSSION

Habitat comparisons

The habitat gradient that exists between riparian and upland habitats is considered to be a primary factor determining patterns of species diversity in riparian zones (Doyle 1990, McComb et al. 1993). Those elements that differ between habitat types may assist in understanding the patterns of vertebrate distribution observed in these systems. At our study sites, structural habitat conditions varied in only a few respects between riparian and upland habitats prior to harvest. These differences included greater dispersion of shrubs in riparian than in upland habitats, woody debris of larger size and greater decay in the riparian, and more natural stumps in the riparian. Down wood is used by many small mammals for cover and as runways. Logs that have decayed on the forest floor may also provide food resources such as fungi, which are used by small rodents (Maser and Maser 1987). The number of deciduous trees was higher in the riparian, although the numbers of conifers and snags were generally the same. There were more conifers and condition 2 snags in the larger DBH classes in the riparian. Large snags are important for cavity nesting birds, bats, and small mammals (e.g., Campbell et al. 1996, Zarnowitz and Manuwal 1985).

The considerable variation in tree species composition across and within sites is typical of second-growth forests in northeastern Washington. This variation reflects the various post-harvest practices used in the past. Few species of shrubs or herbs were unique to either riparian or upland habitats. Most species of shrubs were more abundant in the upland and shrub diversity was greater on the upland point-intercept transects. This pattern was reversed for herbaceous species for which a greater number of species were more abundant in the riparian and at higher levels of diversity on the point-intercept transects. The riparian environment is thus particularly suitable for some herbivorous vertebrates, especially those requiring mesic conditions. Ground surface characteristics varied little between riparian and upland habitats, but number of point-intercepts where litter was recorded was greater in the upland.

Effects of harvest

The Modified sites had wider, but considerably more variable buffers than did State sites (Fig. 2). Otherwise, forest harvest had largely predictable changes on structural characteristics of the habitat. The removal of trees in the upland opened both understory and overstory canopies, and reduced the mean height of trees and snags. Harvesting activities also decreased the shrub layer, regenerating stems, and deciduous trees. Of concern was the loss of snags, which provide various resources to many vertebrate species. These changes accentuated differences between upland and riparian habitats. Fresh down wood and stumps increased, especially in the upland. However, down wood in the older decay classes was generally reduced and remained higher in riparian areas. The numbers of cut stumps increased, particularly in the upland, but natural stumps in older decay classes were lost. Bare ground also increased in the upland. There were few differences in habitat structure between State and Modified sites.

Floristically, there were greater changes on the State sites than on Modified or Control sites. These differences included reductions in the abundance of several shrub species in both upland and riparian habitats of State sites, which were not observed on Modified or Control sites. Additionally, most herbaceous species in the riparian zones of State sites were more abundant prior to harvest. Several weedy species increased in abundance or appeared for the first time after harvest. We anticipate that herbaceous species composition will continue to change over time:.

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Table 1. Comparison of the means (± SE) for the 48 habitat variables measured on 15 riparian and 15 upland plots of the 18 study sites in northeastern Washington prior to logging (df = 1, 17 for all comparisons).

	Riparian		Upland		*F*	*P*
Variable	$\bar{x}$	SE	$\bar{x}$	SE		
Shrub distance	4.08	0.32	3.15	0.4	7.1	0.02
Shrub height	0.8	0.0	0.8	0.1	0.0	0.84
Shrub area	0.3	0.0	0.3	0.0	0.6	0.46
Logs						
> 5 m long and < 15 cm diameter, decay class 1	3.0	0.4	3.6	0.5	0.9	0.36
> 5 m long and < 15 cm diameter, decay class 2	4.2	0.4	6.2	0.7	7.4	0.01
> 5 m long and < 15 cm diameter, decay class 3	1.3	0.2	1.4	0.2	0.1	0.80
> 5 m long and < 15 cm diameter, decay class 4	0.3	0.1	0.3	0.1	0.4	0.54
> 5 m long and 16-24 cm diameter, decay class 1	0.8	0.1	0.7	0.1	0.3	0.61
> 5 m long and 16-24 cm diameter, decay class 2	2.7	0.2	2.8	0.5	0.1	0.82
> 5 m long and 16-24 cm diameter, decay class 3	2.5	0.2	2.1	0.3	1.2	0.30
> 5 m long and 16-24 cm diameter, decay class 4	1.0	0.1	0.8	0.1	1.0	0.33
> 5 m long and > 25 cm diameter, decay class 1	0.2	0.1	0.1	0.0	4.4	0.05
> 5 m long and > 25 cm diameter, decay class 2	0.8	0.1	0.7	0.1	1.7	0.21
> 5 m long and > 25 cm diameter, decay class 3	1.5	0.2	1.1	0.2	3.6	0.07
> 5 m long and > 25 cm diameter, decay class 4	1.0	0.2	0.9	0.2	0.1	0.71
< 5 m long and > 25 cm diameter, decay class 1	0.1	0.0	0.1	0.0	0.7	0.43
< 5 m long and > 25 cm diameter, decay class 2	0.3	0.1	0.2	0.1	2.6	0.13
< 5 m long and > 25 cm diameter, decay class 3	1.0	0.2	0.6	0.1	5.9	0.03
< 5 m long and > 25 cm diameter, decay class 4	1.4	0.3	1.3	0.3	0.3	0.57
Stumps						
Cut, decay class 1	0.2	0.1	0.2	0.1	0.6	0.44
Cut, decay class 2	0.3	0.1	0.5	0.2	0.4	0.53
Cut, decay class 3	0.7	0.2	0.5	0.2	1.6	0.23
Cut, decay class 4	0.0	0.0	0.0	0.0	0.1	0.83
Natural, decay class 1	0.2	0.0	0.1	0.0	1.9	0.19
Natural, decay class 2	0.6	0.1	0.6	0.1	0.3	0.60
Natural, decay class 3	1.4	0.2	0.9	0.2	5.0	0.04
Natural, decay class 4	1.3	0.2	0.7	0.2	8.5	0.01
Regenerating stems	27.5	4.3	32.3	5.1	0.8	0.38
Overstory canopy cover (%)	85.6	1.1	79.5	2.1	6.0	0.03
Understory canopy cover (%)	14.8	2.0	12.6	2.0	0.9	0.35
Tree height	14.0	0.7	13.3	1.0	1.1	0.32
Snag height	23.1	0.9	22.1	0.9	1.2	0.2s

Table 1. Continued.

	Riparian		Upland		*F*	*P*
Variable	$\bar{x}$	SE	$\bar{x}$	SE		
Deciduous trees						
5-10 cm DBH	4.4	1.0	2.1	0.6	4.3	0.05
11-25 cm DBH	2.0	0.6	0.2	0.1	11.2	0.00
26-50 cm DBH	0.2	0.1	0.1	0.0	3.9	0.07
> 50 cm DBH	0.0	0.0	0.0	0.0	1.2	0.30
Coniferous trees						
5-10 cm DBH	22.1	2.8	23.7	4.4	0.2	0.70
11-25 cm DBH	19.7	2.3	18.7	3.1	0.1	0.78
26-50 cm DBH	7.3	0.5	5.8	0.5	4.0	0.06
> 50 cm DBH	1.5	0.3	0.6	0.1	16.7	0.00
Snags, condition 1						
5-10 cm DBH	3.5	0.7	4.0	0.7	0.4	0.56
11-25 cm DBH	2.5	0.4	2.4	0.3	0.1	0.73
26-50 cm DBH	0.6	0.1	0.6	0.2	0.2	0.64
> 50 cm DBH	0.2	0.1	0.1	0.0	2.4	0.14
Snags, condition 2						
5-10 cm DBH	1.2	0.3	1.1	0.3	0.0	0.93
11-25 cm DBH	1.2	0.2	0.9	0.2	3.1	0.10
26-50 cm DBH	0.5	0.1	0.4	0.1	0.2	0.64
> 50 cm DBH:	0.2	0.0	0.1	0.0	16.1	0.00

Table 2A. Tree species present on the 7 Control, 5 Modified, and 6 State sites before harvest. *n* is the number of sites.

Species	Zone	Control					Modified					state				
			No. of stations		Trees per station			No. of stations		Trees per station			No. of stations		Trees per station	
		n	$\bar{x}$	SE	$\bar{x}$	SE	*n*	$\bar{x}$	SF	$\bar{x}$	SE	*n*	$\bar{x}$	SE	$\bar{x}$	SF
Maple species	R	4	3.8	2.4	38.8	35.4	3	3.3	1.9	12.7	7.9	3	1.3	0.3	2.3	0.7
	U	3	3.3	0.3	12.0	1.5	3	1.7	0.7	5.0	2.1	3	2.0	1.0	7.7	4.7
Alder species	R	7	8.9	1.7	55.4	15.5	5	9.2	2.2	79.4	33.5	6	9.3	1.6	100.2	49.5
	U	6	1.7	0.3	6.0	2.1	4	6.5	1.8	36.5	7.2	4	3.0	0.9	13.0	6.7
Black cottonwood	R	2	5.0	4.0	7.:	6.5	3	3.3	1.3	4.3	1.9	3	1.7	0.3	1.7	0.3
	U	1	1.0		1.0		1	1.0		1.0						
Birch species	R	4	2.0	0.7	4.3	2.1	2	9.0	5.0	52.0	41.0	2	1.0	0.0	1.0	0.0
	U	1	2.0		31.0		2	4.5	2.5	17.5	15.5					
Quaking aspen	R						1	6.0		13.0		1	4.0		9.0	
	U	2	2.0	0.0	3.5	1.5	3	2.0	0.6	2.3	0.7	2	3.0	0.0	9.5	3.5
Willow species	R	2	2.5	1.5	4.5	3.5	1	1.0		1.0		4	2.0	0.7	2.5	0.6
	U	5	3.2	1.4	8.4	5.6	4	4.8	0.9	10.3	4.0	5	4.2	1.6	28.6	24.2
Douglas fir	R	7	8.9	1.4	25.4	7.3	5	9.4	1.5	30.0	15.1	6	11.7	1.5	56.2	18.6
	U	7	10.1	1.8	103.4	35.3	5	12.8	1.4	80.6	24.5	6	14.0	0.8	132.3	45.4
Englemann spruce	R	7	7.7	1.5	64.0	48.7	4	12.5	1.0	90.0	40.1	5	11.6	1.6	256.6	143.5
	U	5	5.2	2.5	21.4	17.2	4	8.3	2.3	80.8	71.8	5	5.6	1.5	13.6	3.4
Grand fir	R	6	12.2	0.8	90.8	20.2	5	10.4	1.8	67.2	30.3	6	6.3	2.8	45.2	27.1
	U	6	11.7	1.3	98.0	22.5	5	10.2	0.7	37.2	9.7	6	4.3	1.6	14.7	6.0
Lodgepole pine	R	4	4.5	2.3	22.5	19.5	4	5.0	1.5	8.8	3.5	6	6.0	1.2	32.7	15.0
	U	5	7.8	2.4	96.0	60.8	5	10.0	1.2	55.6	15.7	6	11.2	1.6	101.8	43.4
Ponderosa pine	R						2	1.0	0.0	1.0	0.0	1	1.0		1.0	
	U	4	2.3	0.8	3.8	1.3	2	5.5	4.5	13.5	12.5	3	7.7	3.3	20.3	10.5
Subalpine fir	R	4	7.3	0.9	21.8	8.8	3	4.3	1.2	6.3	1.7	5	8.6	2.7	96.0	55.7
	U	7	4.6	0.8	23.1	9.7	3	1.3	0.3	1.3	0.3	4	6.5	1.8	19.3	7.5

Table 2A. Continued.

Species	Zone	Control					Modified					state				
			No. of stations		Trees per station			No. of stations		Trees per station			No. of stations		Trees per station	
		N	$\bar{x}$	SE	$\bar{x}$	SE	N	$\bar{x}$	SE	$\bar{x}$	SE	N	$\bar{x}$	SE	$\bar{x}$	SE
Western hemlock	R	7	11.9	1.7	179.0	61.7	5	11.4	1.6	87.4	37.8	6	11.7	1.7	327.7	148.7
	U	7	9.0	1.7	82.4	42.6	4	10.3	1.9	76.8	41.0	6	6.3	2.6	317.2	199.9
Western larch	R	7	6.6	1.6	19.6	ii.0	5	X.4	2.4	36.X	17.0	6	10.8	1.8	47.7	17.9
	U	7	8.9	1.9	67.9	39.9	5	10.0	2.0	60.4	36.0	6	10.2	2.2	102.8	4x.4
Western Redcedar	R	7	14.0	0.8	285.7	52.8	5	14.8	0.2	310.4	57.0	6	14.2	0.7	161.0	39.9
	U	6	13.0	1.3	287.7	102.8	5	11.2	2.4	347.6	128.1	5	9.0	2.2	66.8	19.8
Western white pine	R	5	2.6	0.7	3.4	1.0	2	2.0	1.0	2.0	1.0	5	3.2	0.7	5.4	2.4
	U	6	3.5	1.1	7.3	3.3	2	2.5	0.5	2.5	0.5	5	4.8	1.9	32.0	25.8

Table 2B. Tree species present after harvest on the 6 State and 5 Modified Sites. *n* is the number of sites

Species	Zone	Modified					state				
			No. of stations		Trees per station			No. of stations		Trees per station	
		n	$\bar{x}$	SE	$\bar{x}$	SE	*n*	$\bar{x}$	SE	$\bar{x}$	SE
Maple species	R	2	3.5	2.5	3.5	2.5	2	2.0	1.0	2.0	1.0
	U	2	1.5	0.5	5.0	1.0	2	1.0	0.0	2.5	1.5
Alder species	R	4	10.3	1.9	53.3	18.7	6	8.7	2.2	43.3	13.2
	U	4	5.8	1.3	17.0	4.0	3	1.7	0.7	8.0	2.0
Black cottonwood	R	3	3.3	i.9	4.7	3.2	3	1.7	0.3	2.0	0.6
	U	1	1.0		1.0		1	1.0		1.0	
Birch species	R	2	9.5	4.5	36.0	29.0	1	1.0		1.0	
	U	2	2.0	1.0	3.0	2.0					
Quaking aspen	R	2	3.0	2.0	4.5	3.5	2	2.0	1.0	3.5	2.5
	U	2	2.5	1.5	2.5	1.5	2	3.0	1.0	4.5	2.5
Willow species	R	1	1.0		1.0		3	1.7	0.7	3.0	0.6
	U	3	1.7	0.3	1.7	0.3	5	1.6	0.6	3.0	1.8
Douglas fir	R	5	8.6	1.8	27.2	14.0	6	10.0	1.4	42.3	16.8
	U	5	10.4	2.1	41.6	14.2	6	11.5	1.0	58.5	14.4
Englemann spruce	R	4	12.0	1.5	64.5	31.4	5	11.0	1.8	210.6	115.8
	U	4	4.8	2.6	25.5	22.8	4	4.3	0.9	12.8	5.6
Grand fir	R	5	9.8	2.1	52.2	25.6	6	7.8	2.2	46.8	25.0
	U	5	9.0	1.2	24.8	7.x	6	3.7	1.2	8.2	3.2
Lodgepole pine	R	4	3.3	0.8	4.5	1.5	5	6.0	1.0	28.2	16.5
	U	4	6.3	1.4	22.3	7.1	6	8.0	1.9	61.8	29.8
Ponderosa pine	R	2	1.0	0.0	1.0	0.0					
	U	1	9.0		19.0		2	7.5	0.5	16.5	6.5
Subalpine fir	R	3	3.3	1.9	6.3	4.3	4	10.8	2.8	109.5	58.1
	U	1	1.0		1.0		3	8.0	0.6	15.7	0.7

Table 2B. Continued.

Species	Zone	Modified					state				
			No. or stations		Trees per station			No. of stations		Trees per station	
		N	$\bar{x}$	SE	$\bar{x}$	SE	N	$\bar{x}$	SE	$\bar{x}$	SE
Western hemlock	R	5	10.8	1.7	74.8	33.1	6	12.3	1.3	259.7	128.7
	U	4	9.0	1.9	42.8	18.7	4	7.8	3.6	209.8	122.5
Western larch	R	5	6.8	2.4	19.4	10.3	6	9.2	2.1	34.5	15.3
	U	5	7.2	2.3	23.4	15.8	6	9.7	2.2	53.2	20.5
Western Redcedar	R	5	14.8	0.2	237.4	44.3	6	14.3	0.5	131.0	21.6
	U	5	10.0	2.0	131.4	42.5	5	8.0	1.9	39.6	10.8
Western white pine	R	2	2.0	1.0	2.0	1.0	4	2.3	0.X	2.8	0.9
	U	1	1.0		2.0		5	3.8	1.7	15.6	12.9

Table 3. The number of sites at which shrub and herbaceous plant species were found in the riparian or upland zone during the 1992 @e-harvest) and 1995 (post-harvest) sampling periods. There were seven Control, five Modified, and six State sites.

Taxon	Common name	**Control**				Modified				state			
		Riparian		Upland		Riparian		Upland		Riparian		Upland	
		Pre	Post	Pre	Post	Pre	Post	Pre	Post	Pre	Post	Pre	Post
SHRUBS													
Araliaceae													
Oplopanax horridum	Devil's club	2	3			2	3			2	4		
Berberidaceae													
Berberis aquifolium	Oregon grape	4	4	5	5	2	2	3	3	3	3	4	3
Berberis repens	Creeping Oregon grape			2	2			1	2			1	2
Caprifoliaceae													
Lonicera utahensis	Utah honeysuckle	5	5	5	5	3	3	4	3	4	4	2	3
Sambucus cerulea	Blue elderberry	2	2										
Symphoricarpos albus	Snowberry	4	5	2	2	4	4	4	3	2	3	4	4
Viburnum edule	Squash berry												
Celastraceae													
Pachistima myrsinites	Mountain boxwood	5	5	6	6	5	4	4	5	5	5	6	6
Comaceae													
Cornus stolonifera	Red-osier dogwood	4	4			5	5	3	3	3	4	2	1
Elaeagnaeae													
Shepherdia canadensis	Soapberry			3	3			3	3	1		4	2
Ericaceae													
Arctostaphylos uva-ursi	Bearberry	1	1	1	1			2	2	1		4	3
Gaultheria ovatifolia	Slender wintergreen	1		1	1					2	2	1	0
Vaccinium sp.	Huckleberry	7	7	7	7	4	4	3	5	5	5	6	6
Grossulariaceae													
Ribes sp.	Gooseberry	6	6	1	2	5	5	2	4	6	6	1	1

Table 3. Continued.

Taxon	Common name	Control				Modified				State			
		Riparian		Upland		Riparian		Upland		Riparian		Upland	
			Post	Pre	Post	Pre	Post	Pre	Post	Pre	Post	21-e	Post
Hydrangeaceae													
Philadelphus lewisii	Mock orange	1	1										
Rhamnaceae													
Ceanofhus sanguineus	Red stem ceanothus			1								2	2
Ceanothus velutinus	Mountain balm			1	1			1	1				1
Rhamnus alnifolia	Alder buckthom	1	1							1			
Rosaceae													
Amelanchier alnifolia	Western serviceberry	4	5	2	4	3	3	3	3	3	4	4	3
Holodiscus discolor	Ocean-spray	1	1	2	3	1	1	3	2	1		2	2
Rosa sp	Rose	6	6	6	6	4	4	4	4	4	4	5	6
Rubus ideaus	Red raspberry	3	2			3	2				2	2	1
Rubus parviflorus	Thimbleberry	5	6	4	4	5	5	5	5	4	5	3	4
Physocarpus malvaceus	Ninebark	1	1	3	3	1	2	2	2	1		2	2
Prunus virginiana	Common chokecherry	1	1										
Spireae betulifolia	Shiny-leaf spireae	3		4	4	3		3	4	4		4	4
HERBS													
Apocynaceae													
Apocynum androsaemifolium	Spreading dogbane				2		1		2			1	2
Araliaceae													
Aralia nudicaulis	Wild sarsaparilla	4	3	3	3	3	1	1	2	4	4	1	
Aristolochiaceae													
Asarum caudatum	Wild ginger	3	4	1	2	2	3			2	2	2	
Boraginaceae													
Cryptantha toryana													1
Mertensia paniculata	Tall bluebell	1	2			2	3		1	1	2		

Table 3. Continued.

Taxon	common name	Control				Modified				state			
		Riparian		Upland		Riparian		Upland		Riparian		Upland	
		Pre	Post	Pre	Post	Pre	Post	Pre	Post	Pre	Post	Pre	Post
Campanulaceae													
Campanula rotundifolia	Scotch bluebell											1	1
Caprifloliaceae													
Linneae borealis	Twinflower	7	7	7	7	5	5	5	5	6	6	6	6
Lonicera hispidula	Hairy honeysuckle		2				1						
Lonicera ciliosa	Orange trailing honeysuckle	1		2	2				2	1		1	3
Carophyllaceae													
Armemaria macrophylla	Bigleaf Sandwort								1				1
Sangina procumbens	Procumbent pearlwort										1		
Silene menziesii	Menzies' campion/silene				1				1				2
Compositae													
Achillea millefolium	Common yarrow			1	2			1			1	4	1
Adenocaulon bicolor	Trail-plant; pathfinder	4	5	6	5	1	4	2	3	3	4	1	2
Anaphalis margaritaceae	Pearly-everlasting		1			1					1		
Antennaria racemosa	Raceme pussytoes				3				2		1		
Antennaria sp.	Pussytoes		1		1		1		1			1	2
Arnica cordifolia	Heart leafed anrica		1	2	2				1	1		1	1
Aster sp.	Aster					1	1				2		1
Circium arvense	Canada thistle										3		1
Circium vulgare	Bull thistle						1			3	3		3
Erigeron philadelphicus	Daisy fleabane					1		1		2		1	
Erigeron speciosus	Showy fleabane					1					2		
Gnaphalium chilense	Cotton batting												1
Hieracium albiflorum	White-flowered hawkweed	1	2	3	5	1	1	3	3	2	5	5	6
Hieracium canadensis	Canadian hawkweed				1					1			2
Senecio triangularus	Arrowleaf groundsel	4	4			3	5			4	5	1	1

Table 3. Continued.

Taxon	Common name	Control				Modified				state			
		Riparian		Upland		Riparian		Upland		Riparian		Upland	
		Pre	lost	Pre	Post	Pre	Post	Pre	Post	Pre	Post	Pre	Post
Solidago missouriensis	Goldenrod	1											
Taraxacum officinale	Dandelion						2				2		2
Cornaceae													
Cornus canadensis	Bunchberry	6	5	2	2	1	2	2	2	5	5	2	2
Crassulaceae													
Sedum lanceolatum	Lanceleaved sedum				1		1					2	1
Cyperaceae													
Carex sp.	Sedge											1	
Equesetaceae													
Equisetum sp		3	5			1	3			5	4	1	
Ericaceae													
Chimphila umbellata	Pipsissewa, Prince's pine	4	4	6	6	3	2	5	5	6	5	6	6
Monotropa uniflora	Indian-pipe					2						1	
Pterospora andromedea	Pinedrops			1							1	1	
Pyrola asarifolia	Common pink wintergreen	3	7	2	6	4	4	4	4	4	5	3	3
Pyrola chlorantha	Green wintergreen			2						2			1
Pyrola secunda	One-sided wintergreen	2	3	4	5	2	4	5	5	2	5	2	1
Pyrola uniflora	Wood nymph		2			1	2			1	1	2	
Hpericaceae													
Hypericum perforatum	St. Johnswort												2
Labiatae													
Prunella vulgaris	Self-heal; all-heal												
Menthe arvense	Field mint												

Table 3. Continued.

Taxon	Common name	Control				Modified				State			
		Riparian		Upland		Riparian		Upland		Riparian		Upland	
		Pre	Post	Pre	Post	Pre	Post	Pre	Post	Pre	Post	*Pre*	*Post*
Leguminosae													
Lupinus sp.	Lupine			2	2		1	2	2			3	3
Trifolium arvense	Hare's foot clover												1
Trifolium repens	White clover			1	1			1	1			1	3
Trifolium dubium	Least hop clover												2
Vicia gigantea	Giant vetch												2
Liliaceae													
Clintonia uniflora	Queen's cup	7	7	7	7	5	5	5	5	6	6	6	5
Disparum trachycarpum	Wartberry fairy bell	2				4		1		1			
Lilum columbianum	Tiger lily		2	1			1					2	
Smilacina rasmosa	Western Solomon's seal	4	4	4	4	1	1	1	1			2	
Smilacina stellata	Star-flowered Solomon's seal	7	6	5	6	5	5	3	4	6	5	5	5
Streptopus amplexifolius	Fairy lantern	2	6	2	2		5	1	1	1	5	1	1
Trillium ovatum	White trillium	5	6	3	3	4	5	2	1	5	6	1	2
Lycopodium													
Lycopodium annotinum	Stiff clubmoss	1	1			1	1			1			
Onagraceae													
Circaea alpine	Enchanter's nightshade	2	3			4	4			3	4	1	1
Epilogium angustifolium			1		1				3	1	3	2	6
Epilogium glaberrimum	Smooth willow-herb		3				1		1	2	3		
Epilogium paniculatum											1		2
Epilogium watsonii													1

Table 3. Continued.

Taxon	Common name	Control				Modified				state			
		Riparian		Upland		Riparian		Upland		Riparian		Upland	
		Pre	Post	Pre	Post	Pre	Post	Pre	Post	Pre	Post	Pre	Post
Orchidaceae													
Corallorhiza maculata	Spotted coral-root					1	1						
Goodyera oblongifolia	Rattlesnake plantain	1	2	4	4	1	1	4	1	1	3	2	1
Habernaria saccata	Slender bog-orchid	3	4			1	3			1	1		
Habernaria orbiculata	Round leaved rain orchid												
Listera bore&	Listera	1	3	2	2		1		2		1		
Polenmoniaeae													
Collomia grandaflora	Large-flowered collomia											1	1
Polygonaceae													
Rumex acetosella	Sheep sorrel										1		1
Polypodiaceae													
Dryopteris austriaca	Mountain wood-fern	5	6	2	2	5	5	2	1	5	5	2	1
Gymnocarpum dryopteris	Oak-fern	7	7	1	1	5	5	2	3	6	6		1
Polystichum munitum	Holly-fern			2		1	1						
Pferidium aquilinum	Bracken fern; Brake-fern	4	3	3	1	1	3	1	1	3	1	1	2
Portulacaceae													
Claytonia lanceolata	Lanceleafed spring beauty	1	1				1						
Montia cordifolia	Broadleafed montia		3			1	2				1		
Ranunculaceae													
Aconifum columbianum	Monk's hood	3	2			3	3			1	1	1	1
Actaea rubra	Baneberry	4	4			3	5			4	5	1	1
Aquilegia sp.	Columbine					1				1	1	1	
Clematis columbiana	Columbia clematis	1		1	2	1	1	1				1	1
Coptis occident&	Western goldthread					1	1	1	1	1	2		
Ranunculus occident&	Western buttercup	1	1				1	1	1				
Ranunculus uncinatus	Small flower buttercup												1

Table 3. Continued.

Taxon	Common name	Control				Modified				State			
		Riparian		Upland		Riparian		Upland		Riparian		Upland	
		Pre	Post	Pre	Post	Pre	Post	Pre	Post	Pre	Post	Pre	Post
Thalictrum occident&	Western meadowrue	1	1	2	2		2			1	1	2	1
Trautvetteria caroliniensis	False bugbane	2	3		1	3	3		1	3	2		
Rosaceae													
Geum aleppicum	Yellow avens	2		2									
Geum macrophylum	Avens	1	1	2		3	2			3	2	2	
Fragaria virginiana	Wild strawberry	2	3			4	3			2	5	1	
Rubus pedatus	Strawberry bramble	1	1			1	1			3	3	1	1
Rubiaceae													
Galium aparine	Bedstraw; Goose-grass						1		1		3		2
Galium bifolium	Thinleaf beadstraw										1		
Galium boreale	Northern bedstraw	1	3	2	1		1	3	1		2	5	1
Galium trifolium	Fragrant bedstraw	6	5			5	5		4	5	5		4
Saxifragaceae													
Chrysosplenium tetrandum	Golden carpet									1			
Heuchera cylindrica	Roundleaf alumroot				2							1	1
Mitella caulescens	Star-shaped mitrewort	1	5			1	4	1		1	3		
Tiarella trifoleata	Foamflower; coolwort	7	7	6	6	5	5	3	5	6	6	3	2
Scrophulariaceae													
Castilleja miniata	Scarlet paintbrush								1			1	1
Collensia grandiflora	large-flowered blue-eyed mary												1
Melampyrum lineare	Cow wheat											2	1
Mimetanthe moschatus	Musk flower										1	1	
Pedicularis racemosa	Sickletop	1						1	1	1			
Veronia americana	American speedwell		2								1		
Veronia serpyllifolia	Thyme-leaved speedwell												1

Table 3. Continued.

Taxon	Common name	Control				Modified				state			
		Riparian		Upland		Riparian		Upland		Riparian		Upland	
		Pre	Post	Pre	Post	Pie	Post	Pre	Post	Pre	Post	Pie	Post
Urticaceae													
Urtica dioica	Stinging nettle	1	2			1	3				1	1	1
Umbelliferae													
Angelica arguta	Sharptooth angelica	1,	1			1	2			1	3		
Heracleum lanatum	*Cow-parsnip*	2	3			3	4			4	4		
Ligusticum canbyi	Licorice root		1			1	1				1		
Oenanthe sarmentosa	Water parsley	1											
Osmorhiza chelensis	Mountain sweet-root	1	5	4	5	4	5	3	3	2	6	3	4
Violaceae													
Viola spp.		7	7	7	6	5	5	5	4	6	5	6	6

‘Fable 4. Mean number (± SE) of shrub or herbaceous species per point-intercept transect and per site in 1992 (preharvest) and 1995 (post harvest). Significant differences between riparian and upland habitats for preharvest conditions are indicated in bold.

	Shrub species				Herbaceous species			
	Preharvest		Postharvest		Preharvest		Postharvest	
Treatment	Riparian	Upland	Riparian	Upland	Riparian	Upland	Riparian	Upland
Point-intercept transect								
Control	2.8 ± 0.2	3.6 ± 0.3	2.7 ± 0.23	4.3 ± 0.3	7.2 ± 0.4	5.0 ± 0.3	8.8 ± 0.5	5.9 ± 0.4
Modified	2.9 ± 0.2	3.9 ± 0.3	3.4 ± 0.3	4.5 ± 0.4	7.6 ± 0.6	4.8 ± 0.4	10.1 ± 0.6	5.4 ± 0.5
state	**2.4 ± 0.1**	4.9io.3	2.9 ± 0.19	4.8 ± 0.3	**6.8 ± 0.4**	**5.3 ± 0.4**	9.9io.5	6.0 ± 0.4
Site								
Control	8.7 ± 1.4	7.7 ± 1.8	9.3 ± 1.1	8.7 ± 1.9	21.4 ± 2.9	15.7i2.9	25.5i3.5	17.6 ± 2.4
Modified	9.2 ± 1.7	9.8 ± 1.8	10.8 ± 1.5	11.4 ± 2.0	25.2i3.4	15.8 ± 3.4	30.4 ± 3.2	19.4i3.5
State	8.5 ± 0.7	9.7 ± 2.1	9.8 ± 0.6	10.5 ± 1.8	24.3 ± 2.5	20.5 ± 2.5	32.0 ± 2.3	23.8 ± 5.4

Table 5. Relative abundance (measured by mean number of point-intercepts/30-m transect) of common shrubs and herbaceous plants. R, abundance greater in riparian; U, abundance greater in upland zone; Pre, abundance greater in 1992 (pre-harvest); Post, abundance greater in 1995 (post-harvest); =, no difference between riparian/upland or pre/post. Comparisons based on ANOVA, $P < 0.05$ significance level.

	Riparian vs Upland		Riparian			Upland		
Taxon	Pre	Post	Control	Modified	State	Control	Modified	state
SHRUBS								
Araliaceae								
Oplopanax horridum	R	R	=	=	=	=		
Berberidaceae								
Berberis aquifolium	U	U		=	=	=		Pre
Caprifoliaceae								
Lonicera utahensis	=	=				=	=	=
Symphoricarpos albus	U	=		Post	=	=	=	=
Celastraceae								
Pachistima myrsinites	U	U	=	=	=	=	=	Pre
Cornaceae								
Cornus stolonifera	R	R	=	=	=	=	=	=
Elaeagnaeae								
Shepherdia canadensis	U	U			=	=	=	Pre
Ericaceae								
Arctostaphylos uva-ursi	U	U			=	=	=	Pre
Vaccinium sp.	U	U		=	Pre	=	=	Pre
Grossulariaceae								
Ribes sp.	=	R	-	Post	Pre	=	Post	=
Rosaceae								
Amelanchier alnifolia	U	R	=	=	=	=		Pre
Holodiscus discolor	U	U			Pre	=	=	=
Rosa sp.	U	U	=	=	=	=	=	
Rubus ideaus	=			=	=	=	=	
Rubus parviflorus	R	R	=	Post	Post	=		=
Physocarpus malvaceus	U	U	=	=	=	=	=	Pre
Spireae betulifolia	U	U		=	=	=		=

Table 5. Continued.

	Riparian vs Upland		Riparian			Upland		
Taxon	Pre	Post	Control	Modified	State	Control	Modified	State
HERBS								
Araliaceae								
Aralia nudicaulis	R	R	=		Pi-e	=		Pre
Aristolochiaceae								
Asarum caudatum	R	R		=	Pre	=		Pre
Boraginaceae								
Mertensia paniculata	R			=	Pre			
Caprifloliaceae								
Linneae borealis	U	U		=	=		=	Pi-e
Lonicera ciliosa	U	U			=	=		=
Compositae								
Achillea millefolium								Post
Adenocaulon bicolor	R	R			=	=		=
Circium vulgare	=	U			Post	=	Post	Post
Hieracium albiflorum	U	U		=	Post		=	=
Senecio triangularus	R	R		Post	Pre			=
Taraxacum officinale		=			Post	=		Post
Cornaceae								
Corms canadensis	=	=		=	=			=
Equesetaceae								
Equisetum sp	R	R	Post	Post	=			=
Ericaceae								
Chimphila umbellata	U	U			Pre	=		Pre
Pyrola asarifolia	=	R		=	Pre	=		Pre
Pyrola secunda	U	=			Pre	Post		Pre
Leguminosae								
Lupinus sp.	U	U		Post			=	Post
Trifolium repens	=	U					=	Post
Trifolium dubium								
Vicia gigantea								
Liliaceae								
Clintonia uniflora	R	R	=	=	Pre	=	=	
Smilacina rasmosa	=	=		=				
Smilacina stellata	R	R	=		Pre	=		=
Streptopus amplexifolius	=	R	Post	Post	Post		=	=
Trillium ovatum	R	R		Post	Pre	=	=	=

Table 5. Continued.

	Riparian vs Upland		Riparian			Upland		
Taxon	Pre	Post	Control	Modified	State	Control	Modified	State
Onagraceae								
Circaea alpine	R	*R*	Post	Post	Pre			=
Epilogium angustifolium	=	=	=	=	=		Post	Post
Orchidaceae								
Goodyera oblongifolia	U					=	Pre	=
Polypodiaceae								
Dryopteris austriaca	R	R	Post	=	Pre	=	=	
Gymnocarpum dryopteris	R	R	Post	Post	Pre		=	=
Pteridium aquilinum	=	=	=	=			=	=
Ranunculaceae								
Aconitum columbianum	R	R	=	=	Pre			=
Actaea rubra	R	R	Post		Pre			
Thalictrum occidentalis	=					=		
Trautvetteria caroliniensis	R	R				=	=	
Rosaceae								
Fragaria virginiana	U	U	=	=	=	=		=
Rubus pedatus	R	R	=	=	Pre			=
Saxifragaceae								
Tiarella trifoleata	R	R	=	Post	Pre	=	=	
Umbelliferae								
Heracleum lanatum	R	R			Pre			
Osmorhiza chelensis	=	R	Post		Pre	Post	=	Pre

Table 6. Mean (± SE) litter depth, mean (± SE) litter intercepts, and mean (± SE) bare soil intercepts in riparian and upland habitats of Control, Modified, and State sites. * $P < 0.05$; ** $P < 0.001$; *** $P < 0.0001$.

		Litter Depth					Litter Intercepts					Bare Soil Intercepts				
		Riparian	Upland	*F*	df	Sig.	Riparian	Upland	*F*	df	Sig.	Riparian	Upland	*F*	df	Sig.
Overall	Pre	11.9 ± 0.38	11.80 ± 0.41	0.1	1,502	ns	52.6 ± 0.37	54.9 ± 0.334	19.7	1,501	***	0.77 ± 0.18	0.41 ± 0.06	3.5	1,501	ns
	Post	12.0 ± 0.35	8.8 ± 0.30	62.2	1,306	***	58.0 ± 0.26	55.2 ± 0.60	17.6	1,306	***	0.88 ± 0.19	4.66 ± 0.75	23.7	1,501	***
Control	Pre	11.3 ± 0.48	11.9 ± 0.48	0.8	1,166	ns	50.8 ± 0.61	53.3 ± 0.57	8.9	1,194	**	0.45 ± 0.26	0.11 ± 0.04	1.6	1,194	ns
	Post	11.6 ± 0.47	[illegible] ± 0.50	1.7	1,194	ns	58.2 ± 0.32	59.2 ± 0.44	3.1	1,194	ns	.56 ± 0.17	0.19 ± 0.06	3.9	1,194	*
Modified	Pre	12.0 ± 0.70	14.1 ± 0.98	3.1	1,166	ns	53.7 ± 0.70	56.0 ± 0.44	7.1	1,138	**	0.89 ± 0.23	0.43 ± 0.12	3	1,138	ns
	Post	10.7 ± 0.57	9.3 ± 0.54	3.6	1,166	*	58.2 ± 0.39	56.8 ± 0.76	2.6	1,138	ns	1.07 ± 0.33	4.6 ± 1.36	6.4	1,138	**
state	Pre	12.8 ± 0.8	9.8 ± 0.64	8.3	1,166	**	53.9 ± 0.56	55.7 ± 0.62	5	1,166	**	1.05 ± 0.41	0.73 ± 0.16	0.5	1,166	ns
	Post	15.3 ± 0.70	7.5 ± 0.52	80.1	1,166	***	57.9 ± 0.34	54.0 ± 0.89	16.7	1,166	***	0.71 ± 0.22	4.71 ± 0.80	23.3	1,166	***

Table 7. Mean (± SE) values for habitat variables measured on 15 riparian (R) and 15 upland (U) plots on the 5 Modified and 6 State sites for pre- and post harvest. Analysis of variance results are presented for significant comparisons ($P < 0.05$) between treatments, harvest types, and habitat zone.

	Preharvest								Postharvest								Treatment	Harvest	Habitat
	Modified				state				Modified				state						
	R		U		R		U		R		u		R		U				
Habitat Variable	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE			
Shrub distance	3.x	0.3	3.2	0.3	4.1	0.3	2.5	0.2	4.4	0.3	3.3	0.3	4.4	0.3	3.5	0.2	post > pre		
Shrub height	0.8	0.0	0.7	0.0	0.8	0.0	0.9	0.2	0.8	0.0	0.7	0.0	0.8	0.0	0.7	0.0			
Shrub area	0.3	0.0	0.2	0.0	0.3	0.1	0.3	0.0	0.4	0.1	0.2	0.0	0.5	0.1	0.4	0.1			
Logs																			
> 5 m, < 15 cm diam., decay class 1	3.3	0.9	4.9	0.9	3.6	0.X	3.1	0.6	4.9	1.0	7.0	1.0	2.6	0.2	6.6	2.0	post > pie		U > R
> 5 m, < 15 cm diam., decay class 2	4.2	0.8	7.9	1.5	4.6	0.6	5.7	1.2	2.x	0.2	3.4	0.5	2.1	0.7	2.0	0.8	pre > post		
> 5 m, < 15 cm diam., decay class 3	1.6	0.5	1.2	0.3	1.0	0.5	1.4	0.4	1.3	0.4	0.7	0.2	1.7	0.4	0.5	0.1			R > U
> 5 m, < 15 cm diam., decay class 4	0.6	0.2	0.4	0.2	0.2	0.1	0.2	0.1	0.1	0.1	0.1	0.0	0.4	0.2	0.0	0.0			R > U
> 5 m, 16-24 cm diam., decay class 1	1.2	0.3	1.1	0.3	0.6	0.1	0.5	0.1	1.9	0.3	2.3	0.4	1.1	0.3	3.4	1.2	post > pre		
> 5 m, 16-24 cm diam., decay class 2	3.3	0.4	4.5	1.4	2.5	0.3	1.7	0.3	1.x	0.3	2.2	0.4	1.3	0.2	1.3	0.6	pre > post	mod > state	
> 5 m, 16-24 cm diam., decay class 3	3.3	0.5	2.6	1.0	2.1	0.3	1.9	0.3	2.3	0.5	1.2	0.3	2.1	0.4	0.8	0.2	pre > post		R > U
> 5 m, 16-24 cm diam., decay class 4	1.4	0.2	0.8	0.3	1.2	0.3	0.9	0.3	0.5	0.2	0.2	0.1	0.6	0.2	0.2	0.1	pre > post		R > U
> 5 m, > 25 cm diam., decay class 1	0.3	0.2	0.2	0.1	0.1	0.0	0.0	0.0	0.5	0.2	0.7	0.3	0.2	0.1	1.3	1.1			
> 5 m, > 25 cm diam., decay class 2	1.0	0.2	1.0	0.3	0.7	0.1	0.4	0.1	0.6	0.1	0.6	0.2	0.2	0.1	0.2	0.0	pre > post	mod > state	
> 5 m, > 25 cm diam., decay class 3	1.5	0.3	1.1	0.4	1.3	0.2	1.1	0.4	1.9	0.4	0.8	0.2	1.6	0.2	0.6	0.2			R > U
> 5 m, > 25 cm diam., decay class 4	0.9	0.3	1.0	0.4	1.2	0.3	1.2	0.6	1.0	0.5	0.3	0.1	0.9	0.3	0.5	0.2			
< 5 m, > 25 cm diam., decay class 1	0.1	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.4	0.1	0.5	0.2	0.1	0.0	1.2	1.1			
< 5 m, > 25 cm diam., decay class 2	0.5	0.1	0.5	0.1	0.2	0.1	0.2	0.1	0.3	0.1	0.3	0.1	0.1	0.0	0.7	0.6			
< 5 m, > 25 cm diam., decay class 3	1.1	0.4	0.6	0.2	0.X	0.3	0.9	0.3	0.5	0.2	0.4	0.0	0.5	0.1	0.5	0.1	pre > post		
< 5 m, > 25 cm diam., decay class 4	1.8	0.5	1.0	0.4	1.2	0.3	1.7	0.6	0.9	0.3	0.6	0.2	0.8	0.1	0.4	0.1	pre > post		
Regenerating stems	26.1	9.0	34.1	14.1	24.X	5.2	40.9	5.9	17.8	5.7	12.3	3.8	14.8	1.9	12.22.X		pre > post		
Overstory canopy cover (%)	87.1	2.7	77.8	5.3	86.0	2.1	77.6	3.4	89.6	2.9	56.4	5.4	X7.6	1.9	53.4	4.6	pre > post		R > U
Understory canopy cover (%)	16.3	2.5	14.5	4.8	9.9	2.1	12.6	4.5	5.9	2.0	3.X	1.3	X.4	2.3	5.5	1.7	pre > past		

Table 7. Continued

	Preharvest								Postharvest								Treatment	Harvest	Habitat
	Modified				state				Modified				state						
	R		U		R		U		R		U		R		U				
Habitat Variable	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE			
Stumps																			
Cut, decay class I	0.4	0.2	0.3	0.1	0.2	0.1	0.3	0.2	1.8	0.4	5.7	1.3	0.6	0.1	4.6	1.0	post > pre		U > R
Cut, decay class 2	0.4	0.3	0.3	0.3	0.2	0.1	1.0	0.7	0.1	0.1	0.4	0.2	0.1	0.0	0.1	0.1			
cat, decay class 3	0.7	0.5	0.7	0.5	0.3	0.1	0.4	0.1	0.9	0.5	0.5	0.2	0.5	0.2	0.2	0.1			
Cut, decay class 4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.1	0.0	0.0	0.1	0.0	0.0	0.0			
Natural, decay class 1	0.1	0.0	0.2	0.1	0.2	0.1	0.0	0.0	0.1	0.0	0.2	0.0	0.0	0.0	0.1	0.0			
Natural, decay class 2	0.5	0.1	0.7	0.1	0.9	0.2	0.5	0.1	0.1	0.1	0.2	0.1	0.2	0.1	0.1	0.1	pre > post		
Natural, decay class 3	1.5	0.4	1.3	0.3	1.8	0.4	1.0	0.4	0.5	0.0	0.3	0.1	1.3	0.2	0.5	0.2	pre > post	state > mod	R > U
Natural, decay class 4	1.8	0.3	1.1	0.4	1.4	0.4	0.6	0.3	0.2	0.1	0.0	0.0	0.6	0.2	0.3	0.2	pre > post	state > mod	
Tree height	24.5	0.8	21.2	1.9	20.3	1.5	19.9	1.5	20.2	0.9	i6.3	1.0	iX.4	1.1	16.9	i.7	pre > post		R > U
Snag height	13.8	1.8	13.4	1.0	12.5	1.1	10.9	1.3	11.4	0.9	8.3	0.9	9.4	1.3	7.6	1.6	pre > post		
Deciduous trees																			
5-10 cm DBH	4.1	1.1	2.9	0.8	5.3	2.3	2.4	1.5	2.2	0.7	1.1	0.3	2.4	0.7	0.5	0.1	pre > post		R > U
11-25 cm DBH	3.0	1.4	0.3	0.1	1.8	1.0	0.2	0.1	1.5	0.7	0.2	0.1	0.7	0.2	0.1	0.1	pre > post		R > U
26-50 cm DBH	0.6	0.3	0.1	0.1	0.1	0.0	0.1	0.1	0.5	0.3	0.1	0.0	0.1	0.0	0.0	0.0			
> 50 cm DBH	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0			
Coniferous trees																			
5-10 cm DBH	15.7	3.8	22.7	6.2	29.3	5.7	26.3	11.8	12.1	3.2	10.5	2.6	22.2	5.2	12.6	5.0	pre > post		
11-25 cm DBH	16.5	2.X	17.6	5.4	27.1	3.9	20.5	X.3	10.5	1.8	6.6	2.2	19.5	3.1	10.2	3.8	pre > post	state > mod	R > U
26-50 cm DBH	7.3	0.9	7.0	1.3	7.4	0.7	5.2	0.7	7.7	1.2	3.3	0.8	9.5	0.8	3.1	0.6			R > U
> 50 cm DBH	1.6	0.9	0.4	0.2	0.8	0.2	0.4	0.2	I.2	0.5	0.3	0.1	1.4	0.3	0.3	0.1			R > U
Snags, condition 1																			
5-10 cm DBH	2.3	0.4	4.3	1.5	5.0	1.8	4.0	1.3	1.3	0.1	1.5	0.6	3.6	I.4	1.6	0.8	pre > post		
11-25 cm DBH	2.9	0.7	3.0	0.7	3.2	0.7	2.0	0.6	I.6	0.4	I.2	0.4	2.1	0.3	1.0	0.3	pre > post		R > U
26-50 cm DBH	0.4	0.1	0.6	0.2	0.4	0.2	0.3	0.1	0.3	0.1	0.1	0.0	0.4	0.1	0.2	0.1			R > U
> 50 cm DBH	0.1	0.1	0.1	0.1	0.1	0.0	0.0	0.0	0.1	0.1	0.0	0.0	0.2	0.1	0.0	0.0			R > U

Table 7. Continued.

Habitat	Variable	Preharvest								Postharvest								Treatment	Harvest	Habitat
		Modified				state				Modified				state						
		R		U		R		U		R		U		R		U				
Habitat	Variable	$\bar{x}$	*SE*	$\bar{x}$	*SE*	$\bar{x}$	*SE*	X	*SE*	$\bar{x}$	*SE*	$\bar{x}$	*SE*	$\bar{x}$	*SE*	$\bar{x}$	*SE*			
Snags, condition 2																				
	5-10 cm DBH	1.4	0.4	2.0	1.0	1.7	0.6	0.8	0.4	0.9	0.2	0.6	0.2	1.2	0.3	0.4	0.3			R > U
	11-25 cm DBH	1.8	0.5	1.6	0.4	1.5	0.3	0.7	0.4	1.0	0.2	0.4	0.1	1.2	0.3	0.4	0.1	pre > post		R > U
	26-50 cm DBH	0.8	0.1	0.6	0.1	0.4	0.1	0.3	0.1	0.7	0.1	0.2	0.0	0.5	0.1	0.2	0.0			R > U
	> 50 cm DBH	0.2	0.1	0.1	0.0	0.3	0.0	0.1	0.0	0.3	0.1	0.1	0.0	0.1	0.0	0.1	0.0			

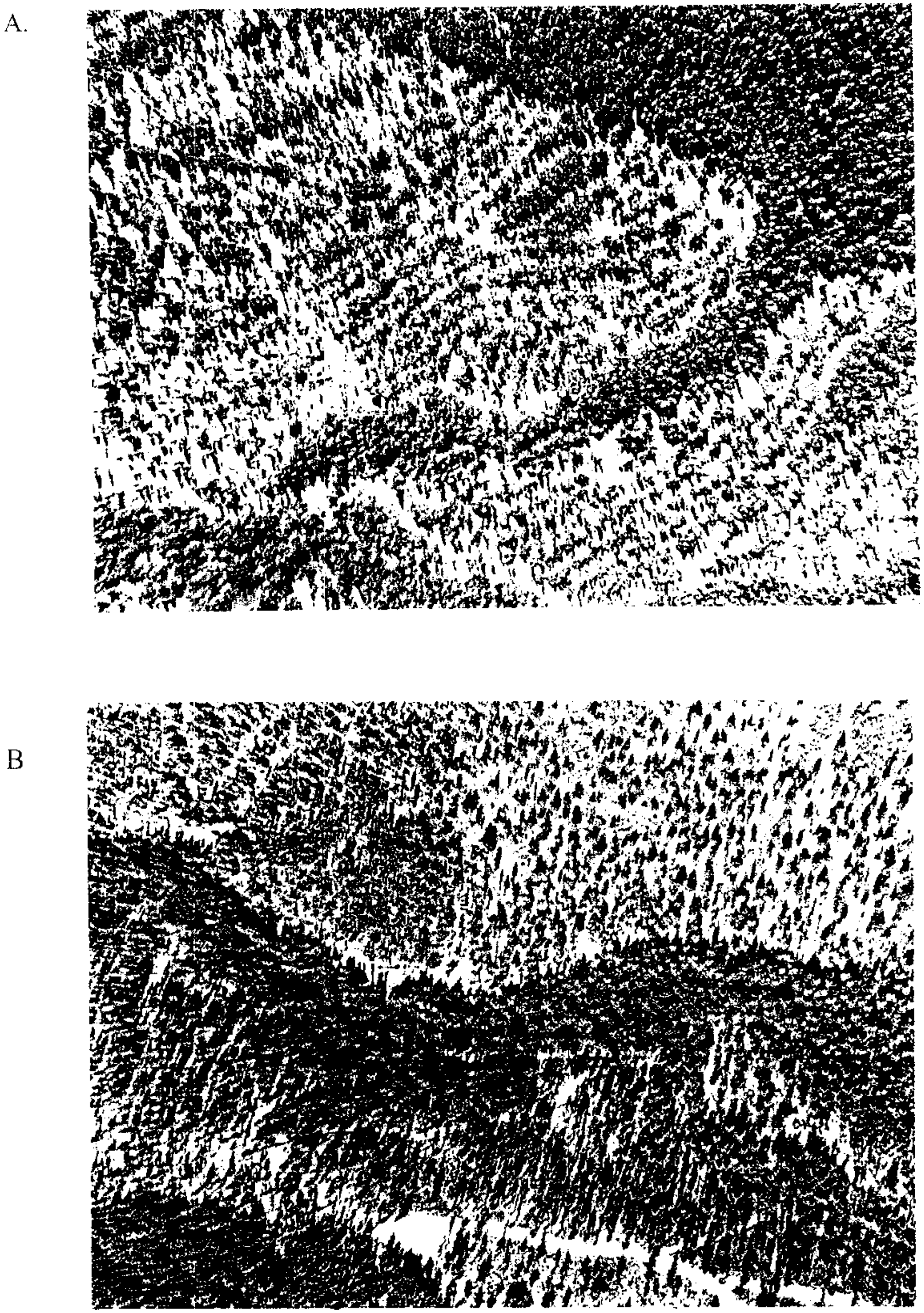

Figure 2. Aerial photographs illustrating the post-harvest buffers of A) a State site and B) a Modified site.

Chapter 5

WEST-SIDE AVIAN SURVEYS

Abstract. Buffer strips of standing trees are often left along rivers and streams after harvesting the adjacent upland to protect water quality and to minimize the adverse effects of harvest on aquatic and terrestrial species associated with riparian habitats. Little information is available on which species depend on riparian habitats or how riparian buffer strips provide habitat for wildlife species. To determine the bird species associated with riparian habitats in western Washington, we compared both individual species abundance and community composition in riparian and adjacent upland habitats before timber harvest. To assess the effectiveness of riparian buffer width to the breeding bird community, we compared individual bird species abundance and community composition on sites where the upland habitat was not clearcut or clearcut leaving either wide (-3 1 m, Modified harvest) or narrow (~14 m, State harvest) riparian buffer along both sides of second and third order streams. We also compared individual species abundance and community composition in the uplands before and after clearcutting.

Before harvest, there were no differences in bird community measures between the riparian and upland habitats. Among species groupings, four groups were more abundant in riparian habitats: Neotropical migrants, resident species, species associated deciduous trees and species associated with shrubs in forested habitats. Total species abundance and four individual species were also ‘more abundant in riparian habitats: American Robin (*Turdus migratorius*), Pacific-slope Flycatcher *(Empidonax difficilis*), Black-throated Gray Warbler *(Dendroica nigrescens*) and Winter Wren *(Troglodytes troglodytes). No* species or species group was more abundant in the upland.

When examining the effect of buffer width on the breeding bird community, we found the number of species on State harvested sites increased from slightly fewer than controls before harvest to an average of 10 more species than controls after harvest. This change was also reflected in an average increase in species turnover of 20% on State harvested sites relative to controls after harvest. Local extinction rate and overall bird abundance did not differ between treatments and controls after harvest. Total bird abundance did not differ between treatments and controls after harvest. Resident species and species associated with coniferous trees declined on both treatments post-harvest. Abundance of Black-throated Gray Warbler, Golden-crowned Kinglet (*Regulus satrapa*) and Brown Creeper (*Certhia Americana)* decreased on one or both riparian treatments. Wilson’s Warbler (*Wilsonia pusilla*) was more abundant on the wider-buffered treatments than narrow-buffered treatments. Dark-eyed Junco *(Junco hyemalis)* and Song Sparrow (*Melospiza melodia*) were more abundant on narrow-buffered treatments than controls or wide-buffered treatments.

When comparing the clearcut uplands in the first year post-harvest with the same habitats before harvest, there was an average increase in local extinction and species turnover of 30%. In the second year post-harvest there was an average increase in species turnover of 27% and an average increase in local extinction of 18% relative to the pre-harvest year. Neotropical migrants, residents, cavity nesters, and species associated with coniferous trees, deciduous trees, and shrubs in forested habitats declined on clearcut uplands when compared

to unlogged controls. In general, individual species associated with forested habitats declined while those associated with open habitats increased.

High species turnover on the State treatment indicates that riparian buffers less 14 m on each side of the stream do not maintain the pre-logging bird community. Neither buffer treatment maintained residents species as well as unharvested controls. Despite small sample sizes, the poor reproductive success of cavity nesters on State treatments suggests that riparian buffers wider than that required by State Forest Practices would benefit cavity nesting species. The Black-tbroated Gray Warbler was the only riparian associate to decline on both State and modified treatments and the abundance of this species was positively correlated with buffer width. This species was not detected on sites with buffers narrower than 30 m on a side. Thus in order to maintain the entire breeding bird community associated with forested riparian habitats in the coastal Northwest, we recommend a minimum buffer of 30 m along both sides of second and third order streams. The dependence of the Black-throated Gray Warbler on deciduous, riparian habitats makes it a good indicator of this habitat type. Habitat features such as deciduous trees (*Alnus rubra* and *Acer macrophyllum*) and berry producing shrubs (especially *Rubus spectabilis* and *Vaccinium spp.*) appear to be important habitat attributes to species associated with riparian zones and should be maintained within forested riparian buffer strips.

This study documents the baseline conditions for long-term research and describes the short-term effects of riparian treatments on the breeding bird community. The breeding bird community may take several years to respond to habitat manipulations; thus, we recommend continued monitoring to assess the long-term effect of buffer width reduction.

INTRODUCTION

Riparian zones are ecotones between the terrestrial and aquatic environments and represent some of the most dynamic portions of the landscape (Swanson et al. 1988). As a consequence, riparian areas are typically more structurally diverse and more productive than the adjacent uplands (Bull 1978). Riparian zones usually support a greater number of plant (Gregory et al. 1991) and vertebrate (Thomas et al. 1979, Oakley et al. 1985) species, In arid regions of the western United States, riparian habitats make up <1% of the landscape, yet 82% of all bird species annually breeding in northern Colorado occur in riparian vegetation (Knopf 1985), and 51% of all bird species in southwestern states are completely dependent upon this habitat type (Johnson et al. 1977).

However, in regions where the contrast between riparian and upland habitats is less pronounced, there may be little or no difference in bird species richness and abundance (Murray and Stauffer 1995). In the relatively wet and lush forests of the Pacific Northwest, bird species richness and abundance may even be higher in upland habitats than in riparian habitats (McGarigal and McComb 1992). Thus, the relative importance of riparian zones to terrestrial wildlife appears to vary geographically.

Throughout most of North America, buffer strips of standing trees are left between clear-cuts and aquatic habitats (Knopf et al. 1988). Buffer strips are left to protect water quality, and to minimize adverse effects of harvest on aquatic species and the terrestrial species associated with riparian habitats. In addition, buffer strips may serve as important connections between fragmented habitats, and consequently, may counteract some of the problems associated with landscape fragmentation (Wilcox and Murphy 1985, Saunders et al 1991). A critical question associated with riparian zone management is how wide should riparian buffers be in order to protect the species that depend on these habitats? Managing buffer width appears to be an effective approach to conserving biological diversity (Spackman and Huges 1995). The width of riparian zones appears to influence species richness or abundance (Stauffer and Best 1980, Darveau et al. 1995, Spackman and Huges 1995, Kilgo et al. 1998, Hagar 1999). Buffer width also appears to effect microclimatic conditions within the riparian zone (Brosofske et al. 1997) which may, in turn, influence the plant and animal community found there.

Investigators have used a variety of approaches to evaluate the effect of buffer width on bird species richness and abundance. Some have looked at the change in species richness as one moves away from the stream in unharvested forests (e.g. Spackman and Hughes

1995). Others have correlated buffer width with bird abundance and richness after timber harvest (Kinley and Newhouse 1997, Hagar 1999, Whitaker and Montevecchi 1999). We know of only one study that has used an experimental approach to examine the effect of buffer width on wildlife (Darveau et al. 1995). For many studies, it is difficult to evaluate the effect of 'buffer width on species that depend upon riparian zones because few have first attempted to identify the species associated with riparian habitats (but see Whitaker and Montevecchi 1999). As a consequence, the species that decline or disappear in narrow riparian buffers may not be species that depend upon riparian zones for reproduction or survival.

This study had two primary objectives: 1) to determine the species, if any, that are associated with riparian habitats in the coastal and Cascade mountains of western Washington; and 2) to assess the effect of riparian buffer width on the breeding bird community. To accomplish these objectives we compared the breeding bird community in riparian and upland habitats before harvest; and compared the riparian breeding bird community in unharvested stands (Controls) with stands where the upland had been clearcut leaving either a wide (Modified) or narrow (State) unharvested buffer along the stream. Because few studies have experimentally examined the effect of harvest on upland habitats (but see Chambers et al. 1999) and because upland habitats may be as important or more important to the avian community than riparian habitats in the Pacific Northwest (McGarigal and McComb 1992), we also compare the upland breeding bird community before and after logging.

METHODS

Bird sampling

Birds were surveyed using 15-m fixed radius point counts (Verner 1985). In each stand, 10 riparian stations were established along the edge of the stream with five stations spaced evenly on each side of the stream. Each riparian station was located 15 m from the usual high water line, 100 m from other stations and at least 50 m from the edge of the stand. Ten additional 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. Censuses usually started 30 min before or after dawn and were completed within 5 h. Upon arriving at a survey point, observers remained stationary and quiet for a minimum of 1 min to allow birds to settle and then recorded all birds heard or seen during a 6 min period. To avoid observer bias, observers were rotated among the 18 study sites. To avoid bias associated with visiting riparian or upland sites first, we alternated travel routes. Each stand was visited 6 times between mid-April and late-June. The surveys were evenly spaced throughout the breeding season to account for differences in breeding phenology among species. No survey was conducted during heavy precipitation or high winds. Every attempt was made to avoid counting individual birds more than once. If the riparian buffer was narrower than the diameter of our point count circle on harvested sites, then we recorded whether the bird was detected in the forested buffer or in the clearcut portion of the circle.

Small radius point counts were used because of the difficulty associated with travel along streams with steep slopes (some >40% slope) and dense vegetation. Small radius point counts eliminated 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 strips of potential habitat post-harvest.

Nests were found by searching the plots thoroughly in the 2 yr after harvest. We searched for nests after completing bird censuses. We standardized nest searching by spending equal time searching in each stand. Because of our primary interest in the effect of harvest on riparian habitats, we concentrated our search efforts on finding nests in riparian habitats. Because nesting success was not originally included in the study design, we were only able to monitor nests every 6 to 10 d throughout the nesting period. Nests were considered successful if at least one offspring fledged. Nests were considered depredated if there was sign of predation or the nest was found empty well before the estimated fledging date.

Data analyses

For all analyses, detections of Hermit (*Dendroica occidentalis)* and Townsend's *(Dendroica townsendi)* Warblers were grouped as one species (hereafter Hermit/Townsend's Warbler) because these species hybridize extensively in this region (Rohwer and Wood 1998) and cannot be distinguished by song in regions of hybridization (Pearson and Rohwer 1998). In addition, we excluded from all analyses individuals that flew over the stand, migrants that did not breed in the area (e.g., Ruby-crowned Kinglet, *Regulus calendula*, and Golden-crowned Sparrow, *Zonotrichia leucophrys*), and all species not adequately sampled by point counts (grouse, raptors, and waterfowl). Finally, to avoid including non-breeders in our analyses, we excluded all species that were not detected on at least three occasions.

Species richness, turnover, and extinction probability

It is often difficult to count all species within any given area. Consequently, counts of species detected often underestimate the numbers of species present and create problems upland sites when compared to controls (Table 3). Species associated with shrubs in open habitats increased on harvested sites relative to controls (Table 3).

Abundance comparisons

There was no effect of treatment on overall abundance in the uplands (Table 1) but treatment effects were detected for seven species. Species that were more abundant on controls than harvested uplands include: Chestnut-backed Chickadee, Golden-crowned Kinglet, Pacific-slope Flycatcher, and Winter Wren (Table 1). Species more abundant or only found on harvested uplands include Dark-eyed Junco, Spotted Towhee (*Pipilo maculatus*) and White-crowned Sparrow (*Zonotrichia leucophrys*) (Table 1).

Species-habitat relationships

Abundance of species associated with open or shrubby habitats (Dark-eyed Junco, Spotted Towhee, and White-crowned Sparrow) was negatively correlated with canopy closure (Table 7) and abundance of species associated with forested habitats (Chestnut-backed Chickadee, Pacific-slope Flycatcher, and Winter Wren) was positively correlated with canopy closure (Table 7). Some species associated with logged habitats are found in areas with more shrubs (e.g., White-crowned Sparrow, Table 7) while others appear to be found in areas with few shrubs (e.g., Dark-eyed Junco, Table 7). Pacific-slope Flycatcher abundance was positively correlated with berry producing shrubs and Chestnut-backed Chickadee abundance was negatively correlated with berry producing shrubs. Berry producing shrubs are more abundant in riparian habitats (Chapter 3).

estimators, we report the jackknife estimator which is derived using a bootstrap approach. Bootstrap variance estimates were calculated using 200 iterations and a random seed. Initial fit of the data to the heterogeneity model was calculated using a goodness-of-fit (GOF) test.

Abundance

To compare individual species abundances we used an index of abundance for each common species. Common species were defined as having >20 detections in the pre-harvest year for comparisons between upland and riparian habitats and >20 detections in at least 1 of the 2 yr post-harvest for buffer width comparisons. The index of abundance was calculated by averaging the number of detections over the six visits to each stand in a given year. A separate index of abundance was calculated for riparian and upland habitats in each stand.

We used paired *t*-tests to compare overall abundance, abundance of individual species and abundance of species groups between the riparian and upland habitats. This analysis included all 18 sites from the pre-harvest year (1993). We examined the effect of buffer width on species abundance using mean abundance of both post-harvest years combined (1995 and 1996); a separate ANOVA was conducted for the riparian and upland habitats. Tukey pairwise comparisons were used to examine treatment effects.

We did not use all 3 yr of data and both riparian and upland habitats in a single repeated measures ANOVA for several reasons: 1) we had small samples for many species and thus, any treatment effect would be lost in an overall ANOVA; 2) we were not interested in the interaction between upland and riparian habitats in the post-harvest years because uplands were treated similarly; 3) we were not interested in the variation between the 2 yrs post harvest but were interested in treatment effects for the period immediately following harvest and intend to census these same sites again in the future (5 and 10 yrs post-harvest) to examine any temporal variation. This approach most directly addresses the questions of interest with the least number of *post-hoc* tests.

We compared abundance between riparian and upland habitats before harvest and the riparian treatment effect post-harvest for the following species groupings: 1) species associated with the canopy of coniferous forests; 2) species associated with deciduous trees; 3) species associated with shrubs and small trees in open habitats; 4) species associated with shrubs and trees in forested habitats; 5) cavity nesters; and 6) species grouped according to migratory status (Neotropical migrants, short distance migrants, and residents). For species group membership see Table 1. Not all species were put into a habitat group (1-5 above) and habitat groupings were based on the primary use of these habitats for breeding or foraging. Cavity nesters include species that only used cavities for nesting. Winter wren was not included in cavity nesters because we found it frequently nesting in root wads and other substrates. We compared treatment effects and associations with riparian and upland habitats for these species groups using the same methods as described for individual species. We used a linear regression to compare the abundance of species associated with riparian habitats to buffer width on treatment stands after harvest.

Abundance data not meeting the assumptions of normality (Kolmogorov-Smimov one sample test) or homogeneity of group variances (Bartlett's F-test, residual scatter plots) were log transformed (Zar 1984).

Nesting success

The probability of nest mortality was calculated using the Mayfield method (Mayfield 1961) as modified by Hensler and Nichols (1981). Nesting success was not compared among treatments because of small sample sizes.

Species-habitat relationships

To determine habitat features that are important to species associated with riparian habitats (Table I), we regressed the detection rate for each species in the pre-harvest year with the habitat variables measured in the pre-harvest year. To determine which habitat features are important to species demonstrating a significant treatment effect in riparian habitats post-harvest (Table 1), we regressed the detection rate in the riparian habitats for the two post-harvest years combined with the riparian habitat variables measured post-harvest. To determine which habitat features are important to species demonstrating a treatment effect post-harvest in the uplands (Table I), we regressed the detection rate in the upland habitats for the two post-harvest years combined with the upland habitat variables measured post-harvest. For all regressions we used a stepwise regression (forward selection). Vegetation variables having a tolerance factors >0.70 were excluded from the model to reduce multicollinearity (Wilkinson 1990). All analyses were performed using SYSTAT (Wilkinson **1990).**

RESULTS

Riparian and upland bird community before harvest

Community comparisons

There was no difference in species richness between riparian and upland habitats before harvest (Table 2). ‘There were 22 species detected in riparian habitats and 26 in upland habitats. The members of one habitat type found on the other are remarkably similar (Table 2). The probability of detecting a species was similar in both habitat types and was quite high (335% for both habitat types; Table 2).

Species group comparisons

Neotropical migrants and residents were more abundant in riparian habitats than upland habitats (Table 3). Species associated with deciduous trees and shrubs in forested habitats were more abundant: in riparian habitats than the adjacent upland habitats (Table 3). ***No*** species group was more abundant in upland habitats (Table 3).

Abundance comparisons

We detected 4,646 individual birds of 62 species within the point count circles over the 3 yr of sampling. Before harvest, 86% of all detections in riparian and upland habitats were of five species: Chestnut-backed Chickadee (*Poecile rufescens*), Winter Wren, Pacific-slope Flycatcher, Golden-crowned Kinglet, and Wilson's Warbler. Total abundance was higher in riparian habitats (Table 1).

As with grouped species comparisons, no individual species was more abundant in the uplands than in the riparian habitat in the pre-harvest year. However, the following species were rarely detected in the pre-harvest year and, when detected, they were found exclusively in the uplands: Cedar Waxwing (*Bombycilla cedrorum*), Dark-eyed Junco (*Junco hyemalis*), and Hermit/Townsend's warbler. Four species were more abundant in riparian habitats than upland habitats: American Robin, Black-throated Gray Warbler, Pacific-slope Flycatcher, and Winter Wren (Table 1).

Species-habitat relationships

All four of the species that were more abundant in riparian habitats demonstrated significant correlations with riparian habitat features; three were positively correlated with berry-producing shrubs and two were positively correlated with deciduous trees (Table 4; for a description of the habitat variables see Table 5).

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Effect of riparian buffer width

Community comparisons

Local extinction probability was similar between buffer treatments and controls (Fig. 1). The number of species on State sites increased from slightly fewer than controls before harvest to an average of 10 more species than controls after harvest (Fig. 2). This change is reflected in an average increase of 20% in species turnover on State sites (Fig. 3).

Species group comparisons

Short-distance migrants and species associated with shrubs in open habitats increased on the State treatment relative to Control and Modified treatments (Table 3).

Abundance comparisons

There was no effect of buffer width on total bird abundance (Table 1). Black-throated Gray Warbler and Golden-crowned Kinglet were more abundant on control sites than treatments (Table 1). Brown Creeper was more abundant on control sites than State harvest sites, Wilson's Warbler was more abundant on modified harvest sites than State harvest sites (Table 1). All four of these species were only detected within the forested buffer on treated sites, Dark-eyed Junco and Song Sparrow were more abundant on the State treatment than the control and modified treatment (Table 1) and were found in both the clearcut and forested buffer on harvested sites. Western Tanager (*Piranga ludoviciana*) was more abundant on the modified treatment than the control and State treatment (Table 1) and was only found in the forested buffer on treated sites. The pattern of Western Tanager abundance among treatments differs among years post-harvest suggesting that combining both years data may not be appropriate. The treatment effect for all other species was similar between years post-harvest.

Although buffer width was different between treatments post-harvest (see above), there was overlap between the widest State buffer (range = 7.3 to 23.2 m) and the narrowest modified buffer (range = 20.6 to 47.9 m). Consequently, we compared species abundance with buffer width on treated sites post-harvest. For this analysis we used only the four species associated with riparian habitats and only the Black-throated Gray Warbler demonstrated a relationship with buffer width (Black-throated Gray Warbler: F = 12.37, df = 1,9, *P* = 0.007; American Robin: F = 4.57, df = 1,9, *P* = 0.06; Pacific-slope Flycatcher: F = 1.61, df = 1,9, *P* = 0.24, Winter Wren: F = 0.82, df = 1,9, *P* = 0.39). The Black-throated Gray Warbler was detected on six of the seven control sites post-harvest but only on two treatment sites post-harvest and both sites had riparian buffers averaging >30 m.

Nesting success

We located and monitored 97 nests of 21 species in riparian and upland habitats. We were able to determine the outcome of 40 nests of 11 species in riparian habitats. These nests with known outcomes were dominated by three species: American Robin (27%), Winter Wren (23%), and Hairy Woodpecker (*Picoides villosus*) (13%). All but 1 of the 14 nest failures were the result ofpredation. There were no cases of nest parasitism by the Brown-headed Cowbird (*Molothrus ater*). Unfortunately, we did not have enough nests to statistically compare nesting success among treatments. We will however, provide a qualitative summary of nesting success among treatments. Assuming an average of 26 d for the nest cycle and using the daily survival probabilities, an average nest (cavity and cup nests combined) would have a 50% chance of survival on control sites, 69% chance of survival on modified harvest sites, and a 30% chance of survival on State harvest sites (Fig. 4). For cup nests, there was a 14% chance of survival on control sites, a 35% chance of survival on modified harvest sites, and a 25% chance of survival on State harvest sites (Fig. 4). For cavity nests, there was a 100% chance of survival on control sites, 92% chance of survival on modified harvest sites, and a 39% chance of survival on State harvest sites,

Species-habitat relationships

Abundance of species associated with open or edge habitats (Dark-eyed Junco and Song Sparrow) was negatively correlated with canopy closure (Table 6) and abundance of species associated with forested habitats (Brown Creeper and Black-throated Gray Warbler) was positively correlated with canopy closure (Table 6). Black-throated Gray Warbler and Wilson's Warbler abundance was correlated with berry producing shrubs (Table 6) which are more numerous in riparian habitats (see Chapter 3). Golden-crowned Kinglet abundance was positively correlated (Table 6) with large Douglas-fir trees and snags which are more typical of upland habitats.

Treatment effects in upland habitats

Community comparisons

The number of species on modified harvest sites compared to controls doubled from 6 more species before harvest to 14 more species in the second year after harvest (Fig. 2). When comparing the pre-harvest year and first year post-harvest, local extinction probability increased by at least 33% (Fig. 1) and species turnover increased by nearly 30% (Fig. 3) on harvested sites relative to controls. When comparing the pre-harvest year with the second year post-harvest, there was at least a 27% increase in species turnover on logged sites relative to controls (Fig. 3); local extinction probability did not differ between the controls and modified harvest sites and increased by 18% between controls and State harvest sites.

Species group comparisons

Neotropical migrants, short-distance migrants and resident species declined on harvested uplands but not on controls (Table 3). Cavity nesters and species associated with coniferous trees, deciduous trees, and shrubs in forested habitats decreased on harvested upland sites when compared to controls (Table 3). Species associated with shrubs in open habitats increased on harvested sites relative to controls (Table 3).

Abundance comparisons

There was no effect of treatment on overall abundance in the uplands (Table 1) but treatment effects were detected for seven species. Species that were more abundant on controls than harvested uplands include: Chestnut-backed Chickadee, Golden-crowned Kinglet, Pacific-slope Flycatcher, and Winter Wren (Table 1). Species more abundant or only found on harvested uplands include Dark-eyed Junco, Spotted Towhee (*Pipilo maculatus*) and White-crowned Sparrow (*Zonotrichia leucophrys*) (Table 1).

Species-habitat relationships

Abundance of species associated with open or shrubby habitats (Dark-eyed Junco, Spotted Towhee, and White-crowned Sparrow) was negatively correlated with canopy closure (Table 7) and abundance of species associated with forested habitats (Chestnut-backed Chickadee, Pacific-slope Flycatcher, and Winter Wren) was positively correlated with canopy closure (Table 7). Some species associated with logged habitats are found in areas with more shrubs (e.g., White-crowned Sparrow, Table 7) while others appear to be found in areas with few shrubs (e.g., Dark-eyed Junco, Table 7). Pacific-slope Flycatcher abundance was positively correlated with berry producing shrubs and Chestnut-backed Chickadee abundance was negatively correlated with berry producing shrubs. Berry producing shrubs are more abundant in riparian habitats (Chapter 3).

DISCUSSION

Riparian and upland bird community before harvest

At the community level, we found no differences in any of the estimators used to compare species richness between riparian and upland habitats. Contrary to many studies in the eastern and western United States, we actually found slightly more (but not significantly more) species in the upland habitats. McGarigal and McComb (1992) found species richness to be higher in the uplands in the Oregon Coast Range and posited three hypotheses to explain this unusual pattern: 1) high tributary density in the uplands; 2) the relatively wet maritime climate of coastal Pacific Northwest reduces the contrast between upland and riparian habitats; and 3) unique upland structural components such as large conifers and snags may be important to bird species diversity. The uplands of our sites contained many tributaries. Thus, many of the vegetational and structural components of riparian zones also occur in the uplands. The maritime Northwest receives a tremendous amount of rain which likely moderates the moisture gradient between riparian and upland habitats especially when compared to more arid regions of the west. The uplands of our sites did contain more snags yet species abundance and richness of cavity nesting species was not higher in the uplands suggesting that unique structural components of the uplands such as snags may not explain this unusual pattern of specie richness.

We found several species and species groups to be more abundant in riparian habitats than upland habitats. Both Neotropical migrants and resident species were more abundant in riparian habitats. The trend for residents appears to be strongly influenced by the association of the abundant winter wren with riparian habitats. The trend for Neotropical migrants appears to be strongly influenced by the greater abundance of Pacific-slope Flycatcher, Wilson's Warbler, and Black-throated Gray Warbler with riparian habitats.

These species group trends can be explained by examining the habitat associations of the species associated with riparian habitats: American Robin, Winter Wren, Pacific-slope Flycatcher, and Black-throated Gray Warbler. Abundance for several of these riparian associates was correlated with large deciduous trees (red alder, *Alnus rubra*, and big leaf maple, *Acer Macrophyllum*) and berry producing shrubs (primarily salmonberry, *Rubus spectabilis*, and huckleberry, *Vaccinium spp.*) suggesting that these may be important habitat features. Winter Wren was also found to be more abundant along streams in the Oregon Coast Range (McGarigal and McComb 1992) which may reflect the greater cover of deciduous shrubs along streams (Barrows 1986). Black-throated Gray Warbler has been found to be associated with deciduous tree cover (Morrison 1982) and is likely selecting riparian habitats because of the greater cover of red alder and big-leaf maple. Pacific-slope Flycatcher frequently builds its nest behind adventitious branches on red alder trees (S. F. Pearson and M. lieu unpubl. 1992-1999. University of Washington). The American Robin is a ubiquitous species and is often found in edge habitats and may find preferable habitat for foraging and nesting in the deciduous tree and shrub-dominated riparian habitats, All four species appear to be associated with either deciduous trees or berry producing shrubs which are more abundant in riparian habitats than upland habitats,

No species was found. to be significantly more abundant in upland habitats. However four species were more abundant or only found in uplands and may show significant trends with larger sample sizes: Brown creeper, Golden-crowned Kinglet, Hermit/Townsend's warbler, and Dark-eyed Junco. McGarigal and McComb (1992) found Brown Creepers, Dark-eyed Juncos, and Golden-crowned Kinglets to be more abundant in upland habitats. Several of these species (e.g., Hermit/Townsend's warbler, Golden-crowned Kinglet, and Brown Creeper) may be responding to high densities of larger conifers in upland habitats (Mannan et al. 1980, Mammal and Huff 1987, Morrison et al. 1987, Mariani and Manuwal 1990, Pearson and Manuwal 2000). The Dark-eyed Junco may be responding to the mixture of open ground with some shrubby patches that occurs in upland habitats as a result of high canopy closure. This species is most abundant in the uplands after clearcutting (Table 1).

Effect of buffer width in riparian habitats

Species richness increased on State harvest sites relative to controls. Because the area censused on State harvest sites included edge habitat, the increase in species richness on these sites is likely the cumulative result of censusing species associated with open habitats, edge habitats, and forested habitats. Species turnover averaged 20% higher on State harvest sites than on controls while there was little difference in species turnover between modified harvest sites and controls. The high turnover on State harvest sites is caused by the loss of species associated with interior coniferous forests (e.g., Golden-crowned Kinglet and Brown Creeper) and the gain of species associated with open habitats (e.g., Dark-eyed Junco, White-crowned Sparrow and Song Sparrow).

Changes in the bird community along riparian zones post-harvest may be the result of several factors. First, the elongated shape of riparian zones creates a high ratio of edge-to-area. Thus, forest interior species and species sensitive to fragmentation are likely to decline in these habitats (e.g. Black-throated Gray Warbler, Golden-crowned Kinglet, and Brown Creeper; Rosenberg and Raphael 1986, McGarigal and McComb 1995). Conversely, species associated with edge and more open habitats are likely to increase in abundance (e.g., Dark-eyed junco and Song sparrow). Second, harvesting the adjacent upland habitats decreases the amount of coniferous forest and consequently, the deciduous forest along the stream becomes a large component of the remaining patch. Thus, species associated with conifers such as Brown creeper and Golden-crowned Kinglet are likely to decline (Manuwal and Huff 1987, Mariani and Manuwal 1990). Third, changes in bird abundance and richness may be the result of changes in vegetation and micro-climatic regimes. Harvest changes the amount of light penetration and the micro-climatic regime (Brosofske et al. 1997) and consequently the vegetation within the riparian zone. These climatic changes may also influence critical food resources for breeding birds such as insect abundance. Finally, the nature of the adjacent upland vegetation may also influence the riparian community (Szaro and Jakle 1985). In our study, the uplands of both treatments were clearcut and thus should influence both buffer treatments similarly.

Despite small sample sizes, there appeared to be a decline in nesting success on State harvest sites when looking at cup and cavity nests combined. Sites with narrow riparian buffers have a high ratio of edge-to-area. Edge habitats are thought to provide better habitat for nest predators (Wilcove 1985) and are recognized as poor nesting sites for forest-dwelling species because of nest predation (Wilcove 1985, Yahner and Scott 1988, Askins et al 1990, but see Tewksbury et al. 1998). We found no increase in the number of nest predators on State harvest sites. Interestingly, the low nesting success on State harvest sites is primarily caused by the low success of cavity nesters. State harvest sites had fewer trees and snags than modified harvest sites leaving fewer potential nest sites for cavity nesters. With fewer potential nest sites to search, nest predators may be more successful without necessarily being more numerous (Martin and Roper 1988). Because we could not check the actual nest of most cavities, the cavity nest loss may actually be abandonments driven by a decrease in food availability as a result of decreased foraging substrates. Unfortunately, we could not test these possibilities and merely present them as hypotheses begging to be tested.

Treatment effects in upland habitats

Species turnover (nearly 30%) and the local extinction rate (33%) were high on harvested. upland sites relative to controls. Not surprisingly, the bird community was dominated by forest interior species prior to harvest (e.g., Chestnut-backed Chickadee, Winter Wren, and Pacific-slope Flycatcher) and was replaced by species associated with early successional habitats after harvest (e.g., Dark-eyed Junco, Spotted Towhee, and White-crowned Sparrow). Neotropical migrants and resident species declined significantly on harvest uplands but not on controls. The decline in Neotropical migrants and residents on logged sites appears to be caused by the loss of migrant species associated with forested habitats. Species associated with coniferous trees, deciduous trees, shrubs in forested habitats, and cavity nesters declined after their habitat is removed. Not surprisingly, species associated with shrubs in open habitats increased in harvested uplands.

Scope and limitations

There are several limitations to our study that highlight the need for additional research and should be considered before applying these results to management prescriptions, We only described short-term effects of our buffer treatments on the bird community. The breeding bird community may take several years to respond to habitat manipulations (Hagan et al. 1996). This may explain why two riparian associated species, the Pacific-slope Flycatcher and Winter Wren did not decline on harvested sites even though they appear to be sensitive to fragmentation (McGarigal and McComb 1995, Hagar 1999). Thus, we recommend censusing these sites again 5 and 10 yrs post-harvest to document longer-term effects of harvest. This study focused on territorial birds during the breeding season. Consequently, these results cannot be used to infer species-habitat relationships for species that use these habitats during other times of the year. In addition, the methods used here do not adequately census species that occur at low densities or that do not defend territories using song or other audible displays (e.g., Pileated Woodpecker, *Dryocopus pileatus*, raptors, corvids, grouse, waterfowl, and shorebirds). The types of stands and riparian areas selected for this study represent only one important ecological community in a vast array of riparian community types and consequently, these results may not be applicable to all riparian communities. Finally, Riparian buffers may serve many critical biological functions not examined by this study. For example, riparian corridors may facilitate fauna1 mixing throughout the landscape; stream corridors connect forest patches and ecological communities and consequently, they may facilitate genetic and ecological exchange (Noss 1983, Gregory et al. 1991, Machtans et al. 1996).

MANAGEMENT IMPLICATIONS

Riparian habitats in the coastal Northwest appear to be important to Neotropical migrants, resident species, four individual species, and species associated with deciduous trees and shrubs in forested habitats. Consequently, these habitats and their unique ecological features deserve careful consideration when considering management alternatives, Modified harvest sites retained nearly all of the species that occurred before logging and nesting success was similar to controls. On State harvest sites, species turnover was higher and nest predation may be higher. Thus, to maintain the breeding bird community we recommend a variable width riparian buffer (averaging at least 30m) be retained along second and third order streams in managed forests. Large deciduous trees and berry producing shrubs appear to be important habitat features to several bird species associated with riparian habitats and should be given special consideration when managing riparian zones.

The Black-throated Gray Warbler was the only riparian associate to decline on both riparian treatments in the first 2 yr post-harvest and was only found on sites with riparian buffers >30 m post-harvest. This species is a Neotropical migrant and is closely associated with deciduous trees in forested landscapes. Although abundance of this species has not demonstrated significant long (1966-1996) or short term trends (1980- 1996) in the Pacific Northwest (Breeding Bird Survey Data), its dependence on unharvested riparian habitats makes it a species of management concern. Maintaining this species within riparian buffers post harvest requires riparian buffers >30 m in width on each side of the stream. The apparent dependence of this species on deciduous riparian habitats may make it a good indicator species for the health of this habitat type. Ironically, it is likely the disturbance caused by the initial logging of our study sites that created favorable habitat conditions for this warbler. The riparian zone was likely clearcut or heavily disturbed during the initial logging of our study sites 45 to 65 yr ago. This disturbance would have created favorable conditions for the establishment of red alder and big leaf maple. Without further disturbance from fire, logging, wind blow, or water erosion conifer species are likely to become a larger component of the riparian habitat thus decreasing the quality of these habitats for the Black-throated Gray Warbler.

This research is a product of Washington State's Timber Fish and Wildlife Agreement of 1987 that recommended guidelines for the protection of fish and wildlife and the need for management policies to be flexible and responsive to new information. A central feature of the Timber Fish and Wildlife Agreement was the introduction of adaptive management to Washington State's natural resources. Adaptive management is the continual evolution of management practices in response to scientific information gained through monitoring of natural resources and experimental studies that evaluate how resources are impacted by management practices (Walters 1986). This approach treats management activities as experiments that in turn provide information that leads to new and improved management prescriptions. As demonstrated by this research, the adaptive management process can provide both critical information about potential impacts of alternative management activities and insights into basic ecological relationships.

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Table 1. Mean (SE) detection rate of common species in riparian and upland habitats pre-harvest and the mean (SE) detection rate in riparian and upland habitats that were either not harvested (Control), harvested with modified riparian buffers (Modified) or harvested with State regulation riparian buffers (State). Statistical comparisons were made between riparian and upland habitats pre-harvest and between treatments in riparian and upland habitats after harvest. Statistical tests were only performed for common species (see Methods).

Species	Migration	Habitat	Preharvest (1993)				Post-harvest (1995 & 1996)									
			Riparian	Upland			Riparian					Upland				
					t	P	Control	Modified	State	*F*	*P*	Control	Modified	State	*F*	*P*
Chestnut-backed Chickadee	R	C,CAV	3.21 (1.32)	3.20 (1.23)	0.03	0.97	0.60 (0.11)	0.630.12)	0.64 (0.28)	0.01	0.99	0.82 (0.14)	0.13 (0.06)	0.02 (0.02)	19.66	0.00
Winter Wren	R	SC	3.05 (1.03)	1.26 (0.62)	6.32	0.00	2.85 (0.28)	2.62 (0.30)	2.26 (0.36)	0.89	0.43	0.84 (0.14)	0.13 (0.16)	0.27 (0.08)	13.34	0.00
Pacific-slope Flycatcher	NTM	D	1.94 (0.55)	1.21 (0.66)	3.60	0.00	2.40 (0.22)	1.92 (0.22)	1.80 (0.23)	2.12	0.15	1.02 (0.15)	0.0	0.0	35.76	0.00
Golden-crowned Kinglet	R	C	1.03 (0.73)	1.37 (0.98)	-1.20	0.24	0.52 (0.18)	0.05 (0.03)	0.05 (0.03)	4.99	0.02	1.11 (0.30)	0.02 (0.02)	0.0	10.49	0.00
Wilson's Warbler	NTM	SC	0.57 (0.74)	0.43 (0.61)	0.63	0.54	0.84 (0.21)	1.04 (0.14)	0.27 (0.17)	4.41	0.03	0.11 (0.06)	0.0	0.02 (0.02)		
Swainson's Thrush	NTM		0.21 (0.29)	0.23 (0.35)	-0.21	0.84	0.33 (0.09)	0.56 (0.22)	0.25 (0.11)	1.11	0.36	0.12 (0.10)	0.03 (0.02)	0.08 (0.06)		
Brown Creeper	R	C	0.16 (0.18)	0.24 (0.22)	-1.28	0.21	0.29 (0.10)	0.08 (0.04)	0.02 (0.02)	3.70	0.05	0.30 (0.07)	0.0	0.0		
Black-throated Gray Warbler	NTM	D	0.25 (0.26)	0.09 (0.11)	2.38	0.02	0.17 (0.03)	0.05 (0.03)	0.0	8.22	0.00	0.04(0.02)	0.0	0.0		
American Robin	SDM	D	0.18 (0.31)	0.02 (0.07)	2.06	0.05	0.22 (0.08)	0.54 (0.18)	0.53 (0.11)	2.26	0.14	0.15 (0.06)	0.16 (0.05)	0.15 (0.04)		
Rufous Hummingbird	NTM		0.08 (0.20)	0.08 (0.12)			0.08 (0.05)	0.21 (0.07)	0.23 (0.12)			0.03 (0.02)	0.23 (0.11)	0.26 (0.05)		
Hermit/Townsend's Warbler	NTM	C	0.0	0.14 (0.16)			0.0	0.0	0.0			0.37 (0.12)	0.0	0.0		
Cedar Waxwing	SDM	so	0.0	0.02 (0.09)			0.0	0.20 (0.07)	0.29 (0.19)			0.12 (0.07)	0.08 (0.04)	0.1 (0.10)		
Dark-eyed Junco	SDM		0.0	0.10 (0.16)			0.08 (0.05)	0.39 (0.07)	0.76 (0.31)	4.76	0.03	0.12 (0.06)	1.64 (0.30)	1.22 (0.20)	16.37	0.00
Hairy Woodpecker	R	CAV	0.04 (0.08)	0.02 (0.07)			0.26 (0.06)	0.29 (0.06)	0.29 (0.11)	0.04	0.96	0.03 (0.02)	0.12 (0.07)	0.06 (0.02)		
Western Tanager	NTM		0.06 (0.13)	0.05 (0.09)			0.03 (0.02)	0.26 (0.10)	0.06 (0.02)			0.0	0.0	0.02 (0.02)		
Warbling Vireo	NTM	so	0.02 (0.09)	0.02 (0.07)			0.11 (0.11)	0.63 (0.22)	0.43 (0.22)	2.33	0.13	0.0	0.02 (0.02)	0.0		

Table 1. Continued

Species	Migration	Habitat	Pre-harvest (1993)				Post-harvest (1995 & 1996)									
			Riparian	Upland			Riparian					Upland				
					t	P	Control	Modified	state	F	P	Control	Modified	State	F	P
song Sparrow	SDM	SO	0.01 (0.05)	0.0			0.0	0.36 (0.16)	0.86 (0.30)	6.30	0.01	0.01 (0.01)	0.50 (0.21)	0.65 (0.33)	2.97	0.08
Red-breasted Nuthatch	R	CAV	0.0	0.0			0.04 (0.04)	0.03 (0.02)	0.0			0.23 (0.16)	0.02 (0.02)	0.0		
Spotted Towhee	SDM	so	0.0	0.0			0.01 (0.01)	0.0	0.03 (0.02)			0.0	0.29 (0.13)	0.35 (0.14)	3.64	0.05
White-crowned Sparrow	SDM	SO	0.0	0.0			0.0	0.02 (0.02)	0.05 (0.03)			0.0	0.41 (0.17)	0.58 (0.18)	5.45	0.02
Mean abundance/			10.8 (0.53)	8.48 (0.63)	2.81	0.01	8.83 (0.47)	9.87 (0.58)	5.84 (1.41)	0.56	0.59	5.42 (0.36)	3.75 (0.63)	3.76 (0.67)	3.35	0.06

Migration is migratory pattern, where NTM = Neotropical migrant, SDM = short-distance migrant, and R = resident. Habitat is habitat association, where C = coniferous trees, D = deciduous trees, SO = shrubs and small trees in open habitats, SC = shrubs and small trees in forest habitats, CAV = cavity nester.

Table 2. Estimates of bird species richness, proportion of shared species, number of species unique to a habitat, and average species detection probability on riparian and adjacent upland habitats in coastal Washington.

Parameter	Estimate	SE	95% Confidence Interval
Riparian species richness	21.66	3.29	21.66-29.83
Upland species richness	25.89	1.90	22.00-27.89
Members of upland habitats present in the riparian habitats	19.94	2.02	16.94-23.89
Members of riparian habitats present in upland ‘habitats	21.83	3.11	16.94-28.89
Proportion of upland habitat species present on riparian habitats	0.95	0.06	0.81-1.0
Proportion of riparian species present on upland habitats	0.99	0.07	0.75-1.0
Relative richness of riparian and upland habitats	1.20	0.12	0.80-1.24
Number of species unique to upland habitats	5.32	2.20	0.00-7.5
Detection probability in riparian habitat	0.97	0.08	0.70-0.97
Detection probability in upland habitat	0.85	0.07	0.79-1.0

Table 3. Mean (SE) detection rate of species groupings in riparian and upland habitats preharvest and the mean (SE) detection rate in riparian and upland habitats that were either not harvested (control sites), harvested with modified riparian buffers (Modified) or harvested with State regulation buffers (State). Statistica! comparisons were made between riparian and upland habitats pre-harvest and between treatments in riparian and upland habitats after harvest. Statistical tests were only performed for common species (see Methods). Species are grouped according to migratory pattern, habitat association, and nest predators.

Species Group	Preharvest (1993)				Post-harvest (1995 & 1996)									
	Riparian	Upland			Riparian					Upland				
			t	P	Control	Modified	State	F	P	Control	Modified	State	F	P
Neotropical migrants	3.12(0.34)	2.25(0.27)	2.02	0.05	3.94(0.41)	4.66(0.40)	3.04(0.77)	2.28	14	1.70(0.27)	0.28(0.11)	0.37(0.11)	16.87	0.00
Short-distance migrants	0.20(0.08)	0.14(0.04)	0.52	0.61	0.31(0.10)	1.50(0.27)	2.53(0.48)	15.43	0.00	0.40 (0.12)	3.07(0.62)	3.05(0.57)	12.21	0.00
Residents	7.48(0.32)	6.09(0.48)	2.41	0.02	4.58 (0.23)	3.71(0.34)	3.27(0.61)	3.08	0.08	3.33(0.34)	0.41(0.10)	0.34(0.09)	52.65	0.00
Coniferous trees	4.39(0.40)	4.94(0.51)	-0.85	0.40	1.42(0.24)	0.76(0.15)	0.71(0.29)	3.13	0.07	2.60(0.43)	0.14(0.08)	0.02 (0.02)	26.04	0.00
Deciduous trees	2.37(0.21)	1.33(0.16)	3.97	0.00	2.78(0.29)	2.50(0.39)	2.33(0.22)	0.51	0.61	1.21(0.21)	0.16(0.05)	0.15(0.04)	19.13	0.00
Shrubs in open habitats	0.03(0.02)	0.04(0.03)	-0.31	0.76	0.12(0.11)	1.20(0.35)	1.66(0.65)	4.60	0.03	0.13(0.06)	1.29(0.45)	1.69(0.52)	5.16	0.02
Shrubs in forested habitats	3.62(0.31)	1.68(0.22)	5.05	0.00	3.69(0.41)	3.67(0.16)	2.53(0.43)	3.11	0.07	0.96(0.11)	0.13(0.06)	0.28(0.07)	23.97	0.00
Cavities	3.25(0.32)	3.42(0.33)	-0.38	0.71	0.91(0.16)	0.95(0.04)	0.93(0.70)	0.12	0.99	1.53(0.41)	0.26(0.11)	0.07(0.02)	7.99	0.00

Table 4. Relationship between species abundance and habitat variables measured in the pre-harvest year (1993) for species associated with riparian habitats (*P* < 0.1, Table 1). Values are correlation coefficients (*P* values) for habitat variables included in the regression model. Habitat variables are defined in Table 5.

Habitat Variable	Species			
	American Robin	Black-throated Gray Warbler	Pacific-slope Flycatcher	Winter Wren
CANOPY				
MAPLE				
DECID <50				0.38(0.02)
DECID >50		0.617(0.003)		
TSHE <50				
TSHE >50				
PSME <50	-0.468(0.089)			
PSME >50				
EVSHR			-0.358(0.006)	
BPSHR	0.408 (0.005)		0.459(0.004)	0.421(0.02)
ODSHR	-0.079 (0.117)			
SNAG				
LOG				
R^2	0.34	0.243	0.383	0.406

Table 5. Description of habitat variables used to examine species-habitat relationships

Habitat Variable	Description
CANOPY	Percent canopy cover
MAPLE	Number of vine maple stems
DECID <50	Number of red alder and big leaf maple trees <50 cm dbh
DECID >50	Number of red alder and big leaf maple trees >50 cm dbh
TSHE <50	Number of western hemlock and western red cedar <50 cm dbh
TSHE >50	Number of western hemlock and western red cedar >50 cm dbh
PSME <50	Number of Douglas-fir trees <50 cm dbh
PSME >50	Number of Douglas-fir trees >50 cm dbh
EVSHR	Percent cover of evergreen shrubs 1-3 m tall
BPSHR	Percent cover of berry producing shrubs 1-3 m tall
ODSHR	Deciduous shrubs 1-3 m tall other than berry producing shrubs
SNAG	Total number of snags >1.5 m tall and >10 cm dbh
LOG	Total number of logs >10 cm diameter

Table 6. Relationship between species abundance and habitat variables measured in riparian habitats post-harvest (1995-1996). Only species demonstrating a treatment effect in the riparian habitats were included ($P < 0.1$, Table 1). Values are correlation coefficients (P values) for habitat variables included in the regression model. Habitat variables are defined in Table 5.

Habitat Variable	Species					
	Black-throated Gray Warbler	Brown Creeper	Dark-eyed Junco	Golden-crowned Kinglet	Song Sparrow	Wilson's Warbler
CANOPY	0.67(0.003)	0.54(0.002)	-0.69(0.002)		-0.59(0.01)	
MAPLE						
DECID <50		-0.30(0.113)				
DECID >50		-0.24(0.155)				
TSHE <50						
TSHE >50						
PSME <50						0.26(0.143)
PSME >50				0.62(0.026)		
EVSHR						
BPSHR	0.56(0.003)					0.43(0.044)
ODSHR						
SNAG				0.71(0.005)		
LOG						
R^2	0.445	0.554	0.470	0.502	0.347	0.294

Table 7. Relationship between species abundance and habitat variables measured in riparian habitats post-harvest (1995-1996). Only species demonstrating a treatment effect in the uplands were included ($P < 0.10$, Table 1). Values are correlation coefficients (P values) for habitat variables significantly correlated with species abundance. Habitat variables are defined in Table 5.

Habitat Variable	Species						
	Chestnut-backed Chickadee	Dark-eyed Junco	Golden-crowned Kinglet	Pacific-slope Flycatcher	Spotted Towhee	White-crowned Sparrow	Winter Wren
CANOPY	0.83(0.000)	-0.82(0.000)		0.91(0.000)	-0.60(0.002)	-0.65(0.001)	0.71(0.001)
MAPLE							
DECID <50		-0.10(0.120)				0.38(0.008)	
DECID >50	0.30(0.002)						0.55(0.099)
TSHE <50			0.76(0.000)				
TSHE >50							
PSME <50		-0.15(0.042)			0.06(0.082)		
PSME >50							
EVSHR							-0.03(0.080)
BPSHR	-0.05(0.006)			0.38(0.105)			
ODSHR							
SNAG							
LOG						0.21(0.037)	
R^2	0.856	0.769	0.584	0.862	0.476	0.678	0.681

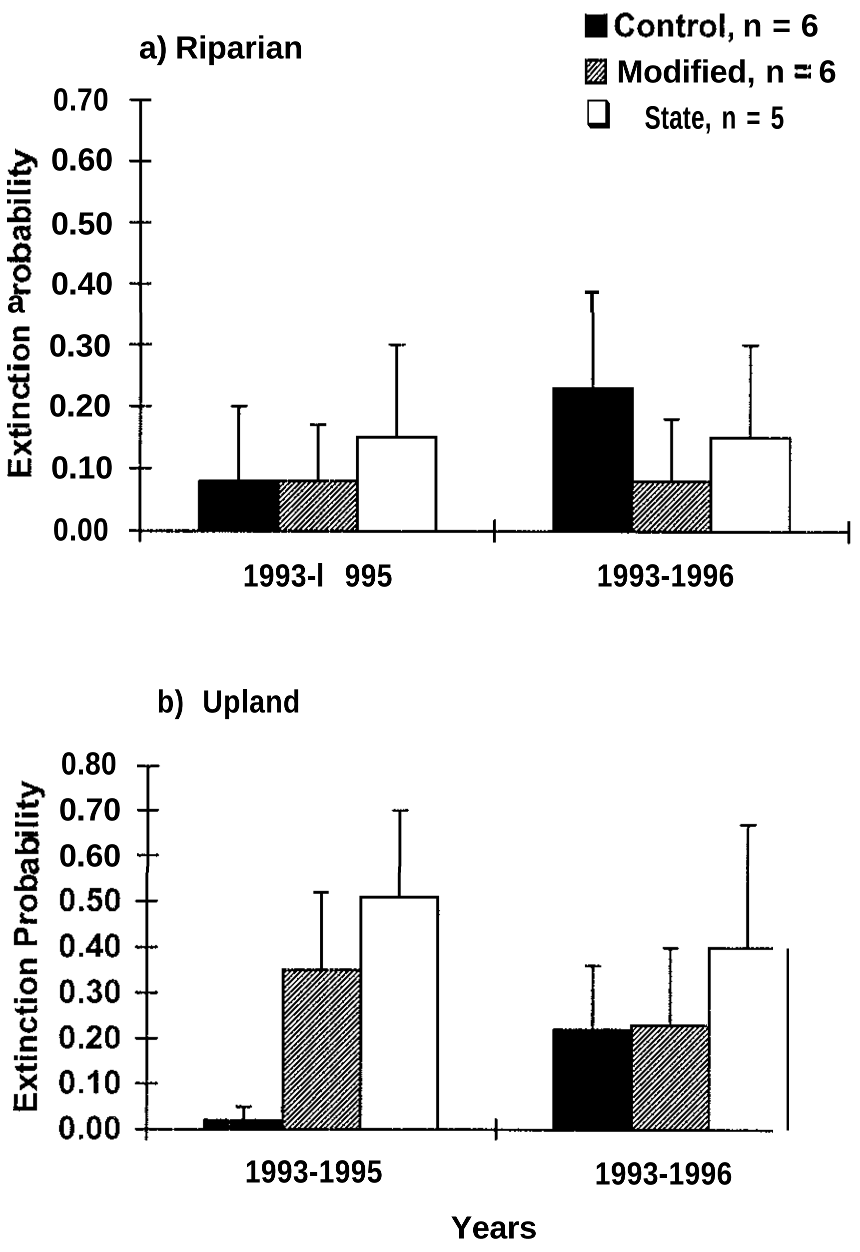
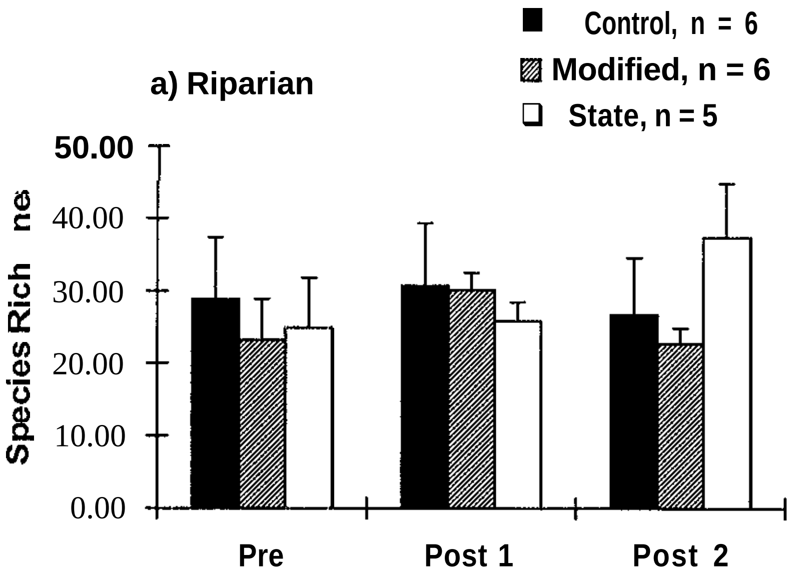


Figure 1. Probabilities of local extinction when comparing the pre-harvest year and the first year after harvest (1993-1995) and the pm-harvest year and the second year after harvest (1993-1996) in a) riparian and b) upland habitats that were either not harvested (Control), harvested with modified buffers (Modified), or harvested with State Regulation buffers (State). Bars are means + 1 SE.



a) Riparian

b) Upland

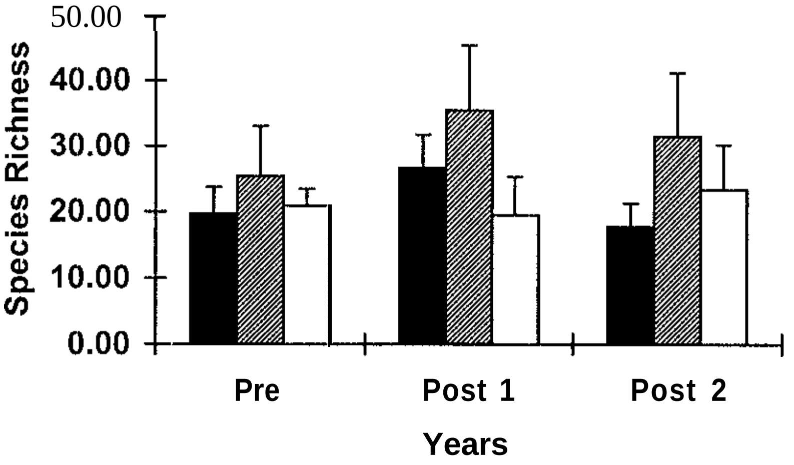
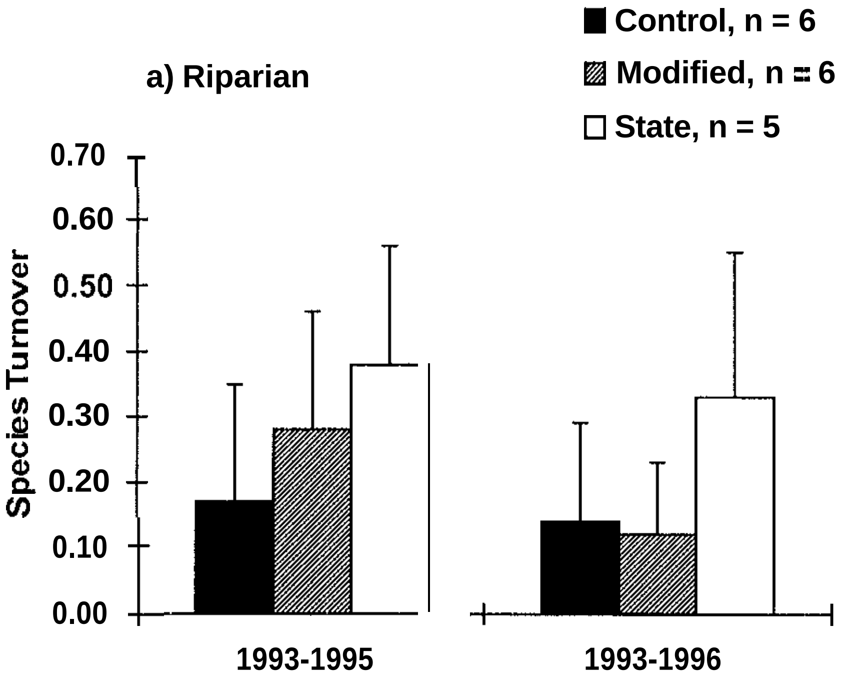


Figure 2. Species richness in the pre-harvest year (Pre), and the first (Post 1) and second (Post 2) years after harvest in a) riparian and b) upland habitats that were either not harvested (Control), harvested with modified riparian buffers (Modified) or harvested with State regulation riparian buffers (State). Bars are means + 1 SE.



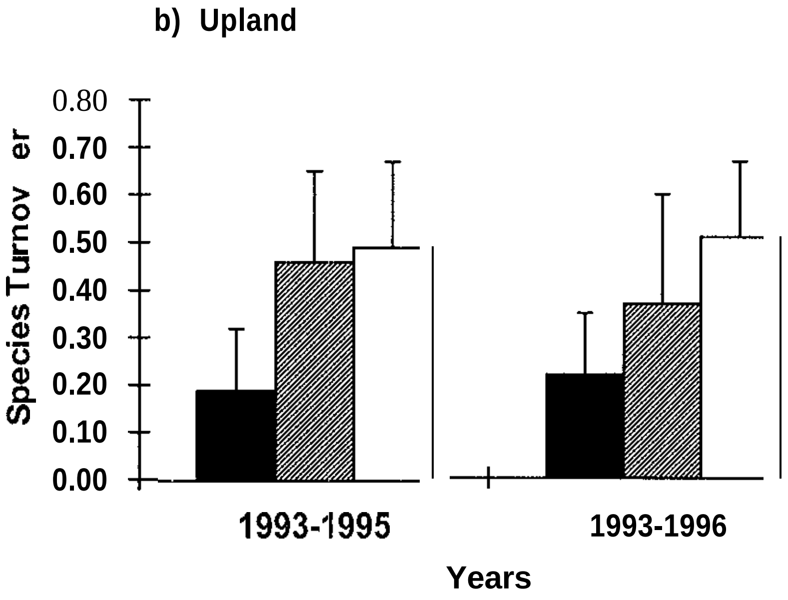


Figure 3. Species Turnover when comparing the pre-harvest year and the first year after harvest (1993-1995) and the pre-harvest year and the second year after harvest (1993.1996) in a) riparian and b) upland habitats that were either not harvested (Control), harvested with modified riparian buffers (Modified), or harvested with State regulation buffers (State). Bars are means + 1 SE.

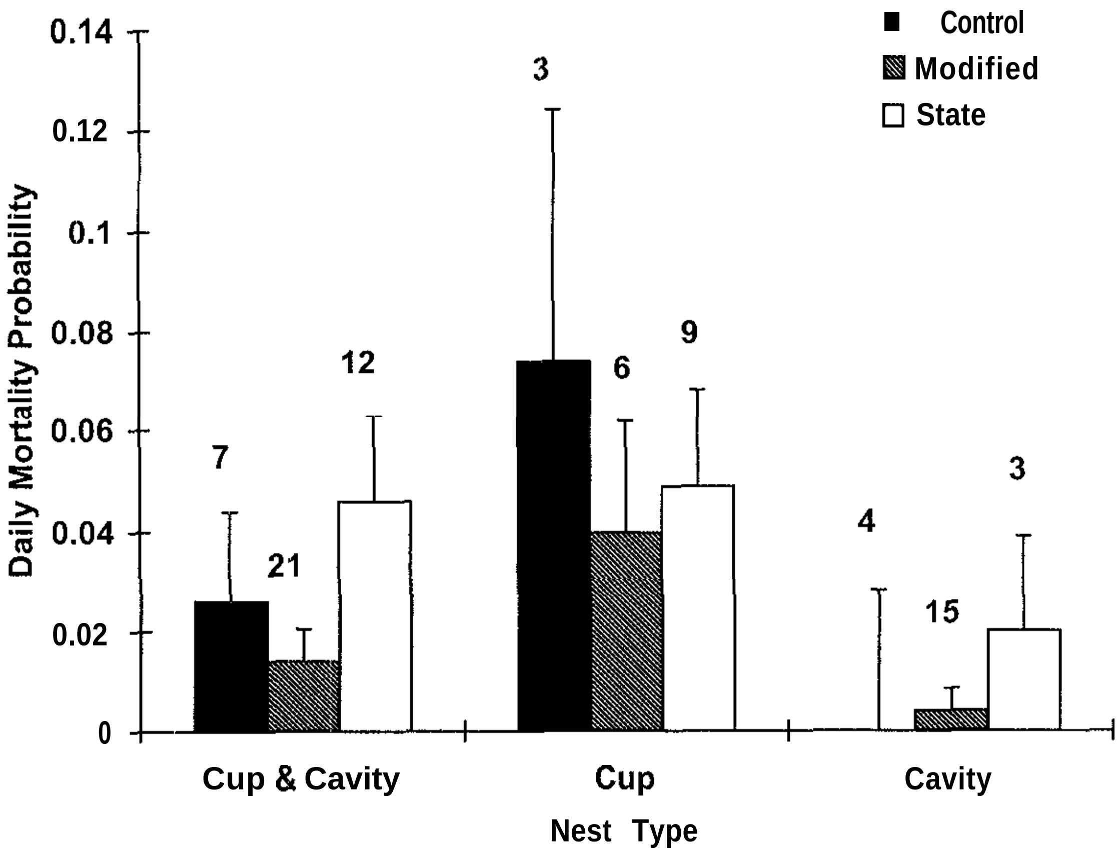


Figure 4. Daily mortality probability of cup and cavity nests on sites that were not harvested (Control), sites harvested with modified riparian buffers (Modified) and sites harvested with State regulation riparian buffers (State). Numbers above bars are sample sizes.

APPENDIX I

We censused one control stand and one modified harvest stand six times in 1997 and 1998. We did not include the results from these censuses in our overall analysis because of possible year effects that cannot be tested with a single control. Censusing only one of the treatments in 1997 and 1998 and using it in our statistical test of treatment could bias our result if there is a year effect. As a result, we present a brief qualitative summary of the results from the 1997 and 1998 censuses.

The general trends for most species appear to be similar to the 1995 and 1996 post-harvest years but the overall detection rate for the most abundant species (Chestnut-backed Chickadee, Winter Wren and Pacific-slope Flycatcher) is lower (compare Tables 1 and 8). The detection rate of species associated with forested habitats is lower in the upland of the modified harvest stand than the control (Table 8). The detection rate of species associated with riparian habitats (Winter Wren and Pacific-slope Flycatcher) on the modified harvest site appears to be similar or higher than that of the control stand (Table 8).

Table 8. Mean detection rate for species detected two or more times in 1997 and 1998. Species were detected in riparian or upland habitats on a unharvested control stand and a stand cut with a modified riparian buffer.

Species	Post-harvest 1997& 1998			
	Riparian		Upland	
	Control	Modified	Control	Modified
Chestnut-backed Chickadee	0.25	0.50	0.67	0.08
Winter Wren	1.58	2.41	0.33	0.67
Pacific-slope Flycatcher	0.58	0.92	0.33	0.0
Golden-crowned Kinglet	0.17	0.0	0.67	0.0
Wilson's Warbler	0.5	1.25	0.0	0.0
Brown Creeper	0.33	0.0	0.33	0.0
Rufous Hummingbird	0.08	0.17	0.0	0.08
Dark-eyed Junco	0.0	0.17	0.0	0.83
Hairy Woodpecker	0.08	0.33	0.0	0.25
Warbling Vireo	0.0	0.17	0.0	0.0
Steller's Jay	0.08	0.0	0.08	0.0

Chapter 6

EAST-SIDE AVIAN SURVEYS

Abstract. We experimentally examined the effects of two prescriptions for riparian buffer strips on bird populations in NE Washington: Washington State guidelines and a modified prescription that buffered snags and seeps in the riparian zone. We studied 18 streams including 7 unharvested Controls, 6 State harvest sites, and 5 Modified harvest sites. Two 800-m transects were established parallel to the stream and at 8 m (riparian) and 100 m (upland) from the stream. Bird surveys were conducted during spring 1992.1994 (pre-harvest) and 1995-1996 (post-harvest) using a modified belt-transect design. Avian species richness, abundance, and diversity were either equal or greater in upland than in riparian habitats, and few species were found predominantly in the riparian. Most species maintained the same habitat associations after harvest. There were no differences due to treatment for species richness, turnover rates, diversity, evenness, or overall abundance in the riparian habitats, Of 22 common species, only four species exhibited a change in abundance in the riparian habitat after harvest. Although overall avian diversity and abundance were comparable, the abundance of individual riparian species was better retained and more positively associated with the Modified prescription as compared to the State prescription.

INTRODUCTION

Riparian habitats have long been considered critical habitat for many wildlife species because of the presence of surface water, complex vegetation and structural features, high productivity, and natural travel corridors and migration routes (e.g., Thomas 1979, Oakley et al. 1985). The importance of riparian habitat to avian populations depends on a variety of factors including climatic conditions, riparian and adjacent upland vegetation, time of year, individual bird species characteristics, stream size and structure, edge to area ratios, and microclimatic conditions (O'Connell et al. 1993). It is therefore not surprising that, as these factors vary, the response of avian populations to riparian habitats might vary. For example, studies in more arid climates of the Southwest consistently report pronounced differences between riparian and upland habitats with respect to avian diversity, species richness, and density, (e.g., Johnson et al. 1977, Strong and Bock 1990), whereas in the more mesic climates of the eastern and Pacific Northwest forests, patterns of abundance and diversity between riparian and upland habitats can be more variable (e.g., McGarigal and McComb 1992, Murray and Stauffer 1995, Sparkman and Hughes 1995, Kinley and Newhouse 1997).

Modifications of vegetational composition and structure in riparian and adjacent upland habitats will impact avian response to riparian habitats. In the Pacific Northwest, where 80-90% of the original mature and old-growth forests have been converted into a mosaic of different successional stands by timber management (e.g., Spies and Franklin 1988), the potential for impacts on avian response to riparian habitats is great. Different silvicultural practices in the upland (e.g., clearcutting, selective harvest, small-patch group-selection) and the riparian (e.g., no-entry reserve buffers, limited-harvest management buffers, no buffers) habitats will impact avian abundance and diversity in riparian habitats.

A growing number of studies have examined avian responses to upland stand conditions created by different silvicultural practices in managed forests of the Pacific Northwest (e.g., Manuwal and Huff 1987, Hagar et al. 1996, Bosakowski 1997, Chambers and McComb 1997, Manuwal and Pearson 1997, O'Connell et al. 1997). Much less attention has focused on the effects of silvicultural practices on avian populations in the riparian habitats in this region. State and federal forest regulations in this region mandate riparian buffer zones that can be either no or limited harvest entry and can vary in width depending upon stream size, location, upland harvest prescription, and land ownership. In Washington, for example, Riparian Management Zones (RMZ) buffering Type 3 streams were established by the Forest Practices Board (Washington Forest Practices Board 1988) to be 8 m wide on clearcut harvests west of the Cascade Crest and 10 to 16.6 m wide on selective and clearcut harvests, respectively, east of the Cascade Crest. Limited harvest entry is permitted in the RMZ's on both sides of the Cascades. Studies examining the effects of riparian buffer zones on avian populations in this and other regions (e.g., Stauffer and Best 1980, Kinley and Newhouse 1997) have focused primarily on the question of buffer width, comparing bird populations in previously created buffers of varying widths. In this study we examined the effects of riparian buffer zones on avian populations through an experimental approach, comparing bird populations in riparian and adjacent upland habitats before and after a partial timber harvest in forests on the east side of the Cascade Crest in Washington, Our goals were 1) to compare avian species richness, diversity, and abundance between riparian and upland habitats, 2) examine how different harvest practices in the riparian zone affect avian species richness, diversity, and abundance, and 3) to examine the habitat correlates that might provide insight into the observed patterns of species richness, diversity, and abundance.

METHODS

Study area

Research was conducted in mixed-coniferous forests in the Selkirk Mountains of northeastern Washington (Stevens and Pend Oreille counties). Forest composition in this region is variable and is affected by slope, aspect, edaphic factors, tire history, and timber management practices. Dominant tree species include Douglas-fir (*Pseudotsuga menziesii*), lodgepole pine (*Pinus contorta),* western redcedar (*Thuja plicata*), western hemlock (*Tsuga heterophylla),* western larch (*Larix occidentalis*), grand fir (*Abies grandis*), and alders (*Alnus incana* and *Alnus sinuata*). Shrubs included gooseberry (*Ribes* spp.), devil's club (*Oplopanax horridum*), Oregon grape (Berberis spp.), mountain boxwood (*Pachistima myrsinites*), red-osier dogwood (*Cornus stolonifera*), ninebark (*Physocarpus malvaceus),* spireae (*Spireae* spp.), serviceberry *(Amelanchier alnifolia*), rose *(Rosa* spp.), huckleberry *(Vaccinium* spp.).

We selected 18 sites that met the following criteria: 1) a minimum 800-m reach of Type 3 or permanent Type 4 stream; 2) >16.2 ha of previously harvested stands at harvestable age on either side of the 800-m reach; 3) ≥610 m and ≤1200 m elevation; 4) mixed coniferous forest; 5) landowners agreed to either leave sites unharvested for 10 yr (controls) or to harvest sites within timeframe and specifications of study design (cut sites). Seven sites were unharvested control sites. The upland areas of 11 sites were selectively harvested in 1994.1995 to yield a 6- to 12-m spacing of trees. The riparian zones of 6 of the 11 sites were harvested according to the Washington State Forest Practices RMZ guidelines (State sites) and 5 of the 11 sites were harvested according to a modified prescription designed for this project (Modified sites). The Modified RMZ incorporated site-specific guidelines. Within 33-m zone of the stream, habitat features such as seeps, snags, and deciduous trees, were identified and protected. For example, 1 snag per 2 acres was buffered by a no-entry zone equal to 1.5 times the height of the snag, and all seeps were buffered by a 10-m no-entry zone that extended to the stream. Following timber harvest, the mean width of the State RMZ buffers was 14.1 ± 3.0 m with a range from 8-22.6 m. and the mean width of the Modified RMZ buffers was 29.7 ± 17.4 m with a range from 12 to 144 m.

Transect design

At each of the 18 study sites we established two 800-m riparian transects. Each transect paralleled the stream, one at 8-m distance from the stream high water mark (about half way from the stream edge to the boundary of a regulatory RMZ) and another 100-m upslope from the first transect, We marked each of these transects with flags at 50-meter increments to serve as reference points during the bird surveys and vegetation studies.

Bird surveys

Bird surveys were conducted during May and June 1992-1996 using a modified belt transect design. All transects were visited six times per year during this period. To maximize the probability of recording all bird species present on a transect, regardless of arrival and breeding times, surveys were scheduled so that each transect was visited at regular intervals throughout the breeding season. In northeastern Washington a period of extensive singing occurs between mid-May and mid-June. A single observer walked both the riparian and upland transects of a site during a survey, alternating which transect was sampled first between visits to increase the probability of observing both early and late morning singers on both transects. Surveys began at 0500, and observers walked the 800-m transects at an average pace of 5 min per 50-m increment of transect. Surveys were not conducted on days of high wind or rain.

The focal areas along the riparian belt were the 8 m between the transect and the stream and the 22 m on the upland side of the transect for a total belt width of 30 m. Birds seen or heard in this 30-m belt were recorded as being either stream side or upland. Birds observed on the opposite side of the stream, regardless of their distance from the transect, were recorded in the across stream zone. Birds observed beyond the 22 m of the upland transect were recorded as out of the riparian area.

On the upland transect, one 30-m wide belt, 15 m on each side of the transect, was the focal survey area. Birds observed in this area were recorded as inside the survey area. Birds observed beyond the 15 m on the stream side of the transect, regardless of distance were recorded as streamside. Birds observed beyond the 15 m on the upland side of the transect were recorded as upland. In addition, birds flying over the transect were recorded as flyovers and the number of birds in flocks were recorded (or recorded as "flock" if number of individuals could not be determined).

Habitat sampling

Habitat features were sampled in 1992 (pre-harvest) and 1994-1995 (post-harvest). At 50-m intervals along each belt transect we established a 16 x 20-m plot that was divided into four 8 x 10-m quadrants.

Trees and snags

Within each 20 x 16-m plot all trees were identified to species and assigned to one of four DBH classes: Class 1 (4-10 cm); Class 2 (11-25 cm); Class 3 (26-50 cm); Class 4 (>50 cm). All snags within each plot were counted and designated as either Condition 1 (bark basically intact) or Condition 2 (bark peeling off to absent). Four representative live trees and two snags were chosen at random and their heights were estimated using a clinometer.

Canopy cover

Percentage ofoverstory and understory cover was measured with a convex spherical densiometer at the center of the 20 x 16 m plot and at the center of each 8 x 10-m quadrant for a total of five measurements per site that were then averaged.

Shrubs and regenerating trees

From the center point of the four 20 x 16-m plots, the distance to the nearest shrub (>0.5 m high) in each of the quadrants was measured and the area of each shrub (length x width) was recorded. In two opposite quadrants, the numbers of regenerating coniferous trees (>0.5 m high; <4 cm DBH) were recorded.

Woody Debris

In two opposite quadrants within each plot the number and decay class of woody debris and stumps were recorded. Logs were assigned to one of four size classes and to one of four decay classes. Size classes were: 1) ≥5m long x <15 cm circumference; 2) ≥5 m long x 16-24 cm circumference; 3) ≥5 m long x >25 cm circumference; 4) <5 m long x >25 cm circumference. Decay classes were defined as: 1) freshly fallen tree with bark essentially intact, wood solid, no decomposition; 2) bark beginning to slough or almost completely gone, decomposition begun with sapwood partially softened but log generally firm; (3) decomposition progressed to the point that wood is generally soft and breaks into chunks, each chunk still as integrity; (4) essentially no integrity to log, wood decomposed to point of soil-like texture. Stumps were assigned as either “natural” or “cut and to one of the four above decay classes. Stumps were differentiated from snags by height; stumps il.37 m high (standard breast height).

Data analysis

Individual stands represent the experimental units for all statistical analyses. We defined species richness as the total number of species detected. We calculated species turnover rates as the proportion of species that were unique at an individual site between two consecutive years. We used Microsoft Excel to calculate Shannon-Wiener Diversity Index and Evenness values. We defined the index of abundance as the average number of detections per riparian or upland transect over the six site visits per year. For example, if the Dark-eyed Junco was detected 36 times during the six visits to one site during 1992, the abundance index would be 6.0. We examined abundance on three levels: I) the total abundance of all species, 2) the abundance of the three migrant types (permanent resident, short-distance migrants, and Neotropical migrants), and 3) the abundance of individual species with >75 detections/species.

For the pre-harvest and post-harvest comparison of species richness, turnover rates, species diversity, and species abundance between riparian and upland habitats we used two way Analysis of Variance (ANOVA) with Tukey's HSD tests to compare means. To examine the treatment effects on species richness, turnover rates, species diversity, and abundance, we used ANOVA with a repeated measure for time. To construct the datasets for the repeated measures ANOVA we either calculated (e.g., total number of species detected) or averaged (e.g., turnover rate) the particular measure for the pre-harvest years and the post-harvest years.

To examine the relationship between the habitat variables and the abundance of individual species with >75 detections/species, we used a stepwise multiple regression. Given that the bird observations were counted along a belt transect, not a point-count station, the detections for two adjacent increments of the belt transect (i.e., O-l and 1-2; 2-3 and 3-4, etc.) were summed and those values were used in the regression with the habitat variables from the mid-point (i.e., 1, 3, etc.). In addition to the habitat variables, we incorporated four additional dummy variables in the regression model. The first dummy variable represented the habitat zone (0 for upland, 1 for riparian), the second represented sampling time (0 for pre-harvest, 1 for post-harvest), the third represented the State harvest treatment (1 for State, 0 for Modified and Control), and the fourth dummy variable represented the Modified harvest treatment (1 for Modified, 0 for State and Control).

To reduce the number of habitat variables used in the analysis we followed several procedures. First, the number of trees was summed by size class for deciduous and coniferous trees rather than individual tree species. Second, we performed principal components analysis on the data for the downed logs. The recent decay class logs were highly correlated ($r > 0.60$) and were positively associated ($r > 0.40$) with the first principal component that explained 20% of the variance. The second principal component explained an additional 15% of the variance and was associated ($r > 0.30$) with logs in the two oldest decay classes. The third principal component explained and additional 10% of the variance and was positively associated ($r > 0.2$) with logs in the second decay class. We therefore summed the counts of all recent decay class logs, those of the two oldest decay classes, and those of the second decay class to reduce the number of variables for downed logs from 16 to 3. Third, in a similar fashion we performed principal components analysis on the data for the stumps. Four principal components explained 60% of the variance and were clearly associated with different stump decay and type classes. The first principal component was positively associated ($r > 0.5$) with the older natural stumps, the second principal component was positively associated ($r > 0.5$) with the medium to older cut stumps, the third principal component was positively associated ($r > 0.4$) with the recent natural stumps, and the fourth principal component was positively associated with ($r > 0.5$) with the recent cut stumps, We therefore summed the counts for these size class and type of stumps to reduce the number of variables from eight to four.

We used the Statistical Analysis System (SAS Institute 1989) to conduct all analyses. All statistical tests were considered significant at $P \leq 0.05$ significance level unless otherwise noted.

Nest predation studies

During 1992 and 1995, we conducted experimental studies to examine the rates of nest predation in the riparian and upland habitats before and after timber harvest. Methods for these studies have been detailed in Entz (1995) and Hackworthy (1996).

Artificial nests constructed from commercially produced open-cup canary nests camouflaged with native grasses to mimic real nests were used. Martin (1987) reported that rates of predation on camouflaged nests were similar to those on natural nests. Nests were placed on the ground and in trees at about 4-m height to mimic the nests of the Dark-eyed Junco and Varied Thrush, two common species. Nests were baited with three quail eggs, a typical clutch size for both the Dark-eyed Junco and Varied Thrush. Ten ground and 10 arboreal nests were placed at 25-m intervals along the riparian and upland transects of 12 study sites. Nest predation studies were conducted during May and June 1992 and 1995. Nests were placed in April and provisioned with eggs at the beginning of each of the 14-d sampling periods. Nests were checked every 7 d, using a pole mirror or climbing gear. Nests were considered preyed upon if the nests were displaced or if eggs were missing, moved, scratched, or pecked. There was a 10-d lag time between sampling periods to minimize any effects of prolonged disturbance.

Rates of nest predation were defined as the proportion of nests disturbed per day. Rates of nest predation were compared between habitat type (riparian or upland), nest type (tree or ground), and between harvest treatments using ANOVA repeated for time and a Tukey’s mean separation test to compare mean rates.

RESULTS

We recorded a total of 17,944 observations of 85 species on all sites combined during the 5-yr study. Of these, 11,745 observations of 78 bird species were within the 30-m width transects and are used in subsequent analyses (Table 1). Species that were recorded in the riparian habitat, but not counted on the transects, included Barred Owl, Black-chinned Hummingbird, and European Starling and in the upland habitat, Canada Goose, Flammulated Owl, Great-homed Owl, Lazuli Bunting, Northern Saw-whet Owl, and Pine Grosbeak, and in both habitats, Lincoln Sparrow, Osprey, and Veery.

Four of the species observed within the transect boundaries (American Dipper, Fox Sparrow, Song Sparrow, and Yellow Warbler) had been identified as riparian obligates in our previous review ofriparian wildlife (O'Connell et al. 1993). Nine species (Black-backed Grosbeak, Dusky Flycatcher,, MacGillivray's Warbler, Northern Waterthrush, Red-eyed Vireo, Warbling Vireo, Western Flycatcher, Warbling Vireo, Western Tanager, and Yellow Warbler) had received moderate to high (>10) sensitivity scores in our ranking of the vulnerability of Washington's riparian wildlife (O'Connell et al. 1993).

There was significant interyear variation in the number of detections (Fig. 1) in both the riparian ($F = 7.81$, df = 4,82, $P = 0.0001$) and upland habitats ($F = 18.19$, df= 4,82, $P = 0.0001$). In the riparian habitat, the number of detections were highest and similar in 1994 and 1995, lowest and similar in 1992 and 1996, and were comparable in 1992, 1993, 1994. In the upland habitat, there were no differences between the mean number of detections per site for 1994 and 1995 and no differences between those for 1992, 1993, and 1996.

Riparian versus upland - pre harvest

Species richness and turnover

Before harvest the number of species recorded per site per year ranged from 5 to 21 in the riparian and from 7 to 25 in the upland. The mean number of species per site was similar between the riparian and the upland habitat of all 3 treatments (Control, State, Modified) (Table 2). Most species were observed in both riparian and upland habitats and relatively few were exclusively found in the riparian habitat (Table 1, Fig. 2). The number of Neotropical and short-distance migrant species was similar between the 2 habitats but there were slightly more species of permanent residents in the riparian habitat (Fig. 3).

Species turnover between years averaged >50% across all sites and habitats (Fig. 4). Before harvest, there were no differences in turnover between riparian ($\bar{x}$ = 0.59 ± 0.025) and upland ($\bar{x}$ = 0.53 ± 0.028) habitats (F = 3.26, df = 1,46, P = 0.08). Nor were there any differences between the three treatments with respect to turnover in the riparian habitat (F = 2.93, df = 2,21, P = 0.076). In contrast, turnover was higher in the upland habitat of the State sites than of the Control and Modified sites (Fig 4; F = 3.47, df = 2,21, P = 0.05).

Species diversity

Across all years, diversity values were consistently higher in the upland than in the riparian habitats and evenness values averaged >0.80, suggesting that the abundance of bird species was distributed relatively even, with no single species dominating (Table 3). During the preharvest years on the Control sites, there were no differences in diversity between the two habitats (Table 3). In contrast, prior to harvest on both the State and Modified sites, avian diversity was significantly higher in the upland habitat (Table 3).

Abundance

Before harvest, the mean detection rate for all species combined was 11 individuals/visit with no differences between the riparian and upland habitats (Fig. 5; F = 0.14, df = 1,82, P = 0.709) and no differences between the three treatments (Riparian: F = 1.20, df = 2,39, P = 0.313 Upland: F = 0.04, df = 2,39, P = 0.961).

The mean abundance of shot--distance migrants was greater than that of either the Neotropical migrants or the permanent residents (Fig. 6). Before harvest, the abundance of the short-distance migrants was greater in the riparian than the upland habitats (F = 5.41, df = 1,8 1, p = 0.022), whereas the abundance of the permanent residents was greater in the upland habitats (F = 6.52, df = 1,81, P = 0.01). There were no habitat differences in the abundance of the Neotropical migrants (F = 0.23, df = 1,81, P = 0.631).

Comparison of abundance between riparian and upland habitats across all sites prior to harvest revealed that of the 22 common species, Golden-crowned Kinglets, Hammond's Flycatchers, and Winter Wrens were more abundant in the riparian than upland habitat and Chestnut-backed Chickadees, Chipping Sparrows, Dark-eyed Juncos, Gray Jays, Nashville Warblers, Red-breasted Nuthatches, Yellow-rumped Warblers were more abundant in the upland habitat (Table 4). Of these, only gray jays, Hammond's Flycatchers, Winter Wrens, and Red-breasted Nuthatches were consistently more abundant in their respective habitats across all sites, the habitat association of the species varied between the Control, Modified, and State sites (Table 4). Although detections of the Red Crossbill were <75, it was only observed in the upland habitat (Table 1).

Observations of the species we had identified as either riparian obligate or sensitive (O'Connell et al. 1993) tended to be either very site specific or year specific. With the exception of the riparian zone of two Control sites, the Fox Sparrow, a riparian obligate, was observed exclusively in a single drainage, Muddy Creek, where it was observed on both riparian and upland transects. Song sparrows, another riparian obligate, were observed primarily on the riparian transects and most of these observations were on one State site.

Detections of five of the nine species that we had classified as vulnerable were sufficient to allow overall habitat comparisons, but not by treatment (Table 4). During the years before harvest, there were no differences in the abundance of Dusky Flycatcher or MacGillivray's Warbler between riparian and upland habitats. The Northern Waterthrush was more abundant in the riparian habitat in the pre-harvest years. Although the Warbling Vireo and Western Tanager have been classified in the literature as species that use riparian habitats, both species were more abundant before harvest in the upland habitats. Indeed, only 6 of the 46 observations of Warbling Vireos were on riparian transects, and 28 observations were from the upland transect of a single State site during both pre- and post-harvest years. Similarly, western tanagers were more observed more often in the upland (n = 45) than riparian (*n* = 14) habitats, but this species was never observed more than a few times at any single site.

Riparian versus upland - post harvest

Species richness and turnover

After timber harvest on the Modified and State sites the number of species recorded per site per year ranged from 7 to 22 in the riparian and from 7 to 29 in the upland. Similar to pre-harvest conditions, there were no differences in mean number of species per site in the riparian habitat of the Control and the Modified sites (Table 2). In contrast to pre harvest conditions, there were more species in the upland habitat of the State sites (Table 2).

After harvest on the Modified and State sites, turnover remained >50% with no differences between the riparian ($\bar{x}$ = 0.56 ± 0.013) and upland ($\bar{x}$ = 0.53 ± 0.013) habitats (Fig. 4; *F* = 3.34, df= 1,94, *P* = 0.07). There were no differences between the three treatments with respect to turnover in either the riparian (*F* = 0.73, df = 2,45, *P* = 0.49) or upland (*F* = 1.50, df = 2,45, *P* = 0.23) habitats (Fig. 4).

Species diversity

Avian diversity was greater in the upland habitats across all treatments (Table 3) Evenness values remained consistently high in both habitats after harvest (Table 3).

Abundance

After harvest, the mean detection rate for all species combined was 12 birds/visit with no differences between the riparian and upland habitats (Fig. 6; *F = 2.84,* df = 1,94, *P* = 0.095) and no differences between the three treatments (Riparian: *F* = 1.38, df = 2,45, *P* = 0.26; Upland: *F* = 1.56 df = 2,45, *P = 0.22).*

After harvest, there were no habitat differences in the mean abundance of Neotropical migrants *(F* = 0.22, df = 1,94, *P* = 0.64; Fig. 7) and short-distance migrants *(F* = 1.61, df = 1,94, *P* = 0.21; Fig. 7). Similar to pre-harvest conditions, the abundance of permanent residents was greater in the upland than riparian habitat (*F* = 10.63, df = 1,94, *P* = 0.001; Fig. 6).

Across all sites, the three species (Golden-crowned Kinglet, Hammond's Flycatcher, and Winter Wren) that had been more abundant in the riparian than upland habitat before harvest remained more abundant in the riparian habitat after harvest (Table 4). Five species, Chestnut--backed Chickadee, Chipping Sparrow, Dark-eyed Junco, Red-breasted Nuthatch, Yellow-rumped 'Warbler, that had been more abundant in the upland habitat before harvest remained more abundant in the upland after harvest (Table 4). Similar to pre-harvest conditions, the Winter Wren and Red-breasted Nuthatch were consistently more abundant in their respective habitats across all sites. Hammond's Flycatcher, which had been consistently more abundant in the riparian habitat across all sites before harvest, was significantly more abundant in the riparian habitat of only the Control sites. The Chestnut-backed Chickadee which had been more abundant in the upland habitat of the Modified and State sites prior to harvest was only found in greater abundance on the habitats of Control sites in the years following harvest (Table 4). The Nashville Warbler, which had been more abundant across all habitats before harvest, was no longer more abundant in the uplands after harvest. The Gray Jay, which had also been more abundant in the upland across all sites before harvest, was more common in the upland of only the Control sites following harvest. Four additional species, Brown Creeper, Mountain Chickadee, Red-naped Sapsucker, and Solitary Vireo, which had been equally distributed between the upland and riparian habitats before harvest, were more abundant in the upland habitat after harvest. However this was only true on the Control and Modified sites for the Brown Creeper and on the State sites for the mountain chickadee (Table 4). The abundance of the solitary vireo was greater in the upland habitats across all treatments, but only significantly when all sites were combined (Table 4). Swainson's thrush became more abundant in the riparian habitat on the State sites but remained equally distributed between habitats on the other sites.

As indicated above, the abundance of the riparian obligate and sensitive species tended to be either site or year specific. The single observation of the American Dipper, a riparian obligate species, was on a low elevation stream in a State site during a single post harvest year. Yellow Warblers, also identified as riparian obligates, were observed on both riparian and upland transects of sites representing all three harvest treatments but only in 1995 when total bird detections were highest.

The abundance of MacGillivray's Warbler remained similar between the riparian and upland habitats in the post-harvest years, that of Northern Waterthrush remained greater in the riparian than upland habitat, and that of western tanager remained greater in the upland than riparian habitat (Table 4). Dusky Flycatcher became more abundant in the riparian habitats during the post-harvest years (Table 4). This species was observed primarily during 1996 (33 of the 74 observations) and this flycatcher was observed most consistently on the riparian transect of one Modified site (n = 25). ‘The abundance of Warbling Vireo became similar between the riparian and upland habitats during the post-harvest years.

Treatment effects - riparian habitat

Species richness and turnover

Although more species of birds were observed along the riparian on all sites during the post harvest years ($F = 7.92$, df = 1, $P = 0.013$), these differences were not due to the effects of harvest treatment ($F = 0.40$, df = 2,25, $P = 0.677$). There were no differences in species richness between the three treatments either before ($F = 0.05$, df - 2,15, $P = 0.951$) or after ($F = 1.01$, df = 2,15, $P = 0.387$) timber harvest. The proportion of species turnover between years prior and post harvest was similar in the riparian habitats in each of the three treatments (Fig. 4; Control: $F = 2.64$, df = 1,26, $P = 0.116$; Modified: $F = 0.24$, df = 1,18, $P = 0.627$; State: $F = 3.23$, df = 1,22, $P = 0.862$).

Species diversity

Shannon diversity values for the riparian habitat were greater across all sites during the years following timber harvest (Table 3; $F = 6.83$, df = 1, $P = 0.02$) but there were no differences due to the effects of the different harvest treatments ($F = 0.05$, df = 2,15, $P =$ 0.95). There were no changes in the evenness values before and after harvest either due to time ($F = 0.53$, df = 1, $P = 0.48$) or the effects of different harvest treatments ($F = 0.05$, df = 2,15, $P = 0.95$).

Abundance

Although the abundance of all species combined was higher in the years following timber harvest (Fig. 7; $F = 6.56$, df = 1, $P = 0.0217$), the increase was observed on all sites and was not due to the effects of harvest treatment ($F = 0.55$, df = 2,15, $P = 0.5855$).

Of the 22 common species, only four species exhibited a change in abundance in the riparian habitat due to the effects of the harvest treatment. The abundance of Golden-crowned Kinglets decreased in the riparian habitats across all sites during the years following timber harvest (Table 4; $F = 14.09$, df = 1, $P = 0.002$) but the decrease was significantly greater on the State sites ($F = 4.62$, df = 2,15, $P = 0.027$) as compared to the Modified and Control sites. Over all sites, the abundance of Hammond's Flycatcher was similar between the pre harvest and post harvest years, but this species was less abundant in the riparian habitat of the post harvest State sites than prior to timber harvest ($F = 3.$, df = 2,15, $P = 0.05$). In contrast, Dark-eyed Juncos were more abundant during the post harvest years across all treatments ($F =$ 11.65, df = 1, $P = 0.004$). Their abundance was significantly greater in the riparian habitats of the State sites ($F = 7.35$, df = 2,15, $P = 0.006$) and this difference was due to the effects of harvest treatment ($F = 4.55$, df = 2,15, $P = 0.03$). Chipping Sparrows were also more abundant during the post harvest years and this increase was due to the effect of harvest treatment ($F = 4.96$, df = 2,15, $P = 0.02$), however this increase was only on the State sites (F = 6.25, df = 2,15, $P = 0.01$).

Treatment effects - upland habitat

Species richness and turnover

More species of birds were observed during the post harvest years in the upland habitats across all sites (F = 20.78, df = 1, P = 0.0004). However, the increase was more pronounced on the Modified and State sites as compared to the Control sites (F = 3.71, df = 2,15, P = 0.0.49). Species turnover rates were similar between the pre harvest and post harvest years on the Control sites (Fig. 4; F = 0.43, df = 1,26, P = 0.52), increased on the Modified sites (Fig. 4; df = 1,18; P = 0.025), and decreased on the State sites (Fig. 4; F = ***6.20,*** df = 1,22, P = ***0.021).***

Species diversity

The Shannon diversity values were greater across all sites during the years following timber harvest (Table 3; F = 17.74, df = 1, P = 0.0008) and the increase was more pronounced on the State sites than on the Control or Modified sites (F = 3.84, df = 2,15, P = 0.045). In contrast, evenness values decreased on the State sites in response to harvest treatment, but did not change on either the Control or Modified sites (F = 3.63, df = 2,15, P = ***0.05).***

Abundance

The abundance of all species combined was greater during the years following timber harvest (F = 23.12, df = 1, P = 0.0002), but, again, this increase was observed on all treatments (Fig. '7) and was not due to treatment effects (F = 0.54, df = 2,15 0.5914).

The abundance of four of the 22 common species, Nashville Warbler (F = 4.7, df = 1, P = 0.47), Solitary Vireo (F = 20.1, df = 1, P = 0.0004) Yellow-rumped Warbler (F = 14.49, df = 1, P = 0.002), Red-breasted Nuthatch (F = 9.60, df = 1, P = 0.007), increased in the uplands during the post harvest years but this increase was observed on all treatments and was not due to treatment effects. Dark-eyed Juncos were more abundant during the years following harvest ($F = 68.79$, df = 1, $P = 0.0001$) and this increase was greater on the State sites than either the Control or Modified sites ($F = 9.43$, df = 2,15, $P = 0.002$). In contrast, the abundance of Winter Wrens also increased in uplands during the post harvest years ($F = 15.37$, df = 1, $P = 0.001$), but this increase was less on the State sites than either the Control or Modified sites ($F = 3.7$, df = 2,15, $P = 0.05$). Differences in the abundance of both Townsend’s Warbler and Hammond’s Flycatcher between pre and post harvest years were due to the effects of harvest treatment. Townsend’s Warbler increased in abundance on the Control sites, decreased on the State sites, and remained similar on the Modified sites (Table 5; $F = 3.98$, df = 2,15, $P = 0.04$). The abundance of Hammond’s Flycatcher increased in response to treatment on the Modified sites but remained similar on the other two treatments ($F = 3.63$, df = 2,15, $P = 0.05$). Of the 22 common species, the abundance of only two species was less during the years following harvest. The Golden-crowned Kinglet was less abundant in the uplands during the years following timber harvest ($F = 14.17$, df = 1, $P = 0.0019$) and this decrease was more pronounced on the State sites than on the Control sites ($F = 6.8$, df = 2,15, $P = 0.0079$). Gray Jays were also less abundant in the uplands during the post harvest years ($F = 12.13$, df= 1, $P = 0.003$) and this decrease was greater on the two harvest treatments than on the Control sites ($F = 3.62$, df= 2,15, $P = 0.05$).

Of the five sensitive species analyzed, only one, the Dusky Flycatcher exhibited a response to harvest treatment ($F = 4.99$, df = 2,15, $P = 0.022$); abundance was greater on the Stale sites during the post harvest years. However, this species was never observed in the upland habitats of any Control or Modified sites. The abundance of the remaining four sensitive species did not exhibit any response to harvest cut (MacGillivray's Warbler: F = 1.53, $P = 0.247$; Northern Waterthrush: $F = 1.04$, $P = 0.379$; Warbling Vireo: F = 1 .00, P = 0.393; df = 2,15 in all cases). The abundance of none of the 'rive sensitive species analyzed changed in response to time.

Species-habitat relationships

The regression of habitat variables on frequency of bird detections revealed that all of the 27 species analyzed were significantly associated with at least one habitat, but that both the individual parameter and model R^2 values were low (Table 6).

With the exception of the varied thrush, the model R^2 values for the other seven common species (American Robin, Black-capped Chickadee, Evening Grosbeak, Hermit Thrush, Pine Siskin, Swainson's Thrush, and Townsend's Warbler) whose abundance was similar between riparian and upland habitats were especially low (<10%). The varied thrush was positively associated with mature forest features (Le., older decay classes of downed wood, stumps, and snags and 25-50 cm DBH conifers; Table 6). The Mountain Chickadee, which also had a very low R^2 value and which was more abundant in the upland habitats of the State sites after harvest, was negatively associated with overstory cover (Table 6). The other common species with a low R^2 value, the Chestnut-backed Chickadee, was positively associated with large (>50 cm DBH) conifers and open overstory. This species was more abundant in the upland forests of the pre-harvest State and Modified sites and of the post-control Control sites.

Four of the common upland-associated species with larger R^2 values (Dark-eyed Junco, Chipping Sparrow, Red-breasted Nuthatch, and Yellow-rumped Warbler) were negatively associated overstory cover, and positively associated with shrubs. These species were also positively associated with the State harvest variable but showed no response to the Modified harvest variable. In contrast, two of the common riparian-associated species, Hammond's Flycatcher and Winter Wren, were negatively associated with the State harvest variable and positively so with the Modified harvest variable. Hammond's Flycatcher was associated with early succession riparian features such as shrubs and deciduous trees whereas the winter wren was associated with more mature forest features such as dispersed shrubs and taller trees. The remaining riparian-associated species, Golden-crowned Kinglet, was also associated with more mature forest features, but showed no response to either harvest variable (Table 6).

Three of the species, Dusky Flycatcher, MacGillivray's Warbler, Northern Waterthrush, that we classified as moderately vulnerable (O'Connell et al. 1993) were positively associated with hahitat features, such as smaller deciduous trees, characteristic of riparian zones. All three of these species were positively associated with the Modified harvest variable and negatively with the State harvest variable. We analyzed the habitat relationships of two other species, Warbling Vireo and Western Tanager, which we had classified in our literature review as moderately vulnerable riparian species. Although R^2 values were low, both of these species were associated with upland habitat features (e.g., shrubs, open overstory) and with recent cuts (reduced snags, recent downed wood and cut stumps). Indeed, both of these species were negatively associated with the riparian zone variable and positively associated with the State harvest variable (Table 6).

Nest predation

The results of the nest predation studies are discussed in detail elsewhere (Entz 1995, Hackworthy 1996). Before harvest, the overall rate of nest predation was 49% and did not differ significantly between the riparian and upland habitats (F = 1.3 1, P = 0.25). Rates of nest predation were significantly greater for ground nests than for arboreal nests (F = 7.61, P = 0.006). The overall rate of nest predation in the post-harvest study was 39%. Again, rates of nest predation did not differ significantly between the riparian and upland habitats (F = 0.05, P = 0.819) but did so between the arboreal and ground nests (F = 4.51, P = 0.037). Post-harvest comparisons of rates of predation between the three harvest treatments revealed that the rates of nest predation were highest on the State sites ($\bar{x}$ = 52%) and were similar between the Control ($\bar{x}$ = 29%) and Modified ($\bar{x}$ = 34%) sites (F = 33.99, P = 0.0001).

DISCUSSION

Pre-treatment comparison between riparian and upland habitats

In the more mesic forests of the Pacific Northwest, evidence suggests that the general trend of greater avian species richness, diversity, and abundance in riparian habitats of other regions might not hold. We observed no differences in species richness, turnover rates, or overall abundance between riparian and upland habitats in the mixed coniferous forests of northeastern Washington and greater species diversity and equitability in the upland habitats. Additionally, of the 22 common species, seven were more abundant in the upland compared to three in the riparian. Finally, we observed no differences in rates of nest predation on artificial nests placed in riparian and upland habitats. In spruce forests of southeastern British Columbia, Kinley and Newhouse (1997) reported similar avian species richness between riparian and upland habitats but greater diversity, equitability, and density in riparian than upland forests, Studies by McGarigal and McComb (1992) found all measures of avian diversity and abundance to be greater in upland than riparian habitats in Oregon coastal forests, Pearson and Manuwal (this report) found no differences in species richness or diversity in the forests on the west slopes of the Cascades of Washington. McGarigal and McComb (1992) suggested that the these similarities between riparian and upland habitats might be explained by 1) the high precipitation and drainage densities, 2) the Maritime environment of the coastal Pacific Northwest ameliorates microclimatic differences between riparian and upland habitats, and 3) the presence of structural components such as large trees and snags in the uplands to support avian richness. Although these explanations are interrelated, the similarities in avian populations between riparian and upland habitats of Inland Northwest forests that we and Kinley and Newhouse (1997) observed, suggest that microclimatic conditions found in Maritime environments are not the sole explanation. Structural similarities between riparian and upland habitats also might not provide a full explanation. McGarigal and McComb (1992) reported greater overstory cover, snag density, low shrub density, and conifer basal area in uplands as compared to riparian habitats. In contrast, in southeastern British Columbia Kinley and Newhouse (1997) found no differences between snag densities, low shrub densities, or CWD densities between riparian and uplands, and greater overstory canopy cover in riparian forests. We observed greater overstory canopy, more deciduous trees and large coniferous trees, and more large older snags in the riparian than upland forests on our sites, suggesting greater structural complexity in the riparian habitat. Nonetheless, avian diversity was greater in the uplands, The density of drainage systems and the proximity to water even in upland forests of the Pacific Northwest might explain the trends in avian richness, diversity, and abundance observed in this region,

Although species richness, diversity, and abundance might be similar between riparian and upland habitats, Kinley and Newhouse (1997) suggested that because riparian habitats represent a smaller area than upland habitats, they have a disproportionate importance in maintaining avian populations in managed forest landscapes. Although the abundance of permanent residents was lower in riparian habitat, the higher number of permanent resident species that we observed in riparian habitats suggests that these habitats might be important for the year-round survival of these species. However, seasonal habitat shifts are certainly possible (e.g., Hagar et al. 1996), and winter studies would be necessary to determine the relative importance of riparian and upland habitats for year-round survival of resident species.

Of the 22 common species, we found three, Golden-crowned Kinglet, Hammond's Flycatcher, and Winter Wren,, to be more abundant in the riparian habitat. Kinley and Newhouse (1997) reported nearly identical results: of 27 species analyzed, the same three were the only species significantly more abundant in riparian habitats of spruce forests of southeastern British Columbia. Although the winter wren was also more abundant in riparian habitats west of the Cascades, it was not uncommon in upland forests (McGarigal and McComb 1992, Manuwal and Pearson 1997, Pearson and Manuwal this report) and McGarigal and McComb (1992) did not consider it a riparian associate. In contrast, east of the Cascades, this species is either uncommon as we observed in this study and O'Connell et al. (1997) or absent (Kinley and Newhouse 1997) from the upland forests. The high R^2 value associated with the variable for riparian zone in our regression model also suggest that the winter wren is a riparian associate in these forests. In contrast, the R^2 values associated with the variable for riparian zone in our regression model for the golden-crowned kinglet and Hammond's Flycatcher were not large. Although these species were more abundant in the riparian forests of southeastern British Columbia (Kinley and Newhouse 1997), they were present in the uplands, In coastal Oregon both were significantly more abundant in the upland than riparian forests (McGarigal and McComb 1992), indeed, Hammond's Flycatcher was never observed along streams. The differences for these species' dependence on riparian habitats between the west and east sides of the Cascades is most likely due to more mesic conditions of the west side.

There was less similarity between the upland birds of our study in northeastern Washington and those of Kinley and Newhouse (1997) in southeastern British Columbia. Whereas seven of the 22 common species were more abundant on the upland habitats of our study sites, only one species out of 27 analyzed, the Dark-eyed Junco, was also more abundant in upland than riparian forests in British Columbia. Our results are similar to those of McGarigal and McComb (1992) who observed a higher proportion of birds that were more abundant in the upland than ~riparian forests in western Oregon. In contrast, Pearson and Manuwal (Chapter 5) found no species to be more significantly more abundant in upland forests in western Washington, although four were more common in the uplands. The response of the seven upland bird species to the habitat variables suggest patterns of habitat association similar to that observed for birds both east (O'Connell et al. 1997) and west (Manuwal and Pearson 1997) of the Cascade crest. For example, the Brown Creeper and Chestnut.-backed Chickadee were associated with features of mature forest such as increased canopy cover and tall trees and the Chipping Sparrow and Dark-eyed Junco were associated with early succession forest features such as open overstory cover. Floyd (1993) compared the nest site habitats of the Black-capped Chickadee, Chestnut-backed Chickadee, and Mountain Chickadee on three of our study sites and found that the nests of the Chestnut-backed Chickadee were in coniferous trees in upland areas.

In our review of wildlife use of riparian habitats (O'Connell et al. 1993) we assessed the sensitivity to disturbance of 132 species identified from the literature as potential inhabitants of riparian habitats. Of the 43 of these species we observed on our sites, 13 had been assigned a sensitivity ranking of >10, suggesting moderate to high vulnerability. Only one, the Northern Waterthrush, of the seven species for which we had sufficient data was actually more abundant in the riparian habitat, all others were either more abundant in the upland (four species) or equally abundant in the habitats (two species). Our literature review was compiled primarily from work conducted outside of northeastern Washington, Based on our findings and comparisons with more recent literature, we would have to reclassify the habitat associations of some species (e.g., Western Tanager, Warbling Vireo). However, the vulnerability ranking of these species points to the need for careful management of upland forests.

Post-treatment effects

We evaluated the effects of harvest treatment by four approaches: the post-harvest associations with riparian versus upland habitats, results of the repeated measure ANOVA, the species' association with the harvest treatment variable in our logistic regression analysis, and comparison of nest predation rates between harvest treatments, The general patterns between riparian and upland 'habitats remained the same after timber harvest: species richness, turnover rates, overall abundance and rates of nest predation did not differ between the two habitats and species diversity and equitability were greater in the upland habitat, However, there were differences in some of these metrics and in individual species' response between harvest treatments.

Species richness, turnover rates, diversity, and equitability remained the same during the pre- and post-harvest time periods in the riparian habitats of all sites and in the upland habitats of the Control sites. In the upland habitats of the Modified sites, species richness and turnover rates increased whereas diversity and equitability remained constant. Changes were most pronounced in the upland habitats of the State sites: species richness and diversity increased and species turnover rates and equitability decreased.

The increase in species richness in the uplands following timber harvest is not uncommon on sites that have been selectively harvested or thinned (e.g., Mannan and Meslow 1984, Hagar, et al. 1996, Chambers and McComb 1997). The decrease in species equitability rates on the State sites is most likely related and due to the increase in the dominance of the Dark-eyed Junco on these sites relative to the other treatments.

The overall similarity in the avian population parameters across the three treatments in the riparian habitats most likely reflects the upland harvest prescription. Most studies that have examined avian populations in riparian buffers have been in more xeric areas or have had clearcut harvests in the adjacent uplands (e.g., Stauffer and Best 1980, Manuwal 1986, Darveau et al. 1995, Kinley and Newhouse 1997, Chapter 5). Although species richness and abundance of birds typically declines immediately following clearcut harvests (Manuwal and Pearson 1997), they often increase following thinning (e.g., Hagar, et al. 1996). The selective harvest in the upland of our study sites might have reduced the impact on the riparian buffers in two ways. First, the negative effects of induced edge might have been lessened because the RMZ boundary was less abrupt. Second, the upland forests were able to retain species, reducing the potential effects of birds shifting use from the upland to the riparian zone (e.g., Darveau et al. 1995). Not surprisingly, the effects of harvest treatment must be examined at a finer scale than the broad patterns of species richness and diversity.

Habitat conditions created by the State treatment resulted in a decrease of the abundance of certain riparian species and an increase of certain upland species in the riparian zone. Two of the species, Golden-crowned Kinglet and Hammond's Flycatcher, more abundant in the riparian zone before harvest exhibited a significant decrease in abundance on the State sites relative to the other treatments. Kinley and Newhouse (1997) observed a decrease in the abundance of these two species on narrower (14 m) as compared to wider buffers (70 m) on streams in southeastern British Columbia. In the boreal forests of Quebec, the density golden-crowned kinglets was lower in 20-m buffer strips than 60-m buffer strips or control plots following clearcutting of adjacent uplands (Darveau et al. 1995). The other species closely associated with the riparian habitat before harvest, the Winter Wren, was more abundant across all sites during the post harvest years, but the increase was significantly less on the State sites. Kinley and Newhouse (1997) observed fewer Winter Wrens in their narrower buffers as compared to wider buffers. Conversely, dark-eyed juncos and chipping sparrows, two upland species became more abundant in the riparian zone of the State sites relative to the other treatments. Swainson's Thrush, a common species that exhibited no habitat association before harvest, was more abundant in the riparian habitat of the State sites after harvest but remained equally distributed between the two habitats on the other sites. Neither the Dark--eyed Junco nor Chipping Sparrow exhibited a response to buffer width in forests of southeastern British Columbia (Kinley and Newhouse 1997). Swainson's Thrush did not exhibit any habitat association in unharvested spruce forests of British Columbia, but was more abundant in the wider buffers (Kinley and Newhouse 1997)). In boreal forests of Quebec, Swainson's Thrush had higher densities on wider buffer strips (Darveau et al. 1995). These similarities between our findings and those of Kinley and Newhouse(1997), and to a lesser extent, Darveau et al. (1995) are striking in that the adjacent uplands in their studies had been clearcut harvested. Darveau - increased on all (similar) than decreased, especially on narrow strips.

Our examination of the relationship between species' abundance and habitat variables revealed several interesting trends. First, upland species such as the Chipping Sparrow, Dark-eyed Junco, and Western 'Tanager, which were associated with open overstory cover exhibited a positive response, to the State RMZ Harvest variable. Second, upland species, such as the Brown Creeper, that were more associated with mature forest variables exhibited a negative response to the State RMZ Harvest variable. Third, riparian species, such as Hammond's Flycatcher, Northern Waterthrush, and Winter Wren, were positively associated with the Modified RMZ Harvest variable and negatively so with the State RMZ Harvest variable. Finally, several species that had been identified as sensitive species (e.g., Dusky Flycatcher, Macgillivray's Warbler, Northern Waterthrush) were positively associated with the Modified harvest variable and negatively so with the State harvest variable.

Our study focused primarily on the abundance of the birds, and, as Vickery et al. (1992) have suggested, abundance might or might not be an accurate measure of nesting success. Riparian buffers, as edge habitat, might experience decreased nesting success due to the effects of nest parasitism (e.g., Gates and Giffin 1991) and nest predation (Yahner and Scott 1988). Indeed, Gates and Giffin (1991) suggested that riparian buffers might represent ecological traps. Brown-headed cowbirds were never common on our sites; however, they were observed only during the years after timber harvest. Our studies on nest predation indicated that the post-harvest rates of nest predation were greater on the State sites as compared to the Modified RMZ or Control sites. This suggests that the Modified RMZ, with its protection of specific habitat features and resulting irregular shape, might support greater nesting success than the State RMZ. In contrast, Darveau et al. (1995) reported a trend towards lower rates of artificial nest predation on 20-40-m buffers as compared to 60-m wide and control areas. Well-designed studies of nest success in riparian buffers within managed forests would be of asset to land managers.

MANAGEMENT IMPLICATIONS

In conclusion, our results indicate 1) avian species richness, abundance, and diversity were either equal or greater in upland habitats as compared to riparian habitats, 2) given the association of certain species with riparian habitat and the relatively restricted area of riparian as compared to upland habitat, protection ofriparian habitats remains important, 3) although the east-side State Riparian Management Zones and our Modified Buffers retained comparable overall avian diversity and abundance following a selective harvest in the adjacent upland, the abundance of individual riparian species was better retained and more positively associated with the Modified buffers. The intent of the Modified buffer was to incorporate a more site-specific approach to riparian management by providing for protection of habitat features of importance to wildlife such as seeps and snags. The importance of upslope habitats in maintaining avian diversity in this region argues for a similar site-specific approach to upland habitat management.

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Table 1. The number of sites at which each bird species was observed on riparian (Rip.) and upland (Upl.) transects and the total number of observations (No.) for each species in each habitat at the 18 RMZ sites for 1992-1996.

Species	Rip. 1992		Rip. 1993		Rip. 1994		Rip. 1995		Rip. 1996		Upl. 1992		Upl. 1993		Upl. 1994		Upl. 1995		Uol. 1996	
	Sites	No.	Sites	No.	Sites	No.	Sites	No.	Sites	No.	Sites	No.	Sites	No.	Sites	No.	Sites	No.	Sites	No.
American Dipper									1	.										
American Kestrel																			1	1
American Redstart	1	1	1	1							1	1								
American Robin	4	11	4	6	5	8	7	17	5	11	4	6	7	9	7	13	9	17	8	13
Belted Kingfisher																	1	1		
Black-backed Woodpecker							1	1			1	1	2	2	1	2	3	5	4	8
Black-capped Chickadee	7	26	7	12	10	19	12	24	5	6	7	23	4	8	14	45	11	27	9	19
Black-headed Grosbeak													2	3	1	1	2	4		
Blue Grouse	1	1	1	1					1	1	2	3	1	1						
Borcal Chickadcc							1	5					1	1			2	3		
Brown Creeper	8	10	8	10	15	38	9	18	6	13	11	27	12	32	18	70	14	43	10	30
Brown-headed Cowbird							3	3	1	1					1	1	1	1	2	2
Calliope Hummingbird													1	1						
Cassin's Finch							1	1							1	1				
Cedar Waxwing	1	4							1	6							1	1		
Chestnut-backed Chickadee	11	57	14	43	14	71	17	65	13	48	12	56	16	80	12	95	15	86	13	119
Chipping Sparrow			1	1	3	4	3	6			3	4	4	7	8	25	8	28	4	11
Clark's Nutcracker																	1	1		
Common Raven	2	3			1	1	5	9	1	1			4	5	1	1	2	4	1	1
Dark-wed Junco	10	31	6	6	11	65	16	95	8	50	11	54	12	42	17	197	17	195	15	169
Downy Woodpecker			1	1					1	1					3	3	1	1		
Dusky Flvcatcher					3	20	2	3	10	31	2	4	1	1	2	3	3	7	2	4
Evening Grosbeak	1	1	6	10	4	10	6	9	2	3			7	13	4	20	7	15	2	3
Fox Sparrow	3	4			3	3			1	1	1	1	1	1	1	3	1	1		
Golden-crowned Kinglet	17	337	18	321	17	289	18	341	18	205	17	187	18	287	18	248	18	257	17	121
Gray Jay	4	6					5	10	2	6	11	39	7	15	10	31	4	11	2	5
Great Blue Heron			1	1	1	1														
Hairy Woodpecker	2	3	1	1	1	1	5	5			3	5	4	4	1	1	7	11	4	6
Hammond's Flycatcher	8	27	11	48	12	31	17	82	7	15	4	4	10	25	3	6	11	38	2	5
Hermit Thrush	4	6	10	12	14	28	10	13	3	6	1	1	10	19	11	21	10	28	3	3
House Wren																	1	1		
Least Flvcatcher					1	1														
MacGillivray's Warbler	6	13	8	20	6	12	9	18	5	11	5	6	10	18	3	7	9	20	6	11
Mallard							1	1												
Mountain Chickadee	4	10	8	11	2	5	9	12	2	3	2	2	9	17	8	22	8	14	7	11
Mourning Dove																	1	1		
Nashville Warbler	3	7	6	8	7	8	6	7	4	5	11	16	11	29	12	35	8	12	3	5
Northern Flicker			1	1			1	1			1	1			1	1	2	2	2	3
Northern Goshawk			1	1									1	1	2	2			1	1

Table 1. Continued.

Species	Rip. 1992		Rip. 1993		Rip. 1994		Rip. 1995		Rip. 1996		Upl. 1992		Upl. 1993		Upl. 1994		Upl. 1995		Upl. 1996	
	Sites	No.	Sites	No.	Sites	No.	Sites	No.	Sites	No.	Sites	No.	Sites	No.	Sites	No.	Sites	No.	Sites	No.
Northern Pygmy Owl													1	1						
Northern Waterthrush	3	4	6	9	6	11	2	6	4	9			2	2	1	2	1	1		
Olive-sided Flycatcher																	1	1		
Orange-crowned Warbler			3	4	2	3					1	1	3	8	4	8	2	5	2	4
Pileated Woodpecker	1	1	2	2	1	1	1	1			3	3	2	2	3	3	1	1		
Pine Siskin	9	26	2	3	11	43	5	15	2	5	13	34	9	23	9	30	10	34	10	26
Pygmy Nuthatch															1	5				
Red Crossbill											1	2	1	4	5	58	1	1	3	6
Red-breasted Nuthatch	7	11	11	24	12	22	12	34	8	16	14	34	17	62	18	137	18	99	17	59
Red-wed Vireo	i	i									i									
Red-naped Sapsucker	1	6	7	9	3	4	3	4	4	7	2	2	4	7	7	16	10	17	10	21
Red-tailed Hawk			1	1													1	1		
Ruby-crowned Kinglet					1	5					2	2			2	4			2	3
Ruffed Grouse	3	3	7	8	7	10	4	6	1	1	4	7	4	8	3	4	4	10	2	5
Rufous Hummingbird					2	2	1	1			1	1					1	1	1	1
Solitary Vireo	2	3	2	5	7	14	13	19	1	1	4	11	3	6	8	31	13	42	7	14
Song Sparrow	2	4			1	1	1	2	1	2							2	2		
Spotted Towhee	6	6	1	1			1	1			1	1							1	1
Spruce Grouse			2	3	2	2							1	1			1	1		
Steller's Jay			1	1											1	1	1	1	1	1
Swainson's Thrush	17	81	15	71	15	66	18	71	18	70	17	56	16	51	16	73	17	81	16	51
Three-toed Woodpecker					2	2	1	1			1	1			3	4	1	3	1	1
Townsend's Solitaire					2	2	2	3	1	1					6	13	8	12	2	2
Townsend's Warbler	15	85	18	182	18	130	18	275	17	96	13	74	18	211	18	161	18	249	15	90
Varied Thrush	6	17	14	35	13	47	11	34	11	24	6	10	12	37	9	21	11	41	7	19
Warbling Vireo							1	2	2	4	1	4	6	11	2	9	2	16	3	10
Western Flycatcher			1	1	1	1	1	1							2	2				
Western Screech Owl	1	1																		
Western Tanager	2	2	3	3	1	1	3	8	1	2	2	2	7	16	5	11	8	13	3	5
Western wood Pewee																	1	1		
White-breasted Nuthatch											1	1								
Wilson's Warbler			1	1	2	4			1	2	1	1	3	3	4	7	1	4	2	2
Winter Wren	16	127	18	194	18	238	18	274	18	175	6	13	10	24	11	37	17	98	11	40
Yellow Warbler							5	5									8	11		
Yellow-rumped Warbler	5	6	6	10	9	20	15	55	4	5	7	17	11	22	15	78	17	108	8	39

Table 2. Comparison of mean (± SE) number of species per site in riparian and upland habitats on Control, Modified, and State sites before and after harvest on the Modified and State sites.

	&-harvest					Post-harvest				
	Riparian	Upland	*F*	df	p	Riparian	Upland	*F*	df	p
Control	12.6±0.9	13.2±0.9	0.24	1,30	0.63	12.7zk0.9	14.5±0.9	2.15	1,34	0.15
Modified	13.3±1.0	15.2±0.8	2.43	1,22	0.13	14.0±1.3	16.8±1.1	2.51	1,22	0.12
State	13.5±1.2	16.3±1.5	2.09	1,20	0.16	15.11±1.2	19.6±1.3	6.31	1,43	0.017

Table 3. Comparison of mean (± 1 SE) Shannon-Wiener Diversity Index values and mean (± 1 SE) evenness values for riparian and upland habitats on Control, Modified, and State sites.

	Shannon Index					Evenness				
	Riparian	Upland	*F*	df	*P*	Riparian	Upland	*F*	df	*P*
Overall										
Control	2.00±0.04	2.16±0.05	6.26	1,68	0.015	0.81±0.009	0.86±0.01	4.66	1,68	0.034
Modified	2.95 0.64	2.34±0.06	12.38	1,48	0.0001	0.81±0.01	0.87±0.01	17.97	1,48	0.0001
state	2.04 0.89	2.43±0.05	14.35	1,58	0.0004	0.8110.01	0.86±0.01	10.05	1,58	0.002
Preharvest										
Control	1.99±0.06	2.09±0.08	0.87	1,32	0.36	0.81±0.01	0.8310.01	0.57	1,32	0.46
Modified	1.98±0.09	2.26±0.08	4.99	1,24	0.035	0.81±0.02	0.86±0.01	9.96	1,24	0.014
State	1.84±0.19	2.36±0.08	6.53	1,22	0.018	0.8010.03	0.87±0.01	6.18	1,22	0.021
Post-harvest										
Control	2.01±0.05	2.2310.06	7.93	1,34	0.008	0.81±0.01	0.85±0.01	5.84	1,34	0.021
Modified	2.12±0.09	2.44±0.06	8.68	1,22	0.008	0.82±0.01	0.87±0.01	13.46	1,22	0.001
State	2.18±0.07	2.49±0.07	9.74	1,34	0.004	0.8210.01	0.85±0.01	3.89	1,34	0.057

Table 1. Abundance rankings of terrestrial amphibians based on raw captures in riparian and upland transects during 1992.1993. P-values reported compare riparian and upland captures for each species. P-values in bold denote significant differences. NA indicates species with <10 total captures, which were not compared due to small sample sizes.

Species	Rank	Total captures	Sites	Total riparian captures (%)	Total upland captures (%)	*P*
PLVE	1	240	18	110 (48)	130 (35)	0.856
ENES	2	180	17	28 (12)	152 (40)	**<0.001**
ASTR	3	59	10	22 (10)	37 (10)	0.053
AMGR	4	39	11	13 (6)	26 (7)	0.544
RAAU	5	3.5	9	20 (9)	15 (4)	0.622
DITE	6	21	10	15 (6)	6 (2)	0.22
TAGR	7	1'7	6	10 (4)	7 (2)	0.411
AMMA	8	a	3	7 (3)	1 (<1)	NA
PLDU	9	4	3	4 (2)	0	NA
RHKE	10	2	2	1 (<1)	1 (<1)	NA
RHCA	11	1	1	1 (<1)	*0*	NA
HYRE	12	1	1	0	1 (<1)	NA
Totals		607		231	376	

SPECIES SUMMARY

AMGR	Northwestern Salamander, *Ambystoma gracile*
AMMA	Long-toed Salamander, *Ambystoma macrodactylum*
ASTR	Tailed Frog, *Ascaphus truei*
DITE	Pacific Giant Salamander, *Dicamptodon tenebrosus*
ENES	Ensatina, *Ensatina eschscholtzii*
HYRE	Pacific Tree Frog, *Hyla regilla*
PLVE	Western Redback Salamander, *Plethodon vehiculum*
PLDU	Dunn's Salamander, *Plethodon dunni*
RAAU	Red-legged Frog, *Rana aurora*
RHCA	Cascade Torrent Salamander, *Rhyacotriton cascadae*
RHKE	Columbia Torrent Salamander, *Rhyacotriton kezeri*
TAGR	Roughskin Newt, *Taricha granulose*

Table 4. Continued

Pre-harvest Species	Overall (df = 1,82) Riparian	Upland	Control (df = 1,32) Riparian	Upland	Modified (df = 1,24) Riparian	Upland	State (df = 1.22) Riparian	Upland
Pine Siskin	0.16±0.05	0.24±0.05	0.10±0.04	0.21±0.07	0.19±0.09	0.18±0.06	0.22±0.13	0.36±0.14
	F = 1.24, p = 0.269		F = 1.7, p = 0.202		F = 0.01, p = 0.909		F= 0.56, p = 0.46	
Red-breasted Nuthatch	0.16±0.03	**0.56**±0.07	0.19±0.05	**0.66**±0.16	0.14±0.06	**0.45**±0.07	0.15±0.05	**0.56**±0.10
	F= 25.48, p = 0.0001		F = 7.84, p = 0.009		F = 11.44, p = 0.003		F= 13.87, p = 0.001	
Red-naped Sapsucker	0.07±0.03	0.05±0.02	0.02±0.01	0.01±0.01	0.17±0.08	0.09±0.06	0.03±0.02	0.06±0.02
	F = 0.34, p = 0.564		F= 0.35, p = 0.559		F= 0.58, p = 0.45		F= 0.85, p = 0.37	
Solitary Vireo	0.07±0.04	0.10±0.03	0.10±0.08	0.088±0.05	0.026±0.02	0.051±0.04	0.08±0.05	1.67±0.09
	F = 0.33, p = 0.569		F = 0.01, p = 0.916		F = 0.35, p = 0.558		F = 0.67, p = 0.42	
Swainson's Thrush	0.7310.78	0.58±0.06	0.6210.14	0.67±0.12	0.68±0.10	0.54±0.08	0.83±0.15	0.51±0.11
	F = 1.98, p = 0.163		F= 0.07, p = 0.796		F = 1.17, p = 0.290		F = 3.10, p = 0.070	
Townsend's Warbler	1.24±0.13	1.48±0.15	1.29±0.17	1.36±0.21	0.94±0.20	1.55±0.28	1.5±0.29	1.58±0.33
	F = 1.53, p = 0.220		F= 0.07, p = 0.80		F= 3.21, p = 0.0857		F = 0.04, p = 0.852	
Varied Thrush	0.30±0.05	0.22±0.05	0.28±0.07	0.2210.09	0.49±0.132	0.23±0.081	0.21±0.08	0.21±0.12
	F = 1.07, p = 0.303		F = 0.39, p = 0.538		F= 1.15, p = 0.295		F= 0.0, p = 1.00	
Winter Wren	1.6910.15	0.2210.05	**1.32**±0.18	0.137±0.06	**2.19**±0.309	0.44±0.124	**1.67**±0.28	0.11±0.04
	F = 84.03, p = 0.0001		F = 38.77, p=O.OOOl		F = 27.82, p = 0.0001		F= 30.21, p = 0.0001	
Yellow-rumped Warbler	0.08310.03	**0.27**±0.07	0.06±0.03	0.32±0.14	0.15±0.064	0.31±0.13	0.08±0.05	0.15±0.07
	F= 5.89, p = 0.017		F= 3.45, p = 0.072		F= 1.70, p = 0.205		F = 0.64, p = 0.433	

Post-harvest	Overall (df = 1,94) Riparian	Upland	Control (df = 1.34) Riparian	Upland	Modified (df = 1,22) Riparian	Upland	State (df = 1.34) Riparian	Upland
American Robin	0.11±0.03	0.14±0.03	0.03±0.02	0.12±0.05	0.17±0.089	0.14±0.05	0.15±0.06	0.1710.05
	F= 0.75, p = 0.389		F = 36.2, p = 0.066		F = 0.09, p = 0.764		F = 0.07, p = 0.800	
Black-capped Chickadee	0.14±0.03	0.23±0.04	0.14 0.03	0.20±0.06	0.11±0.07	0.18±0.08	0.16±0.58	0.28±0.63
	F = 3.37, p = 0.069		F= 0.97, p = 0.33		F= 0.46, p = 0.50		F = 1.97, p = 0.17	
Brown Creeper	0.57±0.09	**0.94**±0.14	0.63±0.12	**1.34**±0.27	0.35±0.11	**0.86**±0.23	0.66±0.20	0.602±0.19
	F = 4.95, p = 0.03		F= 5.97, p = 0.02		F = 4.04, p = 0.056		F= 0.04 p = 0.84	

Table 4. Continued

Post-harvest	Overall (df = 1,82)		Control (df = 1,32)		Modified (df = 1,24)		State (df = 1.22)	
Species	Riparian	Upland	Riparian	Upland	Riparian	Upland	Riparian	Upland
Chestnut-backed Chickadee	0.17±0.03	**0.38**±0.06	0.24±0.05	**0.55**±0.1 i	0.18±0.08	0.35±0.15	0.10±0.04	0.22±0.05
	F = 8.47, p = 0.004		F = 6.19, p = 0.018		F = 0.99, p = 0.331		F = 3.31, p = 0.078	
Chipping Sparrow	0.03±0.01	0.2210.0s	0	0.056±0.04	0	0.07±0.04	0.09±0.03	**0.49**±0.11
	F = 11.77, p = 0.0009		F = 2.13, p = 0.154		F = 3.31, p = 0.082		F= 12.35, p = 0.001	
Dark-eyed Junco	0.69±0.12	1.8110.22	0.38±0.17	0.85±0.20	0.24±0.66	**1.15**±0.19	1.31±0.21	**3.19**±0.34
	F= 19.66, p = 0.0001		F = 3.41, p = 0.073		F= 20.6, p = 0.0002		F= 22.14, p = 0.0001	
Evening Grosbeak	0.07±0.03	0.13±0.05	0.04±0.02	0.20±0.13	0.15±0.10	0.11±0.06	0.06±0.03	0.0610.03
	F = 0.77, p-O.381		F = 1.56, p = 0.22		F = 0.13, p = 0.718		F = 0.00, p = i ,000	
Golden-crowned Kinglet	**2.51**±0.21	1.77±0.18	2.92±0.25	2.6110.27	2.13±0.36	1.35±0.33	2.35-t0.42	1.21±0.20
	F = 7.36, p = 0.008		F= 0.68, p = 0.415		F= 2.55, p = 0.12		F = 5.9, p = 0.021	
Gray Jay	0.06±0.02	0.09±0.03	0.03±0.02	0.13±0.05	0.03±0.03	0.01±0.01	0.10±0.05	0.10±0.05
	F= 1.01, p = 0.317		F= 3.90, p = 0.05		F = 0.20, p = 0.659		F = 0.0, p = 1.0	
Hammond's Flycatcher	**0.39**±0.08	0.17±0.05	**0.19**±0.55	0.009±0.01	0.63±0.15	0.3210.88	0.44±0.15	0.23±0.10
	F= 6.33, p = 0.014		F = 9.82, p = 0.004		F= 3.01. p = 0.096		F = 1.35, p = 0.25	
Hermit Thrush	0.1210.03	0.15±0.03	0.14±0.05	0.176±0.05	0.04±0.02	0.028±0.02	0.16±0.05	0.20±0.06
	F = 0.45, p = 0.505		F= 0.27, p = 0.605		F= 0.23, p = 0.633		F= 0.35, p = 0.560	
Mountain Chickadee	0.06±0.02	**0.13**±0.03	0.08±0.36	0.07±0.39	0.06±0.03	0.07±0.03	0.04±0.02	**0.23**±0.06
	F = 4.86, p = 0.030		F= 0.03, p = 0.86		F= 0.10, p = 0.76		F= 11.14, p = 0.002	
Nashville Warbler	0.06±0.01	0.125±0.03	0.04±0.02	0.14±0.06	0.0710.02	0.11±0.05	0.074±0.28	0.12±0.05
	F = 3.76, p = 0.055		F = 2.62, p = 0.115		F = 0.61, p = 0.44		F= 0.65, p = 0.43	
Pine Siskin	0.18±0.05	0.30±0.05	0.26±0.11	0.33±0.09	0.167±0.08	0.11±0.04	0.10±0.054	**0.39**±0.089
	F = 2.82, p = 0.096		F= 0.26, p = 614		F = 0.36, p = 0.554		F= 7.55, p = 0.009	
Red-breasted Nuthatch	0.23±0.05	0.9310.10	0.21±0.08	**0.80**±0.17	0.21±0.06	0.82±0.20	0.26±0.09	**1.13**±0.15
	F = 41.51, p = 0.0001		F= 9.85, p = 0.004		F= 8.82, p = 0.007		F= 25.48, p = 0.0001	
Red-naped Sapsucker	0.045±0.02	**0.18**±0.03	0.03±0.03	0.83±0.02	0.069±0.03	**0.26**±0.09	0.05±0.02	**0.22**±0.05
	F = 14.16, p = 0.0003		F= 2.27, p = 0.141		F = 4.29, p = 0.050		F = 9.06, p = 0.005	
Solitary Vireo	0.1010.03	**0.28**±0.06	0.06±0.03	0.167±0.08	0.15±0.06	0.49±0.16	0.11±0.05	0.269±0.08
	F = 8.09, p = 0.005		F= 1.63, p = 0.211		F = 3.83, p = 0.063		F= 3.02, p = 0.091	

Table 4. Continued

Post-harvest	Overall (df = 1.82)		Control (df = 1.32)		Modified (df = 1,24)		State (df = 1.22)	
Species	Riparian	Upland	Riparian	Upland	Riparian	Upland	Riparian	Upland
Swainson's Thrush	0.65±0.07	0.63±0.07	0.49±0.07	0.68±0.09	0.47±0.09	0.71±0.16	**0.94**±0.15	0.51±0.10
	F = 0.08, p = 0.778		*F* = 2.69, p = 0.110		*F* = 1.72, p = 0.203		*F* = 5.24, p = 0.0284	
Townsend's Warbler	1.58i0.16	1.48±0.18	1.55±0.22	2.0710.28	1.44zkO.28	1.21±0.27	1.70±0.33	1.06±0.29
	F = 0.18, p = 0.67		*F* = 2.17, p = 0.15		*F* = 0.37, p = 0.551		*F* = 2.14, p = 0.153	
Varied Thrush	0.28±0.05	0.25±0.06	0.3110.08	0.44±0.12	0.14±0.06	0.125±0.07	0.35±0.10	0.15±0.06
	F = 0.17, p = 0.682		*F* = 0.83, p = 0.370		*F* = 0.02, p = 0.886		F = 3.06, p = 0.089	
Winter Wren	**2.11**±0.15	0.55±0.08	**1.90**±0.24	0.49±0.13	2.54io.24	0.85±0.20	**2.05**±0.27	0.42±0.11
	F = 82.53, p = 0.0001		*F* = 26.70, p = 0.0001		*F* = 30.24, *p* = 0.0001		*F* = 30.16, p = 0.0001	
Yellow-rumped Warbler	0.260±0.05	**0.68**±0.11	0.14±0.05	**0.62**±0.22	0.31±0.13	0.39±0.10	0.35±0.10	**0.94**±0.15
	F = 12.49, p = 0.0006		*F* = 4.5, p = 0.041		*F* = 0.26, p = 0.614		*F* = 10.53, p = 0.0026	

Table 5. Comparison of mean abundance (± 1 SE) of sensitive bird species in riparian and upland habitats across all 18 sites before and after timber harvest on the Modified and State sites.

Species	Riparian	Upland	*F*	*P*
Pre-harvest df = 1,82				
Dusky Flycatcher	0.05 ± 0.05	0.03 ± 0.02	0.13	0.72
MacGillivray's Warbler	0.17 ± 0.04	0.10 ± 0.03	2.42	0.12
Northern Waterthrush	0.08 ± 0.02	0.01 ± 0.01	7.00	0.009
Warbling Vireo	0	0.06 ± 0.03	4.81	0.031
Western Tanager	0.02 ± 0.01	0.10 ± 0.04	5.20	0.025
Post-harvest df = 1,94				
Dusky Flycatcher	0.14 ± 0.45	0.05 ± 0.02	3.19	0.07
MacGillivray's Warbler	0.11 ± 0.27	0.13 ± 0.03	0.32	0.57
Northern Waterthrush	0.07 ± 0.03	0.01 ± 0.01	5.19	0.025
Warbling Vireo	0.02 ± 0.01	0.12 ± 0.06	2.60	0.11
Western Tanager	0.04 ± 0.02	0.09 ± 0.03	3.30	0.07

Table 6. Significant variables, the percent variance explained by each variable (R^2), and the total percent variance (Total R^2) in stepwise regressions of habitat variables, time variable, harvest-type variables, and habitat zone variable on the number of detections for 27 bird species for 1992-1996. Sign indicates the direction of association (** $P <$ 0.05; * $P \leq$ 0.10). A positive association with shrub distance indicates a positive association with a close dispersion of shrubs. A negative associated with snag height indicates the absence of snags.

Species	Variable	Sign	R^2	Total R^2
American Robin				5.2
	Shrub height	+**	1.6	
	Moderately-decayed logs	-**	0.9	
	Recent logs	+**	0.8	
	Large-class conifers	-**	0.7	
	Overstory cover	+**	0.6	
	Small-class conifers	-*	0.6	
Black-capped Chickadee				4.0
	Medium-class deciduous trees	+**	0.9	
	Recent cut stumps	-**	0.9	
	Overstory cover	-**	0.6	
	Recent natural stumps	-**	0.5	
	Small-class deciduous trees	+**	0.5	
	Pole-class conifers	+*	0.5	
Brown Creeper				9.0
	Riparian zone	-**	4.5	
	Tree height	+**	2.0	
	State harvest	-**	0.7	
	Small-class conifers	-**	0.7	
	Small-class old snags	+**	0.6	
	Modified harvest	-*	0.5	
Chestnut-backed Chickadee				7.2
	Riparian zone	-**	2.0	
	Old cut stumps	+**	1.2	
	Overstory cover	+**	1.1	
	Large-class conifers	+**	1.1	
	Time	+**	1.0	
	Pole-class conifers	-*	0.6	
	Small-class old snags	+*	0.4	
Chipping Sparrow				26.0
	Overstory	-**	9.5	
	State harvest	+**	6.1	
	Recent logs	+**	3.8	
	Small-class conifers	-**	1.2	

Table 6. Continued.

Species	Variable	Sign	R^2	Total R^2
Chipping Sparrow	Riparian zone	-**	1.1	
	Time	+**	1.1	
	Pole-class recent snags	+**	0.6	
	Understory	+**	0.5	
	Shrub area	+**	0.4	
Dark-eyed Junco				40.0
	Overstory cover	-**	19.6	
	State harvest	+**	8.0	
	Time	+**	6.6	
	Shrub distance	+**	2.0	
	Snag height	+**	0.4	
	Medium-class recent snags	-**	0.6	
Dusky Flycatcher				13.5
	Pole-class deciduous trees	+**	3.8	
	Time	+**	2.2	
	Shrub distance	+**	1.3	
	Modified harvest	+**	1.3	
	State harvest	-**	1.3	
	Overstory cover	+**	0.8	
	Pole-class conifers	-**	1.1	
	Old logs	+**	0.6	
	Old natural stumps	-**	0.6	
	Pole-class old snags	+**	0.5	
Evening Grosbeak				4.1
	Medium-class recent snags	+**	1.5	
	Time	+**	0.8	
	Old logs	+**	0.6	
	Riparian zone	-**	0.5	
Golden-crowned Kinglet				20.9
	Overstory cover	+**	5.7	
	Old logs	+**	2.6	
	Pole-class deciduous trees	+**	2.6	
	Tree height	+**	2.3	
	Small-class recent snags	+**	1.6	
	Medium-class conifers	+**	1.4	
	Shrub distance	-**	1.1	
	Recent cut stumps	-**	0.8	
	Recent natural stumps	+**	0.6	
	Regenerating trees	-**	0.6	
	Time	-*	0.4	
	Large-class old snags	+*	0.4	

Table 6. Continued.

Species	Variable	Sign	R^2	Total R^2
Gray Jay				11.6
	Regenerating trees	+**	4.9	
	Riparian zone	-**	3.7	
	Pole-class conifers	+**	0.7	
	Large-class recent snags	-**	0.6	
	Medium-class old snags	-**	0.5	
	Medium-class recent snags	+**	0.5	
Hammond's Flycatcher				16.9
	Riparian zone	+**	3.8	
	Modified harvest	+**	3.7	
	Recent natural stumps	+**	2.0	
	Pole-class conifers	-**	1.7	
	Pole-class deciduous trees	+**	1.4	
	State harvest	-**	1.4	
	Small-class deciduous trees	-**	0.7	
	Shrub distance	+**	0.7	
	Shrub area	+**	0.7	
	Medium-class deciduous trees	+*	0.5	
	Recent logs	+*	0.5	
Hermit Thrush				3.1
	Small-class conifers	+**	0.9	
	Medium-class old snags	+*	-0.5	
	Small-class recent snags	-*	0.5	
MacGillivray's Warbler				16.1
	Pole-class deciduous trees	+**	7.6	
	Small-class deciduous trees	+**	1.0	
	Modified harvest	+**	0.8	
	Medium-class deciduous trees	-**	0.8	
	Medium-class recent snags	+**	0.6	
	State harvest	-**	0.6	
	Medium-class conifers	-*	0.5	
	Understory cover	+*	0.5	
Mountain Chickadee				3.6
	Overstory cover	-**	1.3	
	Small-class old snags	+**	1.1	
	Modified harvest	-**	0.7	
	Old logs	-*	0.5	

Table 6. Continued.

Species	Variable	Sign	R^2	Total R^2
Nashville warbler	Shrub distance	+**	5.3	11.7
	Riparian zone	-**	3.1	
	Small-class deciduous trees	+**	1.8	
	Old logs	-**	0.8	
Northern Water-thrush				14.2
	Pole-class deciduous trees	+**	6.9	
	Medium-class conifers	-**	2.3	
	Modified harvest	+**	1.2	
	State harvest	-**	0.6	
	Small-class recent snags	+*	0.4	
	Pole-class recent snags	-*	0.5	
	Large-class old snags	+*	0.4	
	Recent natural stumps	+*	0.4	
Pine Siskin				4.7
	Old logs	-**	1.5	
	Riparian zone	-**	1.0	
	State harvest	+**	0.7	
	Small-class recent snags	-*	0.6	
Red-breasted Nuthatch				17.7
	Riparian zone	-**	12.3	
	Time	+**	2.2	
	Shrub distance	+**	1.2	
	Overstory cover	-**	0.8	
Red Crossbill				2.0
	Riparian zone	-**	0.9	
Red-naped Sapsucker				8.7
	Medium-class deciduous trees	+**	2.6	
	Overstory cover	-**	2.3	
	Modified harvest	+**	0.9	
	Time	+**	0.8	
	Old cut stumps	+**	0.8	
	Riparian zone	-**	0.8	
	State harvest	-**	0.5	
Solitary Vireo				10.6
	Over-story cover	-**	5.4	
	Old logs	-**	1.7	
	Time	+**	1.2	
	Shrub distance	+**	0.7	
	Pole-class deciduous trees	-**	0.7	
	Recent natural stumps	+*	0.6	
	Modified harvest	+*	0.4	

Table 6. Continued,

Species	Variable	Sign	R^2	Total R^2
Swainson's Thrush				5.0
	Small-class deciduous trees	+**	2.0	
	Small-class recent snags	+**	0.7	
	Large-class recent snags	-*	0.6	
	Moderately-decayed logs	-*	0.6	
Townsend's Warbler				7.1
	Pole-class old snags	-**	1.0	
	Recent cut stumps	-**	0.9	
	Small-class conifers	+**	0.8	
	Large-class recent snags	-**	0.8	
	Tree height	+**	0.7	
	Time	+**	0.7	
	Shrub height	+**	0.7	
	Small-class deciduous trees	+*	0.5	
Varied Thrush				11.0
	Old logs	+**	5.2	
	Old cut stumps	+**	1.2	
	Small-class old snags	-**	0.9	
	Medium-class conifers	+**	0.7	
	Regenerating trees	-**	0.6	
	Old natural stumps	+*	0.5	
	Small-class conifers	+*	0.5	
	Understory cover	-*	0.5	
Warbling Vireo				5.1
	Pole-class conifers	-**	1.4	
	State harvest	+**	1.4	
	Riparian zone	-**	1.0	
	Shrub distance	+**	0.7	
	Snag height	-*	0.6	
Western Tanager				11.9
	Overstory cover	-**	4.7	
	Recent 'logs	+**	2.9	
	State harvest	+**	1.1	
	Recent cut stumps	+**	1.1	
	Riparian zone	-**	0.7	

Table 6. Continued.

Species	Variable	Sign	R^2	Total R^2
Winter Wren				43.7
	Riparian zone	+**	34.9	
	Time	+**	2.7	
	Modified harvest	+**	2.0	
	Shrub distance	-**	1.8	
	State harvest	-**	0.8	
	Tree height	+**	0.7	
	Large-class recent snags	-**	0.5	
	Large-class deciduous trees	+**	0.3	
Yellow-rumped Warbler				16.3
	Time	+**	6.2	
	Riparian zone	-**	5.5	
	Shrub distance	+**	1.8	
	Medium-class recent snags	-**	0.7	
	Regenerating trees	-**	0.6	
	Small-class conifers	+*	0.5	

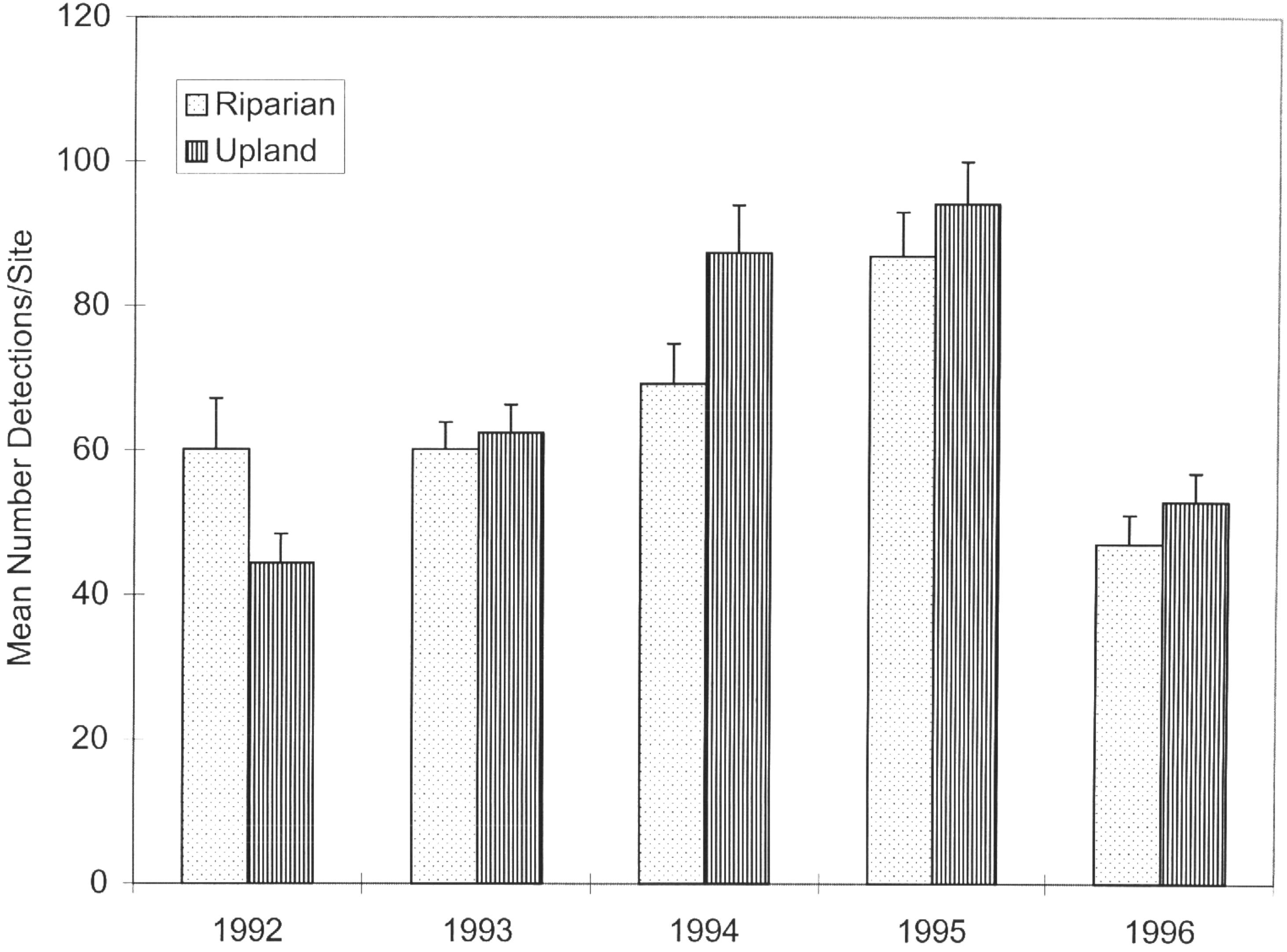

Figure 1. Mean abundance of all birds on riparian and upland transects at 18 study sites during the 5-yr study.

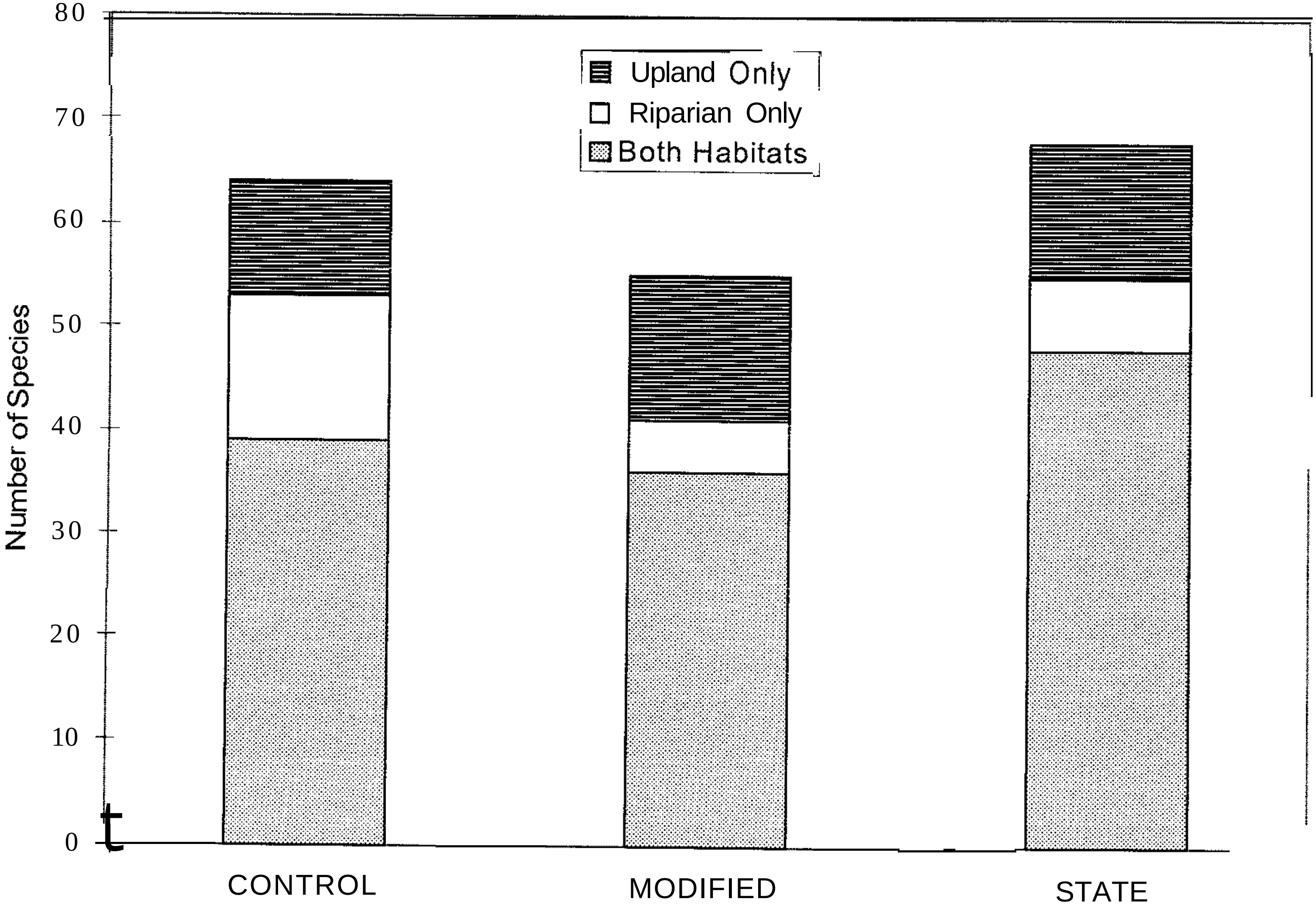

Figure 2. Number of bird species found only in upland, only in riparian, or in both habitats on 18 study sites.

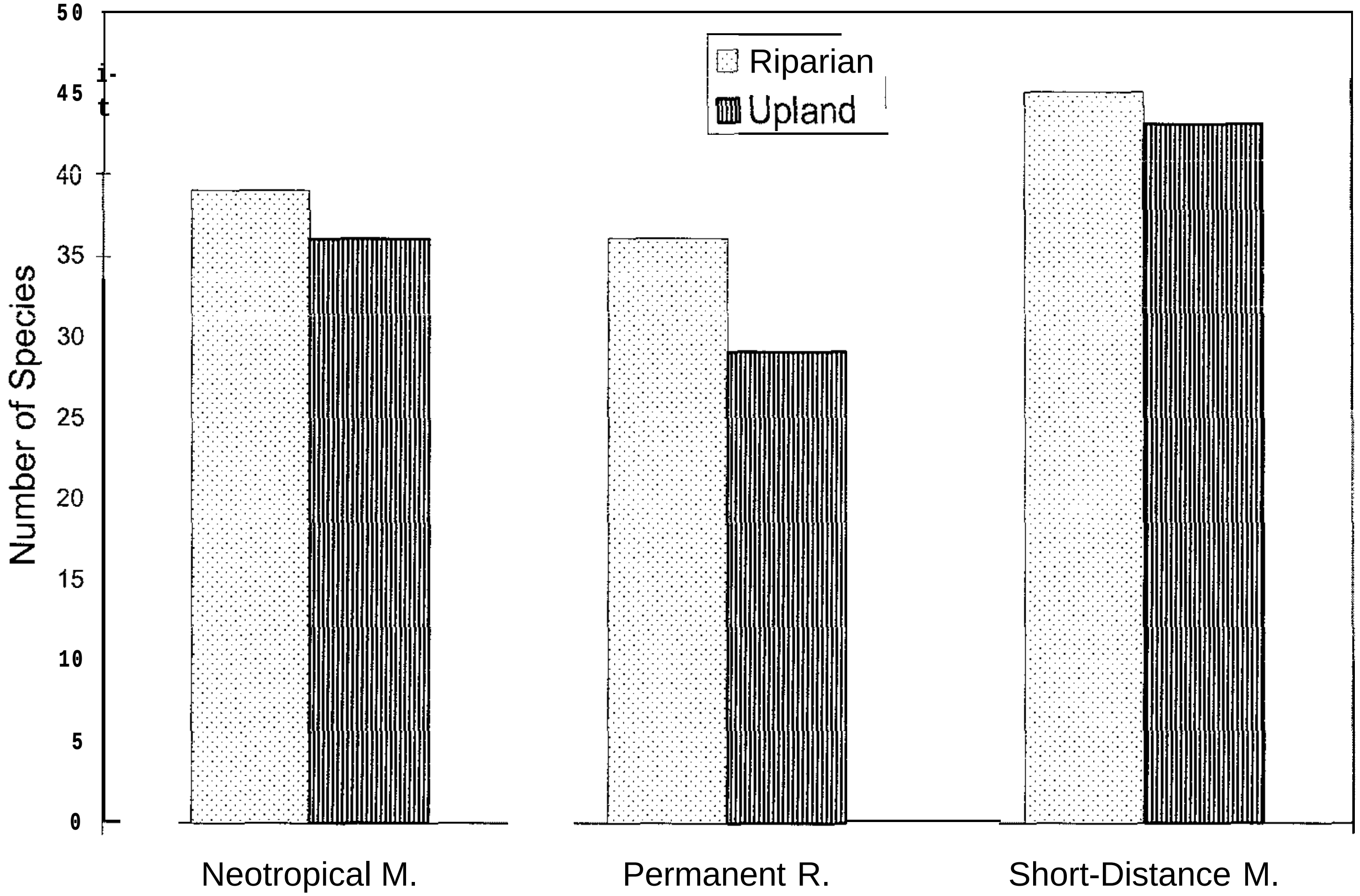

Figure 3. Number of species of Neotropical migrants, permanent residents, and short-distance migrants across all 18 sites.

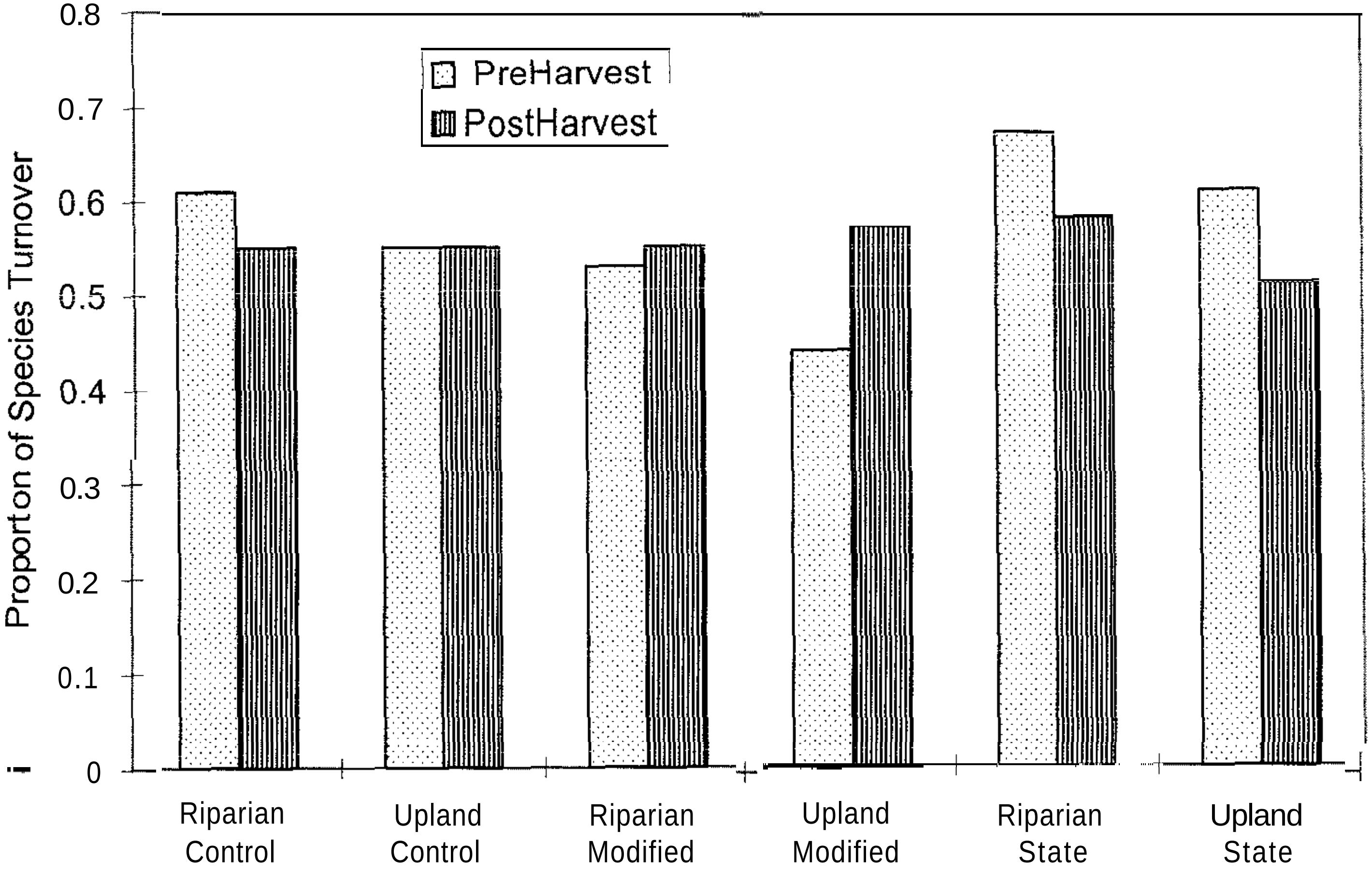

Figure 4. Species turnover rates (defined in text) on riparian and upland transects of Control, State, and Modified sites.

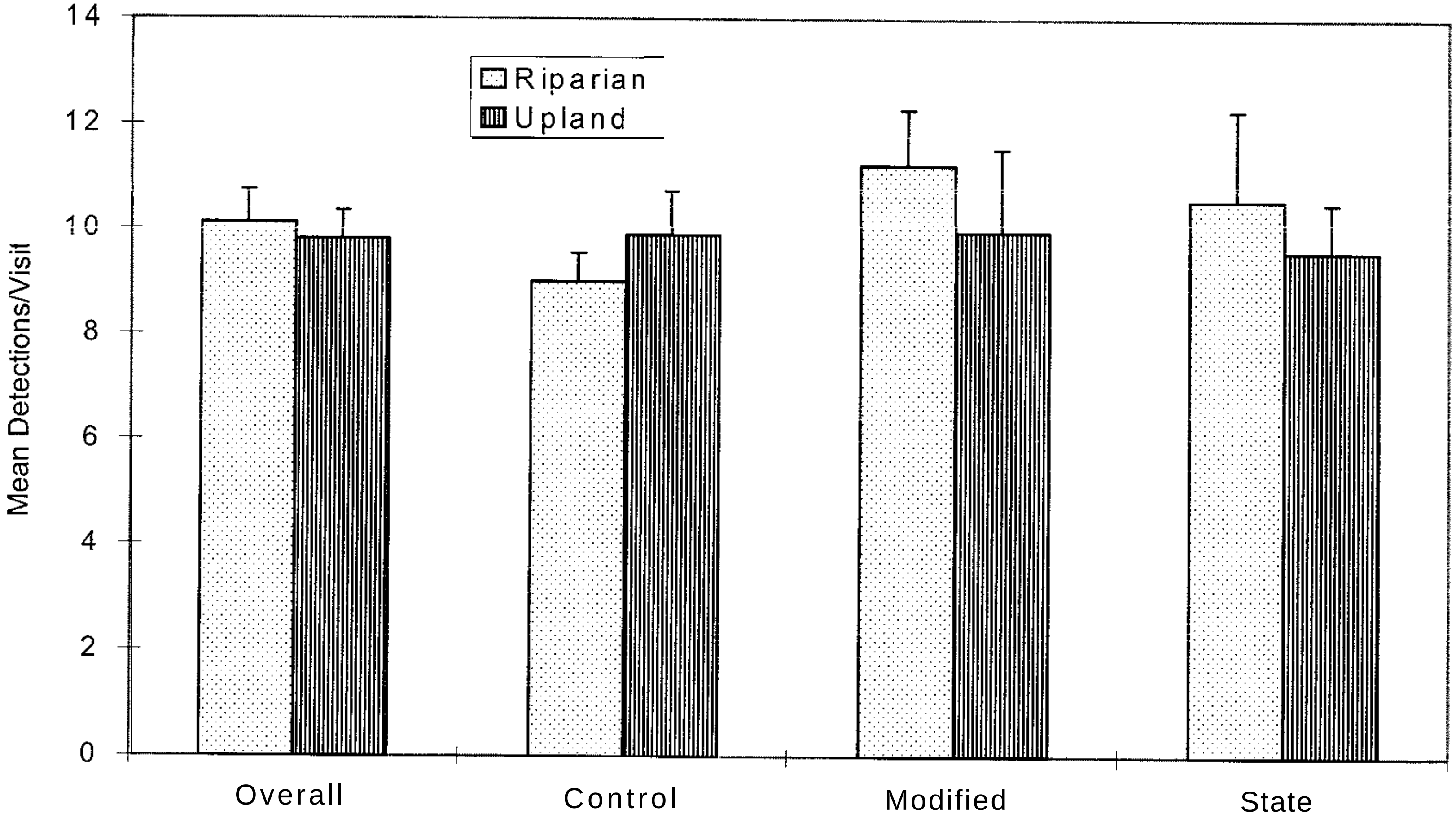

Figure 5. Mean abundance of all species on riparian and upland transects over all sites and on Control, Modified, and State sites before timber harvest on Modified and State sites.

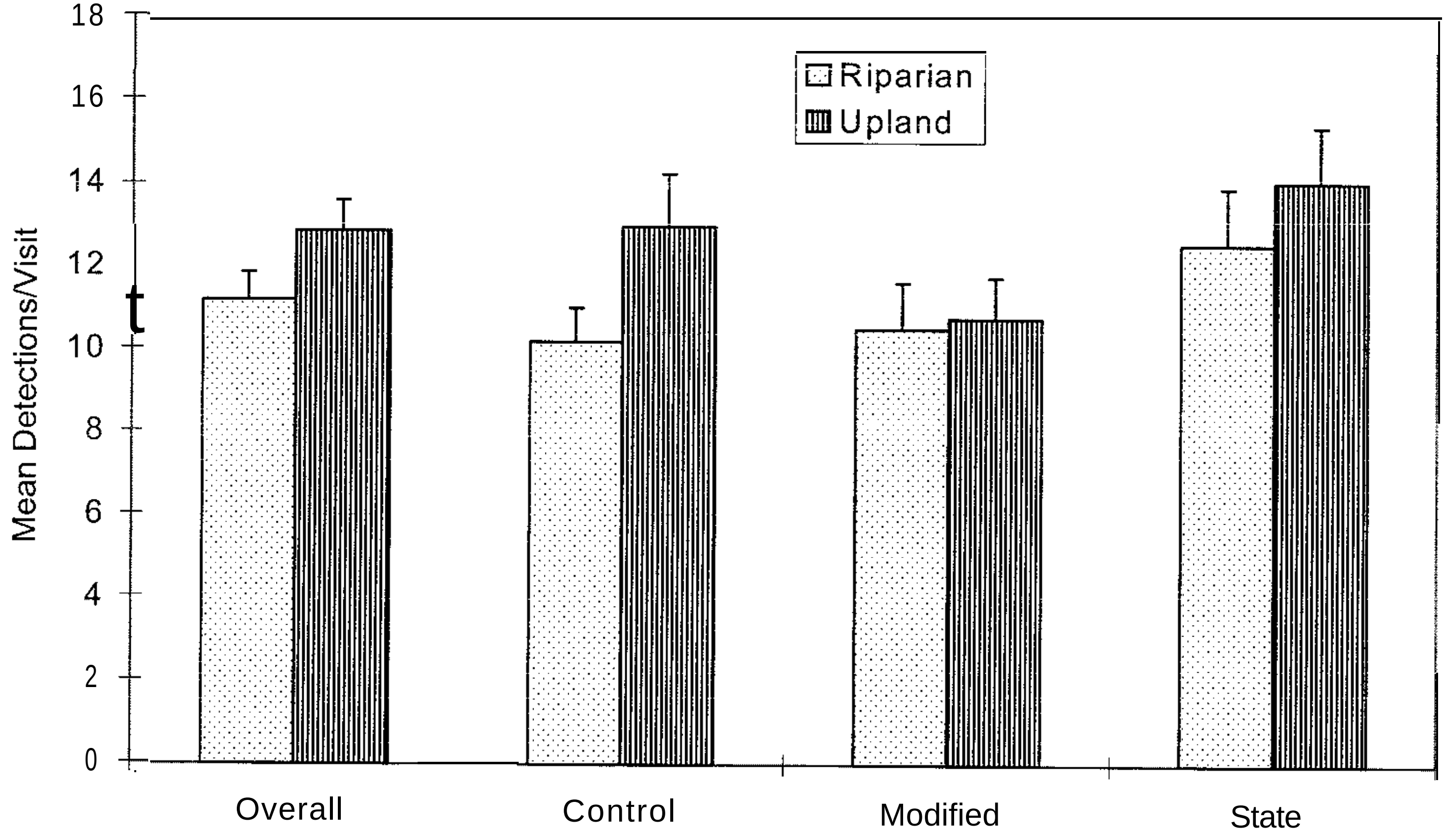

Figure 6. Mean abundance of all species on riparian and upland transects over all sites and on Control, Modified, and State sites after timber harvest on Modified and State sites.

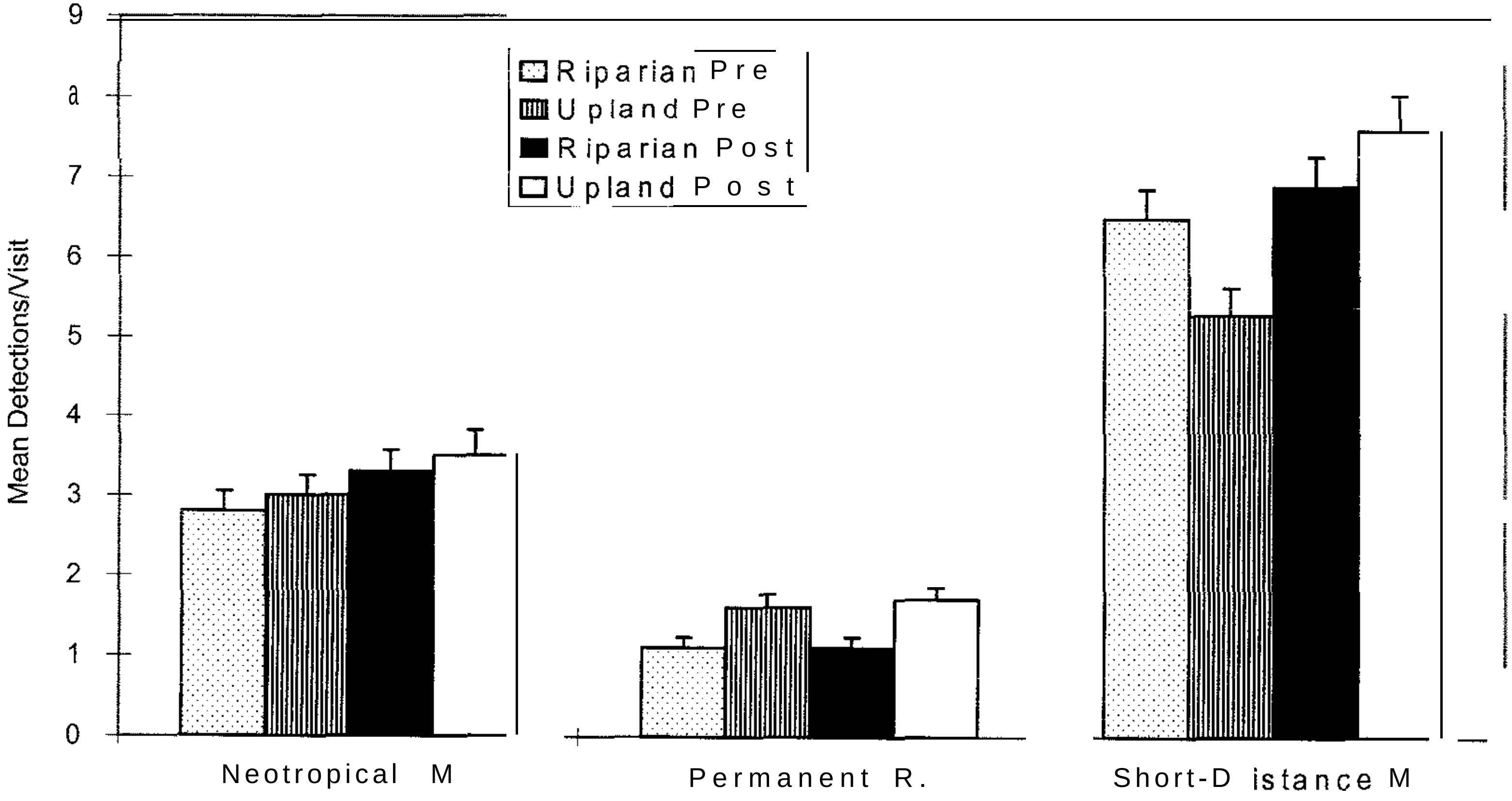

Figure 7. Mean abundance of Neotropical migrants, permanent residents, and short-distance migrants on riparian and upland transects before and after timber harvest.

Chapter 7

WEST-SIDE STREAM AMPHIBIAN SURVEYS

Abstract. We examined stream habitat and densities of tailed frog (*Ascaphus truei*) tadpoles and Pacific giant salamander (*Dicamptodon tenebrosus*) larvae in fish-bearing streams on industrial forest lands in western Washington both before and after clearcut logging. Specifically, the study was designed to detect changes in stream habitat parameters and amphibian densities in streams buffered by the narrowest legal buffer (State sites), a modified buffer containing greater numbers of damaged trees and snags (Modified sites), and forested sites that were not harvested (Control sites). State regulatory width buffers averaged 15.4 m (SD 6.87 m) on one side of the stream. Modified buffers were approximately twice as wide, averaging 30.5 m (SD 10.27 m). Stream habitat and amphibian densities were measured 2 years prior to timber harvest, 1992 and 1993, and 2 years after timber harvest, 1995 and 1996. In all, we captured 1,446 stream-breeding amphibians in 13 of the 18 streams sampled. We found no significant differences among treatment types in stream habitat measures or in tailed frog tadpole and Pacific giant salamander larval densities. The buffer strip configurations tested appear to provide adequate protection of stream amphibians and their habitat during timber harvest and for the two following years.

INTRODUCTION

Western Washington streams provide breeding habitat for six amphibian species: three torrent salamanders (*Rhyacotriton cascadae, R. olympicus,* and *R. kezeri*), two giant salamanders (*Dicamptodon copei* and *D. tenebrosus*), and the tailed frog (*Ascaphus truei*). The community composition of any given stream depends on its geographic location, site-specific geology, and stream history. Of the six species, only the Pacific giant salamander and the tailed frog are distributed throughout the western slope of the Cascade Mountains Stream amphibians use both not&h-bearing as well as fish-bearing streams and adjacent terrestrial habitat. The complexity of their distribution and life history creates a challenge for biologists and land managers working to assure that land-use practices do not result in the extirpation of native species

Of the six species listed above, only two were found in study streams with enough regularity to warrant statistical analyses, the tailed frog and the Pacific giant salamander. Therefore, the primary discussion in this chapter will be limited to these two species.

Giant salamanders are limited to permanent streams where paedomorphic adults can survive and larvae develop over a 2-3 yr period. Larvae and paedomorphs are voracious predators and are often the dominant vertebrate predator in non&h-bearing streams. When forest canopies are removed along streams, primary production, invertebrate density, and biomass increase (Bisson and Davis 1976, Hawkins et al. 1982). These increases most likely contribute to an increase in salamander biomass immediately following timber harvest (Hawkins et al. 1983, Murphy and Hall 1981). However, cumulative effects of timber harvest appear to reduce densities of *Dicamptodon* larvae in streams in forests logged 14 to 40 yr earlier when compared to streams in unlogged forests (Corn and Bury 1989).

Tailed frog tadpoles exhibit unique adaptations to life in fast-flowing streams, Tadpoles maintain position in the stream by creating a negative pressure between their bodies and the surface of a rock, thereby adhering to rocks while expending very little energy (Gradwell 1971). The tadpole then uses rows of teeth to scrape diatoms off the rock. To be effective, this feeding strategy requires stream habitat with cobble or gravel-bottom streams and relatively little silt. Adult frogs have fairly flat, streamlined bodies that allow access to narrow cracks and crevices found between rocks on the streambed and bank. Both tadpoles and adults use the underside of rocks for cover and adult females deposit eggs under rocks and in rock piles or dams where stream water percolates through the rocks. Logging practices that increase siltation and filamentous algal growth in streams reduce the available cover by clogging cracks and crevices. Alga1 blooms that cover rocks block tadpole access and limit tadpole abundance. Several studies have measured lower tadpole densities in streams in logged forests than in unlogged forests both immediately following timber harvest and up to 40 yr after harvest (Corn and Bury 1989, Welsh 1990, Kelsey 1995, Bull and Carter 1996, Dupuis et al. 1997).

Decreases in stream amphibian densities are of concern for several reasons. Stream amphibians of the Pacific Northwest comprise a unique community. Tailed frogs and Pacific giant salamanders occur from northern California to British Columbia (Fig. 1). Torrent salamanders have limited distributions in Washington, Oregon, and California and Cope's giant salamanders are found only on the Olympic Peninsula, southwestern Washington, and the southern Washington Cascades continuing into Oregon (Leonard et al. 1993). If these species are extirpated from Pacific Northwest streams, they will be effectively extinct.

Densities of stream amphibians can be quite high, sometimes exceeding $35/m^2$ (Kelsey unpublished data). Consequently, they provide an important prey base as well as a source of predators in both aquatic and terrestrial habitats. Terrestrial adults that feed away from the stream return to breed and deposit eggs, thus providing a direct link between terrestrial and aquatic habitats in the transfer of energy and nutrients, Because of the uniqueness of the stream amphibian community and their apparent vulnerability to changes in stream habitat following clearcut logging, we endeavored to evaluate the effectiveness of riparian buffer strips along Type 3 streams in protecting stream amphibians and in-stream habitat from effects of clearcut logging. The following chapter presents the results of stream amphibian and stream habitat sampling that was performed from 1992-1998 at 18 western Washington sites.

METHODS

Sampling design

This project implemented standard sampling techniques to quantify physical features of in-stream habitat and stream amphibian communities over the course of the project. The sampling methods described below were repeated in an identical manner at all streams in all years.

Abiotic stream features

Physical attributes of in-stream habitat that could potentially affect stream amphibian densities were measured at all 18 sites in 1993 before timber harvest. These measures were repeated at all 18 sites following timber harvest in 1995 and repeated at 17 sites in 1996. The unsampled site, Ryderwood 1 557, was not harvested until 1996. Consequently, the 2 yr of post-treatment sampling did not occur until 1997 and 1998. These data have been incorporated with the rest of the post-treatment data.

Sampling methods were modified from protocols designed to monitor physical habitat features of streams that influence salmonid abundance (Ralph 1990, Platts et al. 1983). We desired a protocol that would provide information on stream habitat features that influence amphibian distribution and reveal effects of logging on both stream habitat and amphibians. Measurements were compared between Control and treatment sites (State and Modified) before and after timber harvest to identify habitat changes due to timber harvest. In addition to these measures, we included those used by Aubry (1985) in Washington streams in old-growth forests and Corn and Bury (1989) in Oregon streams to generate a comparable data set.

At each site, stream habitat features were surveyed at five transects perpendicular to the stream. Transects were placed at 100-m intervals corresponding with riparian bird point count stations. They were numbered from downstream to upstream, 1-5. Transects were marked with plastic flagging so they could be located in subsequent years, although some flags disappeared between seasons. At each transect, we took identical measures and calculated means for each site.

Stream gradient was measured using a clinometer looking from the transect upstream 10 m. Right and left bank gradients were also measured using a clinometer and looking perpendicular from the stream bank away 10 m. Steep hill slopes send more coarse and tine materials to the stream than flat hill slopes. Larger rocks and cobbles create more varied habitat while fine sediments tend to decrease available habitat by filling interstitial spaces in stream substrate.

Embeddedness of a rock within the streambed indirectly measures the amount of fine sediment in the stream channel. We examined a minimum of five rocks in riffles and estimated the average percent of the rocks vertical dimension buried in the bottom substrate. Rocks sitting on top of the substrate were relatively free from embeddedness and recorded as <5%. If 5-25% of the vertical dimension was below the substrate surface it was recorded as 25%. Likewise, 25-50%, 50-75%, and 75-100% embeddedness were recorded as 50%, 75%, and 100%, respectively.

In-stream habitat was divided into two types: pools and riffles. This distinction is commonly used in stream amphibian studies because tailed frog tadpoles have been positively associated with riffle habitat while torrent and giant salamander larvae are found in both pool and riffle habitats with roughly equal frequencies (Aubry 1985, Bury et al. 1991b). We defined pools as areas with almost no perceptible downstream water velocity and variable depths. Pool substrate was usually covered with a layer of fine silt. Riffles were defined as areas with fast downstream velocities and shallow depths, generally shallower than pools. To measure the amount of habitat in pools, we estimated the percentage of pool habitat that intersected the transect.

Although no stream habitat measurements were made during the winter months when stream stages are highest, the ordinary high water mark indicates the winter channel widths and can be compared between streams regardless of when the measurement is made. We measured stream bankfull width between ordinary high water marks, the point where riparian vegetation meets rock or gravel deposited by the stream. To avoid biases associated with timing of the survey, high flow water depths were measured to the horizontal plane of the ordinary high water width (adapted from Ralph 1990) at three points, 1/4, 1/2, and 3/4 of the distance between shorelines. Average depth was calculated by dividing the sum of these three measurements by four to account for the starting point at the bank where the water surface and the bank meet. Wetted width measures described the horizontal distance along the transect from the edge of the water at one shore to the other. It can be influenced by down wood, rock materials, and debris dams that partially block stream flow, retain water, and create wider channels and more stream habitat.

Volume of large organic debris (LOD) in the stream was calculated by measuring the length and diameter of all pieces of wood >20 cm in diameter that entered or crossed the stream within 5 m upstream or downstream of the transect. Position of LOD was noted as: 1) all of the piece occurred within the high water width that would be covered by water during high water stages; 2) >50% of the length occurred within the stream channel; 3) <50% of the length occurred within the stream channel; or 4) the piece did not enter the channel but was somehow suspended above the channel. Location of log jams or root wads was also noted and length, width, and height dimensions were measured.

Water temperature was measured at each transect to represent average summer water temperatures for these streams. Temperatures are generally highest in July and August (Beschta et al. 1987) when most of the streams were surveyed. Water temperature was measured in either a pool or riffle that did not receive direct sunlight. We measured temperature throughout the course of the day as we moved from the first transect to the fifth transect, 400 m upstream.

Undercut banks provide cover for giant salamanders and identify areas of increased sediment inputs to the stream. In 1993, only the depth, distance excavated by the stream under the bank to the protruding edge of the bank, was measured. In 1995 and 1996, the area of the undercut bank was measured by measuring depth and the length along the stream, Areas of bank failure or bank: soil movements with exposed regions of bare soil were defined as examples of bank slumps. All slumps within 5 m upstream or downstream of the transect were measured so area of the slump could be calculated from the length and height of exposed soil. Frequently, these areas were associated with tree blow down where unearthed root balls exposed large areas of bare soil. To estimate the stability of both the stream bank and valley slope, ocular estimates of vegetation and rock cover were made at each transect. High ratings indicated a stream bank or valley slope less prone to erosion. Four classes were assigned: 1) 0-25% vegetation and rock cover; 2) 25.50% vegetation and rock cover; 3) 50-75% vegetation and rock cover; 4) 75-100% vegetation and rock cover. Estimates were made considering both sides of the stream. To evaluate the amount of stream bank area previously altered by stream processes, animals, or other means, we assigned a soil alteration rating. The condition of the bank just above the water level and within 1 m on either side of the transect was assessed. If bank alteration was slight, 4 was assigned. If 25.50% of the bank appeared altered or eroded, 3 was assigned. Likewise, 50-75% and 75-100% alteration were assigned ratings of2 and 1, respectively.

Following timber harvest at treatment sites, buffer strip widths were measured on both sides of the stream from the ordinary high water mark to the line of trees on the outer edge of the buffer. Slope distance was measured and recorded.

Biotic stream features

Stream amphibian surveys were modified from protocols developed and tested in Oregon and Washington (Aubry 1985, Bury and Corn 1991). Samples were collected at two randomly chosen locations within the 400 m length of stream where physical features were systematically sampled. At both locations, a 10-m length of stream was flagged at l-m intervals and 0.32 cm hardware cloth screens were secured at the downstream end. At each meter interval, microhabitat features were measured so results could be compared with Aubry (1985). Microhabitat measures included estimating percent pool habitat, dominant substrate class, and width of stream and depth at 1/4, 1/2, and 3/4 of the width of the stream at the time of sampling. A rough map was drawn of the 10-m length indicating location of pools, riffles, boulders, undercut banks, LOD, and any other prominent stream feature. Once the microhabitat survey was complete, additional hardware cloth screens were placed between 3 and 4 m and between 6 and 7 m. The bottom edge of the screens was buried in the gravel substrate so the screens would block the passage of stream amphibians.

Following the microhabitat survey and placement of screens, we began to remove all rock and wood from the stream working from downstream to upstream. All rock surfaces were first examined for tailed frog tadpoles and areas underneath rocks were examined for salamanders and tailed frog adults. When an amphibian was located it was captured by hand or with a small aquarium dipnet and transferred to a small ziplock bag filled with stream water. Position of the animal when it was found was noted and the bagged animal was placed to one side, in the shade, until the entire 10 m of stream had been searched. Once the stream had been completely dismantled, it was allowed to settle before being thoroughly reexamined from downstream to upstream for additional amphibians. Some incidental species (lentic rather than lotic breeders) were seen and captured on the stream bank. The stream banks were not systematically searched, only the stream channel itself.

Amphibians were measured (total length, TL, from tip of snout to tip of tail and snout-vent length, SVL, from tip of snout to anterior end of vent, and tadpole rear leg length), weighed using a Pesola scale, and examined for stage of development and sex of adult frogs, Four classes were used to describe the developmental stage of each tadpole (Bury and Corn 1991). Hatchling tadpoles were assigned to the first class and generally range from 20 to 24 mm in total length. Mature tadpoles ranged from 30 to 55 mm total length and showed no signs of initiating metamorphosis, As metamorphosis begins and hind legs develop, tadpoles are classified as developmental stage 3. Stage 4 tadpoles show development of front legs and frog morphology but retain part of the larval tail. Stage 4 tadpoles may be smaller in total length than stage 2 or 3 tadpoles if the tadpoles have begun tail resorption. Once a metamorphosing individual has completely resorbed the larval tail, it is classified as a juvenile frog. Frogs were identified as adults or juveniles depending on their size, Frogs ≥35 mm snout-vent length were considered adults (Aubry and Hall 1991, Bury and Corn 1991). Salamanders were identified as larval, aquatic with gills, wide tail tin and solid brown color, or adult, lacking gills, tail tin and with mottled coloration. The position of the animal in the stream and the type of cover object were also noted. Fish and crayfish were also captured and counted but not measured or weighed.

After processing the animals, hardware cloth screens and flags were removed and all rocks and wood were returned to the stream channel in a configuration alternating pool and riffle habitat. Captured amphibians were gently returned to the stream at approximately the same area they had been found.

Statistical analyses

Site selection processes could not control for specific stream conditions other than stream type and accessibility. Consequently, possible biases in how streams were assigned to treatment types had to be assessed. Differences in stream features between treatment types were compared in 1993 before any timber harvest took place. Streams assigned to different treatment types were compared using single factor analysis of variance (ANOVA) tests.

To assess differences in stream habitat features following timber harvest, comparisons of results across treatment types were completed as described in the overall methods of this project. Briefly, means for each site in each year were calculated using data collected from each of the five transects at a site to describe average conditions of stream habitat by stream, treatment type and year. Graphical examination of the results using standard box plots (Wilkinson 1997) preceded all statistical analyses. Post-treatment means (1995, 1996) were compared using paired t-tests to test the null hypothesis of no difference between years. If the test upheld the hypothesis ($P > 0.05$) then post-treatment data were averaged at each site. If the years were different, only data collected in 1996 were used in the analysis. We subtracted pre-treatment means from post-treatment means for each treatment type and used a single-factor ANOVA to test the null hypothesis of no difference among treatment types when differences between post- and pre-treatment conditions are considered. Percentages (stream and valley gradients) were arcsine transformed; data with heterogeneity of variance were log transformed (Zar 1984).

Numbers of tailed frog tadpoles depend on the time of sampling. In early summer, all age classes are represented. Throughout the summer, second year tadpoles metamorphose and in late summer and early fall, hatchling tadpoles may be found. Because it was impossible to survey all sites simultaneously, tadpoles of various stages were captured. We compared only those that did not have developing leg buds and were ≥ 30 mm. These were assumed to have completed their first year of development and would metamorphose the following summer (Bury and Corn 1991). Tadpole and salamander densities were calculated by dividing the number captured by the stream area for each survey. Densities from both surveys at one stream in one year were averaged to obtain a mean density per site per year. Means were calculated for pm-treatment and post-treatment years for each site. The mean pre-treatment density was subtracted from the mean post-treatment density at each site. Differences in stream amphibian abundance were compared among treatment types using single-factor ANOVA to test the null hypothesis of no difference among treatment types. We set a = 0.05 for all tests.

RESULTS

Abiotic features

Mean and standard deviation of abiotic stream features for pre-treatment surveys (1993) and post-treatment surveys (1995, 1996) are presented in Tables 1-5. For each in-stream and stream bank habitat feature measured, no statistically significant differences were detected in pre-harvest measures among sites assigned to different treatment types.

During the first 2 yr following timber harvest, riparian habitat in harvested sites with buffer strips remained similar to Control sites. In-stream and bank habitat measures showed no significant differences among Control, Modified, and State sites following timber harvest

($P > 0.10$, Tables l-5). The only exception to this was buffer strip width. Buffers were significantly wider on Modified than on State sites ($P < 0.05$, Table 3).

Biotic features

Amphibian community

Altogether, 1,469 individuals of nine amphibian species were captured and released during 4 yr of stream surveys (Table 6). Five of the nine species captured do not breed in lotic habitats, The Pacific tree frog ***(Hyla** regilla*), red-legged **frog** (*Rana aurora*), northwestern salamander (*Ambystoma gracile*) and roughskin newt ***(Taricha** granulosa*) breed in lentic waters. Western redback salamanders ***(Plethodon** vehiculum*) breed in terrestrial habitats. Of these captures, only the red-legged frog was captured in the stream channel. The other captures occurred on the bank adjacent to the stream. The other four species are stream breeders: tailed frogs, Pacific giant salamanders, Cope's giant salamanders and Columbia torrent salamanders. Stream breeders comprised 99% of amphibians captured in the stream surveys.

Locations of study sites did not fall within the distribution of all stream-breeding amphibians. Only 7 of the 18 sites fell within the known distributions of Cope's giant and torrent salamanders, and study streams were generally larger than those where torrent salamanders occur in high densities. Stream amphibian species ***were*** not captured at four of the five Capitol State Forest sites, an area with no historical record of stream amphibian presence (Nussbaum et al. 1983). Two species, tailed frogs and Pacific giant salamanders, have distributions that included all study sites except those in the Capitol State Forest. Tailed frog tadpoles and Pacific giant salamanders were captured frequently enough to permit statistical analysis of the survey results. These two species are discussed separately.

Tailed frog tadpoles

Altogether, 812 tailed frogs and tadpoles were captured and released during pre and post-treatment sampling (Table 6). Only 20 adults and 16 juveniles (recently metamorphosed) were captured during stream surveys between 1992 and 1996 (Table 7). We captured three hatchling tadpoles during all 6 yr of surveys. Over 400 mature tadpoles, 213 and 160 stage 3 and stage 4 tadpoles, respectively, were captured. Stage 2 tadpoles were captured throughout the summer at all study sites where tailed frogs were known to occur except for three sites in the southwestern Washington Coast Range, Abernathy, Ryderwood 860 and Ryderwood 1557. Tadpoles were found in these streams in spring but not late August or September. We suspect that these three sites have tadpole populations that develop in 1 yr whereas larval development appears to occur over 2 yr at the other sites. To avoid biases associated with time of year the sampling occurred and tadpole phenology, we compared only stage 2 tadpoles that would metamorphose the following summer.

Second year tadpoles were captured at 9 of the 18 study sites, five of six Control sites, three of six Modified sites and one of six State sites (Table 8). Densities of second year tailed frog tadpoles ranged from 0 to 2.77 tadpoles/m’ (Table 8). One stream with a Modified buffer strip, Eleven Creek 3 1, had the highest density of tailed frog tadpoles in all years. All other streams had densities < 1 tadpole/m^2. Analysis of tailed frog tadpole densities revealed no treatment effect when differences in pre- and post-treatment densities were compared among treatment types at all 1.8 sites (*P* = 0.88, Table 8; Fig. 2). When the Capitol State Forest sites are removed, we still found no significant difference among treatment types (*P* = 0.88).

Giant salamander larvae

Altogether, 10 Cope's giant salamander larvae and 619 Pacific giant salamanders (610 larvae and five adults) were captured from 1992 to 1996 (Table 6). Because so few Cope's larvae were captured,, they were not included in analyses of treatment effects. Pacific giant salamanders were captured in 13 of the 18 study sites, five Control sites, four Modified sites and four State sites. Stream surveys at Capitol State Forest sites produced no Pacific giant salamander larvae. Adult Pacific giant salamanders were not included in analyses of treatment effects because so few were captured.

Densities of salamander larvae ranged from 0 to 2.36 larvae/m* (Table 9). The State site with the highest density, Kapowsin, (2.36 and 1.36 larvae/m* following timber harvest) contained many large boulders (>30cm in diameter) creating extensive cover areas. As many of the boulders were too large to move, we most likely underestimated actual larval salamander density.

Mean densities of Pacific giant salamander larvae remained the same or increased at all but two sites and showed no differences between treatment types ($P = 0.507$, Table 9; Fig. 3). The largest decrease occurred at a Modified site, Ryderwood 860. At this site, salamander larval density was similar in pre-treatment years and the first post-treatment year, 0.46, 0.49, and 0.49 larvae/m^2, respectively. The following year, the density dropped to 0.15 larvae/m^2. This is the only site that showed more than a slight decrease following timber harvest.

The relative sizes (snout-vent length) of salamanders captured did not change following timber harvest (Table 10). Larval snout-vent length ranged from 22 mm to 100 mm prior to timber harvest with a mean length of46 mm. Following timber harvest, sizes ranged from 12 mm to 149 mm with a mean length of 47 mm and no significant treatment effect ($P > 0.47$).

DISCUSSION

The results of 4 yr of intensive sampling of in-stream habitat and stream-breeding amphibian communities indicate that state-prescribed riparian buffer strips function adequately to maintain pre-harvest habitat conditions and amphibian abundance during and immediately following timber harvest activities. This study found no significant differences between post-treatment and me-treatment conditions among Conrol sites, Modified sites, and State sites. These results do not address the question of changes that may occur in both in-stream habitat conditions and amphibian communities in the next 15 to 20 yr. To obtain this information, sites should be sampled over a 2-yr period at 5-yr intervals.

Within treatment sampling variance exceeded between treatment variance in tailed frog analysis. Because site differences were so great and stream-breeding amphibians were not present at all streams, variation between sites tended to obscure any treatment effects. Sampling stream amphibians continues to be a time and labor intensive task. Devising sampling methods that are both adequate for abundance and variance estimates as well as affordable continues to challenge ecologists (Heyer et al. 1994, Welsh et al. 1997). Our sampling method expanded a design described and tested by Bury and Corn (1991). Because we did not limit our research question to just one species, we chose a broad sampling method with the potential to capture all species of stream-breeding amphibians. We could potentially reduce sampling variances by including a third randomly selected, 10-m sampling segment or by sampling five or six randomly selected 5-m segments,

The absence of stream amphibians from the Capitol State Forest sites further complicates the experiment because the forest appears to occupy a gap in the distribution of the species. The difficulty of identifying suitable study sites for this project forced us to adopt sites without stream-breeding amphibians. Initial surveys of several Capitol State Forest streams revealed tailed frog tadpoles in two streams located on the northern boundary of the forest. Historical records indicate tailed frog tadpoles were found near Summit Lake, north of Highway 8 and the main block of the Capitol State Forest (Nussbaum et al. 1983). No systematic surveys of the forest and surrounding areas have been done to try to understand the reason for this distributional gap. Two broad-based vertebrate sampling studies (USFS DEMO Project and this project) have had 400 pitfall traps installed within the forest and operated for more than 30,000 trap nights since 1992 (West unpublished data). Even with this trapping effort, only one juvenile tailed frog has been captured just inside the forest block on the west side only 2.4 km from the Chehalis river and Highway 12. Three metamorphosing tadpoles were also captured in the stream of this same site. We cannot say why this gap in the distribution of stream-breeding amphibians exists in the Capitol State Forest. Attributing it to logging activity or glacial actions seems misplaced as stream amphibians are found in other areas that were logged or heavily glaciated. A combination of soil types and geologic history may have excluded stream-breeding amphibians from this forest,

Densities of tailed frog tadpoles and Pacific giant salamander larvae tended to be lower in Type 3 streams in this study than Type 4 streams in managed forests surveyed in 1992, 1993, 1994 (Kelsey 1995). Mean (± SD) densities of tailed frog tadpoles in Type 4 streams in harvest-age managed forests were 2.13 ± 2.02 tadpoles/m* versus 0.24 ± 0.17 tadpoles/m^2 in Type 3 forested streams. Mean densities of Pacific giant salamander larvae in Type 4 streams in harvest-age managed forests were 1.10 ± 1.46 larvae/m^2 versus 0.47 ± 0.44 larvae/m* in Type 3 streams. Densities might be higher in smaller, non-fish bearing streams for several reasons. First, survey methods that require dismantling of a stream are easier to perform thoroughly in smaller streams. Stream amphibians escape detection more easily when the water column is higher. Therefore, samples from larger streams may underestimate general abundance more than those from smaller streams. Second, the presence of large vertebrate predators in larger streams, namely fish (Family Cottidae and Salmonidae) and paedomorphic giant salamanders, may limit densities of tailed frog tadpoles and smaller salamander larvae. Third, fish-bearing streams tend to have lower gradients and more sediment than nonfish-bearing streams. Higher stream gradients often limit the migration of fishes upstream and the deposition of silt. Higher sediment loads have been correlated with lower densities of stream amphibians, (Corn and Bury 1989) and may reduce available cover and egg deposition sites in low gradient streams. Higher densities of stream-breeding amphibians in Type 4 streams indicate the need to protect habitat in nom'ish-bearing streams from the effects of clearcut logging.

Changes in terrestrial abundance of tailed frog and Pacific giant salamander adults must also be considered. Although differences were not statistically significant, numbers of both species declined in upland clearcuts and showed little change in riparian areas. If adults decline, breeding could be affected and larval numbers may decrease. This type of trend may not be evident immediately after clearcutting. The lag period depends on the rate of loss of metamorphosed juveniles and adults from terrestrial habitats, numbers of larvae already in the stream and the age at first reproduction. Consequently, stream and terrestrial abundance should be measured periodically over the next 15 to 20 yr, until adjacent forests reached closed canopy. If a decline is observed, periodic monitoring should continue until population recovery is observed.

Differences in the effectiveness of buffer strip configurations are difficult to assess at present. Our results show that both buffer configurations effectively protected stream habitat and amphibians from timber harvest operations. The major weakness in this project is that stream amphibians were not found in all 18 sites. A second weakness is that we could only address the question of effectiveness of riparian buffer strips with two stream-associated amphibian species. We have attempted to assess responses of tailed frog and Pacific giant salamander larvae to clearcut harvesting in western Washington. This study, along with previous work, provides a glimpse at an answer. Continued sampling at these sites, along with further work is merited.,

Impacts of clearcutting on Dunn's, Van Dyke's, Cope's giant, and torrent salamanders require projects with limited geographic scope and intensive sampling methods specifically designed to detect these species. Continued work to address impacts on these species is needed.

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Table 1. Stream width and depth in meters summarized by year and treatment type. N = 6 for each treatment type. Data from 1993 were collected prior to timber harvest; 1995 and 1996 are post-treatment data. One State site (Ryderwood 1557) was harvested in 1996. Pre-treatment data are an average of 1993 and 1995. Post-treatment data were collected in 1997 and 1998 yet are listed in the 1995, 1996 columns, respectively. The ***P-value*** indicates the probability of no treatment effect on each habitat parameter assuming the null hypothesis is true and was calculated using an ANOVA as described in the chapter.

Sites	Bankfull Width (m)			Wetted Width (m)			Average Depth (m)		
	1993	1995	1996	1993	1995	1996	1993	1995	1996
Controls									
Mean	4.44	4.84	6.82	2.49	2.39	2.76	0.24	0.32	0.40
SD	1.91	1.60	2.58	0.94	0.43	1.27	0.16	0.07	0.12
Min	2.62	2.75	2.91	1.44	2.05	1.55	0.10	0.25	0.23
Max	6.83	6.16	10.65	4.06	3.18	4.89	0.55	0.44	0.53
State									
Mean	3.85	5.86	4.84	3.35	4.39	2.40	.98	0.33	0.41
SD	1.22	2.56	2.64	1.94	3.18	1.09	1.69	0.06	0.15
Min	2.18	3.02	2.47	1.57	1.68	1.18	0.17	0.27	0.28
Max	5.64	10.26	8.56	6.68	10.28	3.84	4.42	0.39	0.67
Modified									
Mean	3.82	3.64	4.73	2.17	2.34	2.13	0.18	0.30	0.36
SD	1.66	1.49	1.48	0.66	0.39	0.46	0.06	0.06	0.12
Min	2.64	1.27	3.05	1.44	1.92	1.29	0.13	0.21	0.24
Max	7.04	5.64	6.77	3.12	2.94	2.56	0.26	0.37	0.56
P-value		0.40			0.76			0.33	

Table 2. Mean, standard deviation (SD), minimum (Min), and maximum (Max) percent pool habitat, percent embeddedness of substrate, bankslump area, and area of undercut banks. See Table i for sample size, pre and post-harvest treatment years, and *P-value* calculation.

Sites	Pools (%)			Embeddedness of substrate (%)			Bankslump area (m^2)			Undercut bank area (m^2)		
	1993	1995	1996	1993	1995	1996	1993	1995	1996	1993	1995	1996
Controls												
Mean	42.17	21.03	44.17	45.58	41.58	41.67	0.55	0.00	12.94	0.11	0.31	0.56
SD	35.97	12.37	12.11	7.71	10.58	6.06	1.35	0.000	20.16	0.14	0.41	0.99
Min	0.00	8.00	34.00	35.00	24.00	35.00	0.00	0.00	0.00	0.00	0.00	0.00
Max	94.00	42.00	62.00	55.00	52.00	50.00	3.30	0.00	50.84	0.35	0.83	2.52
State												
Mean	62.88	26.17	50.17	45.79	37.75	50.00	0.00	13.92	4.08	0.10	0.91	0.20
SD	27.38	13.98	17.00	14.92	14.59	21.45	0.000	24.23	9.63	0.10	1.79	0.24
Min	16.00	12.00	36.00	25.00	17.00	15.00	0.00	0.00	0.00	0.00	0.00	0.02
Max	88.00	46.00	80.00	65.00	60.00	70.00	0.00	62.68	23.73	0.26	4.54	0.68
Modified												
Mean	57.20	24.00	51.67	35.73	41.33	40.00	0.00	0.86	12.70	1.55	0.26	0.67
SD	20.96	11.47	20.11	10.68	4.68	14.14	0.00	1.57	31.10	2.64	0.32	0.93
Min	24.00	12.00	26.00	20.00	34.00	25.00	0.00	0.00	0.00	0.00	0.00	0.00
Max	90.00	37.00	75.00	48.00	48.00	60.00	0.00	3.90	76.18	6.80	0.79	2.34
P		0.60			0.25			0.92			0.57	

Table 3. Mean, standard deviation, minimum (Min), and maximum (Max) buffer strip widths and bank stability indices. Buffer strips were measured from the stream ordinary high water mark perpendicular to the stream out to the buffer strip edge. Bank, vegetation and soil stability were rated from 1-4. Higher ratings imply higher stability and less soil erosion. Non-parametric Kruskal-Wallis tests were used to detect treatment effects on the stability indices.

Sites	Right buffer width (m)		Left buffer width (m)		Bank vegetation stability			Valley vegetation stability			Bank soil alteration		
	1995	1996	1995	1996	1993	1995	1996	1993	1995	1996	1993	1995	1996
Control													
Mean					3.17	3.30	2.13	4.00	4.00	3.57	3.12	3.20	2.07
SD					0.59	0.84	0.97	0.00	0.00	0.54	0.65	0.83	0.56
Min					2.60	1.60	1.40	4.00	4.00	2.80	2.40	1.60	1.40
Max					4.00	3.80	4.00	4.00	4.00	4.00	4.00	4.00	2.80
State													
Mean	12.91	16.78	14.46	17.42	3.42	2.67	2.62	3.57	2.57	2.90	3.28	2.30	2.58
SD	6.06	7.79	6.21	8.09	0.34	0.72	0.51	0.46	0.71	0.72	0.33	0.74	0.35
Min	5.38	8.92	6.00	8.32	3.00	1.80	2.00	3.00	1.60	2.00	2.80	1.60	2.00
Max	22.38	28.58	24.60	27.76	4.00	3.80	3.20	4.00	3.60	4.00	3.80	3.40	3.00
Modified													
Mean	28.20	29.08	30.03	34.75	3.60	3.57	2.67	3.93	3.47	3.47	3.60	3.33	2.50
SD	5.89	7.73	12.94	13.88	0.358	0.59	0.45	0.10	0.41	0.55	0.34	0.53	0.37
Min	21.80	22.60	17.20	20.70	3.00	2.40	2.20	3.80	2.80	2.80	3.20	2.40	2.00
Max	38.40	43.34	52.20	57.82	4.00	4.00	3.40	4.00	4.00	4.00	4.00	3.80	3.00
P		0.02		0.03		0.71			0.32			0.25	

Table 4. Mean, standard deviation, minimum (Min), and maximum (Max) percent stream and bankslope gradients summarized by year and treatment type. Gradients were measured by sighting over a 10m distance with a clinometer at five points along the stream. P-values indicate probability of a bias in stream gradient among treatment types. Percentages were arcsine transformed before performing the ANOVA on 1993 data.

Sites	Stream gradient (%)			Right bankslope gradient (%)			Left Bankslope Gradient (%)		
	1993	1995	1996	1993	1995	1996	1993	1995	1996
Controls									
Mean	7.59	5.30	4.35	25.18	23.02	17.63	22.55	21.57	15.18
SD	3.78	3.51	2.80	8.70	12.29	10.55	10.27	11.99	9.51
Min	3.80	2.00	0.80	12.70	12.00	6.60	9.00	5.40	5.60
Max	13.80	11.80	8.30	35.40	44.40	33.00	35.40	34.60	26.80
State									
Mean	11.45	10.97	6.42	22.37	33.47	19.17	31.50	31.40	20.77
SD	10.11	8.40	3.38	16.303	15.01	9.96	16.87	13.29	11.44
Min	2.40	1.80	1.60	3.20	10.60	5.60	10.60	7.40	6.00
Max	26.40	25.80	11.40	44.00	54.80	32.00	48.60	43.60	36.8
Modified									
Mean	7.10	5.70	5.48	14.67	21.91	17.04	15.93	23.09	19.53
SD	4.46	3.80	3.04	7.00	14.16	11.63	6.49	12.24	12.08
Min	2.40	2.60	2.20	8.00	5.00	6.00	5.60	11.80	10.20
Max	14.60	12.40	10.60	26.60	44.75	36.40	23.00	45.75	42.20
P	0.64			0.32			0.15		

Table 5. Mean, standard deviation (SD), minimum (Min), maximum (Max), and total volumes (m^3) of Large Organic Debris (LOD) in streams summarized by year and treatment type. Location of LOD described by "A", completely within stream channel; "B", >50% of length in stream channel; "C", <50% of length in stream channel; "D", completely outside of stream channel, usually suspended above. ***P-values*** were calculated using an ANOVA to test the null hypothesis of no difference among treatment types in volumes of LOD.

Sites	LOD "A"			LOD "B"			LOD "C"			LOD "D"		
	1993	**1995**	**1996**	**1993**	**1995**	**1996**	**1993**	**1995**	**1996**	**1993**	**1995**	**1996**
Control												
Mean	0.12	5.12	12.92	6.76	10.39	10.18	60.8	12.93	15.61	7.65	3.11	10.35
SD	1.22	3.15	12.44	6.46	9.88	9.65	135.27	13.16	17.37	8.47	5.45	13.25
Min	0.00	0.26	1.21	0.00	0.05	0.00	0.00	0.26	0.00	0.00	0.00	0.90
Max	0.55	8.76	34.92	14.38	22.14	23.90	336.49	25.39	39.38	24.03	13.96	36.22
Total	0.72	30.69	77.52	40.54	62.36	61.10	364.81	77.57	93.67	45.90	18.66	62.12
State												
Mean	2.09	9.98	14.02	36.41	17.09	33.41	22.36	22.82	38.36	22.55	12.96	62.20
SD	2.73	10.00	20.59	73.82	17.17	65.44	20.55	8.12	59.77	21.59	9.72	103.88
Min	0.00	0.00	0.00	0.79	0.29	0.54	3.09	12.30	2.26	1.64	0.52	0.53
Max	6.50	27.82	53.69	186.86	49.48	166.31	53.38	31.47	156.61	63.52	28.02	271.28
Total	12.52	59.89	84.14	218.48	102.55	200.46	134.13	136.9	230.16	135.3	77.78	373.21
Modified												
Mean	0.50	18.81	20.69	16.27	6.49	9.64	3.35	8.19	13.26	7.47	4.99	14.12
SD	0.86	20.44	32.23	19.75	5.41	14.83	5.29	5.41	11.33	5.91	4.17	10.59
Min	0.00	0.83	3.45	0.92	0.24	0.72	0.71	3.31	1.69	0.86	0.69	1.92
Max	2.21	53.66	85.61	52.42	16.04	39.58	14.13	18.41	31.88	17.03	10.69	28.60
Total	3.02	112.85	124.14	97.61	38.93	57.86	20.09	49.13	79.53	44.81	29.97	84.71
P		0.55			0.62			0.37			0.75	

Table 6. Total captures of stream-breeding amphibians, fishes and crayfish by year and treatment type. Four streams of each treatment type were surveyed in 1992. Six streams of each treatment type were surveyed in each of the following years.

Sites	Tailed frogs (adults & tadpoles)				Pacific giant salamanders				Cope's giant salamanders				Columbia torrent salamanders			
	1992	1993	1995	1996	1992	1993	1995	1996	1992	1993	1995	1996	1992	1993	1995	1996
Controls	6	105	133	225	2	42	68	56	0	0	0	7	0	1	0	8
Modified	34	74	113	92	13	33	42	52	0	1	2	0	0	0	0	0
State	5	0	23	2	19	49	126	113	0	0	0	0	0	0	0	0

Sites	Salmonids				Sculpins				Crayfish				Lamprey			
	1992	1993	1995	1996	1992	1993	1995	1996	1992	1993	1995	1996	1992	1993	1995	1996
Controls	9	11	19	21	9	12	8	8	127	308	174	99	0	0	0	0
Modified	3	11	31	25	11	12	21	39	53	178	163	201	0	1	0	2
State	2	4	3	33	1	23	9	21	79	204	205	235	0	0	0	0

Table 7. Size and counts of all tailed frogs captured at all sites during 4 yr of sampling. Adult frogs ≥35 mm; juvenile frogs <35 mm. Tadpole developmental stages: Stage 1 = Hatchling; Stage 2 = Mature; Stage 3 = Hind leg development; Stage 4 = hind and front leg development.

	Adult	Juvenile	Stage 1	Stage 2	Stage 3	Stage4
Mean	40.55	21.13	22.33	43.53	46.88	44.20
SD	3.953	3.030	3.214	4.019	4.001	5.754
Min	35	17	20	30	28	23
Max	50	28	26	55	57	61
Count	20	16	3	401	213	160

Table 8. Densities of second year tailed frog tadpoles by site, treatment type and year. Differences were not significantly different among treatment types (P = 0.854).

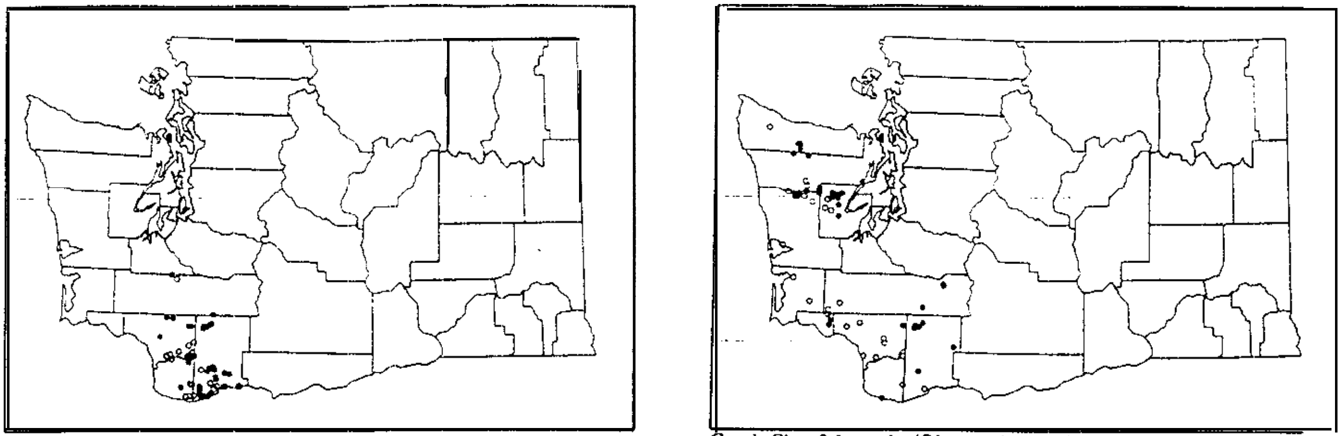
Sites	Tadpole density (#/m^2)			
	1992	1993	1995	1996
Controls				
Abernathy Cr		0.00	0.00	0.02
Elbe Hills		0.84	0.29	0.43
Hotel Cr	0.00	0.03	0.09	0.04
Porter Cr	0.00	0.00	0.00	0.00
Taylor Cr	0.04	0.51	0.58	0.21
Vail	0.06	0.14	0.39	0.37
Mean	0.02	0.25	0.22	0.18
SD	0.03	0.35	0.24	0.10
Modified				
Blue Tick		0.00	0.00	0.00
Eleven Cr 31	2.00	1.24	2.77	0.52
Griffin Cr	0.08	0.61	0.05	0.00
Ms. Black	0.00	0.00	0.00	0.00
Ryderwood 860	0.00	0.03	0.00	0.00
Side Rod		0.00	0.00	0.00
Mean	0.52	0.31	0.47	0.09
SD	0.99	0.51	1.13	0.21
State				
Eleven Cr 32	0.00	0.00	0.00	0.00
Kapowsin		0.00	0.00	0.00
Night Dancer		0.00	0.00	0.00
Pot Pourri	0.00	0.00	0.00	0.00
Rydenvood 1557	0.00	0.00	0.00	0.00
Simmons Cr	0.16	0.00	0.16	0.00
Mean	0.04	0.00	0.03	0.00
SD	0.08	0.00	0.07	0.00

Table 9. Densities of Pacific giant salamander larvae by site, treatment type, and year. Differences were not significantly different among treatment types ($P = 0.507$).

Sites	Salamander larva density (#/m^2)			
	1992	1993	1995	1996
Controls				
Abernathy Cr		0.42	0.24	0.31
Elbe Hills		0.23	0.17	0.29
Hotel Cr	0.00	0.07	0.27	0.21
Porter Cr	0.00	0.00	0.00	0.00
Taylor Cr	0.00	0.10	0.04	0.04
Vail	0.11	0.20	1.39	0.77
Mean	0.03	0.17	0.35	0.27
SD	0.06	0.15	0.52	0.28
Modified				
Blue Tick		0.00	0.00	0.00
Eleven Cr 31	0.07	0.25	0.12	0.34
Griffin Cr	0.08	0.03	0.37	0.53
Ms. Black	0.00	0.00	0.00	0.00
Ryderwood 860	0.46	0.49	0.49	0.15
Side Rod		0.26	0.34	0.38
Mean	0.15	0.18	0.22	0.23
SD	0.21	0.21	0.21	0.22
State				
Eleven Cr 32	0.48	0.17	0.66	1.64
Kapowsin		0.89	2.36	1.36
Night Dancer		0.00	0.00	0.00
Pot Pourri	0.00	0.00	0.00	0.00
Ryderwood 1557	0.47	0.14	0.21	0.31
Simmons Cr	0.26	0.11	0.19	0.16
Mean	0.30	0.22	0.57	0.58
SD	0.23	0.34	0.91 0	0.72 9

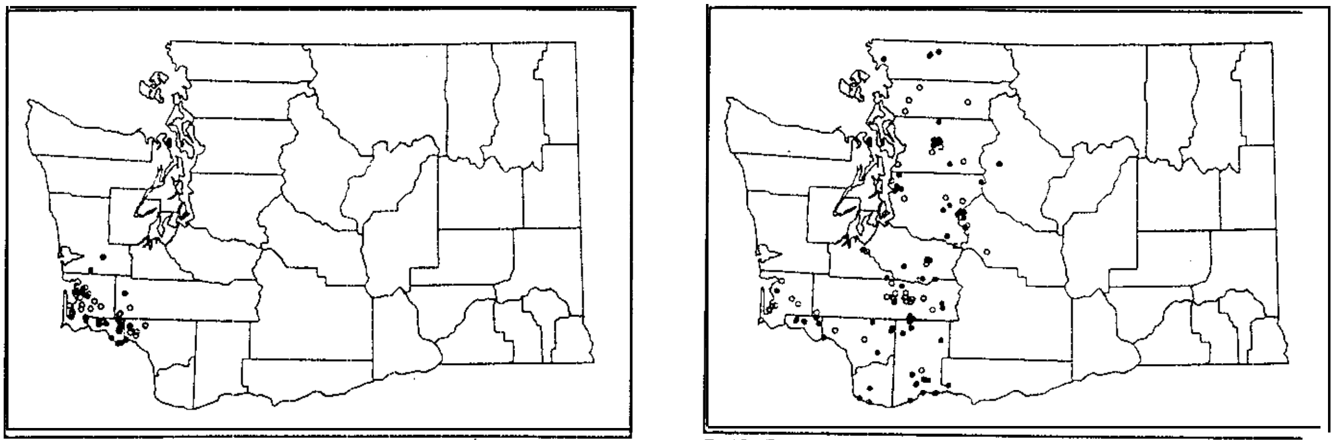
Table 10. Mean snout-vent length (snout to anterior tip of vent) of Pacific giant salamander larvae and paedomorphs by site, treatment type and year. SVL not recorded in 1992. Differences were not significantly different among treatment types (*P* = 0.469).

Sites	Snout-vent length (mm)		
	1993	1995	1996
Controls			
Abernathy Cr	44.1	41.1	33.4
Elbe Hills	57.1	68.6	64.4
Hotel Cr	32.0	37.0	55.2
Taylor Cr	50.0	44.5	32.3
Vail	41.8	34.6	35.6
Mean	45.0	45.2	44.2
SD	9.37	13.64	14.65
Modified			
Eleven Cr 31	50.0	51.5	32.2
Griffin Cr	100.0	52.9	54.9
Ryderwood 860	39.4	43.0	42.2
Side Rod	42.3	48.6	63.2
Mean	57.9	49.0	48.1
SD	28.41	4.37	13.69
State			
Eleven Cr 32	34.0	38.X	42.6
Kapowsin	50.3	52.0	54.2
Ryderwood 1557	28.5	37.43	35.25
Simmons Cr	52.5	56.8	53.1
Mean	41.3	49.2	50.0
SD	11.88	9.33	6.44



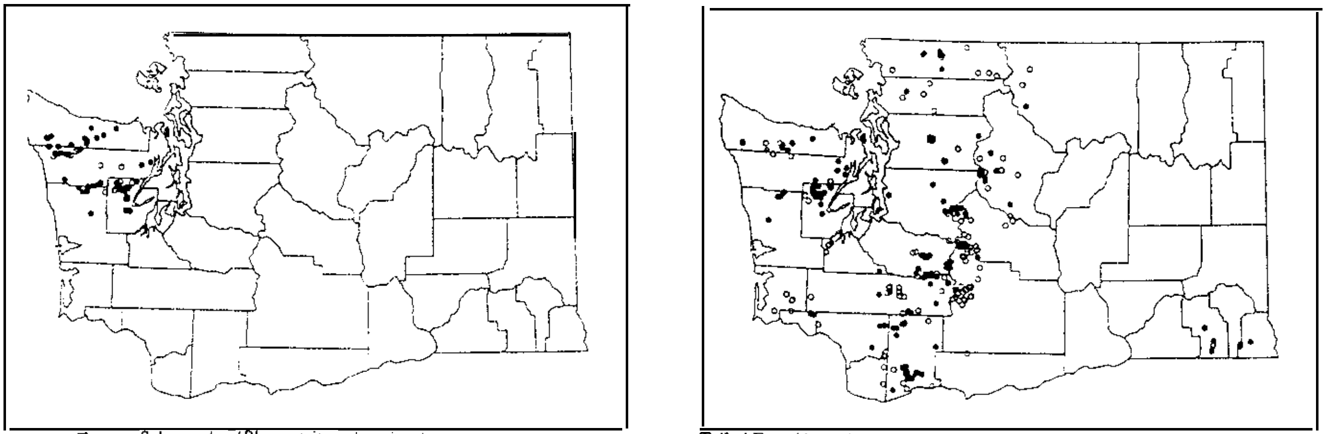
Cascade Torrent Salamander *(Rhyacotriton cascadae)*

Cope's Giant Salamander *(Dicamptodon copei)*



Columbia Torrent Salamander *(Rhyacotriton kezeri)*

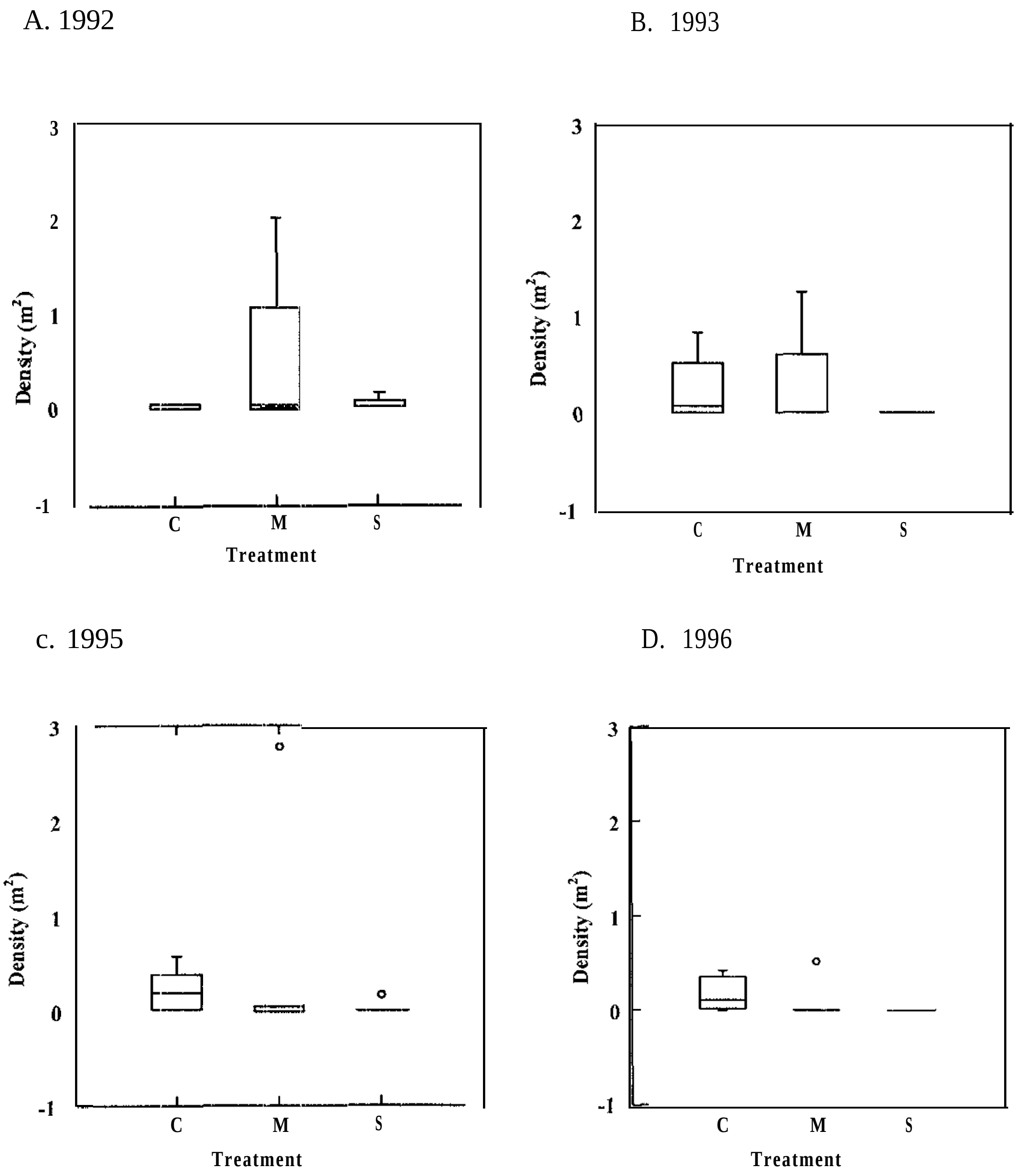
Pacific Giant Salamander *(Dicamptodon tenebrosus)*



ympic Torrent Salamander *(Rhyacotriton olympicus)*

Tailed Frog *(Ascaphus truei)*

Figure 1. Range maps showing distribution of six native, stream-breeding amphibian species. Dark circles represent locations with existing voucher specimens. Open circles represent locations of sightings without voucher specimens. Range maps from McAllister (1995).



A. 1992

B. 1993

c. 1995

D. 1996

Figure 2. Density of second year tailed hog tadpoles in streams on Control (C), Modified (M), and State (S) sites. Surveys in 1992 and 1993 occurred before timber harvest; 1995 and 1996 sampling occurred after timber harvest. Six sites were sampled in each treatment type. Box plots show median (center horizontal line), 2nd and 3rd quartiles around the median, and whiskers that extend to 1.5 times the 2nd and 3rd quartile range. Outliers are shown as an asterisk or open circle.

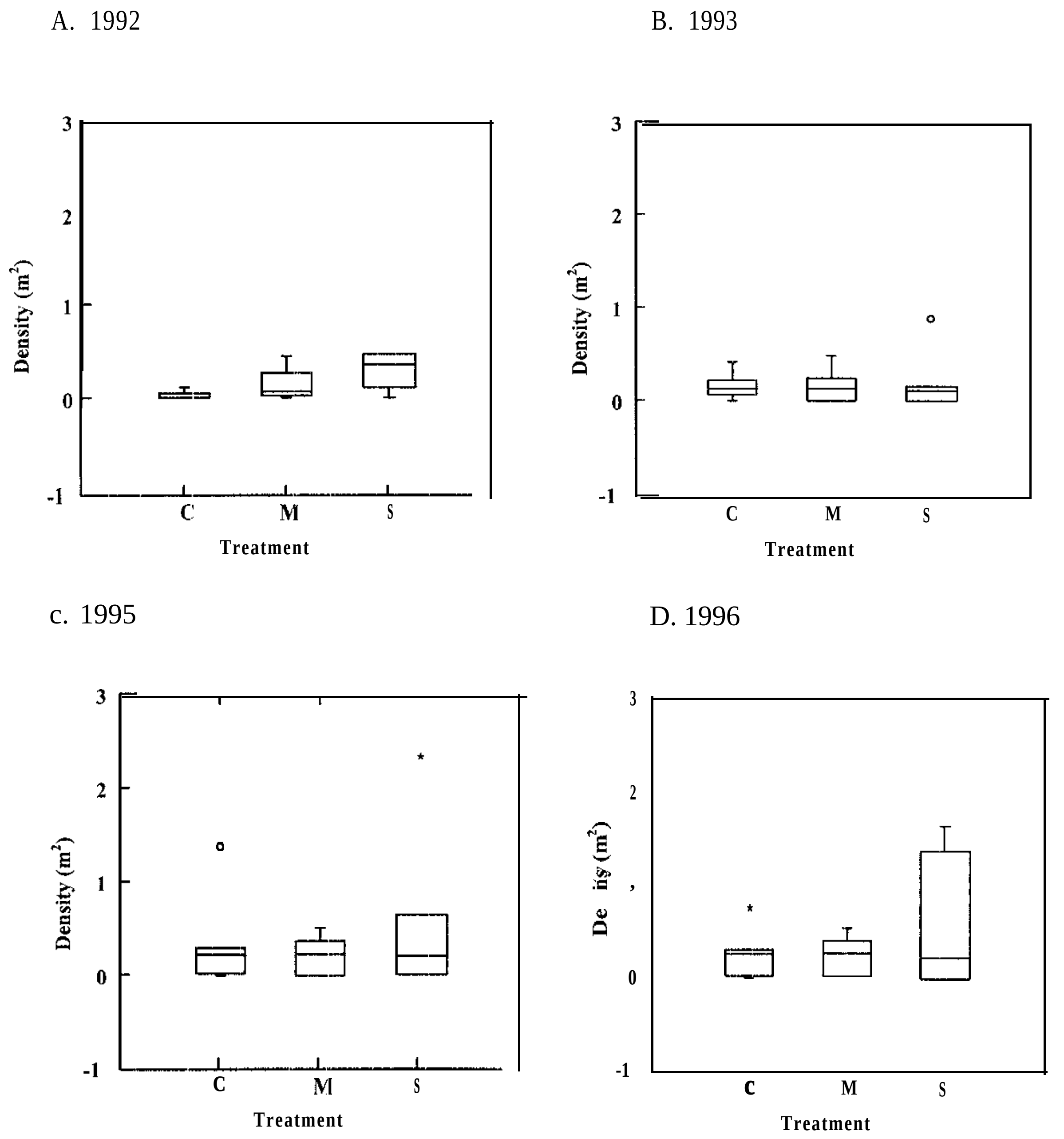


Figure 3. Density of giant salamander larvae in streams on Control (C), Modified (M), and State (S) sites. Surveys in 1992 and 1993 occurred before timber harvesting; 1995 and 1996 occurred after timber harvest. Six sites were sampled in each treatment type. Box plots show median (center horizontal line), 2nd and 3rd quartiles around the median, and whiskers that extend to 1.5 times the 2nd and 3rd quartile range. Outliers are shown as an asterisk or open circle.

Chapter 8

WEST-SIDE TERRESTRIAL AMPHIBIAN SURVEYS

Abstract. We surveyed terrestrial amphibians using pitfall traps in riparian areas and upland forests approximately 100 m from the stream, 2 yr prior and 2 yr following clearcut logging. Captures within two different buffer strip configurations were compared with riparian forest. State regulatory buffers reflected minimum riparian buffer strips required by state law. Modified buffer strips were designed to incorporate more snags, damaged trees, seepy areas and other wildlife habitat. Amphibians were captured in October for a 28-d period in all 4 sampling years. Pre-treatment data were used to compare capture rates between riparian and upland habitats. Pre- and post-treatment data were used to test the null hypothesis of no difference in capture rates among riparian forest, modified buffer strips, and state regulatory buffer strips following clearcut logging in the uplands. Six species were captured frequently enough to perform statistical analyses. These were *Ensatina eschscholtzii, Plethodon vehiculum, Ascaphus truei, Dicamptodon tenebrosus, Rana aurora,* and *Ambystoma gracile.* A total of 607 amphibians of 12 species were captured during pre-treatment sampling *years*. *Ensatina* and *Ascaphus truei* were found in significantly greater numbers in upland forests than in riparian forests. No species were captured significantly more frequently in riparian forests when compared to uplands. Species richness did not differ significantly between riparian and upland forests. A total of 893 amphibians of 13 species were captured during post-treatment sampling years. Captures of *Ensatina* salamanders on both riparian and upland transects were significantly lower at Modified sites than at Control and State sites. Captures of other species did not differ significantly among treatment types following clearcut logging. The results suggest that riparian buffer strip configurations currently implemented in western Washington provide adequate habitat to maintain amphibian populations for 2 yr following timber harvest, Sample sizes for all but two species, *Ensatina eschscholtzii* and *Plethodon vehiculum,* were very small and variances between sites may have obscured possible treatment effects.

INTRODUCTION

Stream and riparian habitat provide distinct ecological conditions, including moist refugia, developed understories and perennial water sources, which adjacent upland forests may lack. These conditions functionally produce idea1 habitat for many amphibian species in western Washington. Understory vegetation, down wood, and rock provide cover while moist soil conditions create suitable refugia from extreme temperatures during summer months Perennial streams provide water throughout the summer when many ponds dry or become choked with algae and emergent vegetation. Stream invertebrates with aquatic development stages emerge as a food source for streamside amphibians. Streams also connect ponds and drainages, offering cool, moist corridors for movement.

Pacific Northwest amphibians have adapted to the moist conditions of regional forests and the interconnectivity provided by riparian areas. Several studies have examined differences in relative abundance in different forest habitat. The USDA Forest Service's Old-Growth Wildlife Habitat Research Program (Ruggiero et al. 1991) compared relative abundance of amphibians in three age classes of unmanaged forests in western Washington, Oregon and northwestern California. McComb et al. (1993a, 1993b) examined relative abundance of riparian and upland amphibian communities in second-growth forests in western Oregon. Donoghue-Stanton (1994) compared riparian and upland amphibian abundance between old-growth forests and clearcut areas with and without buffer strips in southwestern Oregon. Aubry (1998) compared relative abundance of terrestrial amphibians in four structural classes of second-growth managed forests in western Washington, Of these studies, only the Oregon research compared riparian and upland amphibian communities.

The following work is unique in that it compares relative abundance of terrestrial amphibians in riparian and upland habitats before and after timber harvest. Specifically, we examine changes in amphibian community assemblage and relative abundance within riparian buffer strips. The objective of the study was to determine whether amphibian communities are maintained within buffer strips following timber harvest of adjacent forests.

METHODS

Sampling design

Amphibian sampling occurred after the onset of fall rains in all sampling years. This generally occurred during the month of October. Pre-treatment sampling was performed at 13 sites in 1992 and 18 sites in 1993. All but one site were harvested during 1994. The missing site, Ryderwood 1557, was harvested in 1996. Post-treatment sampling at 17 sites occurred in 1995 and 1996. Ryderwood 1557 was subsequently sampled during 1997 and 1998.

Terrestrial amphibian sampling methods were based on protocols developed during the USDA Forest Service's Old-Growth Wildlife Habitat Research Program (Ruggiero et al, 1991). We used pitfall traps constructed according to methods described by Corn and Bury (1990). Field personnel placed 18 traps at 15 m intervals parallel to the stream in the riparian zone and in the adjacent upland forest. Riparian traps were placed within 5 m of the stream's ordinary high water mark. Approximately 100 m separated the riparian and upland transects. Pitfall traps ran along the same transect as small mammal snap traps in the riparian transect and as bird point count stations and small mammal snap traps in the upland transect. Each trap contained an inch of water to assure that amphibians would not desiccate. Traps remained open for 28 nights (4 weeks) and were checked every 5 to 7 d. Live amphibians were identified, measured, held and then released at the capture site at the close of the trapping period. Individuals that died in traps were prepared as museum specimens for the Burke Museum collection. Measurements taken from all amphibians included total length, snout-vent length, and weight.

Statistical analyses

Number of species, mean captures per 100 trap nights, and indices characterizing species-habitat relationships were calculated for riparian and upland habitats at all study sites prior to timber harvest and following timber harvest. We examined species richness in riparian and upland habitats by comparing total numbers of amphibian species using only pre-treatment capture results. Comparisons of species richness among treatment types were calculated using only post-treatment capture data. We used paired t-tests to test the null hypothesis of no difference in species richness between riparian and upland transects. Analyses of treatment effects were performed to test the null hypothesis of no difference in mean captures among treatment types. We tested this hypothesis using a single-factor ANOVA on the difference in mean captures between post- and pm-treatment periods (mean post-treatment captures minus mean pre-treatment captures) at both riparian and upland transects for each site. We used Tukey multiple comparison tests to identify significant differences among treatment types. Snout-vent length of salamanders and tailed frogs were compared using a single-factor ANOVA with 2nd year, post-treatment data. Habitat associations were evaluated using hierarchical clustering of capture data and stepwise linear regression techniques. Habitat variables measured as percentage cover were arcsine transformed. All statistical tests were performed using the statistical tools in Microsoft Excel (1994) and SYSTAT (1989) software packages.

RESULTS

Riparian and upland habitat comparisons

Species richness

We captured 11 amphibian species in riparian traps and 10 amphibian species in upland traps during the pre-treatment sampling period (Table 1). Mean species ($\bar{x}$ ± SD) numbers at each site did not differ significantly between riparian (4.1 ± 1.60 species) and upland habitats (3.5 ± 1.15 species, $P = 0.119$; Fig. 1). Mean species richness did not differ significantly among assigned treatment types on riparian (Fig. 1; $P = 0.348$) or upland transects (Fig. 1; $P = 0.209$).

Relative abundance

We captured a total of 607 amphibians during the pre-treatment sampling period. Ensatina *(Ensatina eschscholtzii*) and western redback salamanders *(Plethodon vehiculum*) accounted for 420 of the 607 amphibians captured (69%, Table 1). Total captures of northwestern salamanders *(Ambystoma gracile*), Pacific giant salamanders *(Dicamptodon tenebrosus*), tailed frogs (*Ascaphus truei*), and red-legged *frogs (Rana aurora)* ranged from 17 to 59 individuals and constituted 28% of the overall captures. Rarely captured species (<3% of total captures) included Dunn's salamander *(Plethodon dunni),* long-toed salamanders *(A. macrodactylum),* torrent salamanders *(Rhyacotriton cascadae, R. kezeri),* Cascade frogs *(Rana cascadae)* and Pacific tree frogs *(Hyla regilla).* Only a third of the individuals captured (231, 38%) were found in riparian traps, a result driven by the large proportion of *Ensatina* salamanders captured in upland transects (Table 1, $P < 0.001$). Tailed frog captures were also significantly greater in upland transects than riparian transects (Table 1, $P = 0.053$) but accounted for only 10% of total upland captures. Differences in captures of other species were not significant, or were untestable due to small sample sizes (<10 individuals, Table I).

Habitat relationships

Hierarchical cluster analyses revealed no obvious patterns in the overall clustering of sites (Fig. 2, riparian, and Fig. 3, upland). Neither geographical location nor assigned treatment type appeared to explain the groupings. Both riparian and upland analyses, however, grouped two sites located on streams that joined less than lkm below the study areas (Pot Pourri, STA4, and. Porter Creek, CON4). Of all 18 study sites, Pot Pourri and Porter Cr. were in closest proximity.

Results of multi-factor linear regressions of pre-treatment surveys showed several significant correlations of ground cover variables and species abundance. Forest mid- and overstory variables were not included in this analysis because of the lack of strong associations with habitat variables in similar studies (Aubry 1998, Corn and Bury 1991, Gilbert and Allwine 1991). The percentage of ground covered with down wood, litter and depth of litter were the most useful variables in predicting abundance of *Ensatina* and western redback salamanders (Tables *2* and *3). Ensatina* salamanders were negatively associated with the amount of rock cover and positively associated with the amount of litter cover and depth. Western redback salamanders were positively associated with litter and coarse woody debris cover and negatively associated with lichen cover. Stream-breeding amphibians, tailed frog and Pacific giant salamander, were found more frequently at sites with more rock, bare soil, and less litter cover. Pond-breeding amphibians, northwestern salamander and red-legged frog, were positively correlated with fern and moss cover and negatively correlated with coarse woody debris and bare soil.

Treatment comparisons

Species richness

The mean number of amphibian species did not differ significantly among the three treatments (Fig. 4, ($P > 0.50$). Riparian transects on Modified sites had a slightly higher average (±1 SD) species richness than those on State and Control sites (3.5 ± 1.1; 3.2 ± 0.8; 3.3 ± 1.4, respectively). Along upland transects, fewer species were found on State (3.5 ± 1.0) and Modified (3.5 ± 1.2) sites than on Control sites (4.7 ± 1.4). The total number of riparian species captured declined with buffer strip area. Nine species were captured in control sites, eight in buffers on Modified sites, and seven in buffers on State sites following timber harvest (Table 4). We captured 10 species in the uplands of Control sites and 11 in upland clearcut areas (Table 5). Six species were captured in clearcut areas on Modified sites and nine were captured in clearcut areas on State sites. The differences, however, were not statistically significant.

Hierarchical cluster analyses revealed no significant patterns with regard to site groupings (Fig. 5 and 6). The analysis did not group riparian and upland species assemblages according to treatment types.

Relative abundance

During the 2 yr of sampling following timber harvest, we captured 893 amphibians (Tables 3 and 4). Once again, roughly one-third of the captures occurred in riparian traps (308) with the remainder in upland transects. Western redback salamanders dominated captures, accounting for nearly 60% of all captures. *Ensatina* salamanders accounted for 22.4% of all amphibians captured. Captures for which sample sizes were large enough to allow statistical analysis included tailed frogs (6.3%), red-legged frogs (3.8%) northwestern salamanders (2.X%), and Pacific giant salamanders (1.9%). We captured very few Dunn's salamanders, roughskin newts, long-toed salamanders, Cascade and Columbia torrent salamanders, Pacific tree frogs, and Cascade frogs (combined 3.8% of all captures).

We observed few changes in relative abundance following clearcut logging. Terrestrial-breeding salamanders showed the greatest change (Fig. 7 and 8). Significant treatment effects were detected in capture rates of *Ensatina* salamanders (riparian: P = 0.019, upland: P = 0.054, Tables 3 and 4). At riparian transects, we found a significant decrease in captures at Modified sites when compared to State and Control sites. At upland transects, significantly more *Ensatina* salamanders were captured at State than Modified sites. All other species showed no statistically significant differences among treatment types in either riparian or upland transects (Fig. 9 and 10). Variance between sites in upland clearcuts was large and possibly obscured any statistically significant differences in numbers of Western redback salamander captured, (Fig. 7).

Although treatment effects were not significant for most species, sample sizes were small and within group variances were high. Post-hoc power analyses revealed a greater than 75% chance of committing a type 2 error for all species except *Ensatina*. *Ensatina* results showed a 45% chance of committing a type 2 error.

Habitat relationships

Responses to timber harvest did not appear to drive species assemblages as shown in hierarchical clusters (Fig. 5 and 6). Sites showed no clear pattern that could be explained by geographic area or treatment type. Amphibian communities in upland clearcut areas did not differ strongly from forested uplands.

Individual species associations with habitat variables changed little (Tables 2 and 3). Litter depth remained strongly correlated with terrestrial-breeding salamander (*Ensatina* and western redback) abundance following timber harvest. Tailed frog captures continued to show negative associations with litter depth and showed a positive association with bare soil and fern cover. Pacific giant salamanders were negatively associated with ferns and positively associated with coarse woody debris. Litter depth and presence of ferns continued to be positively associated with northwestern salamander abundance. Fern cover was also positively associated with red-legged frog captures.

DISCUSSION

Community composition

Differences in amphibian species richness and abundance along riparian and upland transects were slight, both prior to and following timber harvest, suggesting that both upland and riparian areas meet basic habitat requirements for most species and that amphibians move between upland and riparian transects. However, capture rates were low for all species except western redback and *Ensatina* salamanders, making statistical analyses and conclusions difficult.

Abundant rainfall in west-side forests reduces habitat differences between riparian and upland areas (Chapter 3). Amphibian species likely find moist refugia and food resources in both types of forest habitat and, thus, occupy both. Particularly during rainy periods, when fall sampling occurred, riparian buffer, upland forest, and clearcut area microclimates provide moist and cool habitat. Consequently, one might anticipate few differences in species richness immediately following timber harvest.

Amphibian movements

Amphibians are generally considered species of low vagility that move during periods ofprecipitation. Distances covered by various northwest species are not well understood and movements continue to be investigated (e.g., Ovaska 1988, Stringer 1997, Johnson 1999). Amphibians are ectotherms and do not require as much energy as mammals or birds. Consequently, they do not face the same pressures to continually search for food. Spring and fall movements to and from ‘breeding ponds and streams drive the largest amphibian movements. Terrestrial salamander movements are probably the most limited as they do not make breeding migrations. Ovaska (1988) reported movements <2.5 m over an 8-mo period, although the sampling design prevented observations much greater than this.

During the fall trapping period, captures of stream breeders, tailed frogs and Pacific giant salamanders may indicate dispersal of newly metamorphosed individuals, adult movements away from stream breeding areas, or foraging excursions. Captures of stream breeding amphibians in upland traps verify the ability of these species to move >100 m during fall rains. Pond-breeding amphibians captured were most likely moving away from breeding ponds in search of suitable over-wintering habitat. The pond-breeding amphibians moved distances greater than 100 m as there were no ponds within the study site boundaries and pitfall traps were located more than 100 m from site boundaries.

Western redback salamanders

Following timber harvest, capture rates of western redback salamanders in upland clearcuts increased, although not significantly (Fig. 7). The increase does not appear to be a result of an increase in new recruitment as snout-vent length did not differ significantly among treatment types (mean [± SE] SVL in 1996: Controls = 45.3 ± 1.2 mm; Modified =

46.8 ± 1.3 mm; State = 46.2 ± 0.98 mm; $P = 0.660$). It seems unlikely that increases in Western redback salamanders in clearcut areas were due to immigration of salamanders from adjacent forests at least 100 m away. Vertical movements, within the soil column, of terrestrial-breeding salamanders have been well documented in east coast (Test and Bingham 1948, Taub 1961, Burton and Likens 1975) and west coast (Stebbins 1954, Nussbaum et al. 1983) species. As surface conditions become less favorable, individuals move into the soil column through talus, fissures, rodent burrows and root spaces. The additional western redback salamanders found in clearcut areas after timber harvest most likely reflect a difference in the number of surface-active individuals. Grialou et al. (2000) found significantly lower numbers of western redback salamanders on clearcut areas 3-6 yr after harvest when compared to uncut adjacent forest. They speculated that the difference might be due to microclimatic, ground. cover, and soil structural differences. Dupuis et al. (1995) suggest that soil moisture limits activity and densities of western redback salamanders. During periods of high precipitation, soil moisture may reach higher levels in clearcuts because of decreases in overstory interception rates. Certainly other abiotic and biotic factors in addition to soil temperature and moisture influence the number of surface active western redbacks. These factors need to be investigated along with the relationship of surface active individuals to total density within the soil column. Numbers of surface-active individuals may not vary predictably with total density in different habitat types.

Ensatina salamanders

Ensatina salamanders clearly favor well-drained soils away from streams, This result supports previous research where ensatinas were captured at significantly higher numbers in drier sites and away from streams (Aubry and Hall 1991, Gilbert and Allwine 1991, McComb et al, 1993). As discussed above for western redback salamanders, *Ensatina* moves within the soil column and the proportion of surface-active individuals is unknown. It is unclear why riparian and upland captures at State sites would increase while captures at Modified sites would decrease significantly. Particularly in the upland clearcut areas, we expected to find similar patterns regardless of treatment type. Results do not appear to be driven by any single site. Total captures of *Ensatina* salamanders in harvest-age second-growth forests in Washington were significantly greater when compared to pre-canopy forests, and somewhat greater than total captures in clearcut areas (Aubry 1998). Further investigation is warranted to better understand *Ensatina* use of the soil column, total densities and to continue monitoring surface activity in RMZ sites to document whether captures in clearcut areas decrease with time.

Tailed frogs

Tailed frog numbers were significantly greater at upland transects prior to timber harvest than riparian transects. Presumably, all individuals captured in the uplands, moved there from the stream, crossing the riparian transect. These individuals either spend more time moving around in upland areas, thus increasing upland capture probabilities, or move to uplands from tributaries to the study stream as well as from the study stream. These individuals are most likely using upland areas for foraging and juveniles may be dispersing to other streams and drainages.

Following timber harvest, upland capture rates declined. We captured 39 tailed frogs in upland traps following timber harvest; 32 of these captures were from Control sites. Riparian captures remained roughly the same when compared to pre-treatment numbers, Tailed frogs that did not use upland clearcut areas following timber harvest also did not increase their use of riparian buffer strips. They may have moved out of the study site into adjacent forest stands. Snout-vent length of individuals captured throughout the study ranged from 19 to 52 mm and was not significantly different among treatment types during the 2nd year following timber harvest (mean [± SE] SVL: riparian Control 23.7 ± 5.69 mm; riparian Modified 24 ± 1 mm, $P=0.92$; upland Control 32.4 ± 9.2 mm; upland modified buffer 28.0 ± 7.2, $P=0.39$). Very little is known about tailed frog use of terrestrial habitat. Clearly additional studies must be initiated to investigate their movements and use of these areas.

Pacific giant salamanders

Movements of Pacific giant salamanders away from streamside areas appear to be very limited. In 4 yr of trapping, only eight individuals (of 35 total) were captured 100 m away from the stream. Terrestrial Pacific giant salamanders appear to depend primarily on adjacent riparian forest and dispersal distances are limited. Observations by Johnston (1998) support this conclusion. Following 20 radio-tagged individuals, she found the greatest straight-line displacement of only one salamander exceeded 200 m over a 2-mo period. The author did not indicate whether the movement was parallel or perpendicular to the stream. Johnston’s observations suggested very low dispersal probabilities of salamanders, implying that dispersal between streams rarely occurs. For a species with such low vagility, it is important for the riparian buffer strip to continue providing all necessary resources to Pacific giant salamanders. It would be worthwhile to continue sampling riparian transects to determine if buffers continue to provide conditions suitable to terrestrial individuals more than 2 yr following timber harvest.

Northwestern salamanders

Northwestern salamanders move primarily through habitat on their way to and from breeding ponds. As the distance from a breeding pond increases, the salamander density would theoretically decrease. This study did not examine the location of ponds in relationship of study sites. Consequently. capture rates were low, as ponds were more than 100 m from any pitfall trap (outside the study site boundaries).

We found no significant difference in northwestern salamander captures following clearcut logging. This finding supports results from the TFW West-side Landscape Project where capture rates of northwestern salamanders did not differ significantly between clearcut areas and harvest-age stands (Aubry 1998). Closer inspection of the upland captures in our study show declines following timber harvest. Before treatment, 26 individuals were captured in forested uplands. Following treatment, 10 individuals were captured, 7 from forested sites and 3 from clearcut areas. Numbers of individuals captured was very small throughout the study. Because of this, the variance in the data may obscure any treatment effect. Northwestern salamanders use upland areas during migrations to and from breeding ponds, and Stringer (pers. comm.) has documented movements up to 1 km. Stringer also found a preference for northwestern salamanders to select forested habitat preferentially over clearcut habitat. The suitability of clearcut habitat for northwestern salamanders is not clear at this point.

Red-legged frogs

Riparian buffer strips appear to provide adequate habitat for red-legged frog movements. Aubry (1998) notes that red-legged frogs were negatively associated with landscapes with high amounts of edge habitat. Two years following timber harvest, red-legged frog captures in riparian buffers remained similar to those in riparian forest. Perhaps proximity to the stream influences habitat selection by the frogs more than the clearcut edge. Continued monitoring of red-legged frog use of riparian buffer strips is warranted based on their preference for harvest-forests and apparent avoidance of edge habitats.

Trapping efforts were not sufficient to obtain sample sizes that allowed statistical analyses for more than six species. Other studies using pitfall traps to capture amphibians place arrays of 36 traps within one study site. We utilized this same total number of traps but divided them for riparian and upland transects. This reduced the sample sizes and limited our ability to make comparisons. Pitfall transect length was limited to the size of the study site. Moreover, we were hesitant to trap riparian amphibians intensively, particularly if riparian communities proved to be very distinct from upland amphibian communities.

Habitat relationships

Cluster analyses indicate that results were not biased by geographical trends in terrestrial amphibian abundance or in assignment of treatment types (Fig. 2, 3, 5, and 6). Amphibian species with very localized distributions that could influence this type of analysis were captured in such low numbers that effects were negligible (e.g. *Plethodon dunni, Rhyacotriton cascadae, Rhyacotriton kezeri).* Cluster analyses did not group all five Capitol State Forest sites together, as one might expect. The two that were grouped, Pot **Pourri** and Porter Creek (STA4 and CON4, respectively, in Fig. 2, 3, 5, 6), were located within the same drainage. Both sites were located on different streams that joined approximately 1 km below the study areas. Porter Creek was a Control site and Pot Pourri was a State site.

Litter, herbs, ferns, and moss dominated riparian and upland ground cover. These habitat features probably influenced amphibian species distribution more than lichens, grass, bare soil, and rock, which accounted for <5% of ground cover measured. Availability of cover sites appears to have the greatest influence on terrestrial amphibian activity and captures, Terrestrial-breeding species (western redback and ensatina salamanders) which presumably move very little during their lifetimes and spend much of the time in underground refugia, showed strong associations with greater cover and depth of forest litter and down wood (Table 2). Increases in litter cover and depth following timber harvest may explain the increase in captures of western redback salamanders at treatment sites. The increase in litter depth may create a more favorable microclimate at the surface causing an increase in surface activity. Talus areas also provide cover and refugia for terrestrial salamanders (Herrington 1988, Nussbaum et al. 1983, Bury et al. 1991). The negative associations with rock cover in this study probably indicate the preponderance of rocks closely associated with the stream and riparian areas and the lack of rocky, talus areas in upland habitats of these study sites.

Pacific giant salamander and tailed frogs have very different habitat needs although they both breed in small streams. Pacific giant salamander juveniles and adults utilize terrestrial habitats along the stream and have been found most frequently under rocks, down wood in advanced stages of decay, and underground (Johnston 1998). We found a positive association of Pacific giant salamander captures with down wood (Table 2). Upland captures were too few to perform habitat analyses, underscoring the importance of riparian habitat for Pacific giant salamanders.

Tailed frogs tend to move farther from the stream than Pacific giant salamanders and must rely on terrestrial cover sites to maintain a suitable water balance, thermoregulate and escape predation. Strong correlations with the presence of rock and down wood, and a negative association with bare soil and litter cover suggest that tailed frogs do not use underground refugia, including burrowing under litter, as frequently as they use surface cover objects (Table 2). Increases in litter cover and depth following timber harvest may block tailed frog access to surface cover objects. The strong association of tailed frogs with bare soil indicates their association to streamside areas.

Captures of pond-breeding amphibians (northwestern salamanders and red-legged frogs) were too low to draw strong conclusions. Fern cover appeared to be important for both northwestern salamanders and red-legged frogs. Stringer (1997) reported finding radio-tagged northwestern salamanders commonly burrowing under sword fern (*Polystichum munitum*) clumps. Redlegged frogs also showed negative associations with conventional cover objects such as wood and rocks.

Very little information exists on western Washington's stream- and pond-breeding amphibian use of terrestrial habitat. Sample sizes of tailed frog and Pacific giant salamander captures in managed forest studies were too small for analysis (Aubry 1998) while sampling in unmanaged forests suggests that stream- and pond-breeding amphibians are a common member of the terrestrial fauna (Aubry and Hall 1991). Mean captures of amphibians in pitfall traps in unmanaged forest stands compared similarly to captures in this study. Aubry and Hall (1991) report mean capture rates of approximately 0.20 tailed frogs per 100 trap nights in old-growth stands (210-730 yr old), 0.45 per 100 trap nights in mature stands (80-190 yr old), and 0.05 per 100 trap nights in young stands (55-75 yr old). Mean tailed frog captures on upland transects from this study ranged from 0.16 per 100 trap nights before timber harvest to 0.14 per 100 trap nights following timber harvest. Given the proximity of pitfall traps to fish-bearing streams in this project, we expected capture rates higher than those in upland unmanaged forest stands. The low capture rates of tailed frogs in the managed forest study (Aubry 1998) suggest a negative impact from previous logging history. Few Pacific giant salamanders were captured in pitfall traps at unmanaged forest sites (Aubry and Hall 1991). This supports RMZ project results and the speculation that Pacific giant salamanders tend to stay within the riparian corridor.

Aubry (1998) suggests that amphibian communities respond to topographical, zoogeographical or ecological influences operating at a river basin scale (60,000 to 81,000 ha). Thus, to unravel further the reasons for amphibian presence in specific habitats, biologists must look at microhabitat and microclimatic conditions as well as landscape conditions influenced by geology, glaciation, and historical land use patterns. Studies of terrestrial habitat use by pond-breeding species must account for distances to ponds and other bodies of standing water. The limited capture rate of pond-breeding species in this study did not warrant doing surveys for nearby ponds.

Management recommendations

The similarity in amphibian communities among treatment types underscores several important considerations when addressing management issues, First, amphibian abundance is difficult to measure. Amphibians tend to be active at night during breeding seasons and during seasonal rains. Otherwise they are secretive, often fossorial, and difficult to capture. Sample sizes tend to be low, for example when compared with mammal trapping, and the probability of failing to detect a difference when there is one might be high. Community-level amphibian studies in the Pacific Northwest are difficult because so little is known about basic life history. Small-scale projects that focus on a population of amphibians limit inferences, but addressing population-level questions will create a stronger foundation for interpreting community level responses.

A possible response to having found very little change in riparian amphibian captures following clearcut logging is to conclude that both buffer configurations provide habitat similar to pre-treatment riparian conditions. However, we also found very little change in upland amphibian captures following clearcut logging. Is the logical conclusion that clearcut areas provide suitable habitat similar to second-growth forests, 40-60 yr old? Aubry (1998) found amphibian communities in harvest-age stands to be unique when compared with communities in clearcut areas, pm-canopy and closed-canopy stands. Harvest age stands had the highest species richness and overall abundance than the other forest structure classes. The harvest age amphibian communities most closely resembled amphibian communities in unmanaged forests. It would be premature to conclude that riparian buffers and upland clearcut areas, sampled 2 yr following clearcut logging, provide suitable habitat for amphibians for several reasons.

First, detecting population changes in long-lived species that can withstand extended periods of inclement weather requires sampling over a period of time longer than 2 yr immediately after timber harvest. Mean captures of ensatina and western redback salamanders were lower in pre-canopy, second-growth stands, 12-20 yr after timber harvest than in clearcut areas or harvest age stands (Aubry 1998). It appears that amphibian numbers decline following timber harvest, but it is not clear when the actual drop in numbers occurs, nor if there were ways to harvest that would prevent such a decline.

Second, sample sizes for all species except ensatina and western redback salamanders were small and all species had very large differences in captures between sites. Consequently, the ability to detect a difference in captures at different treatment types was low. Variance between sites was high and could potentially obscure significant differences. For that reason, a more in-depth examination of amphibian use of terrestrial habitat is required to ascertain more precisely important habitat features, dispersal patterns and distances, and reproductive success in various habitat types.

Third, sampling during fall rains does not differentiate between dispersing or migrating individuals and residents. Whether buffers provide suitable movement corridors or over-wintering habitat for stream- and pond-breeding species has not been established.

Of greatest concern are changes in tailed frog and Pacific giant salamander populations following timber harvest along streams. Previous studies have shown significant declines in larval densities in streams in second-growth managed forests (Corn and Bury 1989, Kelsey 1995, Dupuis et al. 1997). Although these declines have been attributed to changes in in-stream habitat due to logging, the influence of terrestrial conditions on juveniles and adults may also play a role. Buffer strip designs need to accommodate tailed frog movements away from streams because clearcut areas appear to restrict adult movements. Intensive sampling along buffer strip edges could reveal the effect of edge habitat on tailed frog use of riparian forest and clearcut habitat. Movements of tailed frogs in forested and buffered areas need to be compared to ascertain how terrestrial individuals are affected by timber harvest. Buffer ship design must also accommodate terrestrial Pacific giant salamanders that appear to limit movements to streamside areas. Currently, radio-telemetry studies of Pacific giant salamander movements are being conducted by a group from the University of British Columbia (Neil1 et al. 1997). Results of these studies may provide needed information on riparian habitat needs of Pacific giant salamanders.

Finally, tailed frog and Pacific giant salamander densities in Type 3 streams, those studied in this project, tend to be lower than in Type 4, not&h-bearing streams where riparian areas do not extend as far away from the stream and timber harvest regulations allow clearcut logging without buffer strips. Habitat protection issues along Type 4 streams are far more contentious and a greater concern for stream-breeding amphibians than along Type 3 streams. It is important to direct research of timber harvest effects on terrestrial as well as aquatic habitat in these areas.

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Table 1. Abundance rankings of terrestrial amphibians based on raw captures in riparian and upland transects during 1992.1993. P-values reported compare riparian and upland captures for each species. P-values in bold denote significant differences. NA indicates species with <10 total captures, which were not compared due to small sample sizes.

Species	Rank	Total captures	Sites	Total riparian captures (%)	Total upland captures (%)	*P*
PLVE	1	240	18	110 (48)	130 (35)	0.856
ENES	2	180	17	28 (12)	152 (40)	**<0.001**
ASTR	3	59	10	22 (10)	37 (10)	0.053
AMGR	4	39	11	13 (6)	26 (7)	0.544
RAAU	5	3.5	9	20 (9)	15 (4)	0.622
DITE	6	21	10	15 (6)	6 (2)	0.22
TAGR	7	1'7	6	10 (4)	7 (2)	0.411
AMMA	8	a	3	7 (3)	1 (<1)	NA
PLDU	9	4	3	4 (2)	0	NA
RHKE	10	2	2	1 (<1)	1 (<1)	NA
RHCA	11	1	1	1 (<1)	*0*	NA
HYRE	12	1	1	0	1 (<1)	NA
Totals		607		231	376	

SPECIES SUMMARY

AMGR	Northwestern Salamander, *Ambystoma gracile*
AMMA	Long-toed Salamander, *Ambystoma macrodactylum*
ASTR	Tailed Frog, *Ascaphus truei*
DITE	Pacific Giant Salamander, *Dicamptodon tenebrosus*
ENES	Ensatina, *Ensatina eschscholtzii*
HYRE	Pacific Tree Frog, *Hyla regilla*
PLVE	Western Redback Salamander, *Plethodon vehiculum*
PLDU	Dunn's Salamander, *Plethodon dunni*
RAAU	Red-legged Frog, *Rana aurora*
RHCA	Cascade Torrent Salamander, *Rhyacotriton cascadae*
RHKE	Columbia Torrent Salamander, *Rhyacotriton kezeri*
TAGR	Roughskin Newt, *Taricha granulose*

Table 2. Pre-treatment results of regression models using habitat parameters to predict amphibian species abundance. Habitat variables used in regression analyses were percentages cover of herbs, ferns, moss, grass, lichen, litter, coarse woody debris (cwd), bare soil, rock, and litter depth (cm).

Species	Riparian habitat variables	*P*	R^2	Upland habitat variables	*P*	R^2
E. eschscholtzii	litter (+)	0.002	0.547	litter depth (+)	< 0.001	0.78
	rock (-)	0.053				
P. vehiculum	litter cover (-t)	<.001	0.833	lichen (-)	0.033	0.76
				cwd (+)	< 0.001	
A. truei	lichen (-)	0.057	0.635	cwd (+)	0.009	0.60
	cwd (+)	0.016		litter depth (-)	0.113	
	bare soil (-)	0.003				
	rock (+)	0.024				
D. tenebrosus	herb (-)	0.043	0.626			
	bare soil (+)	0.001				
A. gracile	fern (+)	0.021	0.696	bare soil (-)	0.062	0.52
	moss (+)	0.012		litter depth (+)	0.002	
	cwd (-)	0.002				
R. aurora	cwd (-)	0.025	0.721	fern (+)	0.005	0.45
	bare soil (+)	0.001		bare soil (-)	0.069	
	rock (-)	0.074				

Table 3. Post-treatment results of regression models using habitat parameters to predict amphibian species abundance. Habitat variables used in regression analyses were percentages cover of herbs, ferns, moss, grass, lichen, litter, coarse woody debris (cwd), bare soil, rock, and litter depth (cm).

Species	Riparian habitat variables	P	R^2	Upland habitat variables	P	R^2
E. eschscholtzii	fern (+)	0.021	0.63	litter depth (+)	<0.001	0.71
	grass (+)	0.069				
	rock (-)	0.123				
P. vehiculum	litter depth (+)	1.001	0.88	herb (-)	0.097	0.79
				litter cover (+)	0.001	
A. truei	bare soil (+)	**0.001**	0.52	fern (+)	0.003	0.57
	rock (-)	0.033		litter depth (-)	0.053	
D. tenebrosus	fern (-)	0.113	0.46	rock (+)	0.030	0.25
	cwd (+)	0.013				
	treatment (+)	0.006	0.37			
A. gracile	fern (+)	0.029	0.68	fern (+)	<0.001	0.61
	cwd (-)	0.005		bare soil (-)	0.081	
	litter depth (+)	0.061				
R. aurora	herb (-)	0.016	0.58	fern (+)	0.007	0.36
	fern (+)	0.002				
	grass (+)	0.119				

Table 4. Abundance rankings of terrestrial amphibians from total riparian captures at different treatment sites during 19951996. P-values reported compare capture differences among treatments with pre-treatment captures taken into account. NA indicates species with <10 total captures, which were not compared due to small sample sizes.

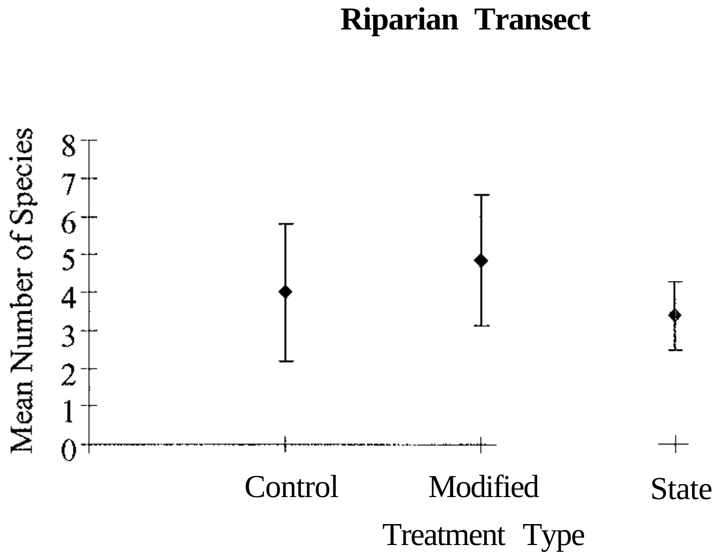
Species	Rank	Total riparian captures	Sites	Control site captures (% of total captures)	Modified site captures (% of total captures)	State site captures (% of total captures)	*P*
PLVE	1	205	18	63(20)	61(20)	81(26)	0.474
ENES	2	30	11	13(4)	5(2)	12(4)	0.019'
RAAU	3	19	7	1(<1)	14(5)	4(1)	0.552
ASTR	4	18	6	9(3)	4(1)	5(2)	0.201
DITE	5	14	7	4(1)	2(1)	8(3)	0.867
AMGR	6	9	7	4(1)	4(1)	1(<1)	NA
PLDU	7	5	1	5(2)	0	0	NA
RHKE	8	3	2	0	2(1)	1(<1)	NA
TAGR	10	2	2	2(1)	0	0	NA
AMMA	10	2	1	0	2(1)	0	NA
HYRE	11	1	1	1(<1)	0	0	NA
Total		308		102(33)	94(3 1)	112(36)	

'Tukey multiple comparison 'test results: Modified < State, *P* = 0.019, Modified < Control, *P* = 0.076.

Table 5. Abundance rankings of terrestrial amphibians based on raw upland captures at different treatment sites during 1995-1996. P-values reported compare capture differences among treatments with pre-treatment captures taken into account. NA indicates species with <10 total captures, which were not compared due to small sample sizes.

Species	Rank	Total upland captures	Sites	Control site captures (% of total captures)	Modified site captures (% of total captures)	State site captures (% of total captures)	*P*
PLVE	1	328	18	69(12)	119(20)	140(24)	0.430
ENES	2	170	18	66(11)	27(5)	77(13)	0.0541
ASTR	3	38	7	32(5)	5(1)	1(<1)	0.288
RAAU	4	15	5	10(2)	5(1)	0	0.125
AMGR	5	10	6	7(1)	0	3(1)	0.391
TAGR	6	9	4	1(<1)	2(<1)	6(1)	NA
PLDU	7	4	2	1(<1)	0	3(1)	NA
DITE	9	3	2	1(<1)	0	2(<1)	NA
AMMA	9	3	1	0	3(1)	0	NA
HYRE	10	2	1	0	0	2(<1)	NA
RHKE	13	1	1	1(<1)	0	**0**	NA
RACAS	13	1	1	1(<1)	0	**0**	NA
RHCA	13	1	1	0	0	1(<1)	NA
Total		585		189	161	235	

[1] Tukey multiple comparison test results: Modified < State, *P* = 0.045, Modified < Control, *P* = 0.524.



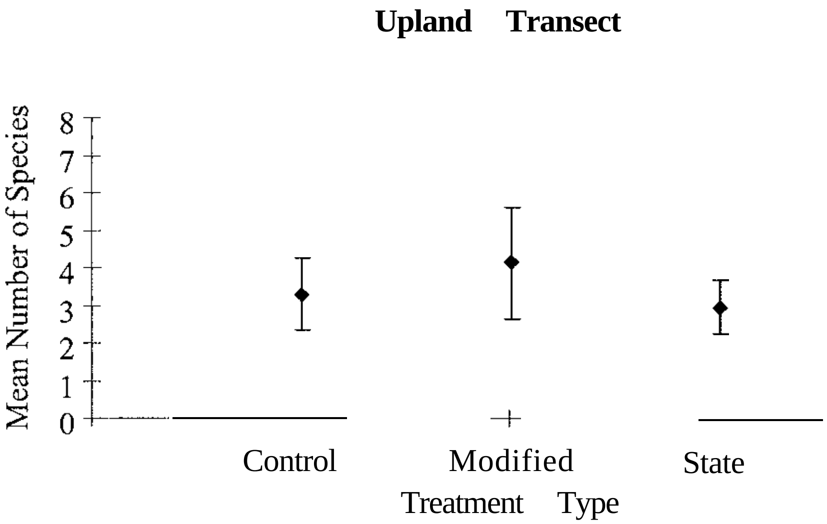


Figure 1. Number of amphibian species (mean ± SD) caught on riparian and upland transects at all treatment types prior to timber harvest. Eleven total species were caught on riparian transects; 10 total species were caught on upland transects. Differences between upland and riparian transects (P = 0.119) and among treatment types were not significant (riparian, P = 0.348; upland, P = 0.209).

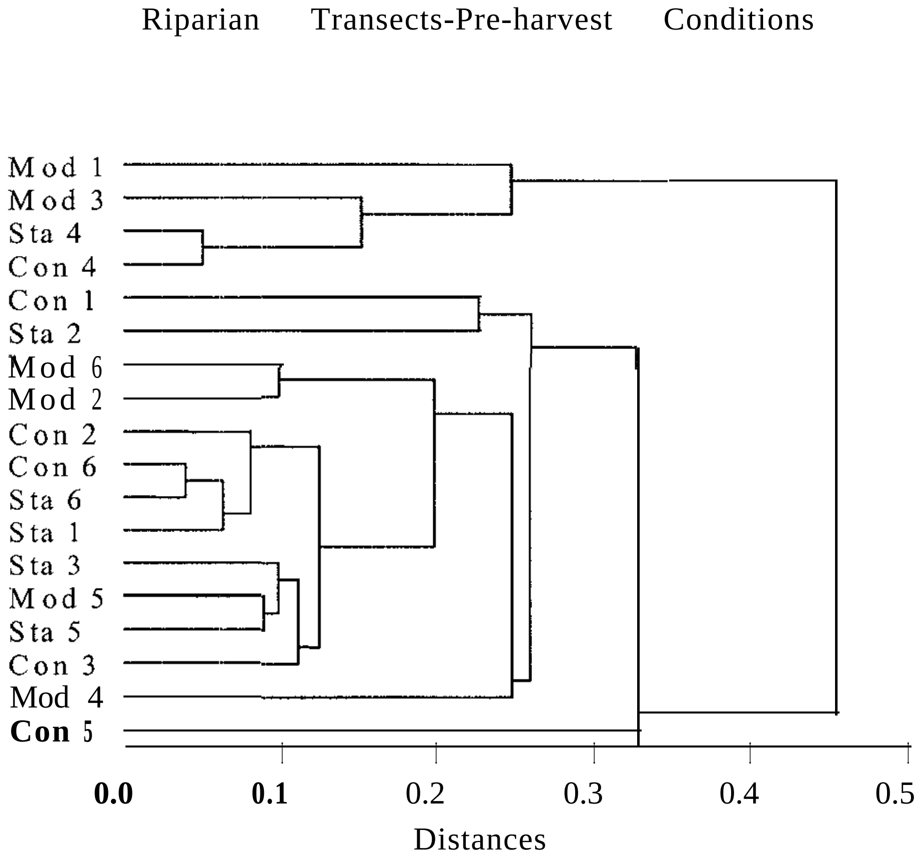


Figure 2. Hierarchical cluster analysis of riparian amphibian community composition prior to timber harvest. Distance is a Euclidean metric. Clustering demonstrates no systematic pattern or differences in riparian amphibian community composition by designated treatment types. CON indicates Control sites; MOD indicates Modified sites; STA indicates State sites. There were six replicates for each treatment type

Con 1 = Abernathy Creek
Con 2 = Elbe Hills
Con 3 = Hotel Cr
Con 4 = Porter Cr
Con 5 = 'Taylor Cr
Con 6 = Vail

Mod 1 = Blue Tick
Mod 2 = Eleven Cr 31
Mod 3 = Griffin Cr
Mod 4 = Ms. Black
Mod 5 = Rydetwood 860
Mod 6 = Side Rod

Sta 1 = Eleven Cr 32
Sta 2 = Kapowsin
Sta 3 = Night Dancer
Sta 4 = Pot Pourri
Sta 5 = Simmons Cr
Sta 6 = Ryderwood 1557

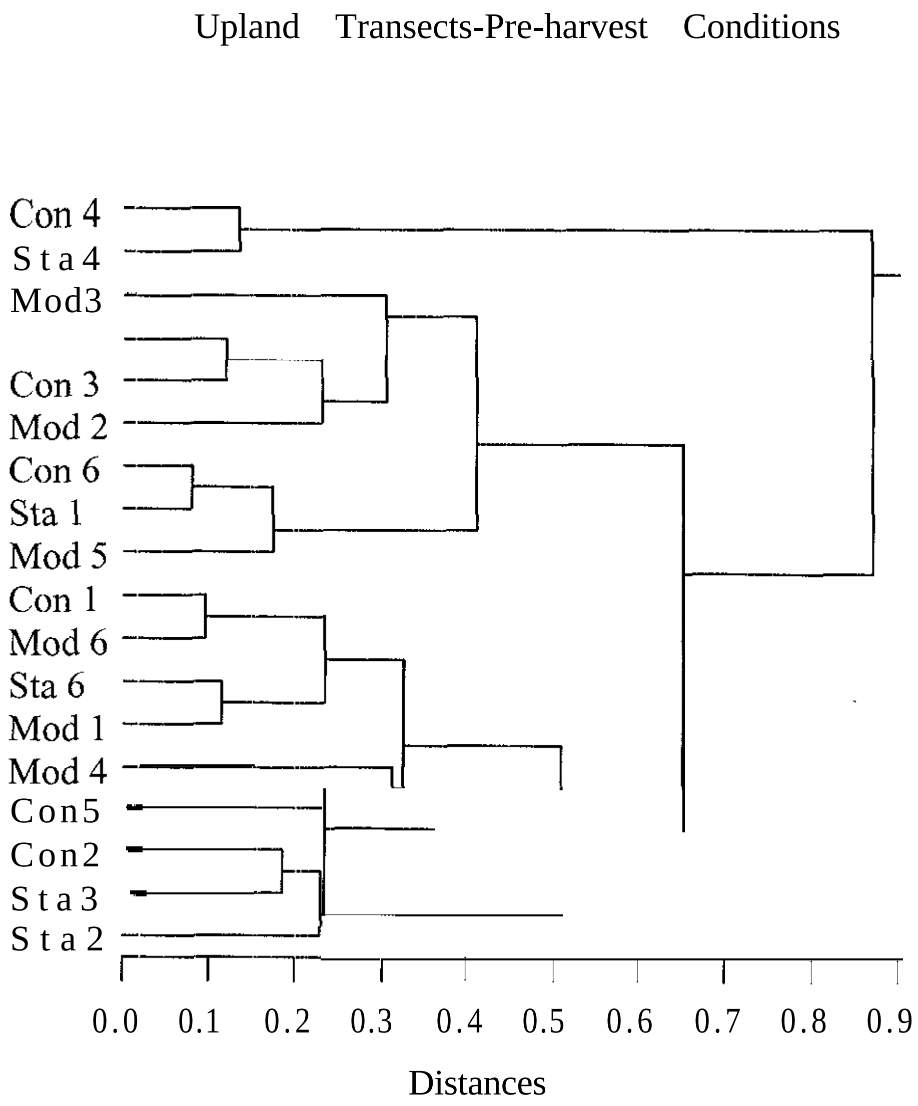


Figure 3. Hierarchical cluster analysis of upland amphibian community composition prior to timber harvest. Distance is a Euclidean metric. Clustering demonstrates no systematic pattern or differences in upland amphibian community composition by designated treatment types. CON indicates Control sites; MOD indicates Modified sites; STA indicates State sites. There were six replicates for each treatment type. See Fig. 2 for key to site designations.

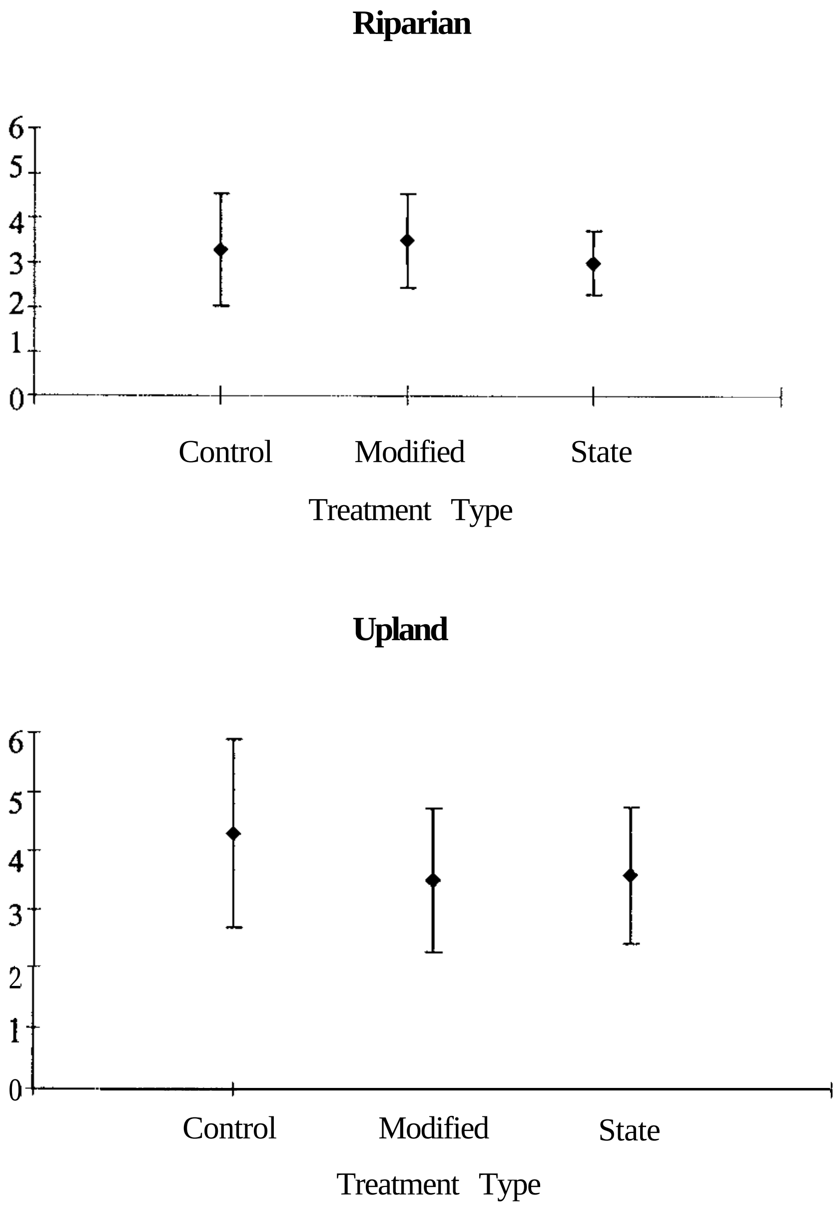


Figure 4. Number of amphibian species (mean ± SD) caught at all treatment types during the two post-harvest sampling years. Nine total species were caught on riparian transects on control sites, 8 on modified buffers, and 7 on state regulation buffer sites. Ten total species were caught on upland transects (lower graph) on control sites, 6 on modified buffers, and 9 on state buffers. Differences among treatment types were not significant ($P > 0.50$).

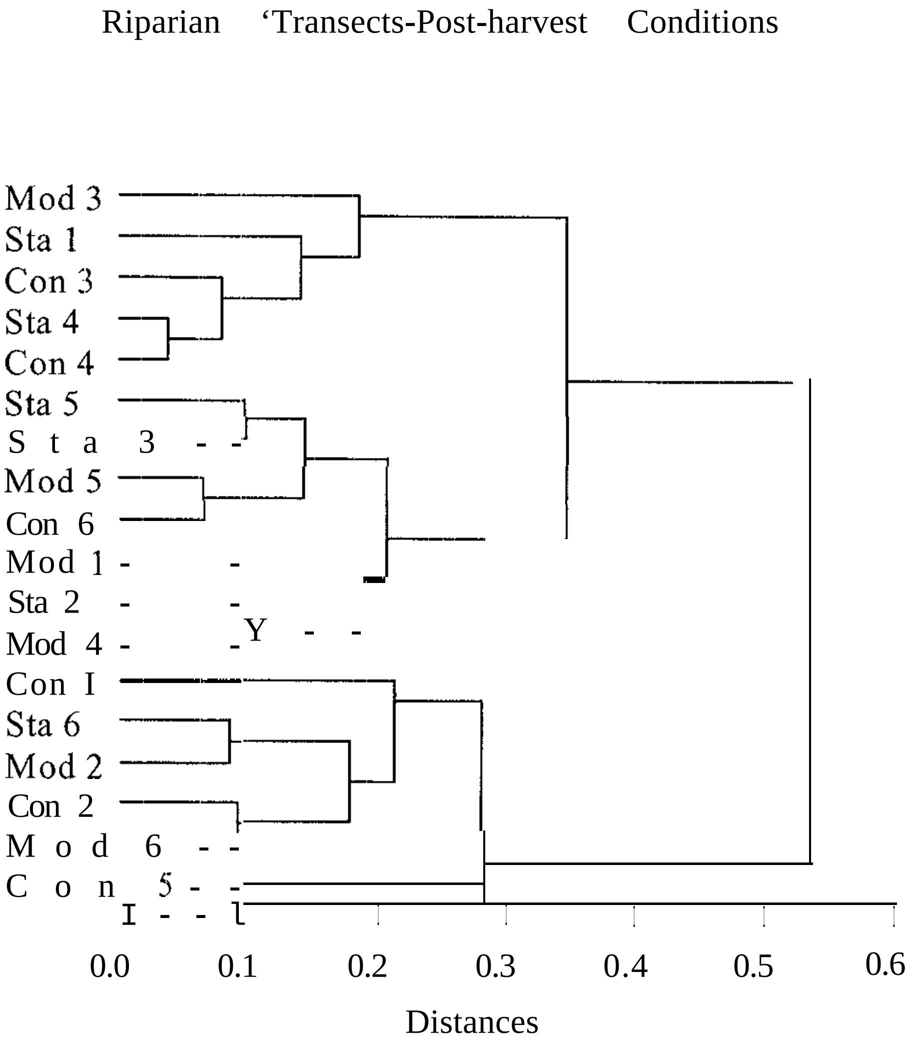

Riparian ‘Transects-Post-harvest Conditions

Figure 5. Hierarchical cluster analysis of riparian amphibian community composition following timber harvest. Distance is a Euclidean metric. Figure shows no clustering of sites, indicating no systematic response of the amphibian community to the treatment type. CON indicates Control sites; MOD indicates Modified sites; STA indicates State sites. There were six replicates for each treatment type.

Upland Transects--Post-harvest Conditions

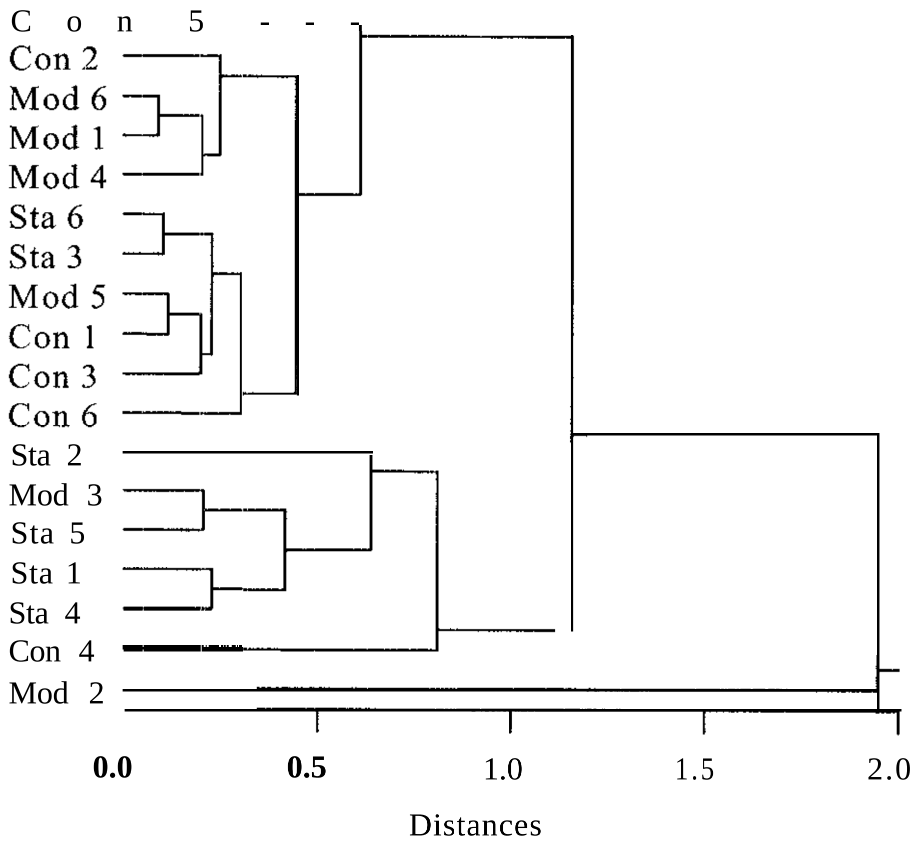


Figure 6. Hierarchical cluster analysis of upland amphibian community composition following timber harvest. Distance is a Euclidean metric. Figure shows no clustering of sites, indicating no systematic response of the amphibian community to the treatment type. CON indicates Control sites; MOD indicates Modified sites; STA indicates State sites. There were six replicates for each treatment type.

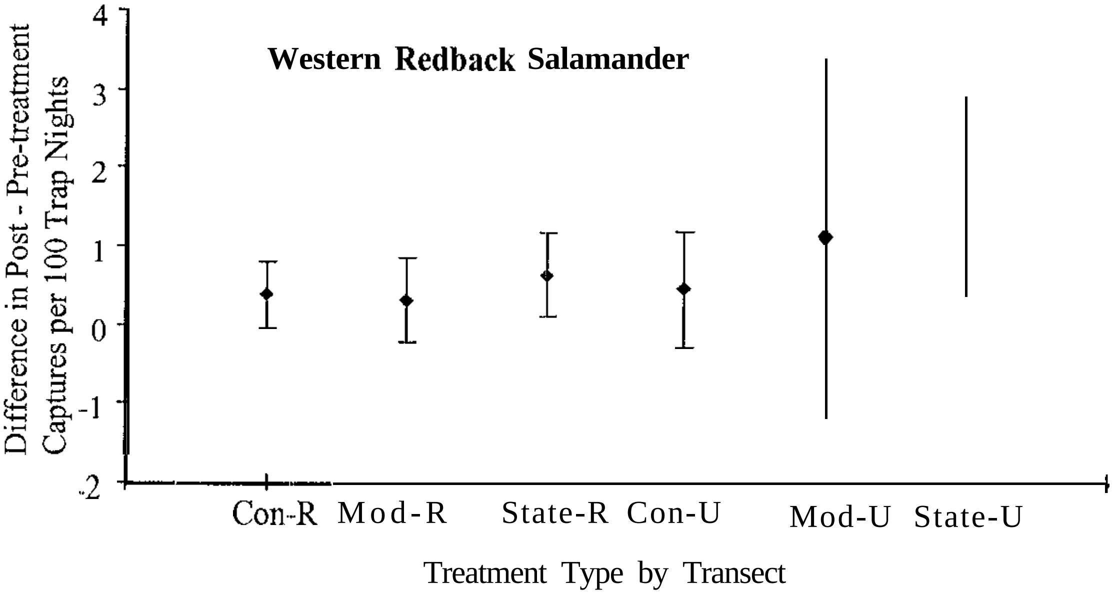


Figure 7. Overall change in abundance from pre-treatment to post-treatment sampling of western redback *(Plethodon vehiculum)* salamanders at riparian (R) and upland (U) transects at all treatment types. Change was calculated by subtracting the mean pre-treatment captures from mean post-treatment captures. Positive values indicate that post-treatment abundance was higher than pretreatment abundance, Negative values indicate that abundance decreased following timber harvest. Vertical bars show standard deviation.

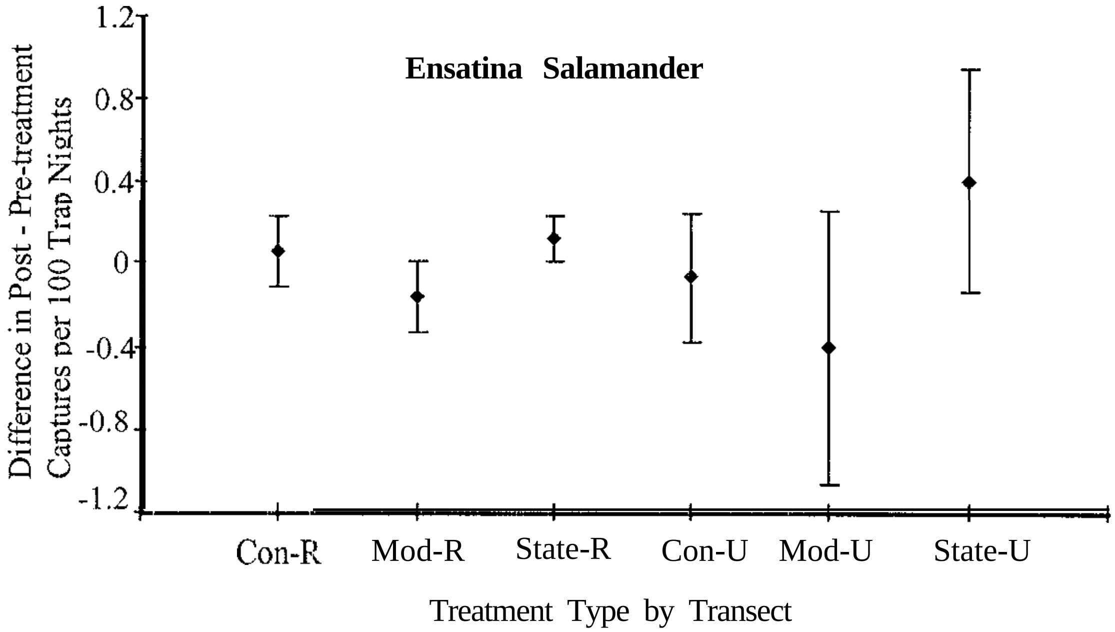
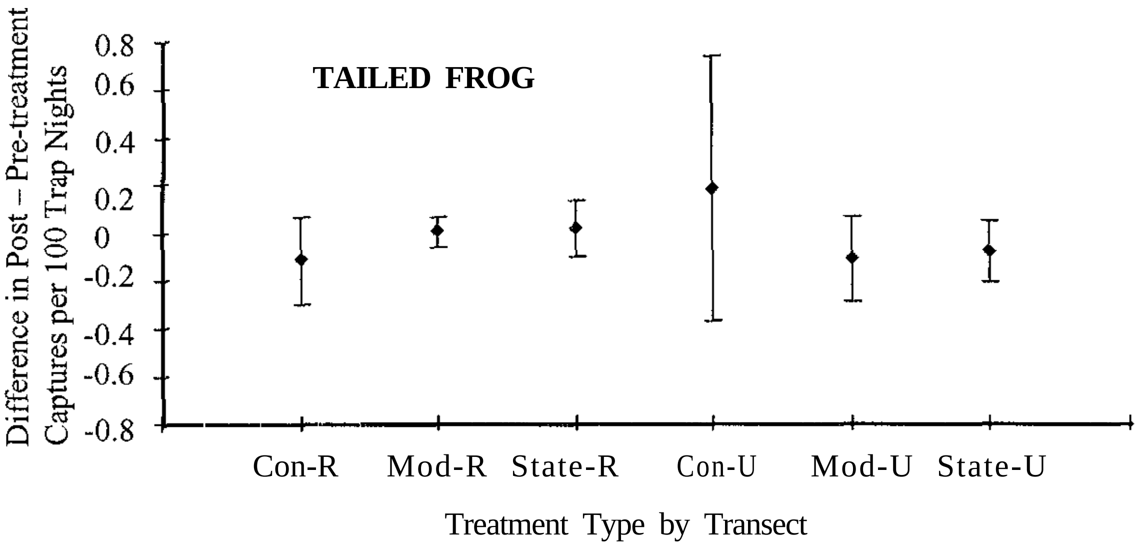


Figure 8. Overall change in abundance from pre-treatment to post-treatment sampling of *Ensatina (Ensatina eschscholtzii)* salamanders at riparian (R) and upland (U) transects at all treatment types. Change was calculated by subtracting the mean pre-treatment captures from mean post-treatment captures. Positive values indicate that post-treatment abundance was higher than pre-treatment abundance. Negative values indicate that abundance decreased following timber harvest. Vertical bars show standard deviation.



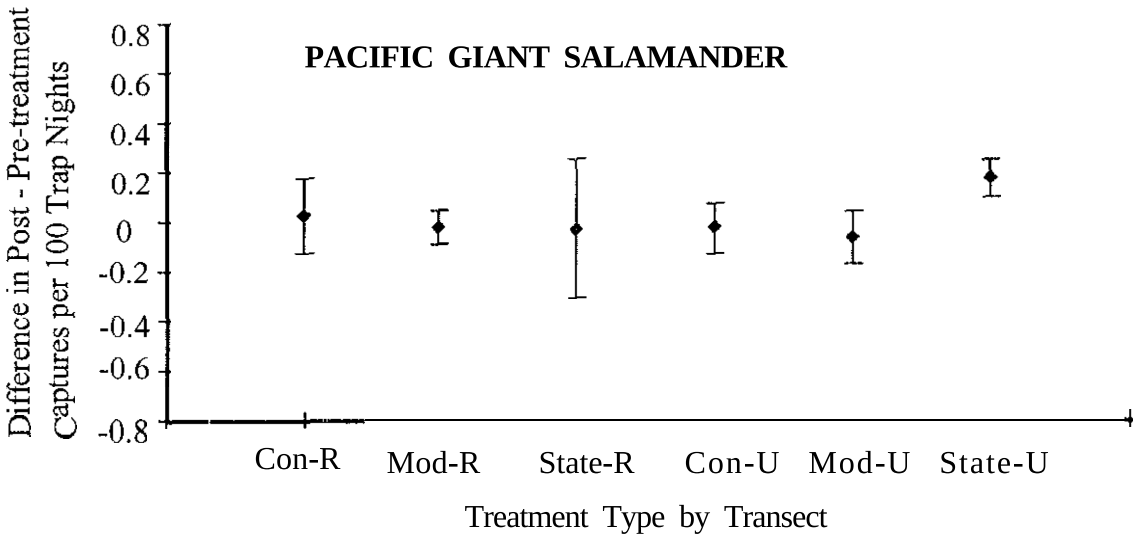
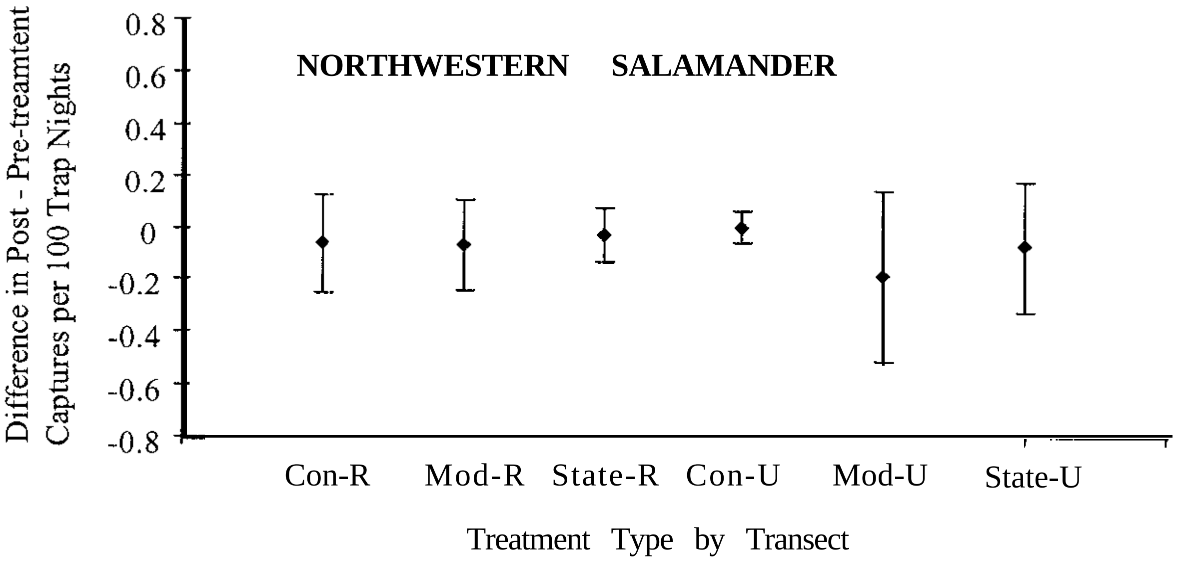


Figure 9. Overall change in abundance from pre-treatment to post-treatment sampling of tailed *frogs* (*Ascaphus truei*) and Pacific giant salamanders (*Dicamptodon tenebrosus*) at riparian (R) and upland (U) transects at all treatment types. Change was calculated by subtracting the mean pre-treatment captures from mean post-treatment captures. Positive values indicate that post-treatment abundance was higher than pre-treatment abundance. Negative values indicate that: abundance decreased following timber harvest. Vertical bars show standard deviation.



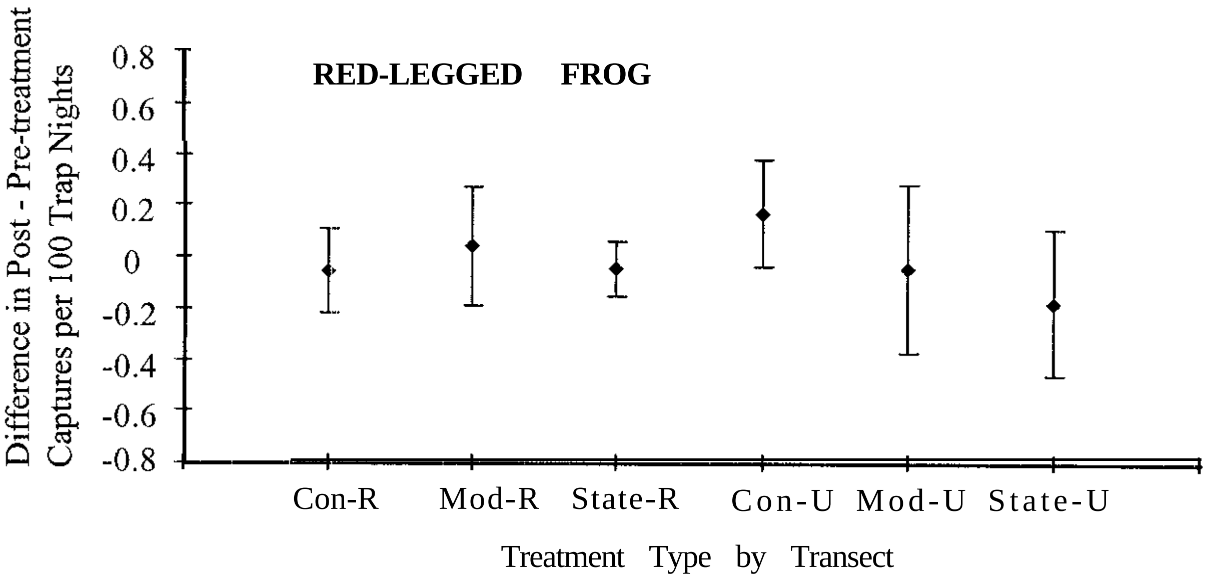


Figure 10. Overall change in abundance from pre-treatment to post-treatment sampling of northwestern salamanders (*Ambystoma gracile*) and red-legged frogs (*Rana aurora*) at riparian (R) and upland (U) transects at all treatment types. Change was calculated by subtracting the mean pre-treatment captures from mean post-treatment captures. Positive values indicate that post-treatment abundance was higher than pre-treatment abundance. Negative values indicate that abundance decreased following timber harvest. Vertical bars show standard deviation.

Chapter 9

EAST-SIDE AMPHIBIAN AND REPTILE SURVEYS

Abstract. Amphibian and reptile populations were studied at 18 riparian and adjacent upland managed forest sites in northeastern Washington before and after logging to examine habitat associations and response to different riparian timber harvest prescriptions, Four amphibian and seven reptile species were found. Overall abundance of amphibians and reptiles was very low (1.8 captures $site^{-1}$ year'). Amphibian abundance was greatest in the riparian habitats whereas reptile abundance was greater in the adjacent upland forests. There were no differences in amphibian and reptile abundance between the unharvested controls, the sites harvested according to the State of Washington's guidelines for riparian management (State), and the sites harvested according to a modified riparian harvest that identified and protected habitat features such as seeps and snags (Modified). Amphibian and reptile abundance remained similar during all years of sampling on the control sites on the Modified sites. Amphibian abundance decreased after harvest in the riparian habitat of the State sites and reptile abundance decreased after harvest in the upland habitats of these sites.

INTRODUCTION

Amphibians and reptiles are important components of the Pacific Northwest fauna (Corn and Bury 1990). A high percentage of the amphibian species are endemic (Nussbaum et al. 1983), and together reptiles and amphibians constitute a large amount of the biomass and numbers of individuals present in the forest habitats of the region (Corn and Bury 1990). The highest species richness is found in the Cascade and Coastal Mountain ranges for amphibians and in the more southern areas of the region for reptiles. In comparison, the forests of northeastern Washington are relatively depauperate in terms of species richness for both amphibians and reptiles and apparently mean abundance of individuals within species as well (Hallett and O'Connell 1997). These forests lack the stream-inhabiting amphibians and terrestrial plethodontid salamanders characteristic of the west-side forests. Moreover, the amphibian and reptile populations of these forests have not been as well documented as elsewhere in the region. The objectives of this study were: 1) to examine species richness and abundance of amphibians and reptiles in riparian and upland habitats of managed forests in northeastern Washington and 2) to compare species richness and abundance before and after different timber harvest treatments of the riparian forests.

METHODS

Study area

Research was conducted in mixed-coniferous forests in the Selkirk Mountains of northeastern Washington (Stevens and Pend Oreille counties). Forest composition in this region is variable and is affected by slope, aspect, edaphic factors, tire history, and timber management practices. Dominant tree species include Douglas-fir (*Pseudotsuga menziesii*), lodgepole pine (*Pinus contorta*), western redcedar *(Thujaplicata),* western hemlock (*Tsuga heterophylla*), western larch (*Larix occidentalis),* grand fir (*Abies grandis),* and alders *(Alms incana* and *Alnus sinuata).*

We selected 18 sites that met the following criteria: 1) 800-m reach of Type 3 or permanent Type 4 stream; 2) >16.2 ha of second growth timber at harvestable age on either side of stream; 3) ≥610 m and ≤1200 m elevation; 4) mixed coniferous forest; 5) landowners in agreement to either leave sites unharvested for 10 yr (Control sites) or to harvest sites within timeframe and specifications of study design (cut sites). Seven sites were unharvested control sites. The upland area of 11 sites was selectively harvested in 1993-1994 to yield a 6- to 12-m spacing of trees. 'The riparian zones of 6 of the 11 cut sites were harvested according to the Washington State Forest Practices RMZ (State sites) guidelines and 5 of the cut sites were harvested according to a modified prescription (Modified sites) designed for this project. The intent of the Modified buffer was to incorporate a site-specific approach to riparian management. Within a 33-m zone bordering the stream, habitat features such as seeps, snags, and deciduous trees, were identified and protected. For example, 1 snag per 2 acres was buffered by a no-entry zone equal to 1.5× the height of the snag and all seeps were buffered by a 10-m no-entry zone that extended to the stream. Following timber harvest, the mean width of the buffers on the State sites was 14.1 ± 3.0 m with a range from S-22.6 m. and the mean width of the buffers on the Modified sites was 29.7 ± 17.4 m with a range from 12 to 144 m.

Sampling

Sampling methods for amphibian and reptile populations that have been developed for the region were based on work in the west-side forests (Corn 1990, Bury and Corn 1991). These methods provided a starting point for designing a sampling protocol for the northeastern Washington forests. Given the lower densities compared with west-side forests, it was appropriate to target locations of expected occurrence. In addition, forest stream surveys for amphibians were inappropriate in northeastern Washington. Amphibians and reptile populations were sampled by pitfall trapping and time-constrained searches. In addition, the field crew maintained lists of all amphibians and reptiles observed on the sites.

Eighteen pitfall traps, consisting of two No. 10 cans with the bottom on one removed and taped together, were placed at 15-m intervals on the riparian and the upland transect for a total of 36 pitfall traps per site. Amphibian and reptile populations were sampled by pitfall trapping for 2 wk per site during June/July 1992-1996. Traps were checked every 2 d. This sampling effort yielded 9,072 trap nights per yr and 45,360 trap nights for the duration of the study.

Time-constrained searches were conducted during late May-mid June in 1992-1996. At six predetermined starting points, spaced at least 100 m apart along each transect, an observer searched for 20 min for a total search time of 120 min per transect and 240 min per site per year. Searches on the riparian transect included the stream and pools adjacent to the streams. Searches consisted of looking in appropriate places such as under rocks, logs, beneath the bark of snags and logs, in the litter layer.

All animals captured were identified, measured, and released. If observers were not positive about an identification, animals were brought back to the field station for positive identification and released at the point of capture at a later date.

Data analysis

Abundance is presented as the number of captures per site per sample year. To examine riparian and upland habitat associations and treatment effects, the mean number of captures per sample year were analyzed using the Kruskal-Wallis analysis of variance (SAS Institute 1989).

RESULTS

The abundance of amphibians and reptiles was very low. We captured a total of 13 1 amphibians of four species, the long-toed salamander (*Ambystoma macrodactylum*), the western toad (*Bufo boreas*), and the Pacific tree frog (*Hyla regilla*), and the spotted frog (*Rana luteiventris*). We captured 30 reptiles of seven species: western skink *(Eumeces skiltonianus*), northern alligator lizard (*Elgaria coerulea*), rubber boa (*Charina bottae),* racer (*Coluber constrictor),* bull snake *(Pituophus catenifer*), common garter snake *(Thamnophis elegans*), and western terrestrial garter snake *(Thamnophis sirtalis*).

Habitat: riparian versus upland

Before harvest, the species richness of amphibians was three times greater in the riparian ($\bar{x}$ = 2.6 ± 0.86) than the upland ($\bar{x}$ = 0.7 ± 0.50) habitats. Mean pre-harvest species richness was 1.2, 1.8, and 2.3 in the riparian habitat and 0.7, 0.6, and 0.8 in the upland habitat of the Control, Modified, and State sites, respectively. The abundance of the four amphibian species during the years before timber harvest was greater in the riparian ($\bar{x}$ = 1.5 ± 0.33) than in the upland ($\bar{x}$ = 0.71 ± 0.23) habitats (Kruskal-Wallis X^2 = 4.98, df = 1, P = 0.026; Fig. 1). Three of the four amphibian species, *A. macrodactylum, B. boreas* and *H. regilla,* were found in both riparian and upland habitats. *R. luteiventris* was found only in the riparain habitat in streams and side pools. *A. macrodactylum* was the most widely distributed species; it was captured at 14 sites and observed on one additional site. *B. boreas* was captured on nine sites and observed on two additional sites. *R. luteiventris* was captured at six sites and observed on an addition two sites. *H. regilla* was captured at four sites and observed at an additional seven sites.

The abundance of the seven reptile species during the years before timber harvest was greater in the upland ($\bar{x}$ = 0.44 ± 0.13) than in the riparian ($\bar{x}$ = 0.12 ± 0.05) habitats (Kruskal-Wallis X^2 = 5.49, df = 1, P = 0.019; Fig. 2). Only two of these species, *E. skiltonianus* and *T. elegans*, were captured in the riparian habitat. Most species were found at only one or very few sites. Three of these species, *C. bottae*, *C. constrictor,* and *E. corulea* were found only in the upland habitat at a single control site, Chewelah Creek. E. *skiltonianus* and *P. catenifer* were captured at *two* sites, *T. sirtalis* was captured at three sites and observed at an additional five sites. Although *T. sirtalis* was captured only on the upland transects, observations at the five additional sites were in riparian habitat. *T. elegans*, found at ten sites and observed on one additional site, was the most widely encountered reptile.

Treatment effects

Mean species richness of amphibians decreased from 2.3 (± 0.75) pre-harvest to 1.2 (± 0.48) post-harvest on the State sites. Species richness of amphibians remained similar on the Control (pre: $\bar{x}$ = 1.3 ± 0.42; post: $\bar{x}$ = 1 .0 ± 0.38) and Modified (pre and post: $\bar{x}$ = 0.8 ± 0.37) sites. The abundance of the amphibians remained the same in the riparian (Kruskal-Wallis X^2 = 1.59, df- 1, P = 0.21) and upland (Kruskal-Wallis X^2 = 1.77, df= 1, P = 0.18) habitats of the Control sites, in the riparian (Kruskal-Wallis X^2 = 1.29, df = 1, P = *0.26)* and upland (Kruskal-Wallis X^2 = 0.15, df = 1, P = 0.69) habitats of the Modified sites and in the upland habitat of the State sites (Kruskal-Wallis X^2 = 0.21, df = 1, P = *0.65;* Fig. 2). In contrast, the abundance of the amphibians decreased in the riparian habitat of the State sites (Kruskal-Wallis X^2 = 4.12, df = 1, P = 0.04; Fig. 2). The decrease was attributable to a decrease in captures of *Rana luteiventris* (1.08 to 0.06 captures per site per year) and of *Bufo boreas* (1 .O to 0.23 captures per site per year). Although not captured within the RMZ, the field crew observed *R. luteiventris* at one site in temporary puddles just upslope in ruts created by the logging equipment.

The abundance of the reptiles was lower across all treatments during the years after harvest (Fig. 2), but decreased significantly only in the upland habitat of the State sites. Results of the Kruskal-Wallis analyses were as follows: Control riparian - X^2 = 1.8, df = 1, P = 0.17; Control upland - X^2 = 0.98, df = 1, P = 0.32; Modified riparian - X^2 = 1.0, df = 1, P = 0.32; Modified upland - X^2 = 1.5, df = 1, P = 0.22; State riparian - X^2= 0.0, df = 1, P = 0.1; State upland - X^2 = 4.03, df = 1, P = 0.045. Four species that had been captured during the pre-harvest years in the uplands of one Control site were never captured during the years post-harvest, explaining the pronounced overall decline in the upland habitats of Control sites (Fig. 3). Two of these species were observed by the field crew during this time but were never captured in pitfall traps or during the time-constrained searches.

DISCUSSION

Of the four genera and nine species of amphibians potentially present in northeastern Washington (Nussbaum et al. 1983), we captured all genera and four species. The absence of the tiger salamander (*Ambystoma tigrinum),* green frog (*Rana clamitans),* leopard frog (*Rana pipiens*), and wood frog (*Rana sylvatica*) is not surprising due the habitat preferences or local geographic distribution of these species (Nussbaum et al. 1983). The bullfrog (*Rana catesbeiana*) has been introduced to the region of our study sites, but its distribution is so far limited to larger aquatic systems. We captured all species of snakes and lizards that have been reported from northeastern Washington (Nussbaum et al. 1983). Thus, despite the managed state of these forests, species richness of amphibians and reptiles in the region of our study sites is consistent with expectations.

However, the species richness and, especially the abundance, of species on individual sites was very low. Given that comparable sampling efforts yield much greater diversity and abundance in other forest types (e.g., Aubry and Hall 1991, Bury et al. 1991), our results probably do reflect the populations of these animals along these streams. Although three of the amphibian species are found in upslope habitats when not breeding, all four species typically breed in ponds or marshes rather than flowing streams. In addition, the presence of fish in the majority of our streams might have further decreased amphibian abundance. For example, Aker (1998) reported ca. 4.5 amphibians per larvae trap in 11 perennial ponds and marshes without either native or introduced fish, ca. 1 amphibian per larvae trap in 9 perennial ponds and marshes habitats with native fish present, and ca. 0.2 amphibian per larvae trap in 12 perennial ponds and marshes with introduced fish present. The mean number of adult *Rana luteiventris* encountered during visual surveys at these same sites was ca. 6.3 per 100 m, 4.0 per 100 m, and 0.5 per 100 m. It is unfortunate that the dearth of information available on amphibian populations in northeastern Washington makes it impossible to examine populations trends on a regional scale to determine whether this region is experiencing the declines in amphibian populations documented elsewhere.

Pre-treatment comparison between riparian and upland habitats

Our limited data support general natural history information on the habitat associations of these species. In our review of wildlife use of riparian habitats (O'Connell et al. 1993) we ranked only one of the amphibian species we captured (*R. luteiventris*) as highly dependent on riparian habitat and the other three (*A. macrodactylum*, *H. regilla*, and B. *boreas*) as only somewhat dependent. Our data support this; only *R. luteiventris* was found exclusively in the riparian habitat. Of the seven lizard and snake species we captured, we (O'Connell et al. 1993) had ranked only one of the snakes, *T. sirtalis*, as dependent upon riparian habitat for feeding and escape habitat. Although captured only in the uplands, this species was observed in both habitats.

Post-treatment effects

The more pronounced declines of *B. boreas* and *R. luteiventris* in the riparian habitat on the State sites and, to a lesser extent of *B. boreas* in both riparian and upland habitats of cut sites, are suggestive of an adverse response to conditions created by timber harvest, but our data only indicate trends. Reduced canopy cover has been one factor associated with decreased amphibian abundance in narrow riparian buffers (e.g., Rudolf and Dickson 1990).

In summary, although we observed most of the amphibian and reptile species potentially present in these forests, the species richness at most and abundance at all sties were very low. Amphibian abundance was greater in the riparian habitat and reptile abundance was greater in the upland habitat. Decreases in abundance of *Rana luteiventris* and *Bufo boreas* following timber harvest on especially the State sites indicate that focused studies in areas supporting higher abundance would be of merit.

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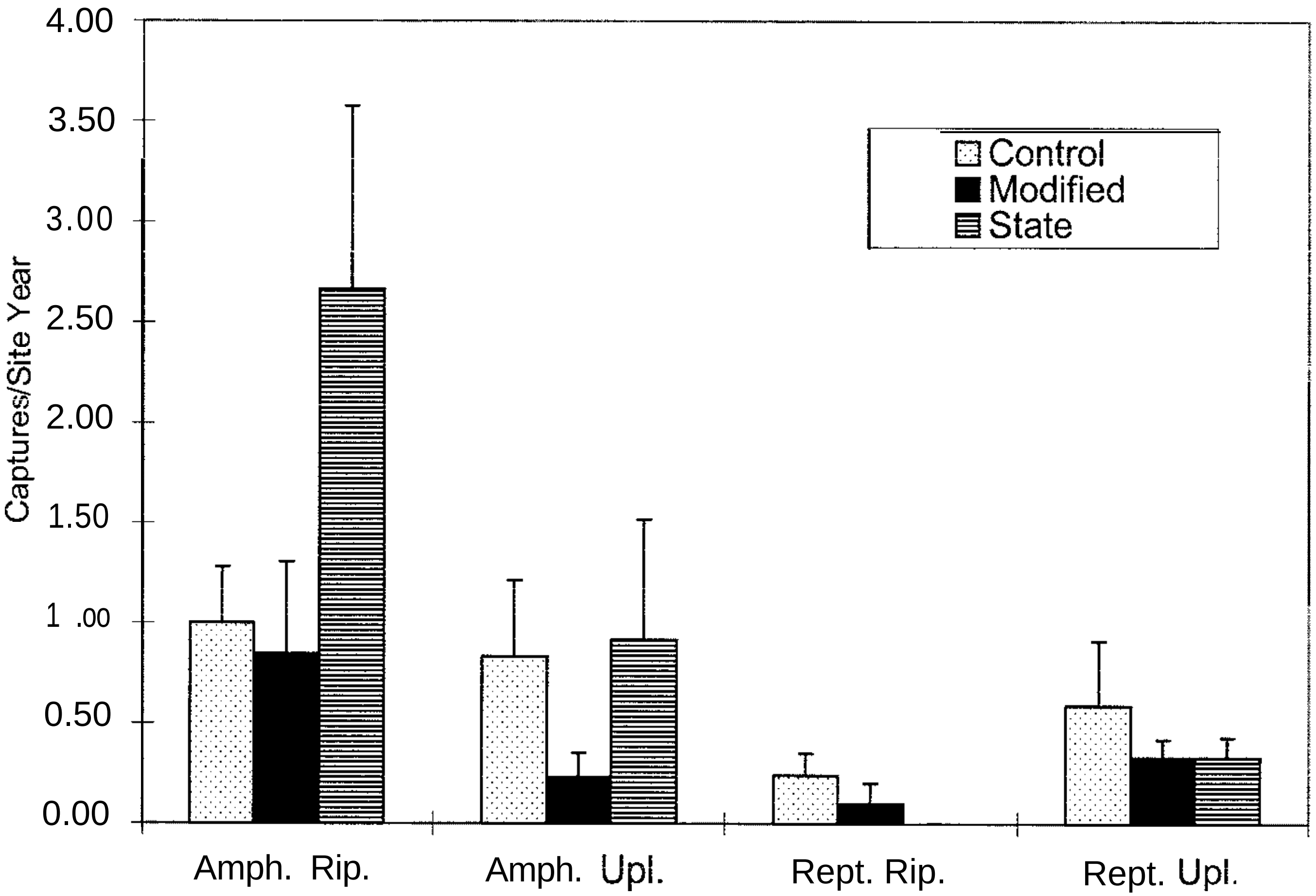

Figure 1. Relative abundance (captures per site per year) of four amphibian (Amph.) and seven reptile (Rept.) species in riparian (Rip.) and upland (Upl.) habitats before timber harvest.

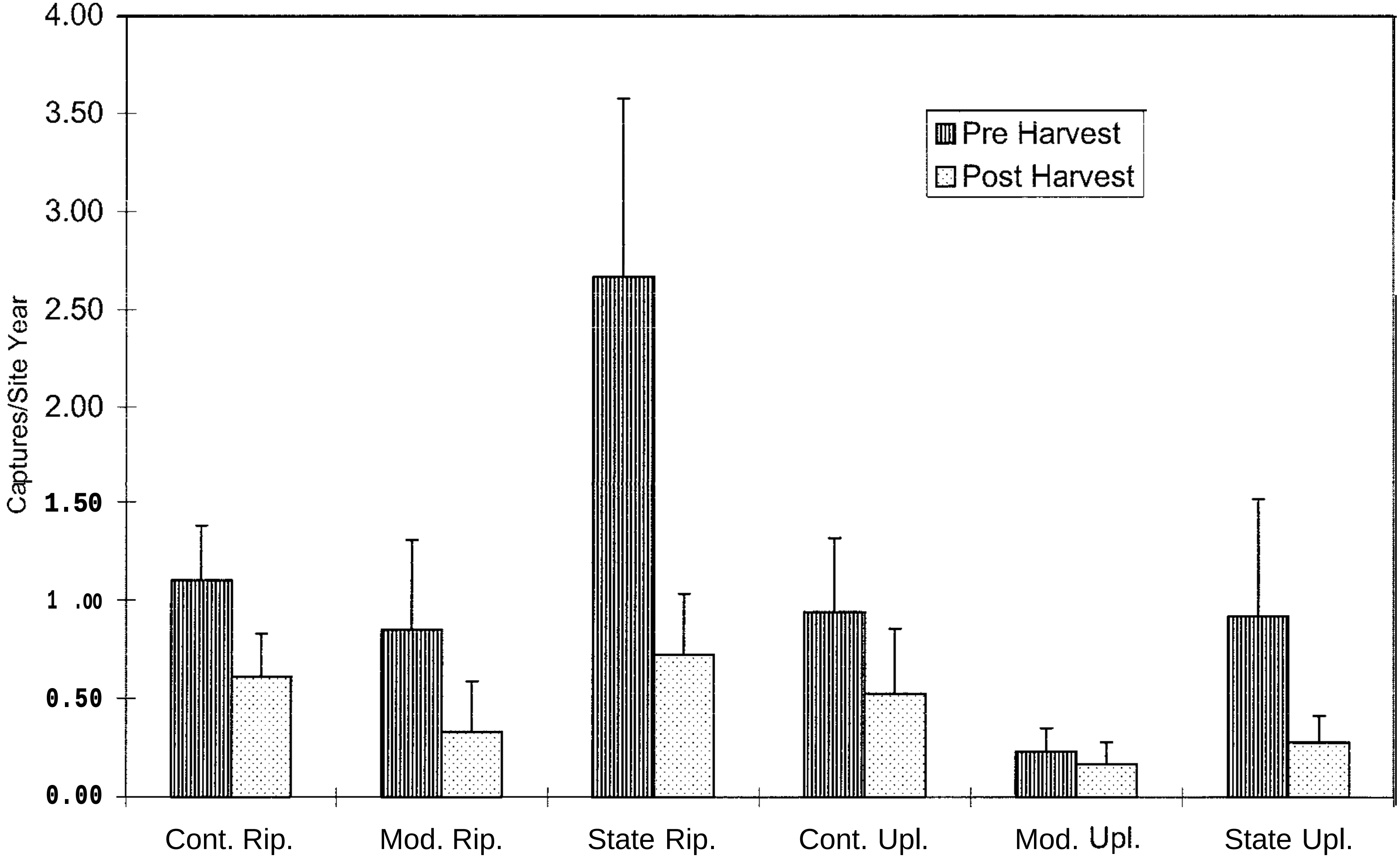

Figure 2. Relative abundance (captures per site per year) of the four amphibian species before and after harvest on the Control (Cont.), State and Modified (Mod.) sites.

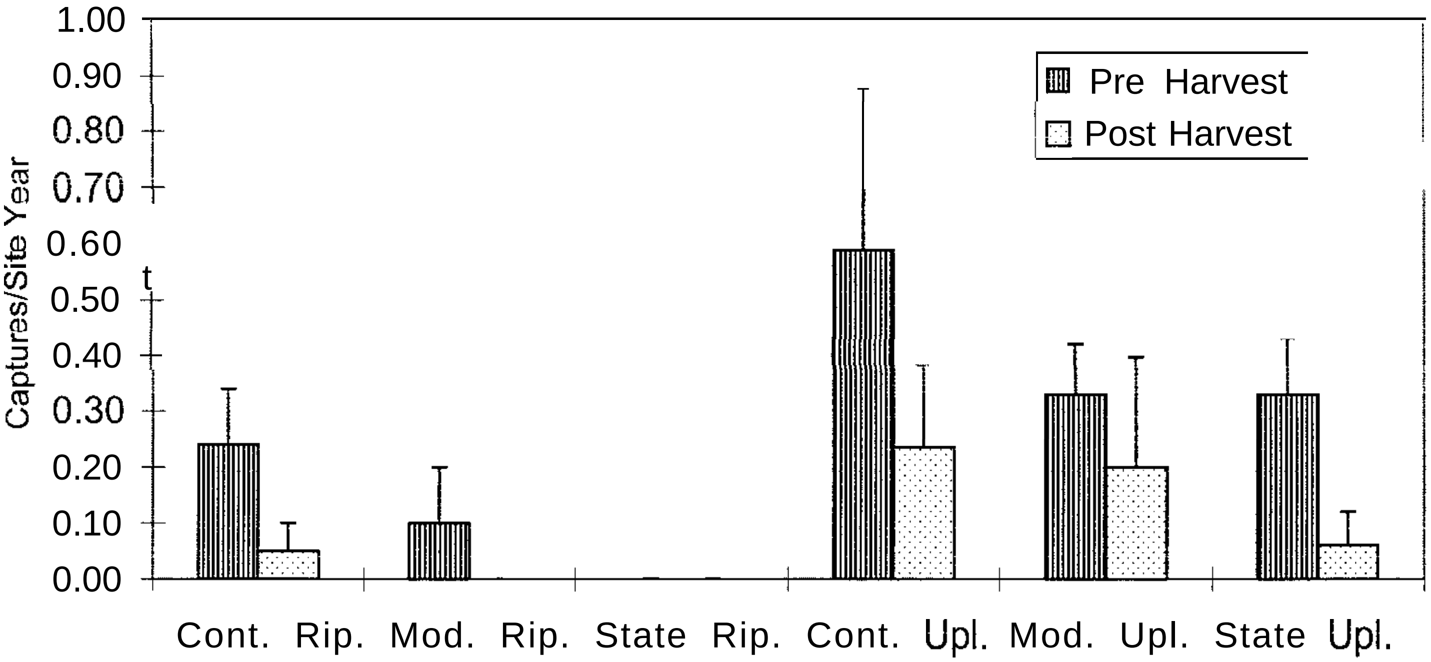

Figure 3. Relative abundance (captures per site per year) of the seven reptile species before and after harvest on the Control (Cont.), State, and Modified (Mod.) RMZ sites in northeastern Washington.

Chapter 10

WEST-SIDE SMALL MAMMAL SURVEYS

Abstract. Habitat occupancy patterns and relative abundance of small mammals within the riparian zone and the associated upland were assessed on unharvested Control sites (Control), on sites harvested under minimal state guidelines for RMZ creation (State), and on sites harvested under guidelines designed as part of this study (Modified). Snap and pitfall traps were used during October- November in 1992 and during October from 1993-1998. For snap traps we sampled with two paired traplines on each side of the stream, one trapline within the riparian zone, and the other well outside the zone about 100 m from the stream. Each trapline consisted of 36 stations set 10 m apart (3.50 m total length) with two Museum Special traps per station. Traplines were centered on the 500-m stream study sites. Traps were baited with peanut butter and whole oats and operated for 4 consecutive d and nights. Pitfall traps were operated for 2 continuous weeks. Traps were checked weekly. Eighteen traps were placed at 15-m intervals on the central portion of each snap trapping transect. The snap and pitfall trapping occurred simultaneously. Trapping totals for each technique were summed to give an overall catch per unit effort index, which was used in statistical testing. To assess the effect of different buffers on capture rates between riparian and upland transects, we used the difference between the pre- and post-harvest mean capture rates as test data, and analyzed for treatment effects using a l-way ANOVA followed by Tukey's HSD test for multiple comparisons. Over the 6 yr of sampling 9,163 individuals of 18 species of small mammals were captured. Species richness before harvest was higher within the riparian zones than in the adjacent uplands. Species evenness and overall abundance were not different,. Species composition was similar between riparian zones and uplands. The montane shrew, the marsh shrew, the Pacific jumping mouse, and the long-tailed vole were caught at greater rates on riparian transects, while the deer mouse, the forest deer mouse, and the southern, red-backed vole were caught more often on the upland transects. There was a trend for the vagrant shrew to favor riparian transects. On riparian transects species richness and evenness did not differ significantly among treatments. Species composition of the riparian transects between harvest treatments was very similar. No species showed a statistically significant change in capture rate with respect to treatment on the riparian transects. The strongest trend toward a statistical difference between treatments on riparian transects was shown by the southern red-backed vole. On upland transects species richness and evenness did not differ significantly. A change in species composition reflected losses of Insectivores (marsh shrew, Trowbridge's shrew, shrew-mole) and the forest deer mouse and gains by the deer mouse and the creeping vole. Capture rates declined significantly for the marsh shrew, Trowbridge's shrew, the shrew-mole, and the forest deer mouse. Capture rates increased for the creeping vole. Over the first 2 post-harvest yr both RMZ treatments provided habitats intermediate in quality for species associated with closed canopy forest. One measure of success for a particular buffer design is whether riparian obligate species and forest-associated fauna will persist within the buffer between the time of harvest and canopy closure. Of the two buffer designs, the modified design appeared to

provide the better chance for persistence. Declines on these sites were less precipitous than the State sites and the species composition of the modified sites more closely reflected that of Control sites. This study has provided a very good baseline from which to evaluate the performance of these buffer designs. An adequate assessment, however, requires future sampling. Several species showed declines over the 2-yr period. Knowing whether they will persist on these sites during the pre-canopy period requires additional sampling.

INTRODUCTION

Until recently there has been little regionally derived information available to forest managers describing the mammalian patterns of habitat use in the riparian zones of managed forests west of the Cascade Mountain crest. Projects investigating the patterns of habitat occupancy in managed forests purposefully have not focused on riparian zones (Ruggiero et al. 1991, Aubry et al. 1998) with the notable exception of the Coastal Oregon Productivity Enhancement Program (COPE) in western Oregon. In the western states most work on habitat relationships in riparian zones was done in arid or semi-arid environments, where the sharp contrast between the physical and biotic features of the riparian zone and adjacent upland resulted in strong habitat use patterns. It is from this perspective that riparian zones are considered habitats of high species diversity and abundance relative to upland habitats,

Given the mild and moist meteorological conditions on the west side, and the general observation that in many instances riparian zone vegetation is limited in extent along small stream courses, one wonders how sharp the contrast in mammalian community composition and species abundance might: be in these forests (Kelsey and West 1998). This study was undertaken in part to describe the patterns of habitat use between riparian zones and associated uplands along small streams of this region, With the decision to center this study on class 3 and 4 streams under the state classification system, data collection is constrained differentially with respect to vertebrate taxa. The riparian zones of interest are quite narrow, often less than 20 m wide, and most clearly for birds and bats but also for some terrestrial mammals provide limited area for sampling and difficulties in assessing habitat use between the riparian zone and the adjacent upland, Terrestrial small mammals and some amphibians, however, may be the best case for describing use of these habitats because these taxa often show sharp patterns of habitat use and have less extensive movement patterns. Many of these taxa also occur at medium to high abundance and can be sampled effectively with generalized sampling methods.

Beyond providing basic information on the associations of small mammals with riparian zones, the primary focus of this study was to assess the usefulness of riparian management zones in maintaining populations of small mammals after timber harvest. I report here on baseline conditions before harvest and the patterns of habitat occupancy during the first 2 yr after harvest. Following the general study design (see Chapter 1) I assess the habitat occupancy patterns and relative abundance of small mammals within the riparian zone and the associated upland with respect to unharvested Control sites, to sites harvested under minimal state guidelines for Riparian Management Zone (RMZ) creation (State), and to sites harvested under guidelines designed as part of this study (Modified).

METHODS

Over the past two decades techniques for sampling small-mammal communities have been developed and used extensively in Pacific Northwest forests. Based on experience gained with these small mammal surveys (Aubry et al. 1991, Bury and Corn 1987, Corn et al.

1988, West 1991) and understanding the biases of two common techniques, this research team (we) decided to sample terrestrial small mammals using a combination of Museum Special snap traps and pitfall traps. Pitfall traps effectively sample small mammals that use tactile and olfactory cues for orientation more than visual cues. They therefore capture insectivores and non-jumping rodents well, but are less effective at capturing deer mice, chipmunks, and jumping mice (Briese and Smith 1974, Williams and Braun 1983, Bury and Corn 1987). The opposite is true for snap traps, which capture large-bodied, agile rodents much more effectively than pitfall traps. Because this study would set the baseline conditions for a set of study sites that could be analyzed regularly over several years, perhaps past canopy closure, we wanted to sample the full complement of species that only a combined field sampling scheme would allow. By capturing large numbers of individuals, this technique allows more reliable identification of species and the assessment of population demography.

Field sampling

As explained in Chapter 1, sampling effort was distributed unevenly across treatment categories and years. Trapping in 1992 was limited to 13 sites: 4 Control, 4 Modified, and 5 State sites. From 1993-1995 sampling occurred on 18 sites: 7 Control, 6 modified, and 5 State sites. Seventeen sites were sampled in 1996, and from 1997-1998 two sites were sampled, finally yielding six sites in each treatment. Sampling effort was limited to one period per year. We began the sampling after the onset of fall rains. The timing of the trapping was set by the need to sample when amphibians were surface-active. In western Washington this occurs in late fall. Consequently, mammals were not sampled during their breeding season. Although a few individuals showed signs of reproductive activity, the populations generally had concluded reproduction at the time of sampling. Populations consisted of high proportions ofjuvenile and sub-adult age classes. In 1992 snap trap sampling extended to mid November and pitfall trapping continued on one site to mid December. During 1993-1998 sampling began in mid October and ended at the end of October or the first week in November.

We sampled with two paired traplines on each side of the stream, one trapline within the riparian zone, and the other well outside the zone about 100 m from the stream. Each trapline consisted of 36 stations set 10 m apart (350 m total length) with two Museum Special traps per station. Traplines were centered on the 500-m stream study sites. Traps were baited with peanut butter and whole oats and operated for four consecutive days and nights (4 trap nights).

Pitfall traps (double deep, two #10 cans) were constructed after the plans of Corn and Bury (1990) and operated for 2 continuous weeks. Traps were checked weekly. Eighteen traps were placed at 15-m intervals on the central portion of each snap trapping transect. The snap and pitfall trapping occurred simultaneously. Animals were frozen and transported to the University for later species identification and measurement. When appropriate, animals were prepared and deposited in the Burke Museum at the University of Washington,

Analytical methods

Capture data were expressed as the number of individuals captured per 100 trap nights. For pitfall traps only the number of days and nights the trap arrays were operated was necessary to compute these values because pitfall traps are multiple capture traps. Snap trap data were corrected to yield traps available per 100 trap nights (Nelson and Clark 1973), recognizing that previously snapped traps could not catch animals. Trapping totals for each technique were summed to give an overall catch per unit effort index. The overall indices were used in statistical testing. Species tested had 87 to 2,258 individuals captured.

To determine the statistical significance of differences in capture rates between riparian and upland transects before harvest I averaged the capture rates across all 4 yr for the Control sites and over the 2 me-treatment yr for the harvested sites. These averages were compared using a paired t-test. Species richness before harvest between riparian and upland was compared using a paired t-test.

To assess the effect of different buffers on capture rates between riparian and upland transects, I averaged capture rates per transect before (1992-1993) and after (1995-1996) harvest, I used the difference between the pre and post mean capture rates as test data (calculated as $\bar{x}_{Post} - \bar{x}_{Pre}$) and analyzed for treatment effects using a 1 -way ANOVA followed by Tukey's HSD test for multiple comparisons. This approach was used in lieu of a more direct ANOVA because of the unequal samples across years in the pre-treatment period. A repeated measure ANOVA was calculated separately for riparian and upland transects to contrast the effects of different buffers on species richness and evenness.

The direction and strength of correlations between selected habitat variables and small mammal captures were assessed with multiple regression (stepwise, backward) using average values for habitat variables and capture rates per site and transect. Correlations were sought for habitat variables found to be significantly different between riparian and upland transect before harvest and among treatments on riparian transects after harvest (Tables 1 and 2, Chapter 3). Snag variables were excluded from these analyses. Habitat variables were transformed as needed prior to regression using log and arcsine transformations.

Hierarchical clustering was used to show similarities in small mammal communities between riparian zones and uplands both before and after harvest. An unweighted group pair algorithm was used based on an Euclidean distance metric and a complete linkage method. Data for the clustering were mean capture rates calculated for riparian and upland transects over the 2-yr pre- and post-harvest periods. Measures of species evenness were calculated using J'(Pielou 1977), which expresses the Shannon diversity index ($\hat{H}' = -\sum_{i=1}^{s}\left[\left(\frac{n_i}{n}\right)\ln\left(\frac{n_i}{n}\right)\right]$) relative to its maximum value ($\frac{H'}{\ln(S)}$) where all species in a sample are represented by one individual. All analyses were done in Systat 7.01 and 8.0 (Wilkinson 1997, 1998). Statistical significance was set at α = 0.05, although I discuss trends between P = 0.05 and 0.10.

RESULTS

Over the 6 yr of sampling 9,163 individuals of 18 species of small mammals were captured (Table 1). Of the 3,528 Insectivores captured 34 individuals of the genus *Sorex* could not be identified to species due to poor condition of specimens. The situation for rodents was more problematic. Juvenile deer mice that had not reached the tail length criterion allowing identification as deer mouse or forest deer mouse could not be assigned to species (Gunn and Greenbaum 1986, Allard et al. 1987, Hogan et al. 1993). There were 1,724 captures of such individuals. In addition, 69 individuals of the genus *Microtus* could not be assigned to species due to poor condition of specimens. These captures were used in statistical calculations where possible, but excluded in analyses for individual species.

On unharvested sites, both on Controls in all years and on treatment sites before harvest, 3,984 individuals of 15 species were captured (Table 2). Of these captures 1,959 individuals of all 15 species were caught on riparian transects and 2,026 individuals of 14 species were caught on upland transects. Two coast moles and nine individuals of the genus *Microtus* that could not be identified to species were caught only on riparian transects (Table 2). The Townsend's chipmunk, Townsend's vole, and ermine were caught only after harvest (Table 1).

Pre-treatment comparisons between riparian and upland habitats

Species richness, evenness, and composition

When all sites were considered, species richness before harvest was higher within the riparian zone than in the adjacent uplands (Fig. 1; $P = 0.011$). An average (± SE) of 8.9 ± 0.36 species were present on the riparian transects compared to an average of 7.8 ± 0.38 on the upland transects. This difference of about one species per site was the result of a higher frequency of occurrence in the riparian zone by five regularly caught species. Infrequently caught species did not contribute materially to the difference. The vagrant shrew (12 riparian vs. 8 upland sites), the marsh shrew (14 vs. 10), the shrew-mole (18 vs. 16), the Pacific jumping mouse (7 vs. 2), and the long-tailed vole (14 vs. 3) were responsible for the pattern (Table 3; Figs. 2A and 2B). Species countering this pattern were the deer mouse (13 riparian vs. 16 upland), the southern red-backed vole (7 vs. 12), and the creeping vole (16 vs. 18). The remaining species (montane shrew, Trowbridge's shrew, and forest deer mouse) showed about equal site occupancy on both transects.

Species evenness (H') before harvest was not statistically different between the riparian and upland transects ($P = 0.17$). Species evenness averaged 0.791 ± 0.019 on riparian transects and 0.752 ± 0.023 on upland transects.

In terms of the small-mammal community before harvest, there were more similarities than differences between riparian and upland transects (Figs. 3 and 4). Most sites were joined at levels around 1-1 .5, a level considerably lower than those observed on post-harvest transects. Sixteen of the 18 riparian transects were similar to each other, although two sites showed different proportions in the capture of deer mice and creeping voles (Fig. 3). At the top of the cluster Griffin Creek was the most dissimilar with a high capture of deer mice relative to the other sites (59 deer mice and 74 forest deer mice). The site Pot Pourri joined the others at a moderate level of dissimilarity at the bottom of the cluster. High captures of forest deer mice (66 individuals) and the highest capture total of creeping voles (44 individuals) during the preharvest years distinguished it. The upland transects showed a similar pattern as the riparian transects (Fig. 4). Griffin Creek was still the most different because of high deer mice captures (38 deer mice and 45 forest deer mice). It joined a cluster of six other sites, which generally showed high captures of deer mice and Trowbridge's shrews.

Species abundance

In terms of average capture rates for all species combined, there was no difference between the riparian and upland transects. Riparian transects averaged 0.55 ± 0.18 captures000 TN and the upland transects averaged 0.56 ± 0.21 captures. Seven species showed statistically significant differences in abundance between the riparian and upland transects (Table 3). The montane shrew, the marsh shrew, the Pacific jumping mouse, and the long-tailed vole were caught at greater rates on riparian transects, while the deer mouse, the forest deer mouse, and the southern red-backed vole were caught more often on the upland transects. The vagrant shrew showed a trend to favor riparian transects (P = 0.099).

Correlations between capture rates and habitat variables

Of the 14 variables found to be either statistically significant or marginally significant different between riparian and upland transects (Table 1, Chapter 3), 13 of these were significantly correlated with the capture rate of at least one of the 11 small mammal species that had sufficient captures for analysis (Table 4). Eleven variables were correlated with more than one species. Multiple correlation coefficients ranged from 0.162 to 0.728 (Table 4). Half of the variables had higher values on riparian transects: percent cover of herbs, bare soil, rock, berry-producing shrubs, other deciduous shrubs, and counts of small alder trees. The other half had higher values on upland transects: percent cover of litter (miscellaneous small cover objects and down wood <10 cm in diameter), canopy, depth of litter (mm), and counts of small western hemlock and small and large Douglas-fir trees.

Post-treatment effects -- riparian habitats

Species richness, evenness, and composition

Species richness did not differ significantly among treatments on riparian transects (Fig. 5). Mean richness ranged from 8.3 to 9.7 species per treatment before harvest to 9.2 to 9.7 species per treatment after harvest. A trend (P = 0.094) toward higher richness on the upland sites after harvest was driven by the difference on Control sites. The slight declines shown by individual species in response to riparian buffer configuration (described below) were offset by increases in species favoring the adjacent harvested area.

Species evenness (H') was also not statistically significant across treatments (Fig. 6). During the pre-treatment period evenness ranged from 0.746 to 0.823 and from 0.792 to 0.799 after harvest.

Species composition of the riparian transects between harvest treatments was similar (Fig. 7A, 7B, 7C). Some differences were noticeable, but as shown above and in the tests for differences in abundance of individual species, these were not strong. There were slight declines in the Insectivores overall with the exception of the shrew-mole. The creeping vole showed a large increase on the State harvest sites (Fig. 7C), but due to high variation among sites this was not statistically significant. The increased proportion of unidentified deer mice (PESP, Figs. 7A, 7B, and 7C) was the result of larger numbers of young animals in the sample. With such subtle differences between the species composition of riparian transects before and after harvest, the similarity of the post-harvest community cluster to the riparian and upland pre-harvest clusters is perhaps not surprising (Fig. 8). As with the earlier two clusterings, Griffin Creek and Pot Pourri appeared as outliers. Ms. Black was also dissimilar to the other sites, Griffin Creek was distinguished again by the large capture of deer mice (43 individuals) and forest deer mice (29 individuals). Pot Pourri repeated its earlier pattern by yielding the highest capture total of creeping voles (104 individuals) and a high number of forest deer mice (39 individuals). It was the most dissimilar site of the group. Ms. Black also had a high capture of creeping voles (46 individuals) and several deer mice (14 deer mice and 28 forest deer mice).

Species abundance

Changes in capture rates of individual species by treatment were assessed by looking at the difference in mean capture rates before and after harvest. No species showed a statistically significant change in capture rate with respect to treatment on the riparian transects (Table 5). The strongest trend toward a statistical difference between treatments on riparian transects (P = 0.071) was shown by the southern red-backed vole which decreased in capture rate between Control and State sites (P = 0.066, Table 5 and Fig. 9). Differences in capture rate for the southern red-backed vole between Control and Modified sites and between Modified and State showed no significant trend.

Post-treatment effects -- upland habitats

Species richness, evenness, and composition

As was the case for the riparian transects, species richness did not differ significantly among treatments in the uplands (Fig. 5). Species richness ranged from 6.8 to 8.8 species before to 7.7 to 9.2 species per site after harvest.

Differences in mean species evenness were also not significant on the upland transects between treatments (Fig. 6). Evenness ranged from 0.713 to 0.754 before and from ***0.692*** to 0.760 after harvest.

Differences in species composition on the upland transects after harvest were quite apparent (Fig. 10A and 10B). The trends shown on the riparian transects were more pronounced in the uplands with statistically significant losses and gains for individual species (below). The overall effects were losses of insectivores (marsh shrew, Trowbridge's shrew, shrew-mole) and the forest deer mouse and gains in deer mice and the creeping vole. A large increase in young *Peromyscus* mice (PESP, Fig. 10A, 10B) characterized the post-harvest period. When the small mammal communities were considered by site, the hierarchical tree consisted of two main clusters and an outlier group of two sites (Fig. 11). Four Control sites constituted one group, and the other consisted of the remaining two Control sites and 10 harvested sites. The sites Ms. Black and Pot Pourri constituted the outlier group which differed from the other sites by continuing to yield very high numbers of creeping voles (84 and 110 individuals respectively).

Species abundance

Five species showed a statistically significant difference in mean capture rate among treatments on the upland transects (Table 5). Capture rates were highest for the marsh shrew on Control sites and declined strongly ($P = 0.004$) when compared with the Modified sites (Table 5 and Fig. 12). Mean captures were not different between the Controls and the State sites ($P = 0.17$), ‘but there was a trend of lower capture rates on the Modified when compared to the State sites ($P = 0.103$). Trowbridge’s shrew had lower capture rates on both Modified and State sites relative to Controls ($P = 0.01$ and $P = 0.022$ respectively; Table 5 and Fig. 13). Capture rates were not different between Modified and State sites. Shrew-mole capture rates were lower on State sites compared to Controls ($P = 0.037$; Table 5 and Fig. 14). Other comparisons for the shrew-mole were not significantly different. Capture rates for the forest deer mouse were lower for Modified sites ($P = 0.011$) and State sites ($P = 0.015$) when compared with the Controls (Table 5 and Fig. 15). Unlike the species discussed previously, captures of the creeping vole increased on Modified sites ($P = 0.037$) and trended up on State sites ($P = 0.097$) when compared with Control sites (Table 5 and Fig. 16). Captures of the deer mouse also increased in the uplands after harvest, but the increase was not statistically significant. However, the increase noted for the young mice (PESP) was highly significant ($P = 0.007$). If these mice could be assigned unambiguously to species, the deer mouse probably would show a statistically significant increase.

Post-treatment effects – riparian and upland habitats

Patterns of site occupancy: riparian and upland transects

There were strong shifts in the pattern of transect occupancy for four species. The southern red-backed vole inhabited two riparian Control transects before and after harvest and three and four upland Control transects before and after harvest respectively. On the Modified sites occupancy fell from two to one riparian transect and from five to two upland transects. On State sites occupancy fell from three to zero riparian transects and from four to zero upland transects. When transect data for the two harvest treatments are combined occupancy for the southern red-backed vole dropped from 14 transects before harvest to three sites afterward. The long-tailed vole showed a different pattern. It occupied the same number of Control riparian and upland transects before and after harvest (four riparian, one upland), and the same number of riparian transects of both harvest treatments before and after harvest (ten for both). On upland transects on the harvested sites, however, it increased dramatically. On upland transects Modified and State sites occupancy rose from one to five transects and zero to five transects respectively. In combination the occupancy of upland transects on harvested sites rose from one to 10 transects. The significant change in abundance shown by the marsh shrew (above) was accompanied by a shift in occupancy pattern as well. Occupancy on riparian transects was similar before and after harvest (Control sites: four vs. five transects; Modified: five vs. three transects; State: five vs. four transects). On upland transects after harvest its occupancy of Control sites increased from two to four transects, but decreased from five to zero transects on Modified sites and from three to one transect on State sites. Altogether, transect occupancy on the upland transects of harvested sites dropped from eight to one transect. The Pacific jumping mouse showed an increase in transect occupancy during the post-harvest years, but the increase was not obviously related to harvest. On riparian transects of Control sites jumping mice occupied two transects before harvest and five afterward. On Modified sites occupancy rose from three to four sites and on State sites occupancy rose from two to five transects. On upland sites they were found on no

Control transects before and one after harvest, on two Modified transects before and on three transects after harvest, and on no State transects before and on two transects after harvest. This resulted in an increase on the Controls from two to six transects and from 7 to 14 transects on harvested sites.

Correlations between capture rates and habitat variables

Of the 10 habitat variables that were significantly different among treatments after harvest (Table 4 of Chapter 3), all had significant correlations with at least one small mammal species (Table 6). All variables but one (fern) were correlated with more than one species. Every small mammal species that had sufficient captures for analysis was correlated with at least one of the habitat variables (Table 6). Values for multiple R^2 ranged from 0.25 to 0.87. In comparison to the correlations obtained during the pre-harvest period, the post-harvest correlations were about the same in strength and the number of variables correlated with the capture rates of individual species.

DISCUSSION

In studies concurrent with this one throughout the Pacific Northwest researchers observed that small-mammal populations underwent fluctuations that yielded a peak of abundance in 1994 (Aubry et al. 1998, West 1998). Consequently, species that experienced high populations were found in a wider array of habitats in 1994 than before and after that year, and the evaluation of habitat occupancy patterns was challenging. Fortunately for the west-side work on this project, 1994 was the year the sites were harvested and small-mammal populations were not sampled. Given relatively similar abundances on the Control sites during 1992.1993 and 19951996 this factor did not have as strong an effect as it might have had during 1994 on habitat occupancy patterns.

Pre-treatment comparisons between riparian and upland habitats

In comparison with other vertebrate taxa, notably the birds (Smith 1977, Stevens et al. 1977, Stauffer and Best 1980, Szaro 1980, Tubbs 1980), relatively little quantitative work has been done comparing small mammal communities in riparian zones with adjacent uplands. Over the past 15 yr work in the Pacific Northwest has been done in southern Oregon (Cross 1985, 1988), the Oregon Cascades (Anthony et al. 1987, Doyle 1990), and the Coast Range of Oregon (McComb et al. 1993a, 1993b). Essentially no work has been done in western Washington during this period, although Rector (1990) compared small mammal communities between riparian and upland habitats along the Nisqually River at Mount Rainier National Park.

Additional information on the use of riparian zones by particular species can be gleaned from natural history accounts or from studies undertaken for other reasons (Oakley et al. 1995). These data, however, suffer from the lack of a sampling protocol designed to compare riparian and upland habitats directly.

On average species richness of small mammals was higher in riparian zones than in adjacent uplands. This was a pattern also seen in riparian vs. upland comparisons made by Doyle (1990) in the Oregon Cascade Mountains and by McComb et al. (1993b) in the Coast Range of Oregon. Rector (1990) found no difference in species richness between riparian zones and uplands along the Nisqually River. Doyle (1990) also found evenness of the small mammal community to be higher in upland than riparian habitat, but McComb et al. (1993b) found the opposite, and Rector (1990) found no difference. In this study evenness of the riparian and upland communities was the same. The small mammal community of the riparian zone was composed of the same species as that of the adjacent uplands with the difference in species number resulting from the more consistent presence of species within the riparian zones. At elevations above about 800 m in the western Cascades of Washington the water shrew becomes a regular part of the riparian small mammal community, although it occurs at low abundance. At subalpine elevations, the water vole (*Microtus richardsoni)* also inhabits riparian zones. In these higher elevation forests the species list of small mammals inhabiting the riparian zone might differ on average from adjacent uplands at least with respect to these species.

In terms of overall abundance of small mammals, there were no differences between riparian and upland transects. This agrees with the findings of McComb et al. (1993a, 1993b) in the Coast Range of Oregon, but not with Doyle (1990) or Anthony et al. (1987) in the Oregon Cascades. Doyle (1990) found higher abundance in riparian than upland areas, while Anthony et al. (1987) found the opposite. At present there seems no consistent trend in the region. It may be that small mammal numbers between the two areas will prove temporally variable, subject largely to conditions promoting variation in local populations of deer mice and microtine rodents.

There were differences in the relative abundance of some species between riparian and upland habitats. Of the seven species that had significantly different capture rates between riparian and upland transects, the positive association with riparian zones has been well documented in the literature for the marsh shrew and the Pacific jumping mouse (Anthony et al. 1987, Doyle 1990, McComb et al. 1993b). The strongest evidence for the positive association of the montane shrew, the long-tailed vole, and the forest deer mouse with riparian zones is provided here. Doyle (1990) reported a significant riparian association for the montane shrew from live sites in Oregon. There is evidence of a riparian association for the long-tailed vole in Oregon (McComb et al. 1993b), but limited evidence for Washington (Dalquest 1948). The southern red-backed vole has been identified as a species found more often in riparian zones than uplands in the Rocky Mountains (Hoffman 1960), but this is not the case in western Washington where it is significantly associated with upland habitats. The related western red-backed vole (C. *californicus*) also favors upland over riparian habitats in Oregon (Anthony et al. 1987, Doyle 1990, McComb et al. 1993b). The forest deer mouse was caught more often on upland rather than riparian transects (574 vs. 458 captures, Table 2). Riparian associations for this species are unknown other than from this study. It appears that it responds similarly to the deer mouse with about 44% of its captures on riparian transects compared with 40% for the deer mouse (Table 3). This is a species that is widely distributed throughout closed canopy forests in Washington (West 1991, 1998) and is the numerically dominant deer mouse in forests at foothill and higher elevations in the Cascade Mountains. Information on riparian vs. upland associations for the deer mouse in Oregon are mixed. Anthony et al. (1987) found them more abundant in uplands, Cross (1985), Doyle (1990), and McComb et al. (1993b) found them more abundant in riparian zones, and McComb et al. (1993a) found no difference in abundance between the two habitats. Part of the difficulty here stems from different spatial arrangements used to sample small mammals, particularly the distance from the stream and riparian vegetation. It is also probable that deer mouse response to riparian habitat varies with local conditions, with greater reliance on the riparian zone in more arid environments. In Washington, the deer mouse becomes more common as forest canopy becomes fragmented and as edge environments become more prevalent (West 1998). Although both species tend to be more abundant in uplands, both are also common in riparian zones.

The patterns of relative abundance for two other species deserve comment. Vagrant shrews were found more often in riparian zones than in the uplands, but the pattern was not statistically significant (P = 0.099). In western Washington this shrew is more common in grassy and brushy pre-canopy conditions than in forest, but present in later seral stages as well. West (1998) showed this pattern clearly. Given the edge environment provided by the riparian zone the trend of high captures on riparian transects tits this general picture of habitat occupancy. General accounts of habitat affiliations of the shrew-mole indicate that it uses moist areas (Dalquest 1948, Ingles 1964, Larrison 1976, Maser et al. 1981), but the strength of the association in forested riparian zones is not particularly well-addressed. Doyle caught 1.6 times as many shrew-moles in riparian than in upland habitat in the Cascade Mountains of Oregon and Cross (1985) caught about twice as many in the riparian zone as in adjacent very dry uplands. Captures of the shrew-mole in this study were essentially equal between the two habitats, very slightly favoring the riparian transects (114 vs. 88 captures, Table 2). As suggested by Cross (1988) the shrew-mole probably has its primary association with well developed forest. In regions where the forest floor remains relatively moist shrew moles may not show strong affiliation with the riparian zone, and where dry conditions prevail, the association would be expected to be stronger.

Differences in the organization of the small mammal community between riparian and upland habitats were slight and limited to differences in relative abundance of species rather than species composition This seems to be a general feature of the vertebrate fauna in the Pacific Northwest (Kelsey and West 1998), where prevailing conditions are moist thereby reducing the contrast between riparian and adjacent upland habitats. Outlier stands identified by the hierarchical clustering analysis were products of differences in relative abundance, primarily due to deer mice and microtine rodents. In both habitats the small mammal communities were numerically dominated by the Trowbridge's shrew and the two deer mice species (Figs. 2A and 2B). Tlne montane shrew, vagrant shrew, shrew-mole, southern red-backed vole, and the creeping vole were of much lower abundance although they were consistently encountered.

Many, but not all of the correlations between mammalian abundance and habitat features reconcile with our present understanding of the natural history of these species and the patterns of habitat occupancy reported above (Table 4). In western Washington the vagrant shrew reaches highest abundance in pm-canopy conditions, and as discussed above, tended to be slightly more common on the riparian transects. The positive correlations with herb and rock cover and counts of alder, variables with high values in riparian zones, and the negative correlations with counts of large Douglas-fir, most prevalent in the uplands, generally agrees with this picture. The strong correlation with litter, while perhaps expected for all Insectivores, does not follow the pattern as litter had high values in upland transects. Other species that tend to favor riparian areas include the montane shrew, the marsh shrew, the Pacific jumping mouse, and the long-tailed vole (Table 3). One expects for these species, as with the vagrant shrew, positive correlations with variables having high values on riparian transects and negative correlations with variables having high values on upland transects. Such was the case for the marsh shrew and the long-tailed vole, although the only significant correlation for the marsh shrew was with rock cover. Four of the six correlations for the montane shrew agreed with this pattern, but the positive correlation with litter depth and the negative correlation with rock cover did not. Two of the three correlations for the Pacific jumping mouse, however, were counter to the pattern (cover of other deciduous shrubs and canopy cover). Species favoring upland areas were the southern red-backed vole and both deer mice. For these species one expects the correlations with habitat variables to be the reverse of species favoring riparian areas. This was seen for three of the five correlations for the southern red-backed vole, for two of the three correlations for the forest deer mouse, and for four of five correlations for the deer mouse. All species showed negative correlations with herb cover that had higher values on riparian transects, In agreement with the pattern, the vole was positively correlated with litter cover and counts of small Douglas-fir, but in contrast, was negatively correlated with counts of large Douglas-firs and small western hemlocks. As expected, the forest deer mouse was positively correlated with counts of small western hemlock. The negative correlation with litter depth was not expected for the forest deer mouse as this variable had high values on upland transects, but this may indicate a microhabitat preference because a negative correlation with litter cover was also found for this species in the TFW Landscape Study (West 1998). Although the southern red-backed vole and the forest deer mouse were captured more frequently in uplands, their contrary response to tree count variables may indicate a differential use of forest habitat. A strong positive correlation with small western hemlock by the forest deer mouse indicates use of heavily shaded conditions in contrast to more open conditions at the forest floor indicated by the positive correlation between the southern red-backed vole and small Douglas-fir and negative correlation with western hemlock. It appears that the forest deer mouse is uniquely capable among the terrestrial rodents in using very dense stands with little production at ground level. In the TFW Landscape Study (West 1998) the forest deer mouse was the only rodent species found in large numbers in young, unthinned forest stands. The ability of this species to use such stands may relate to its climbing ability and use of canopy-level resources. The deer mouse correlations with litter depth, rock cover, and counts of small Douglas-firs fit the expected pattern, but the negative correlation with canopy cover did not. A negative correlation with canopy cover, however, tits the habitat selection pattern for the species over forest seres where the deer mouse is most abundant in pre-canopy and broken canopy conditions. The remaining three species, Trowbridge's shrew, the shrew-mole, and the creeping vole did not show statistically significant preferences for either riparian or upland habitats. 'The four negative correlations between the cover of other-deciduous shrub, berry-producing shrub, canopy, and counts of large Douglas-firs and Trowbridge's shrew appear enigmatic, While the negative correlation with the shrubs might be understood as use of areas with dense canopy cover, this clearly is not the case. Further, Trowbridge's shrew is the most common Insectivore once the forest canopy closes. I suspect that these correlations are the result of lower captures on sites with well-developed shrub layers along riparian zones and on upland sites with very dense canopies, There was a tendency toward lower captures for the Trowbridge's shrew on unthinned sites in the TFW Landscape Study (West 1998). The shrew-mole showed positive correlations with litter and counts of small western hemlocks and alder. The association with litter is expected given this species' foraging mode, but the association with small hemlocks and alder is unclear. The negative correlation with canopy cover may parallel the case for the Trowbridge's shrew, where use of very dense, unthinned sites may be low. The shrew-mole also showed low captures in such conditions in the TFW Landscape Study (West 1998). The negative correlations between fern cover and counts of small western hemlocks for the creeping vole may be understood in terms of its preferences for graminoid vegetation and more open canopy conditions.

Post-treatment effects -- riparian habitats

Although some species showed declines in captures after harvest, these were compensated by species showing increases, resulting in little consistent difference in richness. Species that favored early successional and edge conditions increased site occupancy in the post-harvest period, notably the Pacific jumping mouse, the deer mouse, and Townsend's vole. Because site occupancy increased for the Pacific jumping mouse on Control sites from two to five sites after harvest, it appears that the increase may not have been entirely due to the effects of harvest. In fact, the sampling in 1992 was done in November rather than October. This was apparently after the mice entered hibernation as none were caught. The zero catch in 1992 probably resulted in an underestimate of site occupancy for the pre-harvest period, thus overestimating the change in site occupancy between periods. Nonetheless, this is a species known to respond positively to edge environments and the expanded use of harvested sites is no doubt real, but in this case difficult to estimate. Consistent with this view is that it increased its occupancy of upland sites after harvest (below). Fortunately, it is the only hibernating species in this fauna1 assemblage. The pattern for the Townsend's vole and the deer mouse are less equivocal. Both species do best in early successional conditions, and Townsend's vole was only caught after harvest.

Species evenness was relatively unremarkable with values in a rather narrow range. Species composition was similar between harvested and Control riparian transects. Because most captures of Insectivores were of Trowbridge's shrew, declines of this species in response to harvest will lead to overall declines in Insectivores after harvest. Even so, the Trowbridge's shrew still accounted for slightly less than 20% of total captures after harvest. The forest deer mouse declined by about 50% on average after harvest, while the deer mouse (to which most of the unidentified deer mice might be attributed) increased. It is probably the case that the majority of young *Peromyscus* that are placed in the PESP category are deer mice. This asymmetry in classification is due to the tail length criterion used to distinguish the species. While many young forest deer mice have tails long enough to be correctly classified, young deer mice that have short tails remain ambiguous. At any rate, both the decrease in forest deer mice and the increase in deer mice after harvest were expected. Also expected were increases in the creeping vole, which were clearly observed. Increases by other *Microtus* species might have been expected, but these were not seen. Densities of long-tailed voles generally do not approach the magnitude of abundance shown by other congeners (Randall and Johnson 1979). In addition, the elevations for most of the sites in this study were probably too high for the Townsend's vole to realize large increases in abundance.

As described above, differences in species composition that distinguished certain sites before harvest (Griffin Creek, Pot Pourri, and Ms. Black) were evident after harvest as well. No obvious clusters were identified by the hierarchical clustering procedure that corresponded to treatment after 2 yr. One would expect the small mammal communities within treatments to become more similar over time, but so far, this has not happened.

Differences between the two harvest treatments were also very similar. Comparing Figs 7B and 7C, one sees minor differences for all species except the Pacific jumping mouse and the creeping vole, species that would be expected to respond positively to increasingly open habitats. The similarity between treatments was an unexpected result. Based upon our current understanding of the habitat relationships of the small mammal fauna we expected to see a small mammal community in the State transects that might more closely resemble a community found in clear-cut forest habitat. Despite the fact that forest cover was limited on the State transects the sites retained many of the forest associated species.

Given the subtle responses to the two riparian treatments by most small mammals, one might expect rather mild patterns of correlation between small mammal captures and vegetation variables. For the most part the correlations, although slightly fewer in number, were similar in direction and magnitude to the pre-harvest period. Ten vegetation variables showed significant differences among treatments, but six of these were also significant before harvest. Because the vegetation sampling was designed to capture post-harvest differences in the two quadrants farthest from the stream under State guidelines in effect during 1994 (see Chapter 3), virtually all post-harvest vegetation measurements were taken within RMZ boundaries. Strong treatment effects were not captured in the RMZ vegetation data sets simply 'because the harvests on State sites were not close enough to the streams. Variation in the vegetation variables is therefore more related to variation among intact RMZs than harvest effects. Percent canopy cover and counts of large Douglas-fir and western hemlock trees did show treatment effects, declining from Control to Modified and State RMZs. Most correlations can be interpreted from knowledge of small mammal natural history, as was the case for the pre-harvest correlations. For example, correlations for canopy cover were positive for those species typically found under continuous canopy (montane shrew, shrew-mole, and the forest deer mouse), but negative for the deer mouse, a species favoring pre- or broken-canopy conditions. Mammal species favoring pre-canopy or upland conditions generally (except the shrew-mole) showed negative correlations with large western hemlock (vagrant shrew, forest deer mouse, creeping vole), and those favoring conditions very near the water (Pacific water shrew) showed negative correlations with large Douglas-fir. Understanding the nature of the post-harvest riparian vegetation data set, the best indicators oftreatment effects are the ANOVA tests. Relatively few insights would be expected from such correlations of vegetation and small mammal captures.

With the trends in abundance shown between the two treatments (although smaller than expected) and projecting future trends from a general understanding of natural history, one might expect the following patterns. Shrew numbers may fall over the next few years largely as a function of declines by the Trowbridge's shrew. Populations of the vagrant shrew, the Townsend's chipmunk, the Pacific jumping mouse, the deer mouse, the creeping vole, and the long-tailed vole should increase over the next few years and begin to decline as the canopy closes in the adjacent uplands. The southern red-backed vole might be lost in the buffer strips over the next few years and not reestablish until canopy closure of the surrounding forest.

Only the southern red-backed vole showed a statistical trend in abundance on riparian transects after harvest. The lack of statistical significance is surprising. Although the number of sites was limited due to availability and cost, six sites would pick up dramatic differences in relative abundance. These simply were not observed on the riparian transects, although they were seen between treatments on the uplands (below). It appears that the buffers have retained sufficient structures to result in an intermediate response in community organization between that expected in forested and clearcut-harvested habitats. How long the forest-associated species will inhabit the riparian buffers is unknown. The time course of such change (if it happens at all) requires resampling these sites in the future.

Post-treatment effects -- upland habitats

Despite clearcut harvesting the upland habitats, species richness and evenness were not significantly different among treatments. Of course, neither measure, unlike the hierarchical clustering below, revealed the turnover of species that characterized the shift in the small mammal community after harvest. These average species richness values (6.8 to 8.8 species/transect before harvest; 7.7 to 9.2 species/transect after harvest) are slightly higher than those seen in unmanaged young (6.7 species in 55-75 yr-old forest), mature (7.4 species in 80-190 yr-old forest), and old-growth forest (7.8 species in 210-720 yr-old forest), which were sampled using the same methods as the present study (West 1991). As was the case on the riparian transects, species favoring early successional stages and edge environments tended to increase while those favoring closed canopy forest declined. This can be seen clearly on Fig. 10A and 10B. Unlike the riparian transects, however, several species did show statistically significant changes in abundance as will be discussed below.

After harvest, the small mammal community of the uplands is numerically dominated by rodents, particularly the deer mouse and the creeping vole. In forest, Insectivores are either more abundant than rodents or they have roughly equal abundance. Most of the captures of young *Peromyscus* (PESP) probably can be attributed to the deer mouse as discussed above. Over the next few years, abundance of the Trowbridge's shrew would be expected to continue its decline. Declines also would be expected for the southern red-backed vole, the forest deer mouse, and the shrew-mole. Species expected to increase would include the vagrant shrew, the Pacific jumping mouse, the deer mouse, and the creeping vole.

Abundance of the creeping vole can be highly variable during the first 10 yr or so after harvest when it finds optimal conditions. As grasses and herb cover are reduced by shrub cover, abundance of the creeping vole probably will decline. The deer mouse will reach its highest abundance during the years of high brush cover. As the canopy closes some 1 8-20 yr post-harvest, these species will give way to those species characteristic of closed canopy forest.

Similarities in richness and evenness notwithstanding, the distinctiveness of the pre- and post-harvest small mammal communities is obvious in the hierarchical clustering of the upland sites (Fig. 11). Interestingly, an outlier group consisting of the sites Ms. Black and Pot Pourri join the Controls and harvested sites at a relatively high dissimilarity. This was driven by very high populations of creeping voles present on these sites, illustrating the tendency of this species to reach very high numbers under the right conditions.

Four of the five species with statistically significant differences after harvest declined on upland transects. The upland habitats after harvest were essentially the same on both harvest treatments, although standing trees associated with the riparian zone were somewhat closer to the upland transect on Modified than on State sites. Responding to the removal of forest, the decline in abundance of these three species was expected. All four will probably experience further declines on the upland portion of the site over the next few years. All four will begin increasing in abundance about the time of forest canopy closure. In contrast, the creeping vole increased after harvest, and probably will increase or at least maintain moderate numbers throughout the early post-harvest period. It will become less common as the forest canopy closes, although it will remain the most common *Microtus* species in the resulting forest.

CONCLUSIONS

The primary functions of a buffer are to protect the integrity of the aquatic environment over all timeframes and provide critical habitat elements for terrestrial wildlife. The primary goals of this study were to begin the identification of the elements that current guidelines provide and to begin the assessment of their adequacy. With the exception of the aquatic amphibians, the focus of this study is the provision of habitat for terrestrial wildlife. As such, the remaining discussion will center on this issue and specifically on terrestrial mammals, At the outset, buffers are stopgap devices to provide habitat during the post-harvest years, particularly during the first two decades (western Washington) before the young forest on the adjacent uplands develops a closed canopy. At canopy closure, the riparian zone is once more buffered by the surrounding forest and at lower risk from weather extremes and the negative biotic effects associated with high-contrast edges. The basic strategy is to design a riparian buffer that will maintain the biota of the riparian zone through these early post-harvest years. An added benefit, if the buffer is to remain largely unharvested through forest rotations, is the addition of much needed structure to the riparian zone and to managed forests generally in the form of large trees, snags, and down wood.

One measure of success for a particular buffer design is whether riparian obligate species and forest associated fauna will persist within the buffer between the time of harvest and canopy closure. It may not be necessary for population abundance to remain at pre-harvest levels, but at least a consistent presence by these species within the buffer would allow populations to recover quickly once the canopy of the surrounding forest closed. Species of generalized habitat requirements or those associated with early seral conditions should find ample habitat in -the lands adjacent to the buffer. Judged from this perspective, a buffer for a riparian obligate or a species associated with closed canopy forest represents a habitat of intermediate quality-somewhere between well-developed and recently clearcut forest, The design challenge is to provide sufficient structure to allow persistence and yet provide economic return from the harvest.

The findings

Over the first 2 post-harvest yr both buffer treatments provided habitats intermediate in quality for these species. This was clearly the case because four species (marsh shrew, Trowbridge's shrew, shrew-mole, and forest deer mouse) showed statistically significant declines on the adjacent clearcut uplands, but none showed such strong declines on the buffer transects. Only the southern red-backed vole showed a strong trend of decline ($P = 0.071$) within the buffers. In fact, it may be that the riparian zone is suboptimal habitat for the southern red-backed vole as it was found significantly more often in upland than riparian habitat, Reductions of the riparian zone canopy may have been sufficient in this circumstance to elicit a decline in abundance. Of the two buffer designs, the Modified design appeared to provide the better chance for persistence. Declines on these sites were less precipitous than the State sites and the species composition of the Modified sites more closely reflected that of Control sites.

Data from the 2 preharvest yr supported the contention that in western Washington the small mammal communities of the riparian zone and adjacent uplands are similar. Differences in the fauna are subtle and, with the exception of the water shrew which occurs primarily at elevations above this study, characterized by differences in relative abundance between the two habitats. Species richness averaged about one species higher on riparian vs.

upland transects, but the difference was due to the more regular capture of common species on riparian transects rather than the capture of uncommon species or riparian obligates. Species evenness was the same. Total abundance of small mammals was also about the same. The montane shrew, marsh shrew, Pacific jumping mouse, and long-tailed vole were caught significantly more often in riparian zones than in the uplands. The vagrant shrew showed a strong trend of higher captures in riparian zones as well. The deer mouse, forest deer mouse, and southern red-backed vole were caught significantly more often in the uplands than riparian zones. The high degree of fluidity between these two habitats and the fact that some species favor the uplands argues for replacing some of the structure in the uplands that repeated harvest has removed. While attention to the riparian zone is appropriate, the structural complexity and habitat diversity of the uplands also plays a large role in maintaining our native fauna.

After harvest, only the southern red-backed vole showed a trend of declining captures on riparian transects as discussed above. On the uplands, statistically significant declines were observed for the marsh shrew, Trowbridge's shrew, shrew-mole, and forest deer mouse. Statistically significant increases were observed for the creeping vole. It is likely that the deer mouse also increased on the uplands, but difficulty in identifying juvenile deer mice to species obscured the objectivity of the test.

Because species favoring early successional and edge habitats replaced species associated with closed canopy forest during the post-harvest period, measures of species richness and evenness not particularly useful in evaluating treatments. They indicated no change when in fact the small mammal communities were quite distinct. To assess change in the small mammal community one must measure the performance of individual species over time and document the shifting composition of the small mammal communities.

Overall, the buffers did make a difference. Over the first 2 yr after harvest they retained a high proportion of the fauna associated with closed canopy forest. Whether they will continue to do so over the next several years remains to be seen.

MANAGEMENT IMPLICATIONS

There are really two major concerns. The first is for the well being of the riparian obligate species—the water shrew and to a lesser extent the marsh shrew. The second is that the timeline of persistence within buffers for species associated with closed canopy forest has not been described fully. This must be done if the adequacy of buffer designs is to be evaluated.

The shrews present different situations. In western Washington the water shrew is found mainly at elevations in excess of about 800 m. Much of the State and private forest lands lie below this elevation, so this does not present a consistent problem throughout the region. Only four individual:; were captured in the 4 yr of this study. The species is, however, completely dependent upon the riparian zone. It feeds above and below water, lives in streamside habitat, and rarely ventures far from the riparian zone. Measures that would maintain the aquatic environment in an acceptable State for long-term persistence of native fish and invertebrates would suit the water shrew. They require a successful aquatic strategy. The marsh shrew is not so closely tied to water as is the water shrew. It is often caught at distances of a kilometer from riparian environments and is much more common than the water shrew at lowland elevations. It may move farther from riparian zones in moist rather than dry forest stands, and uses upland habitats rather extensively. As such, it is directly subject to the effects of harvest in the uplands. We captured 91 marsh shrews in this study. The marsh shrew responds negatively to forest canopy removal. Harvested uplands most likely will not provide appropriate habitat until canopy closure. During this time inhabitation of the buffer will be reduced sharply from pre-harvest conditions. Whether marsh shrews will persist in buffers beyond the first 2 yr remains to be seen. The answer to this question requires additional sampling during the pre-canopy period.

A very good baseline has been established from which to evaluate the performance of these buffer designs. An adequate assessment, however, requires sampling at intervals during the pre-canopy years. Several species showed declines over the 2-yr period. Knowing whether they persist on these sites during the pre-canopy period requires additional sampling. We suggest a return to the sites at least twice at 5-yr intervals from the last sample (1996 for most sites). The first 10-yr period will be the most difficult for forest associated species. Should they persist through the first period, it seems likely that they could survive the second decade, when conditions should be improving.

During subsequent buffer assessments, the structural dynamics of the buffers should be investigated. One of the more challenging features of buffer design is creating buffers that will retain their initial configurations for several years. Blow down is a major problem of narrow leave strips, and this is pronounced within riparian zones where soils may be saturated and tree fall is part of the normal functioning of the system. It is even more difficult when buffers are first created as the trees are least wind firm. Once a buffer is established and the older trees become wind firm, subsequent rotations should experience less blow down. It would be very useful to have the buffers mapped for trees, snags, and large down wood at each wildlife assessment. This would provide a basis for tracking changes in the buffers over time. We have mapped a third of the sites, but all of them should be done fairly soon so that the time line of recent changes could be reconstructed

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Table 1. Total number of small mammals captured on all 18 sites. Species acronyms are used in subsequent figures.

Taxa	Number caught
Insectivores	
Vagrant shrew (*Sorex vagrans*) SOVA	247
Montane shrew (*S. montanus*) SOMO	487
Water shrew (*S. palustris*) SOPA	4
Marsh shrew (*S. bendirii*) SOBE	95
Trowbridge's shrew (*S. trowbridgii*) SOTR	2,258
Masked shrew (S. *cinereus*) SOCI	2
Unidentified shrew SOSP	34
Shrew-mole (*Neurotrichus gibbsii)* NEGI	396
Coast mole (*Scapanus orarius*) SCOR	5
Total insectivores	3,528
Rodents	
Townsend's chipmunk (*Tamias townsendii)* TATO	11
Northern flying squirrel (*Glaucomys sabrinus)* GLSA	6
Pacific jumping mouse (*Zapus trinotatus*) ZATR	105
Deer mouse (*Peromyscus maniculatus*) PEMA	821
Forest deer mouse *(P. keeni*) PEKE	1,515
Unidentified deer mouse PESP	1,724
Southern red-backed vole (*Clethrionomys gapperi*) CLGA	87
Creeping vole (*Microtus oregoni*) MIOR	1,015
Long-tailed vole (*M. longicaudus*) MILO	268
Townsend's vole (*M. townsendii)* MITO	3
Unidentified vole MISP	69
Total rodents	5,624
Carnivores	
Ermine(*Mustela erminea*) MUER	11
Total mammals	9,163

Table 2. Total number of small mammals captured on riparian and upland transects on unharvested sites in all years.

Taxa	Number caught		
	Riparian	Upland	Total
Insectivores			
Vagrant shrew (*Sorex vagrans*) SOVA	89	28	117
Montane shrew (S. *montanus*) SOMO	177	124	301
Water shrew (S. *palustris*) SOPA	1	1	2
Marsh shrew (*S. bendirii)* SOBE	29	24	53
Trowbridge's shrew (*S. trowbridgii*) SOTR	573	609	1,182
Masked shrew (*S. cinereus*) SOCI	1	1	2
Unidentified shrew SOSP	11	9	20
Shrew-mole (*Neurotrichus gibbsii)* NEGI	114	88	202
Coast mole *(Scapanus orarius*) SCOR	2	0	2
Total Insectivores	997	884	1,881
Rodents			
Northern flying squirrel (*Glaucomys sabrinus)* GLSA	1	3	4
Pacific jumping *mouse* (*Zapus trinotatus*) ZATR	12	2	14
Deer mouse *(Peromyscus maniculatus*) PEMA	133	209	342
Forest deer mouse *(P. keeni*) PEKE	458	574	1,032
Unidentified deer mouse PESP	98	181	279
Southern red-backed vole (*Clethrionomys gapperi)* CLGA	9	59	68
Creeping vole (*Microtus oregoni*) MIOR	141	111	252
Long-tailed vole (*M. longicaudus*) MILO	99	4	103
Unidentified vole MISP	9	0	9
Total Rodents	960	1143	2,103
Total Mammals	1,957	2,027	3,984

Table 3. Mean (SE) number of mammals caught per 100 traps nights on riparian and upland transects before harvest. Indices are averaged over all years on Control sites and over the 2 pre-treatment yr for treatment sites. Indices combine pitfall and corrected snap trap data.

Species	Riparian	Upland
Vagrant shrew	0.54 (0.16)	0.25 (0.08)
Montane shrew	0.99 (0.17)**	0.67 (0.12)
Marsh shrew	0.29 (0.04)**	0.14 (0.03)
Trowbridge’s shrew	3.35 (0.23)	3.66 (0.22)
Shrew-mole	0.68 (0.11)	0.60(0.12)
Pacific jumping mouse	0.188 (0.03)***	0.034 (0.02)
Deer mouse	0.97 (0.43)**	1.45 (0.32)
Forest deer mouse	2.48 (0.50)*	3.141 (0.38)
Southern red-backed vole	0.06 (0.02)**	0.44 (0.17)
Creeping vole	0.753 (0.25)	0.716 (0.13)
Long-tailed vole	0.651 (0.11)***	0.055 (0.03)

* = $P < 0.05$; ** = $P < 0.01$; *** = $P < 0.001$; paired t-test

Table 4. Multiple correlations between mean capture rate and habitat variables that were significantly different between riparian and upland transects in the pre-harvest period. Tabled values are standardized correlation coefficients. Species acronyms as in Table 1.

Variables'	SOVA	SOMO	SOBE	SOTR	NEGI	ZATR	PEMA	PEKE	CLGA	MIOR	MILO
Herb	0.34*	0.31*					-0.44**	-0.40*	-0.033*	0.56***	
Fern										-0.49*	
Litter	0.70***				0.51**				0.30*		
Litter Depth		0.38**					0.41*	-0.54***			
Soil	0.26*										
Rock		-0.3 1*	0.51*				-0.34*				0.22*
OD Shrub				-0.46**		-0.32*					
BP Shrub				-0.43**		0.66**					
Canopy		-0.29*		-0.55**	-0.69***	0.40*	-0.56**				
PSME Small							0.32*		0.54***		
PSME Large	-0.48***	-0.32**		-0.31*					-0.29*		0.32**
TSHE Small					0.65***			0.47**	-0.49***	-0.43*	
ALRU Small	0.81***	0.30*			0.58**						0.29*
Multiple R^2	0.683	0.703	0.162	0.497	0.603	0.419	0.506	0.465	0.728	0.292	0.67

Percentage cover: Herb, Fern, Litter, Soil, Rock, OD Shrub (Other deciduous shrub), BP Shrub (Berry-producing shrub), Canopy
Counts: PSME Small (Douglas-fir <50 cm diameter), PSME Large, (>50 cm), TSHE (western hemlock), ALRU (red alder)
Litter depth measured in mm.
* = $P < 0.05$; ** = $P < 0.01$; *** = $P < 0.001$.

Table 5. Differences by treatment in mean (se) capture rates (number caught per 100 trap nights) before and after harvest. Indices are derived from mean capture rates before and after harvest. Tabled values are $\bar{x}_{\text{Post}} - \bar{x}_{\text{Pre}}$. Indices combine pitfall and corrected snap trap data. Superscripts indicate significant differences among treatments for the indicated riparian or upland transect. Indices without superscripts or with shared superscripts are not significantly different. * = $P < 0.05$; ** = $P < 0.01$; *** = $P < 0.001$; Tukey ***HSD.***

Species	Control	Modified	State
Vagrant shrew			
Riparian	-0.505 (0.341)	-0.322 (0.157)	0.149 (0.210)
Upland	-0.199 (0.248)	0.543 (0.252)	0.144 (0.142)
Montane shrew			
Riparian	-0.330 (0.331)	-0.649 (0.271)	-0.420 (0.329)
Upland	-0.153 (0.321)	-0.284 (0.163)	-0.024 (0.210)
Marsh shrew			
Riparian	0.034 (0.190)	-0.164 (0.071)	-0.135 (0.078)
Upland*”	0.063 (0.047)^{a}	-0.279 (0.047)^{b}	-0.099 (0.074)^{ab}
Trowbridge’s shrew			
Riparian	1.010 (0.454)	-0.167 (0.518)	-0.901 (0.805)
Upland**	1.060 (0.543)”	-1.141 (0.495)^{b}	-0.892 (0.288)^{b}
Shrew-mole			
Riparian	-0.085 (0.132)	0.202 (0.557)	-0.135 (0.149)
Upland*	0.375 (0.251)^{a}	-0.265 (0.140)^{ab}	-0.611 (0.342)^{b}
Pacific jumping mouse			
Riparian	0.036 (0.173)	0.277 (0.152)	0.971 (0.573)
Upland	0.197 (-)	0.201 (0.175)	0.267 (0.140)
Deer mouse			
Riparian	0.169 (0.156)	0.045 (0.534)	0.606 (0.254)
Upland	0.732 (0.265)	0.803 (0.336)	1.218 (0.585)
Forest deer mouse			
Riparian	0.371 (0.424)	-1.616 (1.067)	-1.044 (1.123)
Upland**	1.434 (1.124)^{a}	-2.829 (0.743)^{b}	-2.647 (0.782)^{b}

bat flying within the range of the detector multiple times, or multiple bats flying over the detector once. Detection rates, therefore, can only provide an index of relative use by bats at different sites (Thomas and West 1989).

Ultrasonic detection has been used successfully in certain regions to identify bats based on species-specific call characteristics (Fenton 1970, Fenton 1982, Fenton and Bell 1981). However, intraspecific variation in search phase echolocation calls makes identification tentative for many species (Obrist 1995, Thomas et al. 1987). Evidence suggests that there can be substantial variation in pulse characteristics emitted by an individual bat (Schrumm et al. 1991) as well as among bats of the same species (Thomas et al. 1987). Unfortunately, several bat species in the Pacific Northwest have similar echolocation calls making species identification difficult if not impossible, especially within the *Myotis* genus. For analyses in this study, we identified detections as *Myotis* (*M. californicus, M. evotis, M. keenii, M. lucifugus, M. thysanodes, M. volans* and M. *yumanensis*), or non-*Myotis* (*E. fuscus, L. noctivagans, L. cinereus* and *P. townsendii*) due to considerable overlap in call characteristics and the predominance of *Myotis* detections. Members of the *Myotis* group are small, slow flying, agile bats that have steep frequency modulated echolocation calls with lowest frequencies ranging from 35 to 55 kHz (Fig. la). The larger non-*Myotis* bats are characterized as faster, less maneuverable species with lower echolocation frequencies generally below 35 kHz (Neuweiler 1989) (Fig. lb). Separation between these groups, although coarse, does provide insight into ecological differences based on morphology and echolocation call design. In spite of its limitations, ultrasonic detection is a valuable tool for surveying free-flying bats and it is the most appropriate method for simultaneously assessing patterns of bat activity at several sites.

Table 6. Multiple correlations between mean capture rate and habitat variables that were significantly different between treatments in the post-harvest period. Tabled values are standardized correlation coefficients. Species acronyms as in Table 1.

Variables'	SOVA	SOMO	SOBE	SOTR	NEGI	ZATR	PEMA	PEKE	CLGA	MIOR	MILO
Fern								0.59*			
Soil	0.74***			0.50*		-0.49		0.46'	0.50*	0.64*	
Moss	0.61**					0.67**					
BP Shrub						-0.99***	0.66*				
Canopy		0.49**			0.86**		-0.55*	0.84*			
Sm. Log DC3						0.31*					
0.72**											
ACCI Small		0.40**		0.48*		0.30*					
TSHE Small		-0.56**				-0.78***				0.48*	
TSHE Large	-0.84***		0.71**	0.50	-0.78*			-0.89**		-1.05**	
PSME Large		-0.33*	-0.55*								
Multiple R^2	0.588	0.826	0.463	0.602	0.472	0.870	0.309	0.630	0.251	0.487	0.512

Percent cover: Fern, Soil, Moss, BP Shrub (Berry-producing shrub), Canopy, Small Log DC3 (decay class 3)
Counts: ACCI Small (vine maple <50 cm diameter), TSHE Small (western hemlock < 50 cm diameter), TSHE Large, (> 50 cm), PSME Large (Douglas-fir > 50 cm diameter)
* = $P < 0.05$; ** = $P < 0.01$; *** = $P < 0.001$.

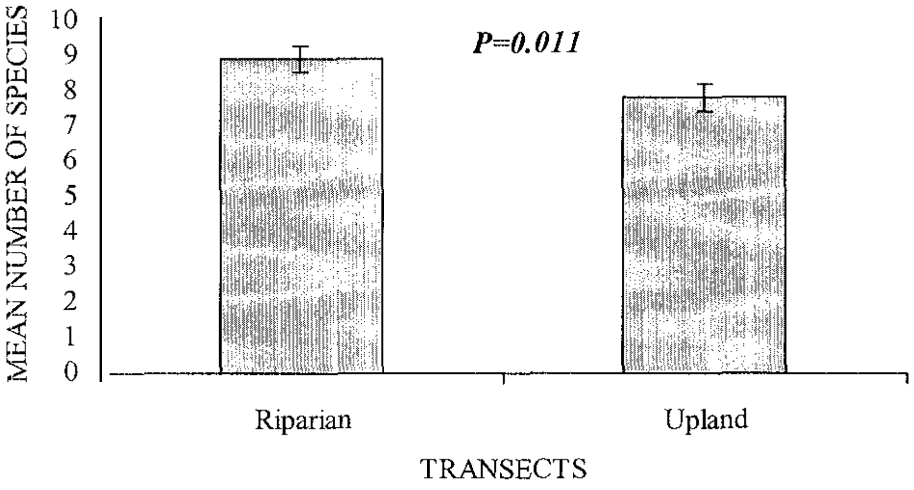
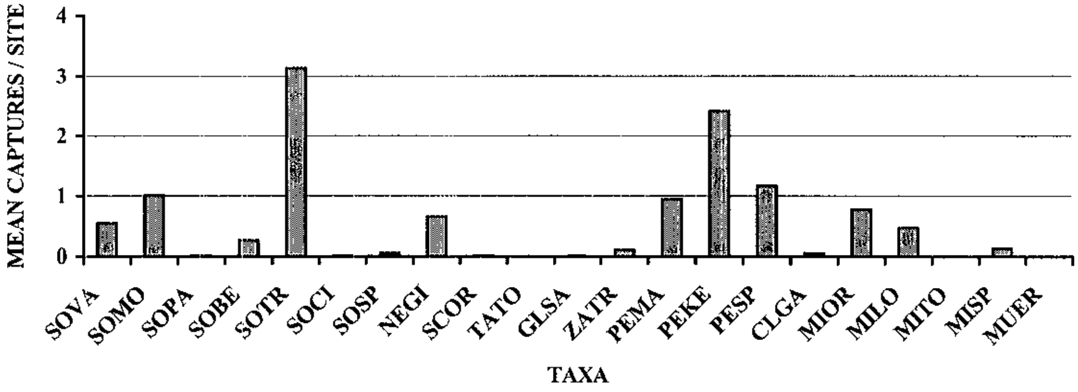
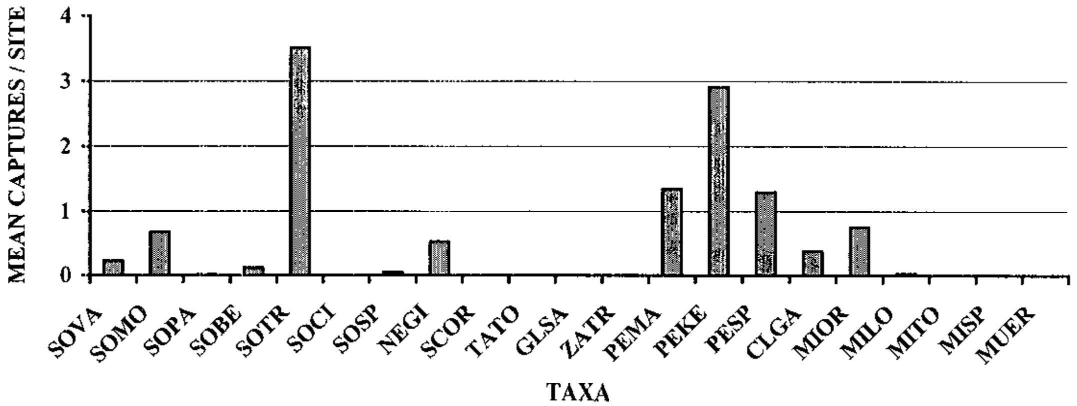


Figure 1. Species richness (mean ± se) on riparian and upland transects for all sites before harvest. Richness calculated from combined pitfall and snap trap data.



A. Riparian Transects-Pre-harvest



B. Upland Transects-Pre-harvest

Figures 2A and 2B. Species composition (mean captures per 100 trap nights per site, n = 18) on riparian and upland transects before harvest. Species acronyms as in Table 1.

Riparian Transects-Pre-harvest Conditions

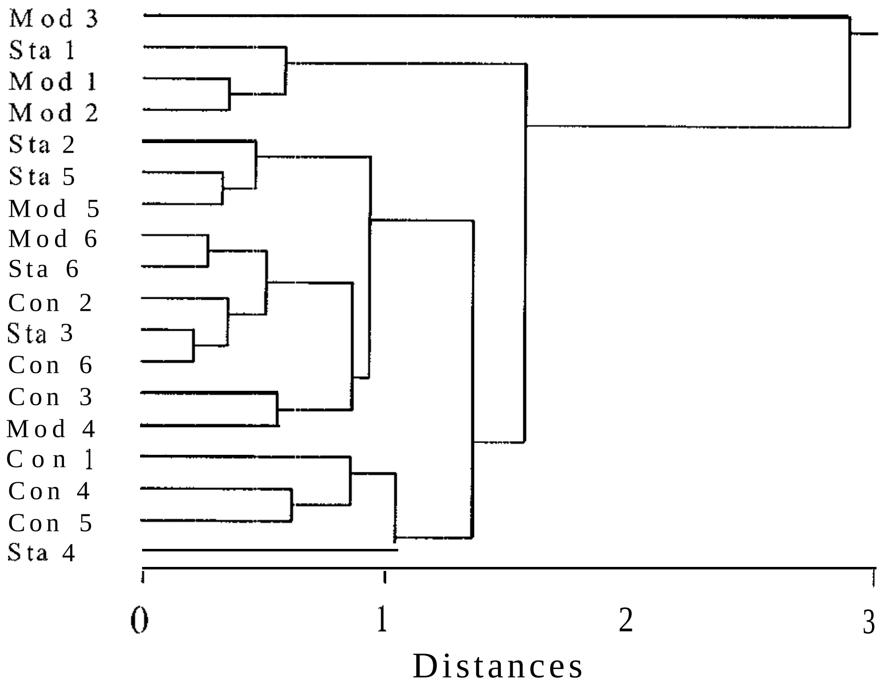


Figure 3. Hierarchical clustering of sites based upon pre-harvest mean capture rates of all small mammal species on riparian transects. Sites with similar small mammal communities join to the left; sites with dissimilar communities to the right,

Con 1 = Abernathy Creek
Con 2 = Elbe Hills
Con 3 = Hotel Cr
Con **4** = Porter Cr
Con 5 = Taylor Cr
Con 6 = Vail

Mod 1 = Blue Tick
Mod 2 = Eleven Cr 31
Mod 3 = Griffin Cr
Mod 4 = Ms. Black
Mod 5 = Ryderwood 860
Mod 6 = Side Rod

Sta 1 = Eleven Cr 32
Sta 2 = Kapowsin
Sta 3 = Night Dancer
Sta 4 = Pot Pourri
Sta 5 = Ryderwood 1557
Sta 6 = Simmons Cr

Upland Transects-Preharvest Conditions

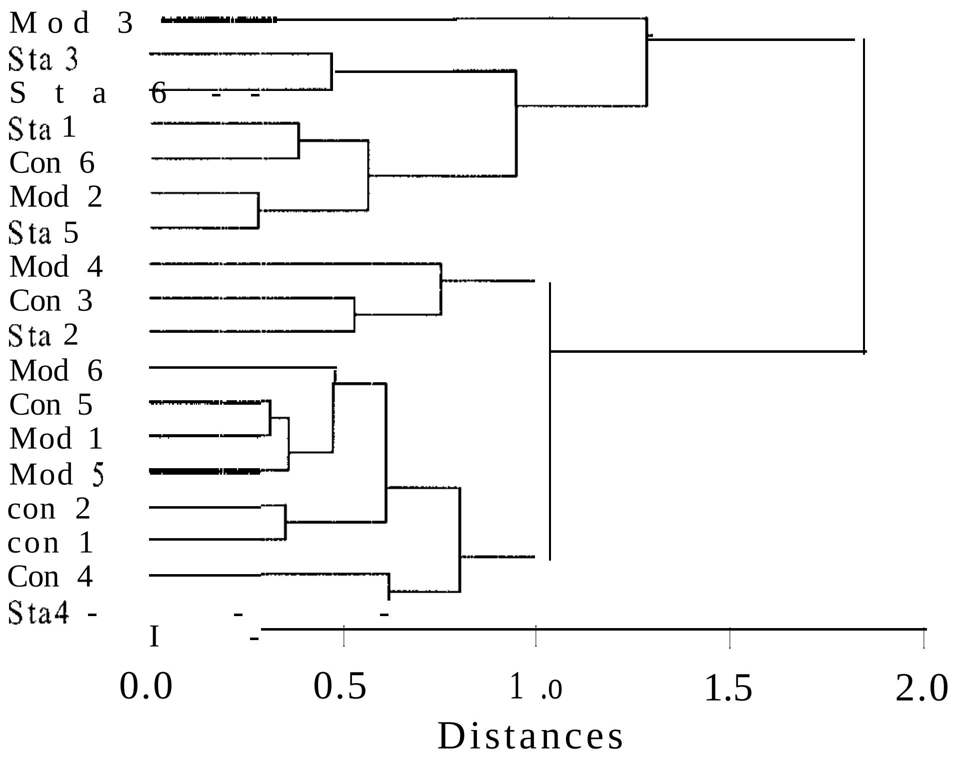


Figure 4. Hierarchical clustering of sites based upon pre-harvest mean capture rates of all small mammal species on upland transects. Sites with similar small mammal communities join to the left; sites with dissimilar communities to the right. Site identities as in Figure 3.

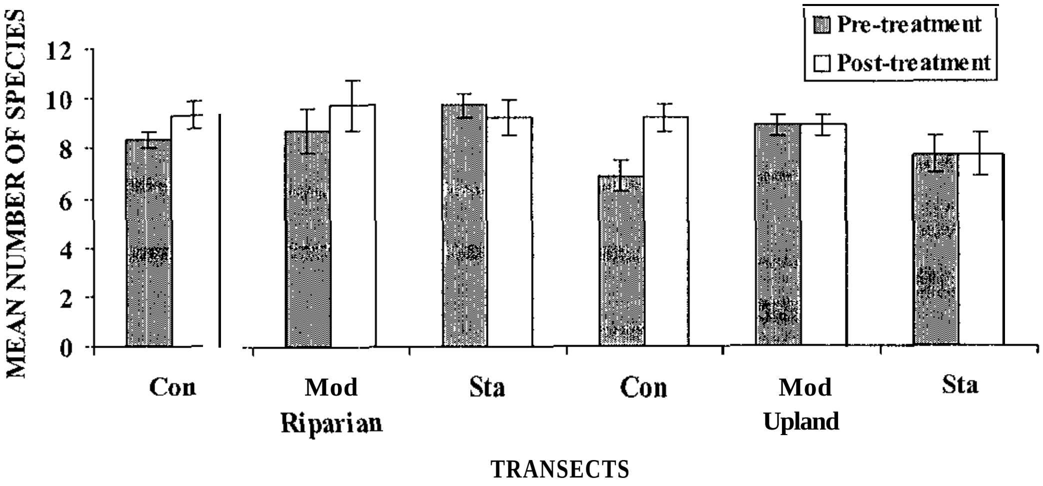


Figure 5. Species richness (mean ± SE) before and after harvest on riparian and upland transects by treatment. Richness calculated from combined pitfall and snap trap data. Mean richness was tested separately for riparian and upland transects and found to be not statistically different among treatments.

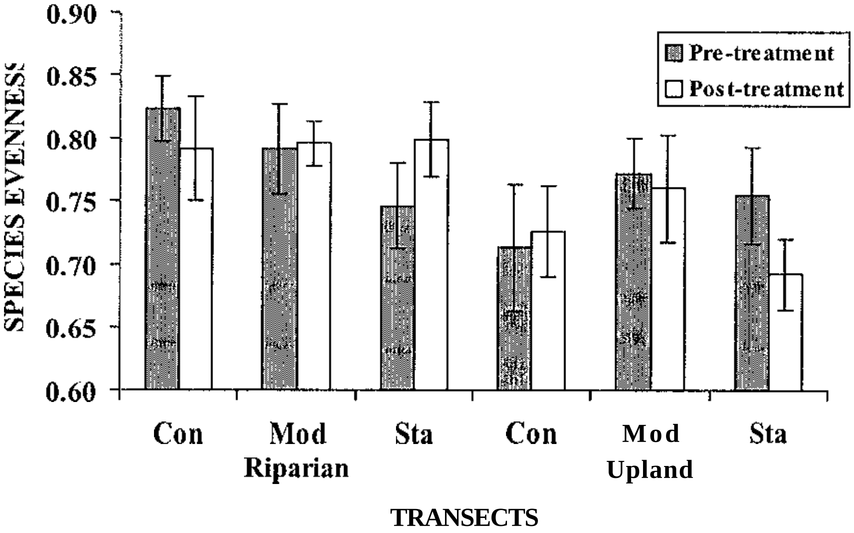
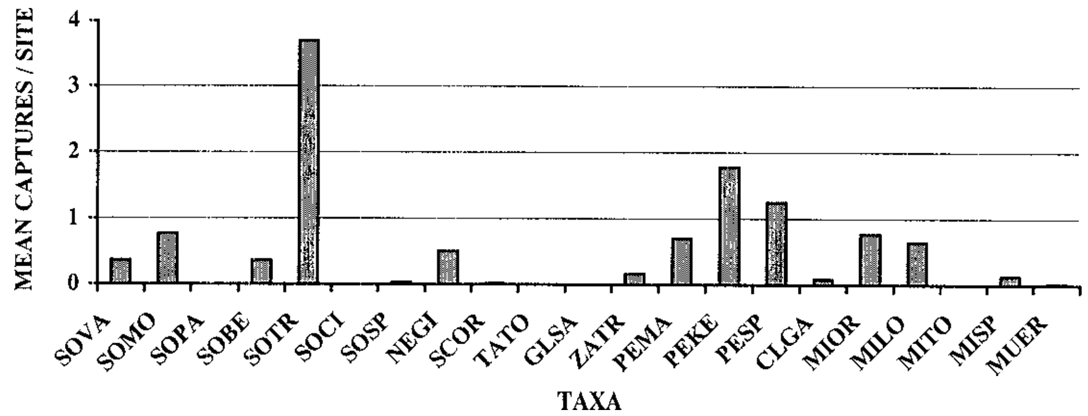
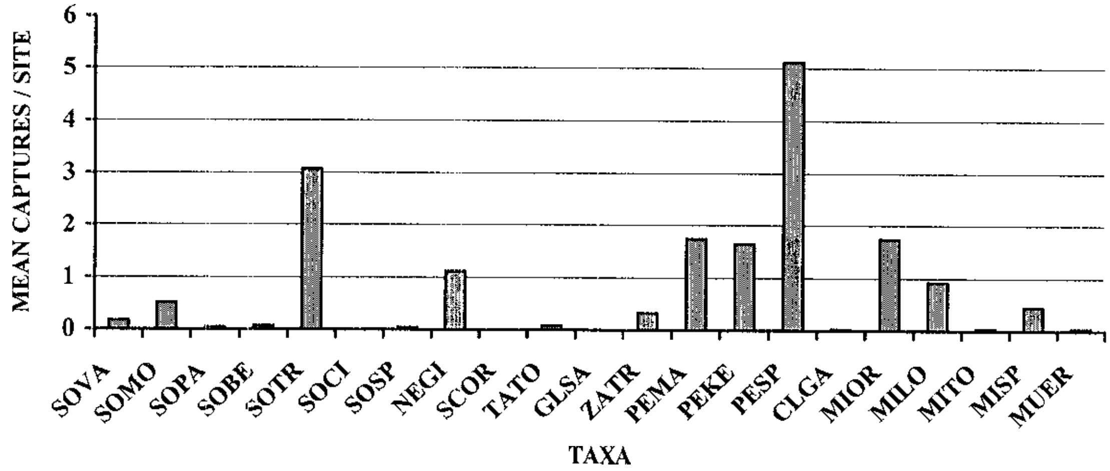
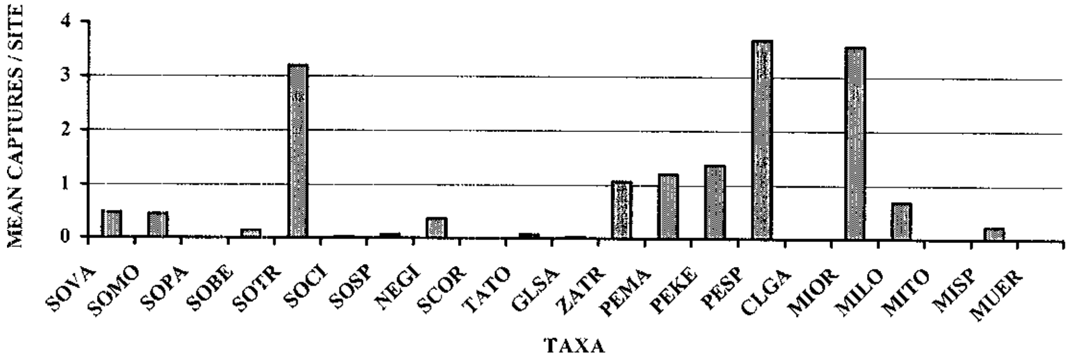


Figure 6. Species evenness (mean *H′*± SE) before and after harvest on riparian and upland transects by treatment. Evenness calculated from combined pitfall and snap trap data. Mean evenness was tested separately for riparian and upland transects and found to be not statistically different among treatments.


A. Riparian Transects Post-harvest--Controls


B. Riparian Transects Post-harvest-- Modified


C. Riparian Transects Post-harvest-State

Figures **7A** – 7C. Species composition (mean captures per 100 trap nights per site, $n = 6$) on riparian transects after harvest. Species acronyms as in Table 1.

Riparian Transects-Post-harvest Conditions

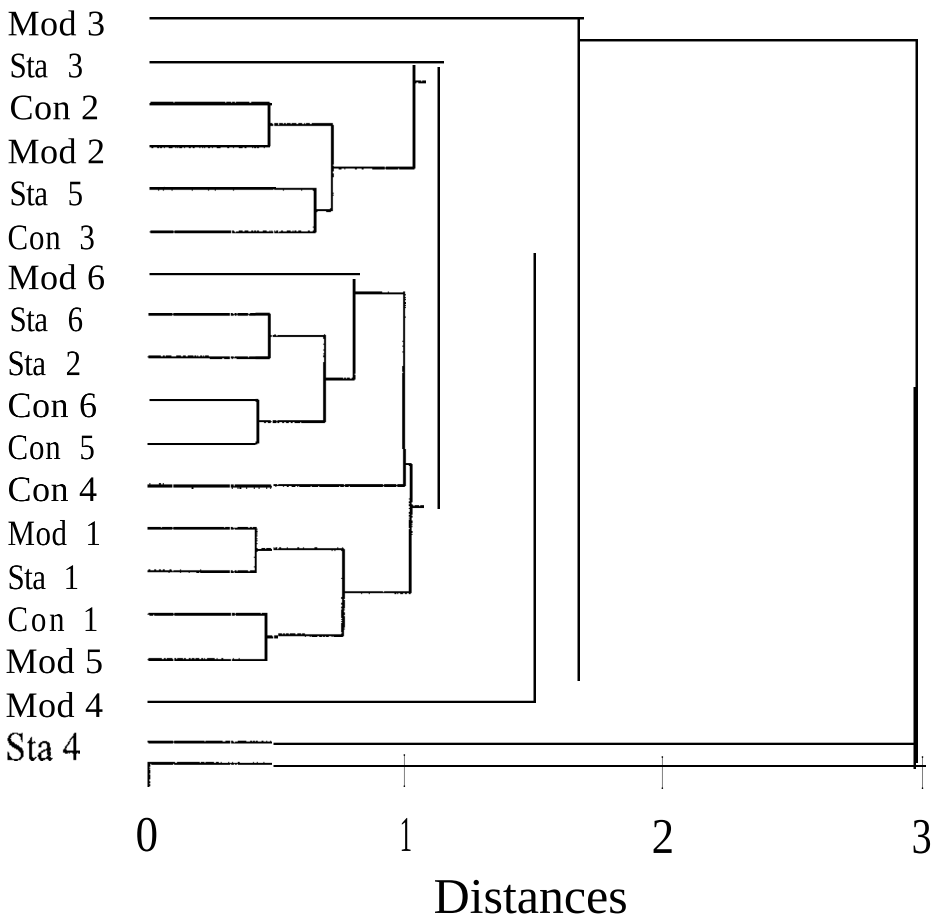


Figure 8. Hierarchical clustering of sites based upon post-harvest mean capture rates of all small mammal species on riparian transects. Sites with similar small mammal communities join to the left; dissimilar sites to the right. Site identities as in Figure 3.

Southern Red-backed Vole *(Clethrionomys gapperi)*

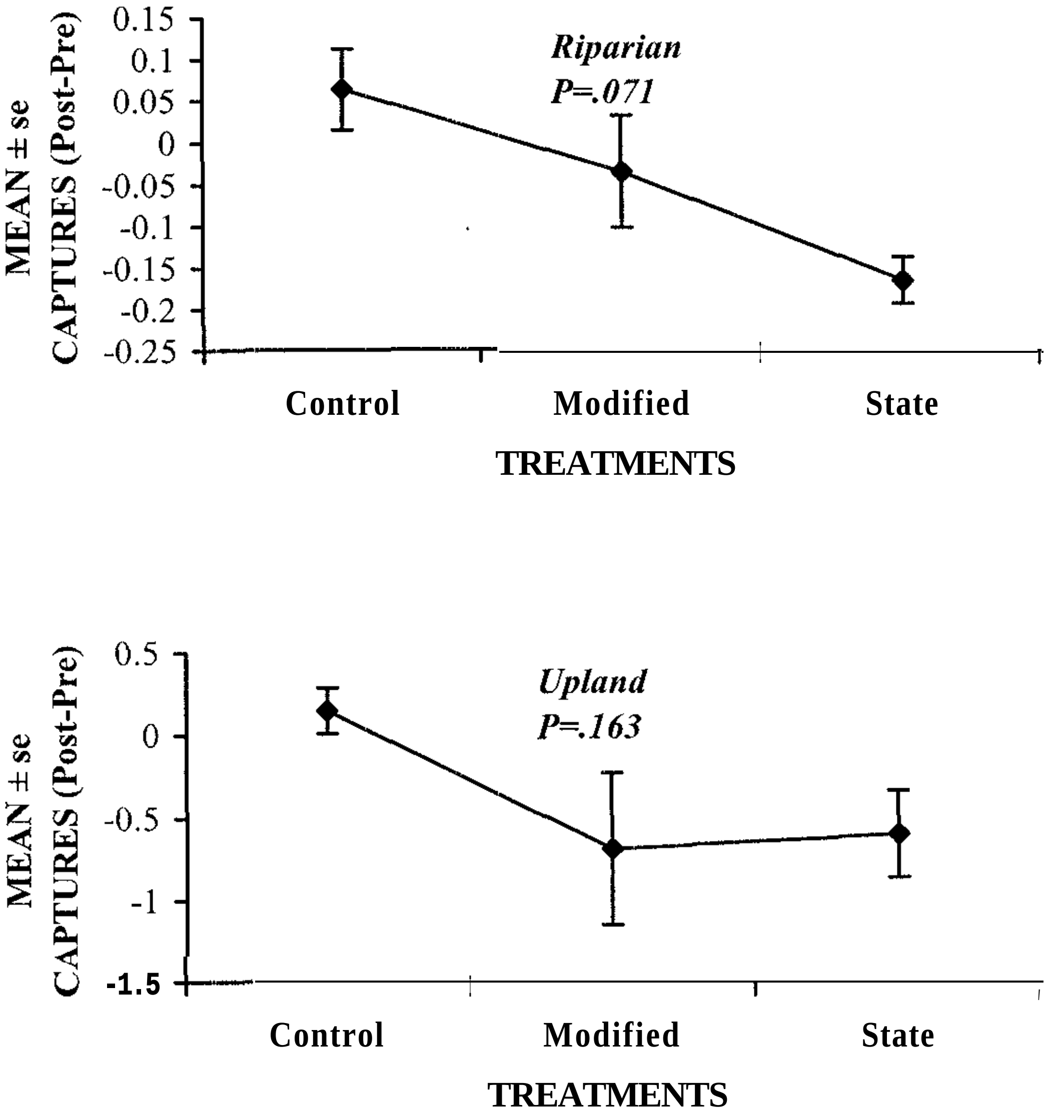
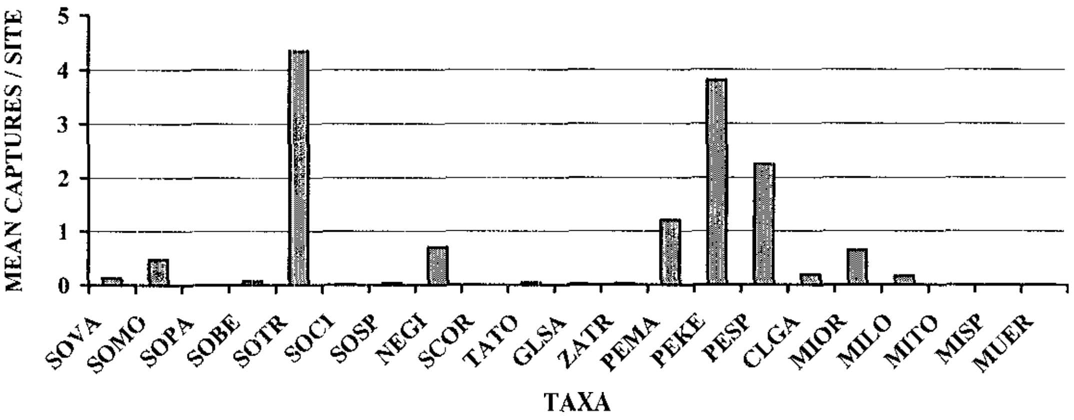
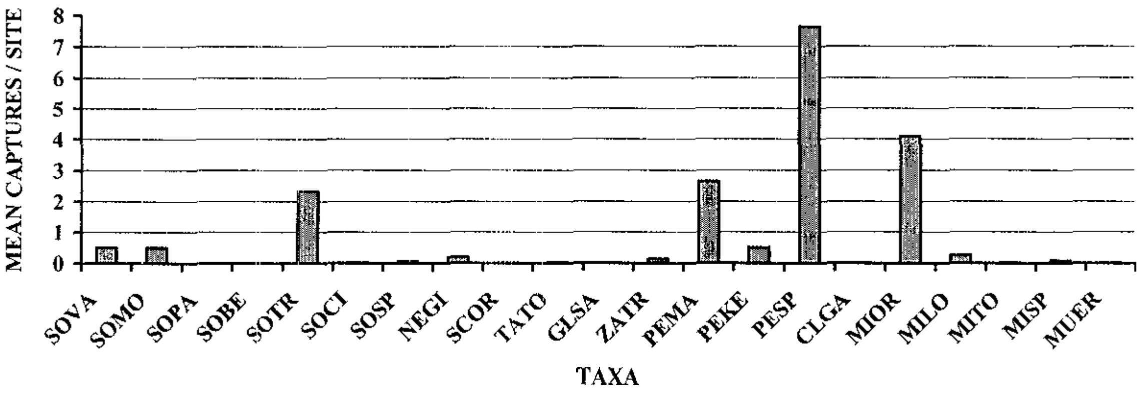


Figure 9. Differences by treatment in mean (± SE) capture rates (number caught per 100 trap nights) for the southern red-backed vole before and after harvest for riparian (above) and upland transects (below). Indices are derived from mean capture rates before and after harvest. Charted values are $\bar{x}_{\text{Post}} - \bar{x}_{\text{Pre}}$. Indices combine pitfall and corrected snap trap data.

A. Upland Transects Post-harvest--Controls



B. Upland Transects Post-harvest-Harvested Sites



Figures 10A and 10B. Species composition (mean captures per 100 trap nights per site) on upland transects after harvest. Control sites, *n* = 6; Harvested sites, *n* = 12. Both harvest treatments combined. Species acronyms as in Table 1.

Upland Transects-Post-harvest Conditions

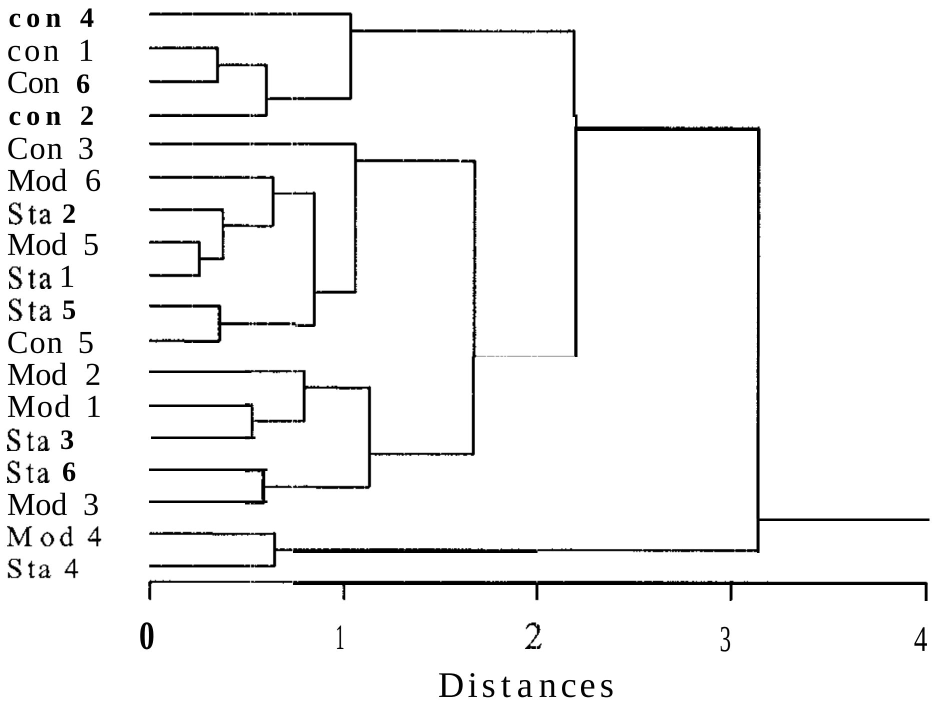


Figure 11. Hierarchical clustering of sites post-harvest mean capture rates of all small-mammal species on upland transects. Sites with similar small mammal communities join to the left; dissimilar sites to the right. Site identities as in Figure 3.

Marsh Shrew (*Sorex bendirii*)

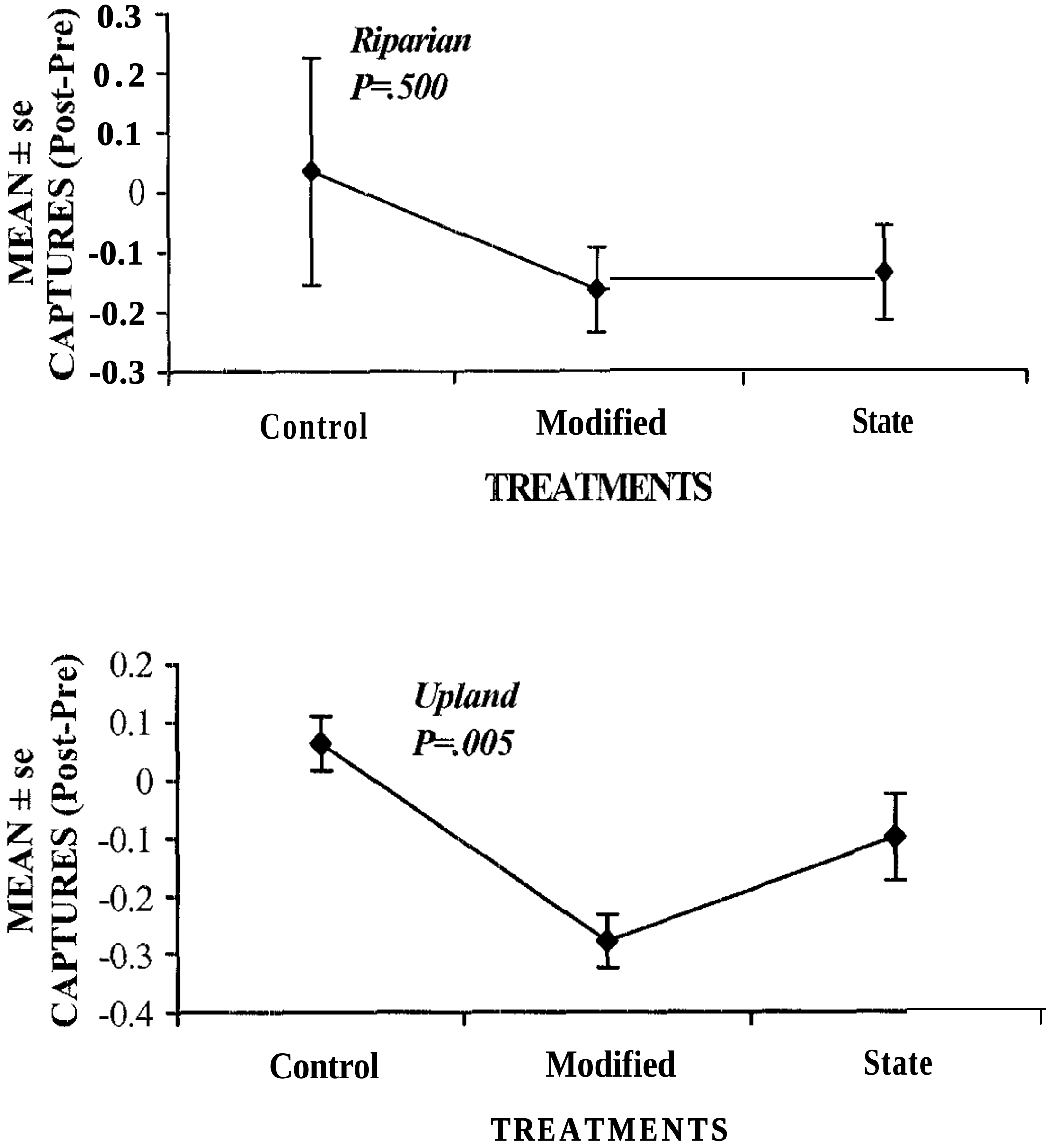


Figure 12. Differences by treatment in mean (SE) capture rates (number caught per 100 trap nights) for the marsh shrew before and after harvest for riparian (above) and upland transects (below). Indices are derived from mean capture rates before and after harvest. Charted values are $\bar{x}_{Post}-\bar{x}_{Pre}$. Indices combine pitfall and corrected snap trap data.

Trowbridge's Shrew (*Sorex trowbridgii*)

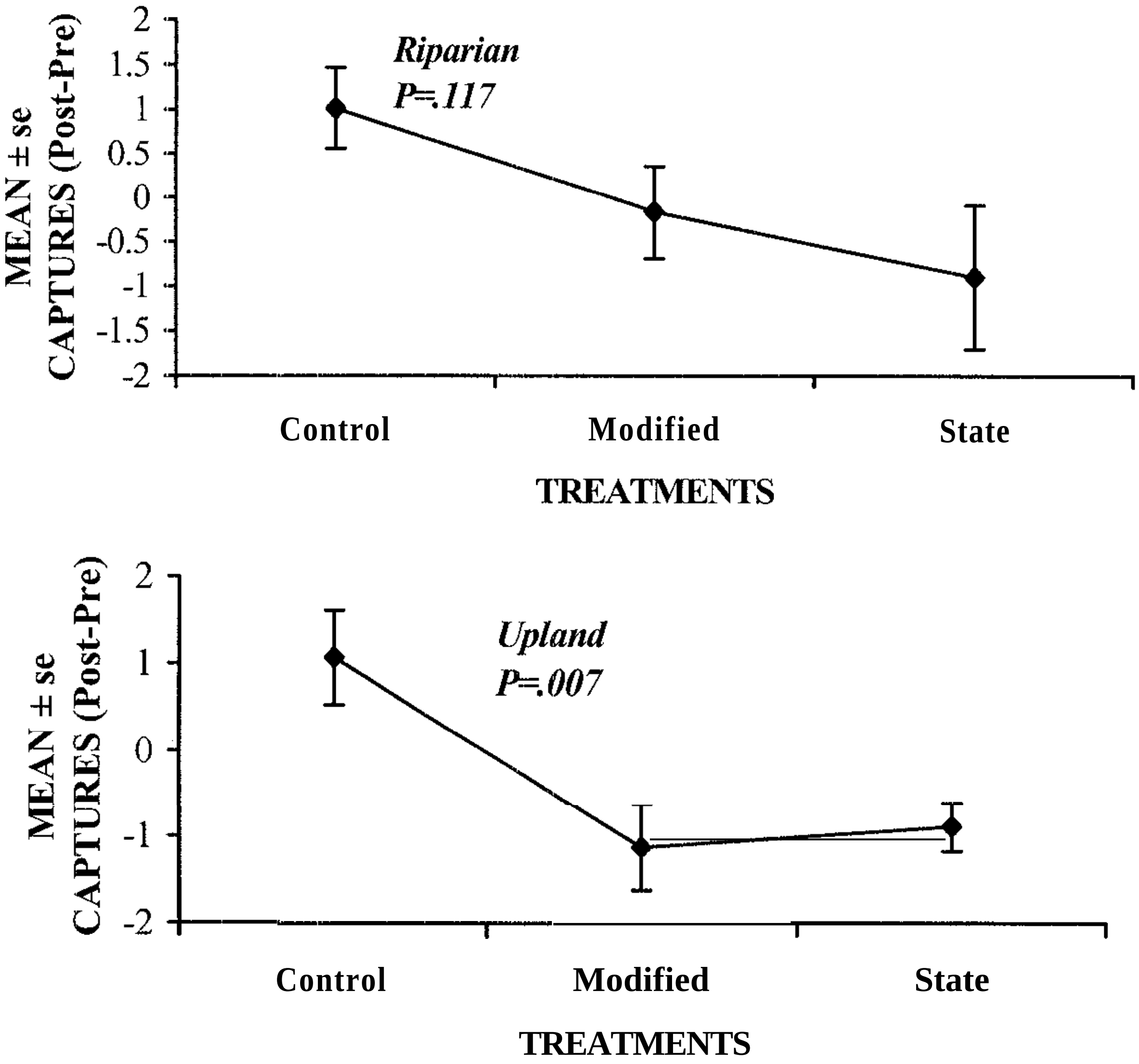


Figure 13. Differences by treatment in mean (se) capture rates (number caught per 100 trap nights) for the marsh shrew before and after harvest for riparian (above) and upland transects (below). Indices are derived from mean capture rates before and after harvest. Charted values are $\bar{x}_{\text{Post}} - \bar{x}_{\text{Pre}}$. Indices combine pitfall and corrected snap trap data.

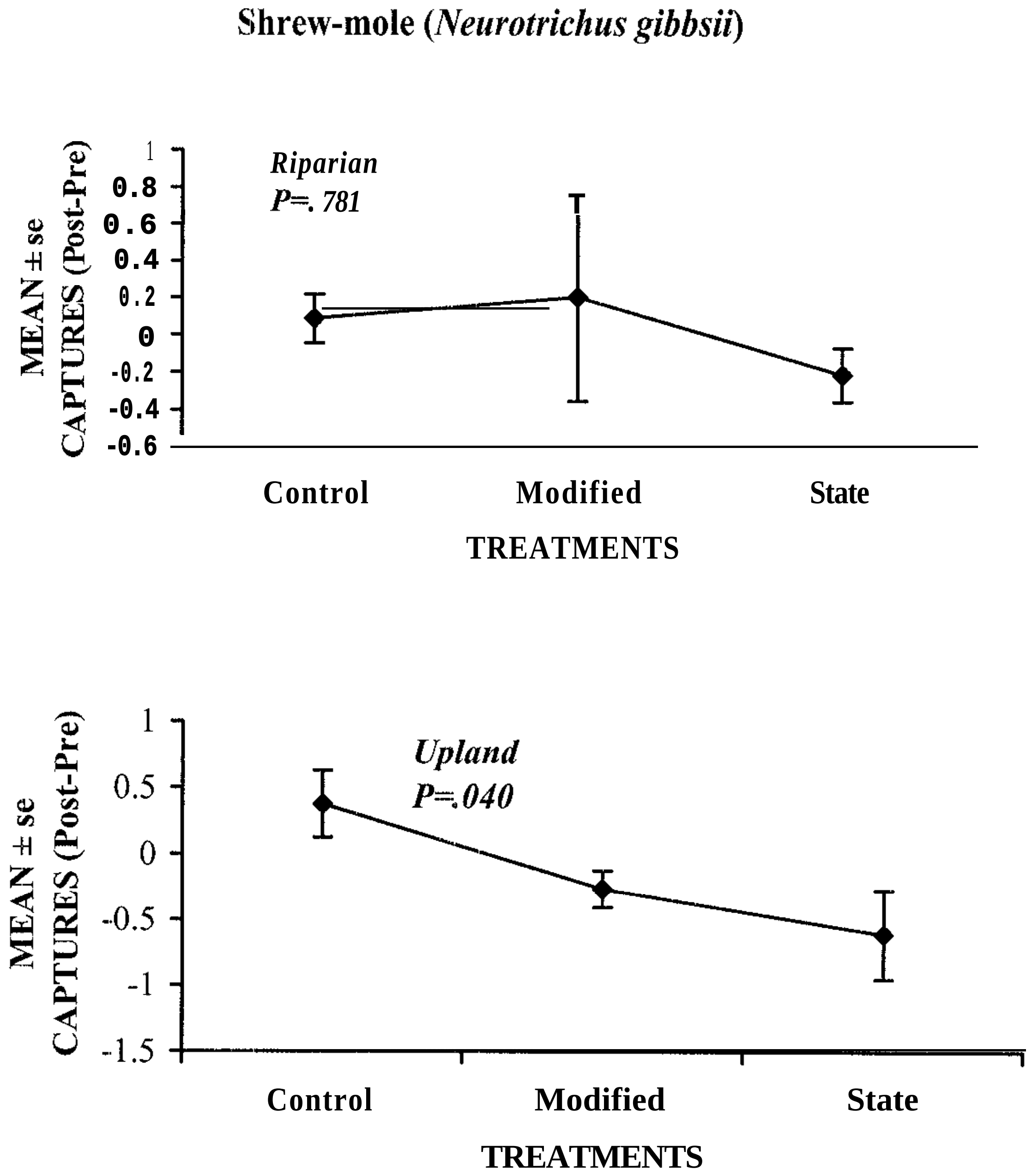
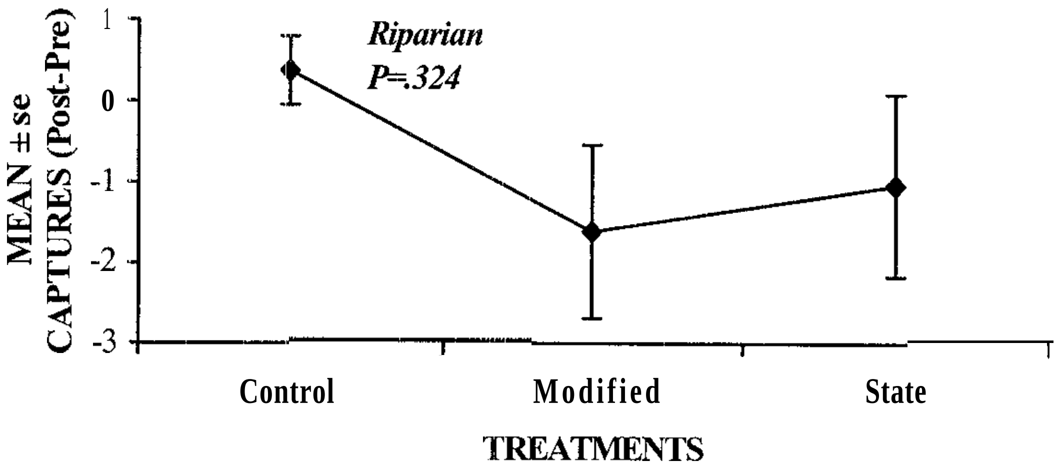


Figure 14. Differences by treatment in mean (se) capture rates (number caught per 100 trap nights) for the shrew mole before and after harvest for riparian (above) and upland transects (below). Indices are derived from mean capture rates before and after harvest. Charted values are $\bar{x}_{\text{Post}} - \bar{x}_{\text{Pre}}$. Indices combine pitfall and corrected snap trap data.

Forest: Deer Mouse (*Peromyscus keeni*)



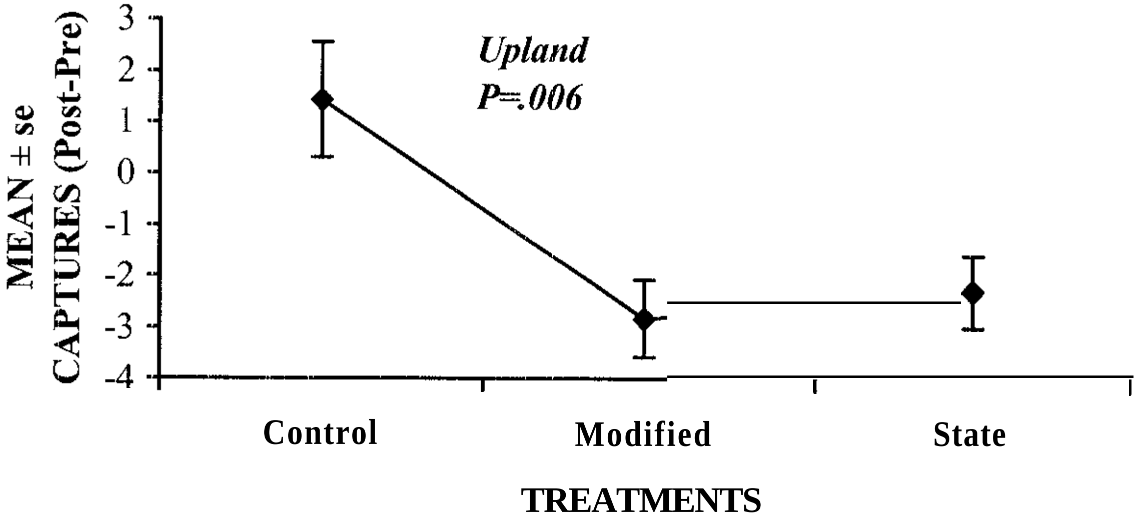


Figure 15. Differences by treatment in mean (se) capture rates (number caught per 100 trap nights) for the forest deer mouse before and after harvest for riparian (above) and upland transects (below). Indices are derived from mean capture rates before and after harvest. Charted values are $\bar{x}_{Post} - \bar{x}_{Pre}$. Indices combine pitfall and corrected snap trap data.

Creeping Vole (*Microtus oregoni*)

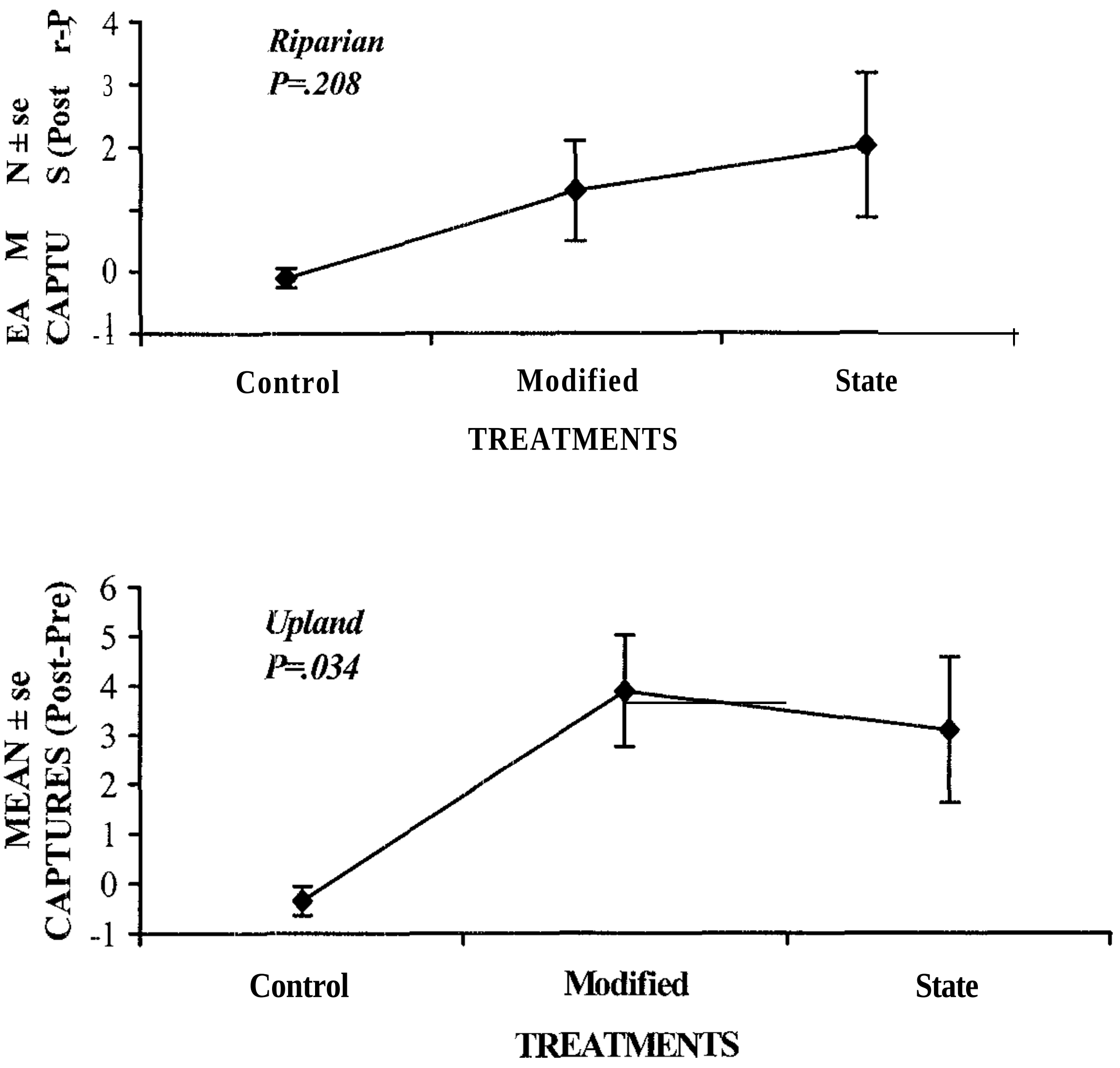


Figure 16. Differences by treatment in mean (se) capture rates (number caught per 100 trap nights) for the creeping vole before and after harvest for riparian (above) and upland transects (below). Indices are derived from mean capture rates before and after harvest. Charted values are $\bar{x}_{Post} - \bar{x}_{Pre}$. Indices combine pitfall and corrected snap trap

Chapter 11

EAST-SIDE SMALL-MAMMAL SURVEYS

Abstract. Current forest management practices require retention of trees along streams to create riparian buffer strips, which may maintain unique microhabitats for some vertebrate species. We experimentally examined the effects of two prescriptions for such strips on small-mammal populations in northeastern Washington: current Washington state guidelines for Riparian Management Zones (State) and a modified prescription that buffered snags and seeps in the riparian zone (Modified). We studied 18 streams including 7 unharvested Controls, 6 State harvest sites, and 5 Modified harvest sites. Two 710-m transects were established parallel to the stream in riparian and upland habitats. Small mammals were sampled during spring 1992-1993 &e-harvest), 1994 (harvest), and 1995-1996 (post-harvest) with pitfall and snap traps. Pour species (*Clethrionomys gapperi*, *Sorex vagrans*, *Peromyscus maniculatus*, and *Sorex cinereus*) were most broadly distributed and were captured most frequently (91.5% of 13,081 captures of 21 species). In all years, overall abundance was greater in riparian than in upland habitat. Before harvest, species diversity, evenness, and richness were similar in both habitats. In 1994, overall abundance increased sharply, but declined by 1995 and remained relatively constant in 1996, near pre-harvest levels. These changes in abundance were paralleled by the mean body mass and mean number of embryos per female for *C. gapperi*, which were both significantly greater in 1994 than in the pre- or post-harvest periods. Evenness was greater in the upland and increased between 1994 and 1996. Species richness peaked in 1994 with the pulse in abundance and decreased by 1995, and was greater for harvested sites and for riparian habitat. Abundance after harvest was significantly greater on modified sites. The modified prescription appears to have a greater potential for maintaining species because of the greater population sizes in the riparian zone.

INTRODUCTION

Second-growth coniferous forests are managed for timber production over much of the Pacific Northwest (e.g., in Washington, about 40% of the land base is in commercial forest; Washington State Department of Natural Resources 1998). Harvesting of these forests over the past 30 yr has created a landscape comprised of forest stands that vary in age and size. At current levels of fragmentation, diversity of small mammals although temporally variable has remained high (Aubry et al. 1998). Riparian habitats may be critical to maintaining this diversity by acting as source populations (Doyle 1990) or as travel corridors that allow recolonization of forest stands (Harris 1984).

Composition and abundance of the small-mammal fauna in riparian habitats may differ from that of adjacent upland habitats. Doyle (1990) found that both species richness and abundance were higher on riparian than on upland transects in mature and old-growth forests in the western Cascades of Oregon. McComb et al. (1993) made similar comparisons in mature Douglas-fir forests in western Oregon. They found higher species diversity in riparian habitats, but no difference in species richness between riparian and upland habitats. Capture rates were higher for some species in the riparian and higher for others in the upland.

Differences in the composition of small-mammal assemblages between riparian and upland habitats rnay depend on several factors. First, some elements of the species pool may require resources found only within the riparian zone (e.g., *Sorex palustris*). Second, the degree of habitat change on the gradient between riparian and upland habitats may affect the ability of some species to use both habitats (McComb et al. 1993). Third, resources may be of higher quality in riparian than in upland habitats as suggested by Doyle (1990).

The few studies that bave examined small-mammal assemblages in riparian areas of managed forests indicate their importance (Cross 1985, Anthony et al. 1987, Doyle 1990, McComb et al. 1993). Consequently, an understanding of how forest practices might affect riparian and adjacent upland assemblages is essential. State and federal forest regulations in the Pacific Northwest mandate riparian buffer zones that can be either no or limited harvest entry and can vary in width depending upon stream size, location, upland harvest prescription, and land ownership. In Washington State, for example, Riparian Management Zones (RMZ) buffering Type 3 streams were established by the Washington Forest Practices

(1987) to be 8-m wide on clearcut harvests west of the Cascade Crest and 10 to 16.6 m wide on selective and clearcut harvests, respectively, east of the Cascade Crest. Limited harvest entry is permitted in the RMZs on both sides of the Cascades.

Because the consequences of the creation of these buffers have not been examined for small mammals, we designed an experimental study to compare small-mammal populations in riparian and adjacent upland habitats before and after timber harvest in managed forests in northeastern Washington. Our objectives were: 1) to determine differences in the distribution and relative abundance of small mammals in riparian and upland habitats, and 2) to evaluate changes in small-mammal populations when riparian buffer zones are created during harvest of adjacent upland forest.

METHODS

Study area

Research was conducted in mixed-coniferous forests in the Selkirk Mountains of northeastern Washington (Stevens and Pend Oreille counties). Forest composition in this region is variable and is affected by slope, aspect, edaphic factors, tire history, and timber management practices. Dominant tree species include Douglas-fir *(Pseudotsuga menziesii),* lodgepole pine (*Pinus contorta*), western redcedar *(Thuja plicata*), western hemlock *(Tsuga heterophylla*), western larch (*Larix occidentalis),* grand fir (*Abies grandis),* and alders (*Alnus incana* and *Alnus sinuata).* Shrubs include gooseberry *(Ribes* spp.), devil's club (*Oplopanax horridum*), Oregon grape (Berberis spp.), mountain boxwood *(Pachistima myrsinites*), red-osier dogwood (*Cornus stolonifera*), ninebark (*Physocarpus malvaceus*), spireae (*Spireae* spp.), serviceberry *(Amelanchier alnifolia*), rose *(Rosa* spp.), and huckleberry *(Vaccinium* spp.).

We selected 18 sites that met the following criteria: 1) 800-m reach of Type 3 or permanent Type 4 stream; 2) >16.2 ha previously harvested stands of harvestable age on either side of stream; 3) ≥610 m and ≤1200 m elevation; 4) mixed coniferous forest; 5) landowners agreed to either leave sites unharvested for 10 yr (controls) or to harvest sites within timeframe and specifications of study design (cut sites). Seven sites were unharvested control sites. The upland areas of 11 sites were selectively harvested for timber in 1993-1994 to yield a 6 to 12-m post-harvest spacing of trees. The riparian zones of 6 of the 11 cut sites were harvested according to the Washington State Forest Practices RMZ (State) guidelines and 5 of the cut sites were harvested according to a modified prescription (Modified) designed for this project. The intent of the Modified treatment was to incorporate a site-specific approach to riparian management. Within a 33-m zone of the stream, habitat features such as seeps, snags, and deciduous trees, were identified and protected. For example, one snag per 2 acres was buffered by a no-entry zone equal to 1.5 times the height of the snag, and all seeps were buffered by a 10-m no-entry zone that extended to the stream. Following timber harvest, the mean width (± 1 SE) of the State buffers measured at 50-m intervals (*n* = 17 per stream) was 14.1 ± 3.0 m with a range from 8 to 22.6 m and the mean width of the Modified buffers was 29.7 ± 17.4 m with a range of 12 to 144 m.

Small-mammal sampling

Trapping was conducted in May-June from 1992 to 1996. The 1992-1993 samples represent pre-harvest conditions, and the 1995-1996 samples represent post-harvest conditions. Although most sites were harvested in late 1993, three sites were not completely harvested until after sampling in 1994.

Pitfall and snap-traps were used to sample small-mammal populations on the 18 riparian and adjacent upland sites. Two parallel transects 720 m in length, were placed 8 m from the stream and 100 m upslope. A total of 72 snap-trapping stations was spaced at 10-m intervals along each transect. Two snap-traps were placed within 3 m of each station, baited with a mixture of oats and peanut butter, and checked for 4 consecutive d for a total of 20,736 trap nights/yr. Eighteen pitfall traps, constructed of two No. 10 coffee cans taped together and buried in the soil, were placed at 15-m intervals on each transect. About 5 cm of water was placed in each can. Pitfall traps were checked every other day for 2 wk for a total of 9,072 trap nights/yr. Captured animals were weighed, measured, numbered, labeled, and frozen.

Specimens were later autopsied to determine reproductive condition. Reproductive data collected for females included size of nipples, number and crown-rump length of embryos, and number of placental scars and corpora lutea. Females were considered reproductive if embryos, corpora lutea, or placental scars were present. Determination of male reproductive condition was based on size of testes and epididymis. Species identification was based on dental characteristics, relative body measurements, and pelage. Museum study skins and skeletons were prepared and deposited in the Conner Museum of Washington State University.

Vegetation sampling

Habitat features were sampled in 1992 (pre-harvest). All cut sites were resampled 2 yr post harvest in either July 1995 or July 1996. Control sites were resampled in July 1995. At 50-m intervals along each transect we established a 16 x 20-m plot that was divided into four

8 x 1 O-m quadrants. We measured vegetation characteristics on 15 riparian and 15 upland transects per site.

Trees and snags.-~-Within each 20 x 16-m plot all trees were assigned to one of four DBH classes: 1) 4-10 cm; 2) 11-25 cm; 3) 26-50 cm; 4) >50 cm. All snags within each plot were counted and designated as either Condition 1 (bark basically intact) or Condition 2 (bark peeling off to absent).

Canopy cover.-Percentage of overstory and understory cover was measured with a convex spherical densiometer at the center of each 20 x 16 m plot and at the center of each 8 x 10-m quadrant for a total of five measurements per site that were then averaged.

Shrubs and regenerating trees.-From the center point of the four 20 x 16-m plots, the distance to the nearest shrub (>0.5 m high) in each of the quadrants was measured and the area of each shrub (length x width) was recorded. In two opposite quadrants, the numbers of regenerating coniferous trees (>0.5 m high; <4 cm DBH) were recorded.

Woody debris.-In two opposite quadrants within each plot the number and decay class of woody debris and stumps were recorded. Logs were assigned to one of four size classes and to one of four decay classes. Size classes were: 1) ≥5m long x <15 cm circumference; 2) ≥5 m long x 16-24 cm circumference; 3) ≥5 m long x >25 cm circumference; 4) <5 m long x >25 cm circumference. Decay classes were defined as: 1) freshly fallen tree with bark essentially intact, wood solid, no decomposition; 2) bark beginning to slough or almost completely gone, decomposition begun with sapwood partially softened but log generally firm; (3) decomposition progressed to the point that wood is generally soft and breaks into chunks, each chunk still as integrity; (4) essentially no integrity to log, wood decomposed to point of soil-like texture. Stumps were assigned as either "natural" or "cut and to one of the four above decay classes. Stumps were differentiated from snags by height; stumps were 4.37 m high (standard breast height).

Statistical analysis

Small-mammal distribution.---We tallied the numbers of individuals of each small-mammal species for each site and habitat type for captures from both snap and pitfall traps. Species richness, species diversity, evenness, and abundance were compared between years and between habitat types by analysis of variance (ANOVA). We performed contingency table analysis of each species abundance in riparian and upland habitats for each year of the study. Furthermore, we conducted a principal components analysis on the abundance data from before and after harvest for all sites and both habitat types. This analysis was conducted on the covariance matrix to determine if any shifts in patterns of species cooccurrence took place following harvest.

Demography.—We compared body mass of reproductive and nonreproductive animals using ANOVA for both males and females of each species. For this analysis, we considered two groups of reproductive females: pregnant (i.e., embryos were present) and mature (Le., characteristics, such as corpora lutea, indicated that they were or recently had been reproductive). The proportion of reproductive males and females of each species was determined for each study site. We also examined differences in the number of embryos present in pregnant females by conducting factorial ANOVA with time period and buffer treatment as classification variables.

Habitat relationships.—To examine the relationship between capture frequency and habitat structure we used stepwise multiple regression with capture frequency of a species at each transect as the dependent variable. The mean values of the habitat variables for each transect were included as independent variables. In addition to the specific habitat variables, we included a dummy variable to represent the habitat types, and two dummy variables to represent the Modified and State treatments. We conducted this analysis for both the pre-harvest and post-harvest periods for each species that had >20 captures in that period. Because of the large number of habitat variables used to describe the transects, we report only those variables that increased the overall R^2 by >5% when entered into the regression model.

All analyses were conducted using the Statistical Analysis System (SAS Institute 1989). All statistical tests were considered significant at $P < 0.05$ unless otherwise noted.

RESULTS

Pitfall and snap-trapping yielded 13,081 specimens of 21 species. Another 121 and 5 specimens could only be identified as *Sorex* or *Microtus,* respectively, and these were not included in any analyses. Difficulties in identification were usually due to predation on specimens while in the trap. Because of difficulties in assigning specimens to *Tumias amoenus* or *T. ruficaudus* for 1992 and 1993, we treated these species as a single taxon.

Species richness across all sites varied from 14 to 18 species per year. Some species were observed infrequently because our trapping techniques were not designed to sample them adequately (i.e., *Lepus americanus,* 8 captures; *Mustela erminea*, 4, *Tamiasciurus hudsonicus*, 2; *Neotoma cinereus*,13; *Glaucomys sabrinus,* 13). Excluding these species, species richness varied from 11 to 14 species across all sites.

About 91.5% of all captures consisted of just four species: *Clethrionomys gapperi* (32.5%), *Sorex vagrans* (24.1%), *Peromyscus maniculatus* (20.0%), and *Sorex cinereus* (14.9%, Table 1). These species also had the broadest distributions (Fig. 1). The total number of captures per year for all species combined varied considerably over the study period, but we captured more individuals in the riparian zone in each year (Fig. 2). This result is partly explained by the greater occurrence of some numerically dominant species in the riparian (e.g., *Sorex vagrans* and *Clethrionomys gapperi;* Table 1). As expected, 69% of the shrews were captured in pitfall traps, and 82% of the rodents were caught in snap-traps (Table 1). The total number of captures from snap-traps per site and habitat type was positively correlated ($P < 0.05$) to that from pitfall traps for all taxa except *Tamias* spp.

Mean body mass of mature males and females was significantly greater than for immature males and females:, respectively, for all species with adequate sample sizes (Table 2). The mean number of embryos per female did not differ with harvest treatment for any species. A significant temporal change in number of embryos was observed only for *Clethrionomys gapperi* (Table 3). The mean number of embryos peaked in 1994, when overall abundance was greatest and did not differ between pre- and post-harvest periods, Similarly, significant differences in body mass of mature females (excluding pregnant individuals) were only observed for *C. gapperi.* For this species, mean body mass differed between all periods, but also was highest in 1994 (pre-harvest, $\bar{x} = 22.2 \pm 0.3$, $n = 263$; harvest, $\bar{x} = 25.4 \pm 0.3$, $n = 329$; post-harvest, $\bar{x} = 23.2 \pm 0.2$, $n = 274$).

Riparian vs. upland habitats

In the 2 pre-harvest yr (1992-1993), there were no differences in species diversity, evenness, or species richness between upland and riparian habitats (for all cases: $F < 0.67$, $P > 0.4$, df = 1,67). There were significant differences between treatments for species diversity and richness (diversity-17 = 3.48, $P = 0.04$, df = 2,67; richness—$F = 8.1$, $P < 0.001$, df =

2,67). Diversity was greater on Modified sites than on Controls (Fig. 4), whereas richness was greater on State and Modified sites than on Controls.

Total abundance of small mammals was greater in the riparian than in the upland (F = 19.3, $P < 0.0001$, df= 1,67), and increased between 1992 and 1993 (F= 9.1, P-0.004, df= 1,67). Abundance did not differ among the three treatments (F = 0.3, $P > 0.74$, df = 2,67).

The red-backed vole, vagrant shrew, western jumping mouse, water shrew, and bog lemming were all significantly associated with the riparian zone prior to harvest (Tables 1 and 4). The few captures of the water vole occurred only in the riparian zone. The masked shrew, montane shrew, and chipmunk species were all upland associates (Tables 1 and 4). The pygmy shrew occurred significantly more often in the upland in 1992, but showed no association in any subsequent year.

Treatment effects

Most sites were harvested by spring 1994, which coincided with a sharp increase in abundance of most small-mammal species on all sites (Fig. 6). Total captures of small mammals declined significantly by 1995 (F= 105.7, $P < 0.0001$, df = 2,102) and remained relatively constant through 1996, near their level in 1992-1993. These changes in abundance corresponded to significant changes in species richness and evenness, but not species diversity (Fig. 4). Species richness peaked in 1994 with the pulse in abundance, and decreased by 1995 (Fig. 3). Evenness increased significantly between 1994 and 1996 (Fig. 5).

Species richness was greater in riparian than in upland habitat (F = 7.7, $P < 0.007$, df = 1,102), and on State and Modified sites than on Controls after harvest (F = 11.2, $P < 0.0001$, df= 2,102). In contrast, species diversity after harvest did not differ between habitats (F= 0.01, P = 0.91, df = 2,102), but was significantly greater on State and Modified sites than on Controls ($F = 13.4$, $P < 0.0001$, df= 2,102). Evenness was greater in the upland than in the riparian ($F = 8.2$, $P = 0.005$, df = 1,102; Fig. 4). Abundance after harvest was again greater in the riparian ($F = 3'7.1$, $P < 0.0001$, df = 2,102), and was significantly greater on the Modified sites with no differences between Control and State sites ($F = 5.6$, $P < 0.005$, df = 2,102).

Habitat associations of those species that were associated with the riparian prior to harvest remained the same following harvest (Table 4). The deer mouse shifted from no association to a greater association with the riparian in 1994 and 1995. This shift corresponded to greater abundance in 1994, and was more pronounced on a few sites. Similarly, species that were associated with the upland prior to harvest maintained this association. Species that showed no association with riparian or upland continued not to show any association after harvest (Table 4).

The principal components analysis also indicated little change in species associations after harvest (Fig. 7). Species were generally aligned with one of the first two principal components, which accounted for about 67% of the variance in distribution and abundance. Strong riparian associates (e.g., C. *gapperi*, *S. vagrans*, and *Z. princeps*) grouped together with high values on principal component 1 for both time periods; whereas upland associates (*Tamias* spp., *S. cinereus)* generally grouped at low values on principal component 2. *Phenacomys intermedius* did not show strong habitat associations to either riparian or upland habitats. *Peromyscus maniculatus* grouped by itself.

For all species examined, structural components of the habitat were strong predictors of the number of captures (Table 5), with each equation explaining 45 to 89% of the variance. Although the types of variables entered in the models were similar between pre- and post harvest periods for most species, the individual variables often changed. In large part, this reflects the structural changes to the habitats that occurred during harvest. Associations with riparian habitat were significant for C. *gapperi*, *Z. princeps*, and S. *vagrans*, whereas *S. cinereus* was significantly associated with the upland. Following harvest, *C. gapperi* and *P. intermedius* were positively associated with the Modified sites, and S. *vagrans* was negatively associated with the State sites.

DISCUSSION

Riparian zones in northeastern Washington contain a diverse assemblage of small-mammal species. Although we considered 16 of the 21 species encountered, most captures consisted of just four species, which occurred at most of our study sites (Fig. 1). The remaining 15 species were not as widely distributed, and consequently the composition of small-mammal assemblages varied among streams. Significant temporal variation in abundance and distribution of almost all species also altered composition of these assemblages over the 5 years. Despite the spatial and temporal variation in small-mammal populations, there were consistent patterns of habitat use by species. These patterns point to the importance of riparian zones.

Prior to the initiation of this study, there had been little work done to compare riparian and upland small-mammal faunas in forested ecosystems of the Pacific Northwest (e.g., Doyle 1990, McComb et al.1993), and none that had examined the consequences of upland harvest and riparian zone management. Based on West's (Chapter 10) work on the west side of the Cascade Crest, it is clear that one should not extrapolate patterns of habitat use for small mammals from the more mesic forests of western Washington and Oregon to northeastern Washington. For example, *Clethrionomys gapperi* is an uncommon upland species on the west side, but is a numerical dominant in the riparian on the east side. Similarly, *Sorex monticolus* is a riparian species on the west side, and an upland species on the east side. Riparian zones may be of ecological importance both east and west of the Cascade Crest, but the small-mammal species and their responses may differ.

Pre-harvest patterns

The lack of any initial differences between riparian and upland habitats in small-mammal diversity, richness, or evenness prior to harvest was surprising. Doyle (1990) and McComb et al. (1993) found that either species diversity or richness was higher in the riparian. The higher relative abundance in the riparian zone was the only relationship that remained constant throughout the study. Doyle (1990) also found greater abundance in the riparian than in the upland. Although McComb et al. (1993) did not find overall abundance to be greater in the riparian, they did find that some species were more abundant in the riparian and others in the upland. In our study, most species showed very clear differences in their relative abundance between habitats. These differences were very consistent from year to year.

We did observe differences between sites assigned to different treatments. In particular, the Control sites had lower species diversity and richness. This result partly reflects the greater age of some of the Control sites, which generally had reduced understory.

Post-harvest patterns

Although large increases and decreases in abundance for small-mammals are not unusual (e.g., the large literature on microtine fluctuations), the increases that occurred in 1994 were not confined to any particular taxonomic group, but included rodents and insectivores alike. Species richness rose dramatically (Fig. 3) in 1994 with the greater abundance and distribution of some less common species (e.g., *Sorex hoyi,* several microtine species). 'The subsequent decline in species richness in 1995 corresponded to the decline in overall abundance and the loss of the less common species at most sites. It is significant that the increases in species richness were greater in the riparian than in the upland. This suggests that riparian areas may provide suitable habitat for travel for some species that are not persistent residents. During a pulse in abundance as observed in 1994, movement of dispersers through the riparian zone may allow recolonization of habitat patches. Mech and Hallett (in *press)* presented genetic evidence that narrow strips of closed-canopy forest acted as corridors for movement of C. *gapperi* across managed-forest landscapes.

In contrast, there was no temporal change in species diversity subsequent to harvest or differences between riparian and upland habitats. Because species diversity accounts for the relative abundance of each species, the much greater contributions of really common species to this measure result in much smaller changes despite the greater richness. The increase of less common species in 1994, however, is reflected in the reduction in evenness in that year. As these species were lost in 1995 and 1996, evenness increased. The lower evenness in the riparian corresponds to the presence of less common species. Overall abundance remained greater in the riparian than in the upland after harvest across all sites.

As before harvest, both species diversity and species richness were greater on State and Modified sites than on Controls. The significant difference was the greater overall abundance on Modified sites following harvest, Greater abundance increases the probability of population persistence (e.g., Stacy and Taper 1992).

Management recommendations

Riparian zones provide important habitat for small mammals in northeastern Washington. Several species have their greatest abundance or occur only in the riparian zone (e.g., *Sorex palustris*). Maintenance of the riparian area will help to retain species that have specific requirements for elements found only in the riparian. Abundance is the key to long term persistence of populations, and consequently, the Modified RMZs appear to have a greater potential for species persistence.

The use of riparian areas as corridors is inferred from our work, and supported by Mech and Hallett's (in press) study of corridor use in managed forests of northeastern Washington. The use of riparian buffers as corridors may change temporally. Pulses in abundance of small mammals as we observed in 1994 may be of great importance for recolonization of vacant habitats. Riparian buffer strips may also be source habitats for dispersers.

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Table 1. Number of captures of small-mammal species in snap and pitfall traps on all 18 sites in riparian and upland habitats from 1992 to 1996. Percentage of captures in each trap type is given for each habitat. Probabilities for χ^2-tests for differences in capture frequency with trap type between habitats are provided when significant. Species acronyms are used in subsequent figures and iables.

Taxon	Riparian				Upland				*P*
	snap		Pitfall		snap		Pitfall		
	n	%	*n*	%	*n*	%	*n*	%	
Insectivores									
Masked shrew (*Sorex cinereus*) SOCI	223	28.1	570	71.9	346	30.1	803	69.9	ns
Pygmy shrew (S. *hoyi*) SOHO			17	100.0	1	3.2	30	96.8	ns
Montane shrew (*S. monticolus*) SOMO	4	7.0	53	93.0	24	18.1	109	81.9	0.05
Water shrew (S. *palustris*) SOPA			25	100.0			1	100.0	
Vagrant shrew (S. *vagrans*) SOVA	814	32.8	1669	47.2	239	36.2	421	63.8	ns
Total Insectivores	1041		2334		610		1364		
Rodents									
Chipmunks (*Tamias* spp.) TASP	75	97.4	2	2.6	195	100.0			ns
Western jumping mouse (*Zapus princeps*) ZAPR	188	63.5	108	36.5	41	83.7	8	16.3	10.01
Deer mouse (Peromyscus maniculatus) PEMA	1329	89.9	150	10.1	998	88.4	131	11.6	ns
Southern red-backed vole (*Clethrionomys gapperi*) CLGA	1952	78.5	534	21.5	1415	80.7	338	19.3	ns
Long-tailed vole (*Microtus longicaudus*) MILO	11	44.0	14	56.0	19	54.3	16	45.7	ns
Montane vole (*M. montanus*) MIMO	3	60.0	2	40.0	5	83.3	1	16.7	ns
Meadow vole (*M. pennsylvanicus*) MIPE	5	25.0	15	75.0	9	81.8	2	18.2	co.01
Water vole (*M. richardsoni*) MIRI			2	100.0					
Heather vole (*Phenacomys intermedius*) PHIN	35	60.3	23	39.7	31	53.5	27	46.5	ns
Bog lemming (*Synaptomys borealis) SYBO*	4	50.0	4	50.0					
Total rodents	3602		854		2713		523		
Total mammals	4643		3188		3323		1887		

Table 2. Comparison of mean body mass (g) for reproductively mature and immature individuals of the small-mammal species across all treatments and years. Reproductive age females are split into pregnant and mature groups. Different superscripts for means of females indicate significant differences based on Tukey's Studentized Range Test.

Species	Males								Females										
	Mature			Immature					Pregnant			Mature			Immature				
	$\bar{x}$	SE	*n*	$\bar{x}$	SE	*n*	*FP*		$\bar{x}$	*SE*	*n*	$\bar{x}$	*SE*	*n*	$\bar{x}$	SE	*n*	*F*	*P*
Insectivores																			
Masked shrew	4.9	0.0	568	3.5	0.0	541	2252	<0.01	6.0”	0.1	60	5.1^{b}	0.1	154	3.7^{c}	0.1	528	57	co.01
Montane shrew	7.0	0.2	34	4.3	0.1	66	286	<0.01	6.2^{a}	0.4	9	6.2^{a}	0.4	12	4.3^{b}	0.1	62	23	<0.01
Pygmy shrew	3.6	0.1	16	2.6	0.1	17	57	co.01				3.7^{a}	0.1	3	2.7^{b}	0.1	11	20	<0.01
Vagrant shrew	7.9	0.0	527	5.2	0.0	1178	5156	<0.01	8.2^{a}	0.1	110	7.3^{b}	0.1	266	5.2^{c}	0.0	948	532	<0.01
Water shrew	14.7	0.2	4	8.9	0.3	15	101	co.01	14.5^{a}		1				8.8^{b}	0.3	6	49	co.01
Rodents																			
Bog lemming	33.3		1	15.3	2.5	4	10.1	0.05	26.7		1	27.5	1.2	2					
Chipmunk species	56.0	0.8	34	52.7	0.6	112	7.6	<0.01	63.0”	1.7	12	63.1^{a}	0.6	76	57.5^{b}	1.5	16	7	co.01
Deer mouse	20.6	0.1	641	14.6	0.1	758	1397	<0.01	22.9^{a}	0.3	166	20.8^{b}	0.2	435	13.4’	0.1	462	671	<0.01
Heather vole	22.2	1.1	25	16.3	0.1	24	16.7	<0.01	26.6”	1.0	28	26.0^{a}	1.4	17	14.5^{b}	1.5	12	24	<0.01
Long-tailed vole	29.6	1.3	27	17.8	2.3	9	19.7	<0.01	33.5^{a}	2.0	6	27.3^{a}	1.9	9	14.1^{b}	2.3	4	17	10.01
Meadow vole	27.9	1.2	19	16.0		1	4.9	0.04	27.4	2.4	5	21.3		1	20.1		1	1.2	0.4
Montane vole	34.5	2.7	5	15.3	1.6	3	26	<0.01							11.5	0.2	2		
Southern red-backed vole	24.5	0.1	1430	16.2	0.2	789	2346	<0.01	26.8^{a}	0.2	536	23.7^{b}	0.2	866	11.8^{c}	0.2	279	1046	<0.01
Water vole									109.8		1	101.0		1					
Westemjumping mouse	23.1	0.4	56	22.1	0.3	156	3.7	0.057	25.3^{a}	0.5	51	23.6^{a}	0.6	51	19.7^{b}	0.9	28	16	<0.01

Table 3. Mean number of embryos (± SE) from pregnant females of each species, and ANOVA results for differences by time period (preharvest, 1992-1993; harvest, 1994; and post harvest, 1995-1996). Different superscripts for means indicate significant differences based on Tukey's Studentized Range Test.

Species	Preharvest			Harvest			Post harvest				
	$\bar{x}$	SE	n	x	SE	n	$\bar{x}$	SE	n	F	P
Insectivores											
Masked shrew	4.8	0.5	13	5.8	0.2	27	5.2	0.3	23	1.71	0.19
Montane shrew	5.0	0.8	5	6.0	0.0	2	5.5	0.5	2		
Vagrant shrew	4.9	0.1	33	4.7	0.2	38	5.2	0.2	47	1.32	0.27
Water shrew							6		1		
Rodents											
Chipmunk species				3.5	0.5	2	4.0		1		
Deer mouse	4.8	0.1	46	5.3	0.3	51	5.1	0.1	86	1.2	0.30
Heather vole	4.2	0.2	13	4.4	0.4	11	3.9	0.2	8	0.55	0.58
Long-tailed vole				4.0	0.7	6	4.7	0.3	3		
Meadow vole				5.3	0.6	4	7.0		1		
Southern red-backed vole	4.0^a	0.1	203	4.3^b	0.1	261	3.9^a	0.1	214	18.2	<0.001
Water vole	8.0		1								
Western jumping mouse	5.4	0.4	5	4.8	0.3	12	5.0	0.2	16	0.97	0.39

Table 4. Habitat associations of the small-mammal species based on contingency analysis of captures in the two habitat types across all treatments. R and U indicate associations with riparian and upland habitats, respectively. N indicates no association. Unless noted with "ns", comparisons were significant at $P < 0.05$. *Sorex palustris*, *Synaptomys borealis*, and *Microtus richardsoni* were only captured in the riparian zone, but sample sizes were small for these species.

Species	Year				
	1992	1993	1994	1995	1996
Riparian					
Southern red-backed vole	R	R	R	R	R
Vagrant shrew	R	R	R	R	R
Western jumping mouse	R	R	R	R	R
Water shrew	R ns	R	R	R ns	Rns
Bog lemming	R		R ns	Rns	
Water vole		Rns		Rns	
Deer mouse	N	N	R	R	N
Upland					
Chipmunk species	U ns	U	U	U	U
Masked shrew	U	U	U	N	U
Montane shrew	Uns	Uns	U	U	U
No association					
Pygmy shrew	U	N	N	N	N
Montane vole			N	N	N
Meadow vole	N		N	N	
Long-tailed vole	N		N	N	N
Heather vole	N	N	N	N	N

Table 5. Results of regression models predicting numbers of captures on riparian and upland transects of the 18 sites before and after harvest. Regressions were conducted for species with >20 captures in each time period. Sign indicates the direction of the relationship. All variables were significant at $P < 0.05$.

Species	R^2	Overstory	Shrub/regeneration	Woody debris’	Zone/Treatment
C. gapperi					
Preharvest	73.1	+4C1, 4C2	+Shrub area	+S1D1, S1D3, -S3D1	+Riparian
Postharvest	70.9	+2D, -4D	+Shrub distance	+S3D4, -S4D4, N1	+Modified
P. maniculatus					
Preharvest	76.4	- 2D, 1C	+Shrub height, - Shrub area	+N1, N3, -LS1, S4D3	+State
Postharvest	75.6	+Overstory	+Shrub area	+S3D1, N1, N2, -S3D4	
P. intermedius					
Preharvest	80.2		+Shrub area	+C2, -C3, S4D3	
Postharvest	45.6	+1D, -Overstory	+Shrub area	-S2D2	+Modified
Tamias spp.					
Preharvest	67.2	-3C, 4c		+SF2	
Postharvest	61.6	-Overstory		+S2D4, C1, N3, -S2D2	

Table 5. Continued.

Species	R^2	Overstory	Shrub/regeneration	Woody debris	Zone/Treatment
Z. princeps					
Preharvest	61.5	+Understory		+C3, -S3D3	+Riparian
Postharvest	75.3	+3C1, 4C1	-Regeneration	+S4D2, -N4	+Riparian
S. cinererus					
Preharvest	56.7	+4C1, - 3c		-C2	+Upland, Modified
Postharvest	63.2	-4C, 4C2		+S1D2, C3, -S2D2	+ Modified
S. monticolus					
Preharvest	65.0	-4C1, 4C2		+S4D2, -C3, S3D4	
Postharvest	80.3			+S4D1	
s. *vagrans*					
Preharvest	88.6	+1D, 4C2		+S4D2, N2, -S1D3	+Riparian
Postharvest	75.0	3D		+S3D2	+Riparian, - State

[1] Overstory variables include percentage of overstory and understory cover and counts of trees. Tree count variables are given as the DBH class (1-4), tree category (D = deciduous, C = coniferous), and snag condition class (1 or 2) where appropriate.

[2] Woody debris includes: logs denoted by size (S) and decay (D) class and stumps denoted as cut (C) or natural (N) and decay class. Decay and size classes are described in the text.

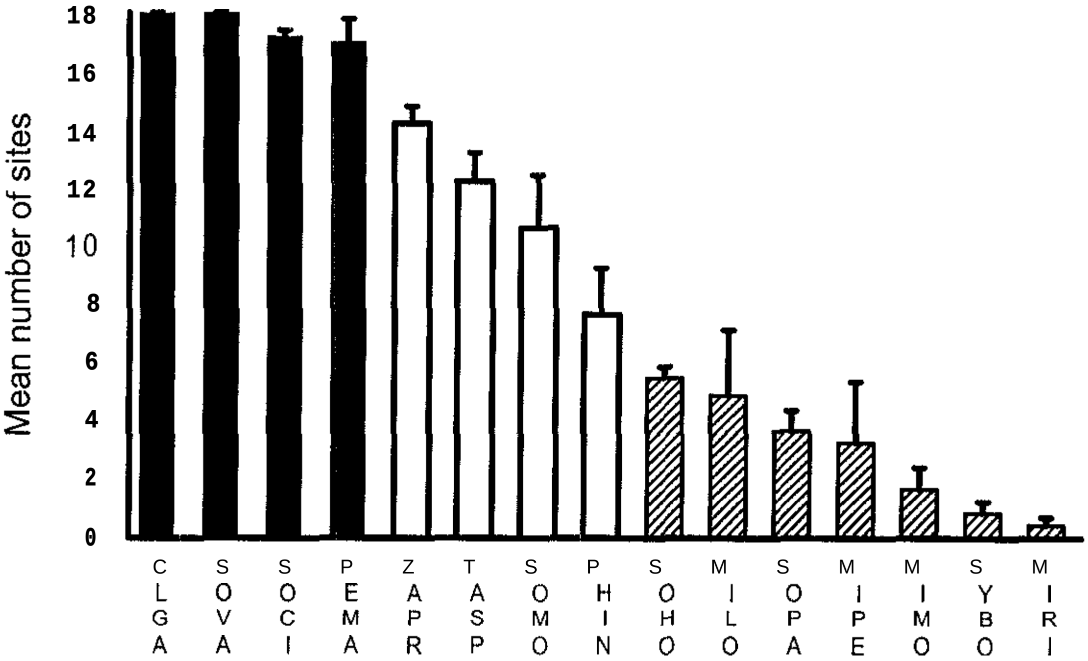


Figure 1. 'The mean number of sites at which each species was observed over the 5 years. Species codes are presented in Table 1.

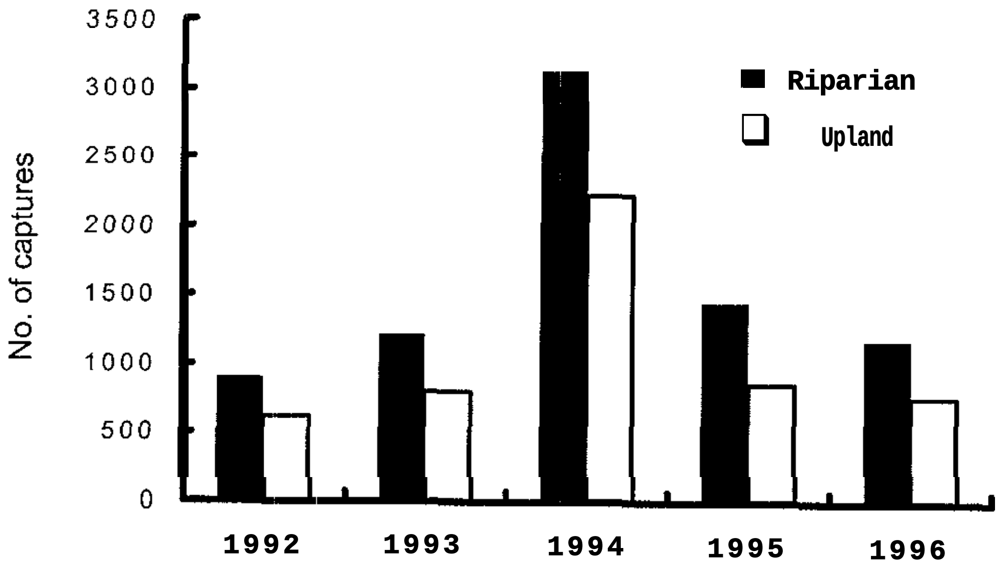


Figure 2. Number of individuals of small mammals captured in riparian and upland habitats over the 5 yr.

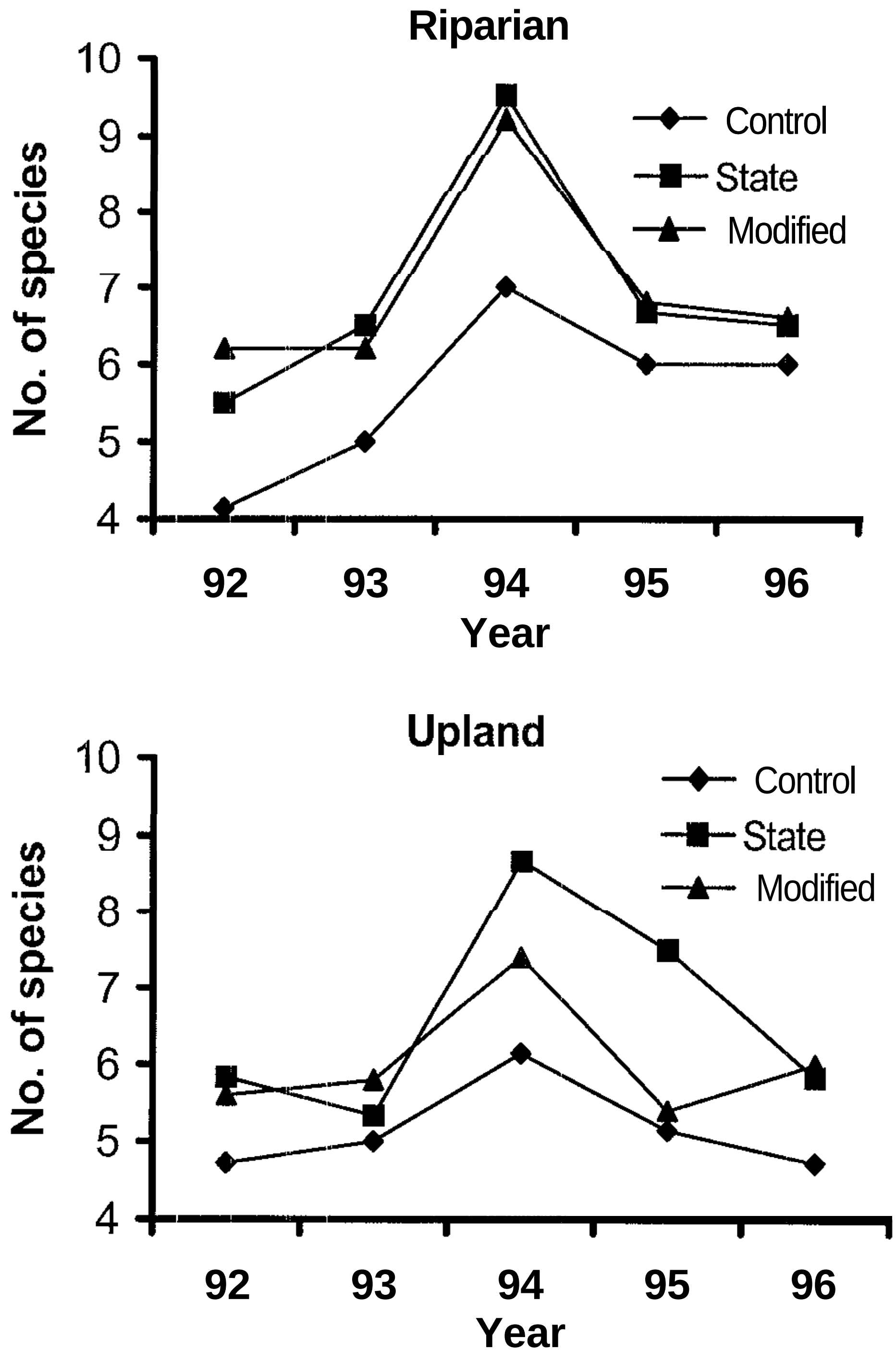
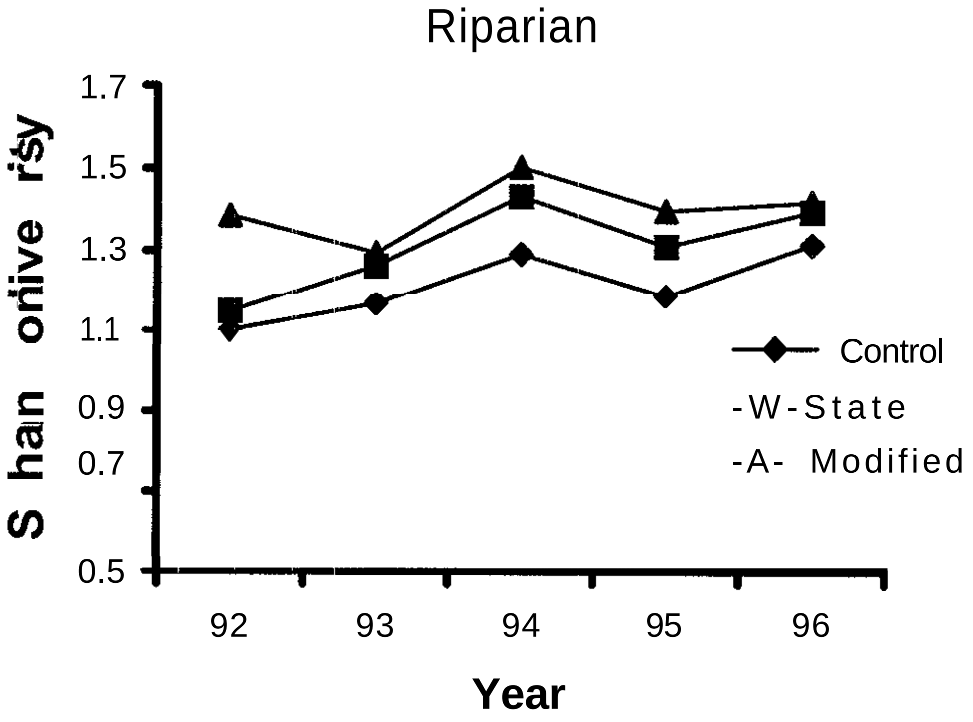


Figure 3. Species richness of small mammals in riparian and upland habitats from 1992-1996.



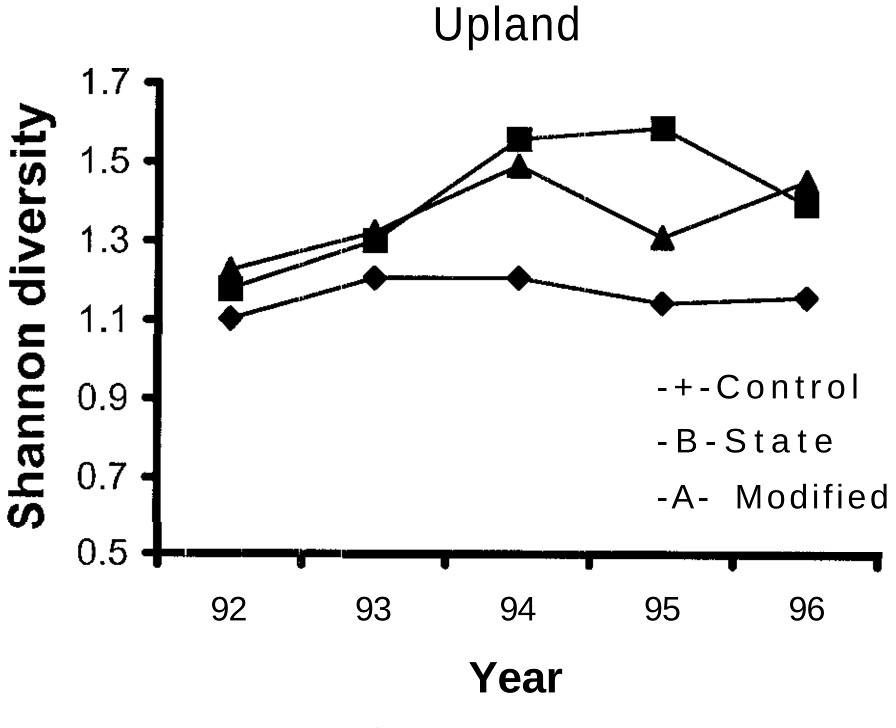


Figure 4. Shannon diversity for small mammals in riparian and upland habitats from 1992-1996.

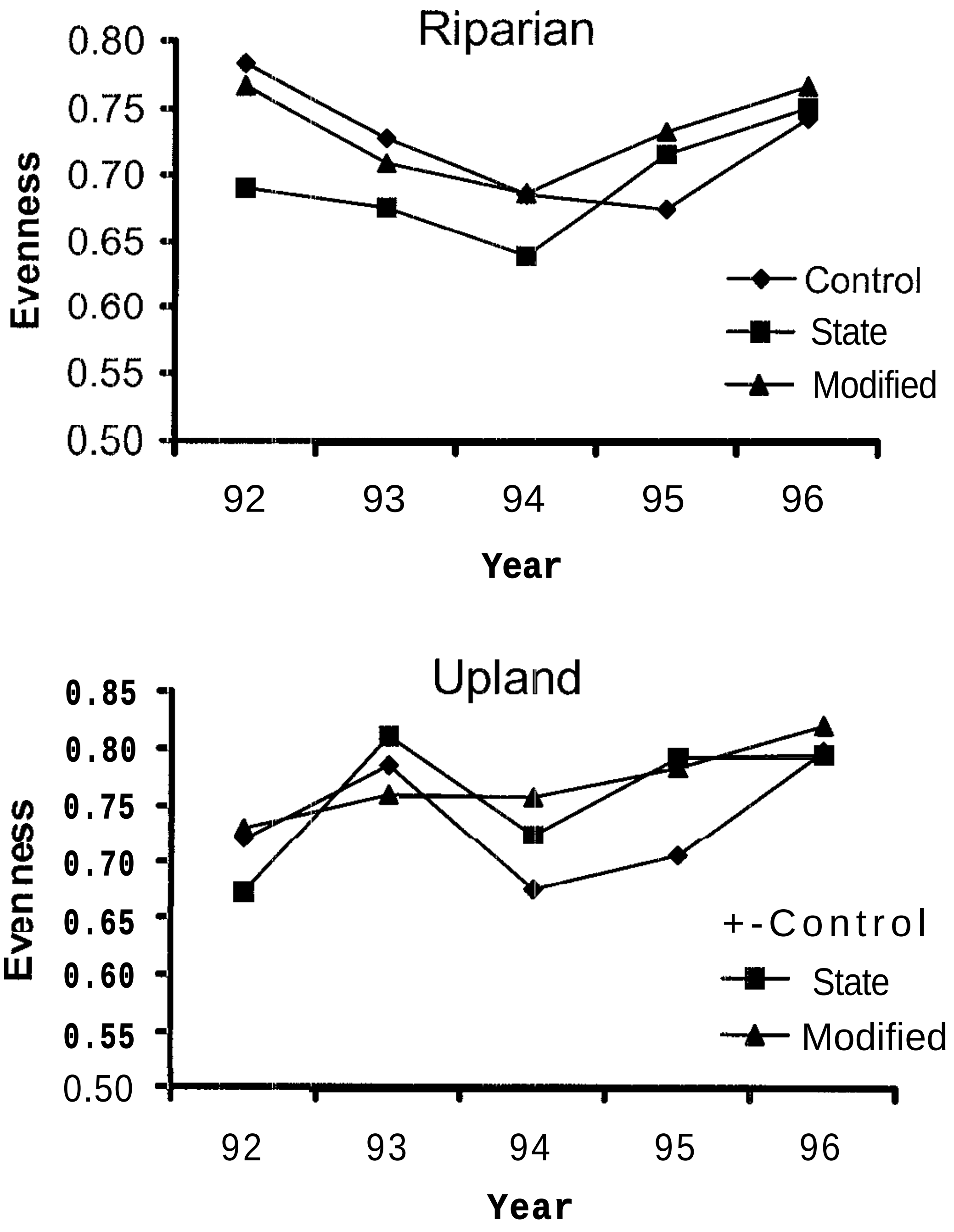


Figure 5. Evenness for small mammals in riparian and upland habitats from 1992-1996

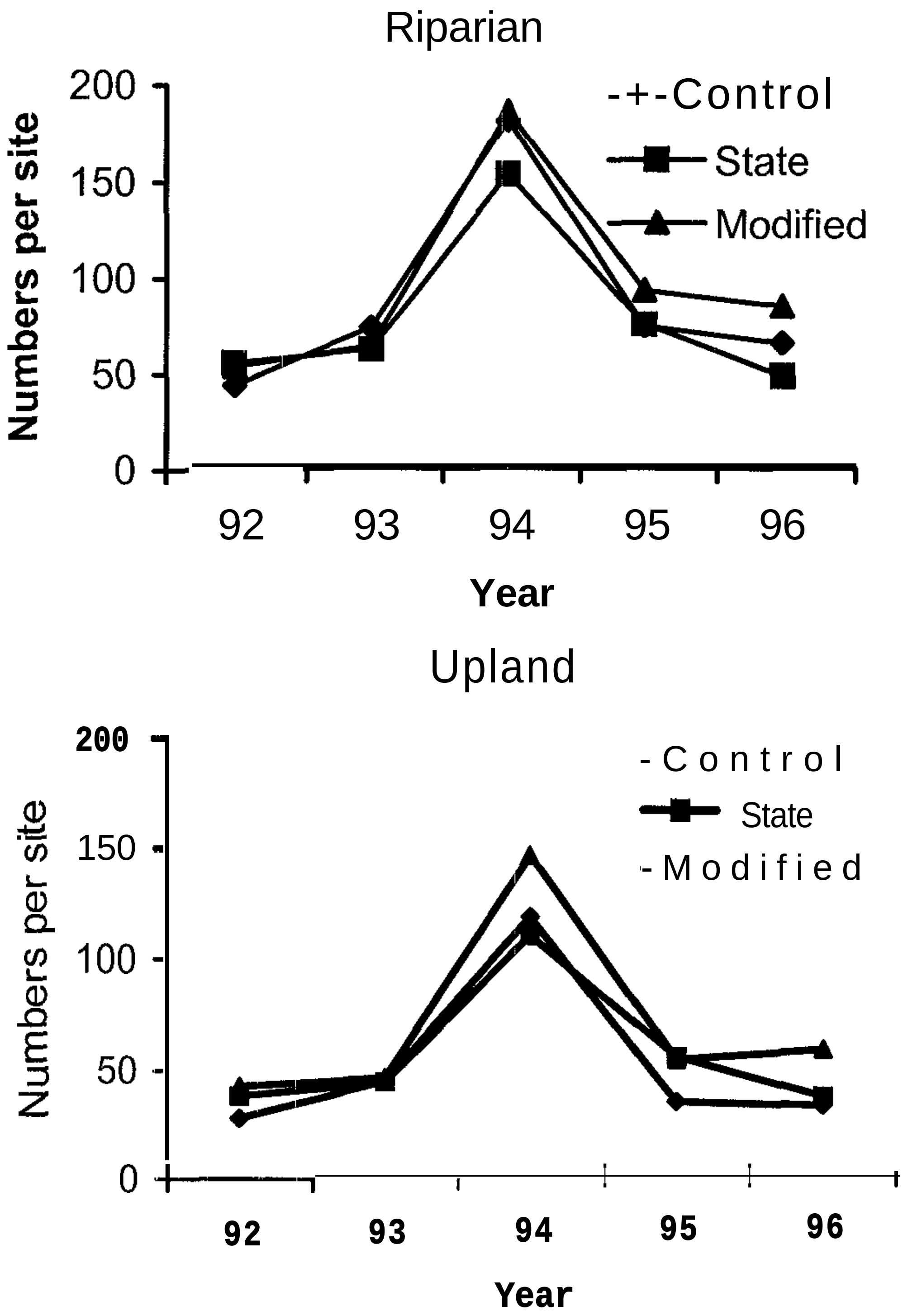


Figure 6. Changes in abundance for small mammals in riparian and upland habitats, 1992.1996.

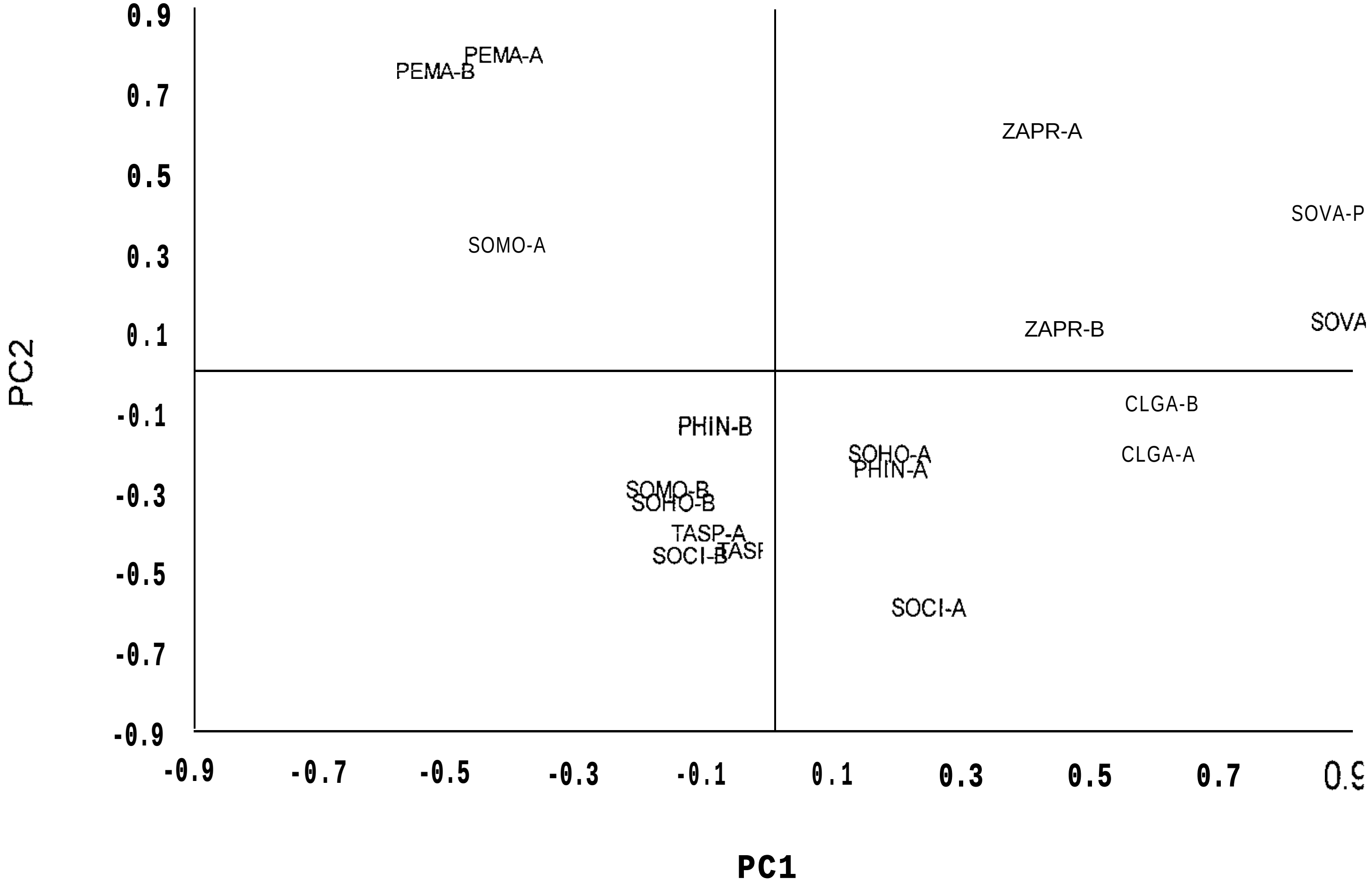


Figure 7. Principal component loadings for small-mammal species captured >20 times in both pre- (A) and post-harvest periods (B). Species abbreviations are defined in Table 1.

Chapter 12

WEST-SIDE BAT SURVEYS

Abstract. To investigate the influence of different riparian buffer configurations on forest-dwelling bats, we compared activity patterns and species composition of bats within the riparian zone and the associated upland with respect to unharvested Control sites (Control), to sites harvested under minimal State guidelines (State), and to sites harvested under guidelines designed as part of this study (Modified). Detection rates of bats were highest during the post-harvest period across all treatments. During the pre-treatment period, 2,433 calls were detected (1992: 10 sites; 1993: 17 sites) with an average of 15 calls recorded per night. During post-treatment sampling, 5,937 calls were detected (17 sites both years) with an average of 30 calls per night. We did not detect differences in activity for *Myotis* bats among treatments. Differences among buffer treatments for non-*Myotis* detection rates were significant with highest activity levels recorded within sites harvested under State regulations. *Myotis* bat detection rates differed significantly between riparian and upland habitats with the highest detection rates in the riparian buffer. This pattern did not vary by treatment. For non-*Myotis* bats, there was a significant difference in activity levels among treatments with highest activity recorded in State sites, as well as a significant interaction between habitat and treatment. However, non-*Myotis* activity levels were not significantly different between riparian and upland habitats. Our findings suggest that the influence of riparian buffers on bat activity differs among taxa. Differences in morphology and echolocation call structure among bat species may partially explain differences in activity levels found among treatments.

INTRODUCTION

Bats use a variety of habitats to meet their daily requirements for food, water, and cover. In forested landscapes, roosting sites are generally located within older stands where large trees provide shelter in the form of cracks, crevices and exfoliating bark (Erickson 1998, Krusic and Neefus 1996, Thomas 1988). Foraging sites are often located in areas of high insect concentrations such as forest gaps, edges, ponds, and streams (Christy and West 1993, Crome and Richards 1988, Fenton 1990, Grindal 1996, Lunde and Harestad 1986, Thomas and West 1991).

Riparian areas in particular provide bats with multiple resources. The importance of riparian habitats as foraging areas and sources of water for drinking has been documented in several studies (Brigham et al. 1992, Furlonger et al. 1987, Hayes and Adam 1996, Lunde and Harestad 1986, Thomas 1988, Thomas and West 1991). In addition, riparian corridors may be used as flight routes through which bats travel when commuting between distant roosting and foraging sites (Law et al. 1998, 1999). When comparing activity levels in riparian, old growth, clearcuts and second growth sites in southeastern Alaska, Parker et al. (1996) found riparian habitat to have the highest proportion of nights in which bats were detected, the highest number of bat calls per night, and the highest proportion of calls containing feeding buzzes. Similarly high activity levels have been observed in riparian habitats from other temperate regions (Law et al. 1998, Thomas 1988, Thomas and West 1991).

Although bat activity appears to be disproportionately high in riparian areas (Cross 1988), few studies have addressed the impact on bats of forest management activities in riparian zones. One notable exception is the study by Hayes and Adam (1996) that evaluated differences in bat activity in logged and unlogged riparian areas in western Oregon. Bat activity was found to be four to eight times higher in wooded than in partially logged areas even though logged openings were small in size (90-l 80 m). These findings indicate that forest management activities in riparian areas can have important consequences for habitat use by bats (Hayes and Adam 1996).

Knowledge of the response by bats to various forest management activities in riparian areas is important to land managers who must manage for wildlife as well as wood production. The primary objective of this study was to assess the usefulness of riparian management zones in maintaining habitat for bats after timber harvest. We report here on baseline conditions before harvest (1992-1993) and the patterns of activity during the first 2 yr (1995-1996) after harvest. We assessed the activity patterns and species composition of bats within the riparian zone and the associated upland with respect to unharvested Control sites, to sites harvested under minimal State Riparian Management Zone (RMZ) guidelines, and to sites harvested under guidelines designed as part of this study (Modified guidelines).

METHODS

The bat fauna

The area west of the Cascade Range crest in Washington is believed to support 11 species of bats (Barbour and Davis 1969, Thomas and West 1991). These include seven species of *Myotis (M. californicus, M. evotis, M. keenii, M. lucifugus, M. thysanodes, M. volans* and *M. yumanensis*), big brown bats *(Eptesicus fuscus),* silver-haired bats *(Lasionycteris noctivugans),* Townsend's big-eared bats *(Plecotus townsendii)* and hoary bats (*Lasiurus cinereus*). At present, all *of the Myotis* species and the big brown bat are on the Washington State Priority-Habitats and Species List (Washington Department of Fish and Wildlife 1996). In addition, Townsend's big-eared bat is designated as a species of special concern and is being considered for federal listing under the Endangered Species Act. Basic natural history information for many of these species is lacking, however, all are believed to use riparian habitats to some degree (Cross 1988).

Sampling design

Using ultrasonic detectors, patterns of habitat use by bats were monitored for 2 yr prior to treatment (1992-1993) and 2 yr immediately following harvest (1995-1996) in Control and harvested buffers. Due to the difficulty in locating sites, monitoring in 1992 was limited to 10 sites: 3 Control, 4 Modified, and 3 State sites. From 1993-1996 sampling occurred on 17 sites: 6 Control, 6 Modified, and 5 State sites. An additional site (Ryderwood 1557) was monitored for pretreatment bat activity in 1993 and 1995 but was not harvested until 1996. Post-treatment monitoring of this site occurred in 1997 and 1998. A Control site, Abernathy, was also sampled during these years for comparison (1993, 1995, 1996, 1997, 1998).

To examine differences in detection rates between riparian and upland habitats following harvest, an additional sampling station was established at each site 100 m upland from the stream. In 1996, 17 sites were sampled at both riparian and upland locations. Of these sites Abernathy was also sampled in 1997 and 1998, as was Ryderwood 1557.

Bats were surveyed using remotely deployed ultrasonic detectors (Anabat II bat detectors and delay switches, Titley Electronics, Ballina, N.S.W., Australia) coupled with tape recorders to record echolocation calls of bats. The automated detectors consist of a divide-by-n circuit board which counts the waves in the ultrasonic signal (i.e., echolocation call) and constructs a new wave at the rate of one-for-n. This effectively brings the signal into the range of human hearing and is compatible with cassette tape storage. We used a divisor of 16 for this study. The sound-activated tape recorder stores the bat passes as they occur along with time announcements entered at the time of detection.

Within the riparian zone, a detector was placed 1 m from the ground along the stream bank with the microphone directed upstream. The detector was left in place for 2 consecutive nights then rotated to another site. Sampling for echolocation calls began in late June or early July (depending upon the timing of warm weather) and continued through early September. No sites were sampled in heavy rain due to the decrease in bat activity associated with precipitation (Erkert 1982) and continual triggering of the detector system by raindrops.

For purposes of this study, a detection, or bat pass (Ruggiero et al. 1991), was operationally defined as the sequence of pulses recorded as a bat flies through the airspace sampled by the microphone. High repetition rate "feeding buzzes", known to be associated with prey capture, were identified as feeding activity (Griffin 1958). Activity was indexed as the average number of detections per night within each site. Because most of the bat activity is concentrated within the first four hours following sunset, differences in total night length over the sampling period were not considered influential on nightly detection totals.

The analysis of detections occurred at two levels. First, calls were summed regardless of species to determine the general distribution of detections among sites. Second, detections were grouped into two main categories, *Myotis* and non-*Myotis* bats, based on similar call characteristics using zero-crossing analysis and signal processing software (Anabat V, Titley Electronics, Ballina, N.S.W., Australia). Some calls were too brief or weak to identify and were categorized as "unknown". The presence of feeding activity was assessed by listening to recorded calls and visually inspecting call sonograms for increased pulse rate and drops in the terminal frequency (Jones and Corben 1993).

Limitations of ultrasonic detection

Ultrasonic detection, as with other survey methods, has limitations. The maximum detection distance of a commuting or foraging bat depends on the direction, intensity, and the rate of attenuation of its echolocation call. High frequency calls have a greater rate of attenuation than calls of lower frequencies. As a result, species with different call designs are likely to be differentially detected because of the physics of sound transmission. These differences in detection negate any simple one-to-one relationship between detection rates and absolute population density. In addition, detection rates can not differentiate between one bat flying within the range of the detector multiple times, or multiple bats flying over the detector once. Detection rates, therefore, can only provide an index of relative use by bats at different sites (Thomas and West 1989).

Ultrasonic detection has been used successfully in certain regions to identify bats based on species-specific call characteristics (Fenton 1970, Fenton 1982, Fenton and Bell 1981). However, intraspecific variation in search phase echolocation calls makes identification tentative for many species (Obrist 1995, Thomas et al. 1987). Evidence suggests that there can be substantial variation in pulse characteristics emitted by an individual bat (Schrumm et al. 1991) as well as among bats of the same species (Thomas et al. 1987). Unfortunately, several bat species in the Pacific Northwest have similar echolocation calls making species identification difficult if not impossible, especially within the *Myotis* genus. For analyses in this study, we identified detections as *Myotis* (*M. californicus, M. evotis, M. keenii, M. lucifugus, M. thysanodes, M. volans* and *M. yumanensis*), or non-*Myotis* (*E. fuscus, L. noctivagans, L. cinereus* and *P. townsendii*) due to considerable overlap in call characteristics and the predominance of *Myotis* detections. Members of the *Myotis* group are small, slow flying, agile bats that have steep frequency modulated echolocation calls with lowest frequencies ranging from 35 to 55 kHz (Fig. la). The larger non-*Myotis* bats are characterized as faster, less maneuverable species with lower echolocation frequencies generally below 35 kHz (Neuweiler 1989) (Fig. lb). Separation between these groups, although coarse, does provide insight into ecological differences based on morphology and echolocation call design. In spite of its limitations, ultrasonic detection is a valuable tool for surveying free-flying bats and it is the most appropriate method for simultaneously assessing patterns of bat activity at several sites.

Statistical analyses

Detection data were expressed as the mean number of detections per night for *Myotis*, non-*Myotis* and "all bats". To assess the effect of different treatments on detection rates, we averaged the number of calls detected per night before (1992-1993) and after (1995-1996) harvest. We used the difference between the pre and post mean detection rates as test data (calculated as $\bar{x}_{Post} - \bar{x}_{Pre}$) and analyzed for treatment effects using the non-parametric Kruskal-Wallis test followed by the Dunn test for multiple comparisons. Because of different sampling years and large yearly variations in detection rates, Ryderwood 1557 could not be included In this analysis. However, it was examined in comparison to the Abernathy Control site which was sampled concurrently.

To assess the activity patterns and species composition of bats within the riparian zone and the associated upland with respect to treatment, a repeated measures Analysis of Variance (ANOVA) was calculated using the average detections per night for each upland/riparian pair surveyed. Detection rates were averaged across years when multiple years were surveyed. Because the counts of bat detections had a strongly non-normal distribution, a log transformation was used before testing.

All analyses were done in Systat 7.0 (Wilkinson 1996). Statistical significance was set at a = 0.05.

RESULTS

Riparian habitats

Summary ***of pre-treatment*** *results* **(1992-1993)**

During the pre-treatment period (1992: 10 sites; 1993: 17 sites), a total of 2,433 calls was detected. The *Myotis* group accounted for >97% of these detections with an average of

14 calls recorded per night (Table 1). Other species were rarely detected. Feeding activity was also low within all sites. Of the 2,433 detections, only 12 were identified as feeding activity (≈ 0.5%).

Post-treatment Effects (1995-1996)

A total of 5,937 calls was detected during the post-treatment sampling (17 sites both years). The *Myotis* group accounted for 95% of these detections. Non-*Myotis* bats accounted for 5% of the total detections (n = 281). Forty-nine percent of the non-*Myotis* detections were attributed to big ‘brown or silver-haired bats. Most of the remaining non-*Myotis* detections (36%) could not be identified to genus. Of the 5,937 detections, 72 (≈1%) were identified as feeding activity.

Detection rates were higher during the post-harvest period across all treatments (Fig. 2). During pre-treatment sampling, sites averaged 14.7 (± 1.9) calls per night. During post-treatment sampling, sites averaged 27.9 (± 2.5) calls per night (Table 1). The greatest activity levels were recorded in 1996 with over 36 calls per night.

Changes in detection rates by treatment were assessed by looking at the difference in mean detection rates per night before and after harvest (Fig. 3). Detection rates for “all bats” and *Myotis* bats did not vary significantly with respect to treatment ($P = 0.61$ and $P = 0.70$, respectively), despite the upward trend in mean difference before and after harvest (Fig. 3a, 3b). Differences by treatment for non-*Myotis* detection rates were significant ($P = 0.04$) with highest activity levels recorded within sites harvested under State regulations (Fig. 3c). Differences in detection rates for non-*Myotis* bats between Control and Modified sites and between Modified and State sites were not significant ($0.20 > P > 0.10$ and $P > 0.50$). Significantly different detection rates were found between Control and State sites ($P < 0.05$).

Post-harvest feeding rates were similar among treatments. Control sites had 19 feeding buzzes. State and Modified sites had 26 and 27 feeding buzzes, respectively. Feeding rates were not significantly different among treatments ($P = 0.55$).

Ryderwood 1557

A similar pattern of bat activity was observed in Ryderwood 1557. Pre-treatment sampling (1993 and 1995) resulted in an average of 15 calls recorded per night with 95 % of these being attributed to the *Myotis* group. Post-treatment sampling (1997-1998) had six times greater activity with over 86 calls recorded per night. *Myotis* calls accounted for 85% of these calls. However, Abernathy (Control site) also had higher activity levels recorded during 1997-1998 with three times the activity recorded during 1993 and 1995 (Table 2).

Comparisons between riparian and upland Habitats

For *Myotis* bats, detection rates differed significantly between riparian and upland habitats ($F = 145.72$, df = 1, $P < 0.01$; Fig. 4a). On average, 44 calls were detected per night within the riparian zone and three calls were detected in the uplands (Table 3). This pattern did not vary by treatment ($F = 0.02$, df = 2, $P = 0.98$). For non-*Myotis* bats, there was a significant difference in activity levels between treatments ($F = 8.35$, df = 2, $P = 0.004$; Fig. 4b), as well as a significant interaction between habitat and treatment ($F = 4.62$, df = 2, $F = 0.03$; Fig. 4b). Non-*Myotis* activity levels were not significantly different between riparian and upland habitats ($F = 0.13$, df= 1, $P = 0.72$).

Less than 2% of the calls recorded were identified as feeding activity. For “all bats”, feeding activity was not significantly different between treatments ($F = 0.30$, df = 2, $P = 0.74$). However, there was a trend toward higher feeding rates in riparian habitats as compared to upland habitats. ($F = 3.99$, df = 1, $P = 0.06$).

DISCUSSION

There are many factors that might affect the attractiveness of rivers and streams to bats (Racey 1998). Use of a particular riparian area may in part depend on the amount of physical clutter associated with the stream corridor. Mackey and Barclay (1989) demonstrated that artificial clutter (Styrofoam "rocks") added to the surface of calm water, reduced the foraging activity of *Myotis lucifugus*, a species that tends to fly very close to the water's surface. They hypothesized that the clutter increased acoustic complexity making it difficult to discriminate prey echoes from background echoes. Vegetative clutter is also known to reduce bat activity over streams by making flight and the maneuvers required to capture prey more difficult (Zahn and Maier 1997). In their study of the Kruger National Park bat fauna, Aldridge and Rautenbach (1987) observed that foraging habitat fell into different categories varying in the degree of clutter. Each habitat was occupied by one to three common species according to wing morphology: narrow-winged, less maneuverable species foraged in open areas while broad-winged species with greater maneuverability foraged in cluttered areas.

A similar morphological separation between *Myotis* and non-*Myotis* bats may have occurred in this study. While members of the *Myotis* group are small, slow flying, agile bats with steep frequency modulated echolocation calls, the larger bodied non-*Myotis* bats are generally characterized as faster, less maneuverable species (Neuweiler 1989). Based on these characteristics, non-*Myotis* bats would be predicted to forage within open habitats rather than cluttered areas (Fenton et al. 1980). Not surprisingly, this study found very low detection rates for non-*Myotis* bats in forested uplands compared to harvested uplands. Similarly, significantly higher detection rates of non-*Myotis* species were detected in State harvested sites than Control sites. These differences in activity indicate a favorable response by non-*Myotis* bats to increasingly open habitats, which is a pattern that has been previously documented in this region (Erickson 1997, Hayes and Adam 1996).

While the higher detection rates for non-*Myotis* bats within harvested uplands agree with the hypothesis that larger, less maneuverable bats prefer to fly in open habitats, it is less clear why detection rates were also higher within the adjacent riparian corridor (presumably still a "cluttered" habitat). We suggest that these higher detection rates may be explained by the detectors recording large bats that flew near but outside of the riparian corridor (along the newly created buffer edge or above the riparian canopy). Because larger bats generally produce lower frequency calls than smaller species, their detection distances are greater (Griffin 1971). As a result, it is likely that larger bats flying outside the buffer strip were also detected within the riparian corridor. A similar conclusion was drawn by Brigham et al. (1997) when explaining the lack of an anticipated effect of artificial clutter on activity patterns of large bats.

In contrast to the non-*Myotis* species, *Myotis* bats had significantly lower detection rates in the uplands as compared to rates in the riparian zone. On average, 44 calls were detected per night in the riparian corridor and less than four were recorded in the uplands. These differences in activity demonstrate the importance of riparian habitats to *Myotis* bats, However, detections of *Myotis* species followed a pattern similar to that observed for non-*Myotis* bats with respect to treatment. Activity levels increased from Control to State regulation sites although the increase was not significant.

Even though the response to the different treatments was similar for both *Myotis* and non-*Myotis* bats, the reasons for the observed patterns likely differ to some degree. While creation of the riparian buffers may have resulted in greater amounts of open habitat used by foraging large-bodied bats, it: also created greater amounts of edge. Edge habitats are regularly used by a variety of bat species, however, Limpens and Kapteyn (1991) proposed that small bats in particular prefer flying along edges. Hypotheses suggested to explain this phenomenon include the greater availability of insects, shelter from wind, protection from predators or a reliance on these features for orientation within the landscape (Verboom 1998). Because of their greater reliance on edges, small bats may have had greater commuting and feeding opportunities on the Modified and State sites. In addition, the reluctance to fly over harvested areas may have served to concentrate *Myotis* activity within the riparian corridor.

Another possibility for the differences in activity among treatments could be due to differences in prey availability within the riparian corridors. Hayes and Adam (1996) found that the number and biomass of insects differed between wooded and logged riparian habitat. They hypothesized that these differences may have influenced the species composition of bats within the two habitat types. Similar alterations in prey abundance and composition may have occurred in this study, but insect sampling was not part of the study.

Finally, roost availability may also influence the attractiveness of a riparian area to bats (Racey 1998). Because of the evaporative water loss experienced by day roosting bats (particularly lactating females), the need to drink on emergence from the roost may be one of the factors determining the proximity of roost sites to water (Racey 1998). Additionally, bats may select roost sites close to foraging areas (i.e., streams and rivers) to minimize energy expenditures associated with commuting flight (Christy and West 1993). In the Pacific Northwest, bats appear to preferentially choose roost sites in the vicinity of streams.

Campbell et al. (1996) found roosts of the silver-haired bat to be within 100 m of streams while Ormsbee (1995) found the long-legged myotis to roost within 270 m from class IV streams (stream size reduces from class I to IV) and 2 km from class I streams. Because of the cooler temperatures associated with riparian areas, bats are thought not to roost as readily within the riparian zone. However, it is possible that the harvesting of the associated upland trees increased roost suitability within the uncut riparian zone by providing greater sun exposure on potential roost trees. Further research is needed to assess potential impacts of various buffer configurations on roost suitability.

Management Implications

Despite the high variation in the mean number of calls per night, the Control sites collectively had the lowest detection rates. This suggests that logging of the adjacent forest may not be particularly deleterious to bat populations. For non-*Myotis* bats, creating openings in continuous forest does appear to improve site access within the riparian corridor and its associated upland. Other studies in this region have demonstrated similar increases in detection rates o:f non-*Myotis* species in recently clearcut areas (Erickson 1998, Hayes and Adam 1997). However, the increased activity of *Myotis* bats in the riparian buffers of Modified and State sites, coupled with the low detection rates within the adjacent harvested uplands may be a result of bats avoiding flying over large clearings. Such behavior suggests that some species of bats may be sensitive to habitat fragmentation or conversion. Additional research is needed on individual bat species to determine more precisely the ecological consequences of altering landscape structure, including riparian buffers (Verboom 1998).

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Table 1. Mean number of bat detections for *Myotis*, non-*Myotis* and all bats on 17 sites before (1992-1993) and after (1995-1996) harvest on Control, Modified and State regulation sites (means for unidentified calls are not shown). Ten sites were sampled in 1992 (sites indicated by an *) and 17 in 1993, 1995, and 1996.

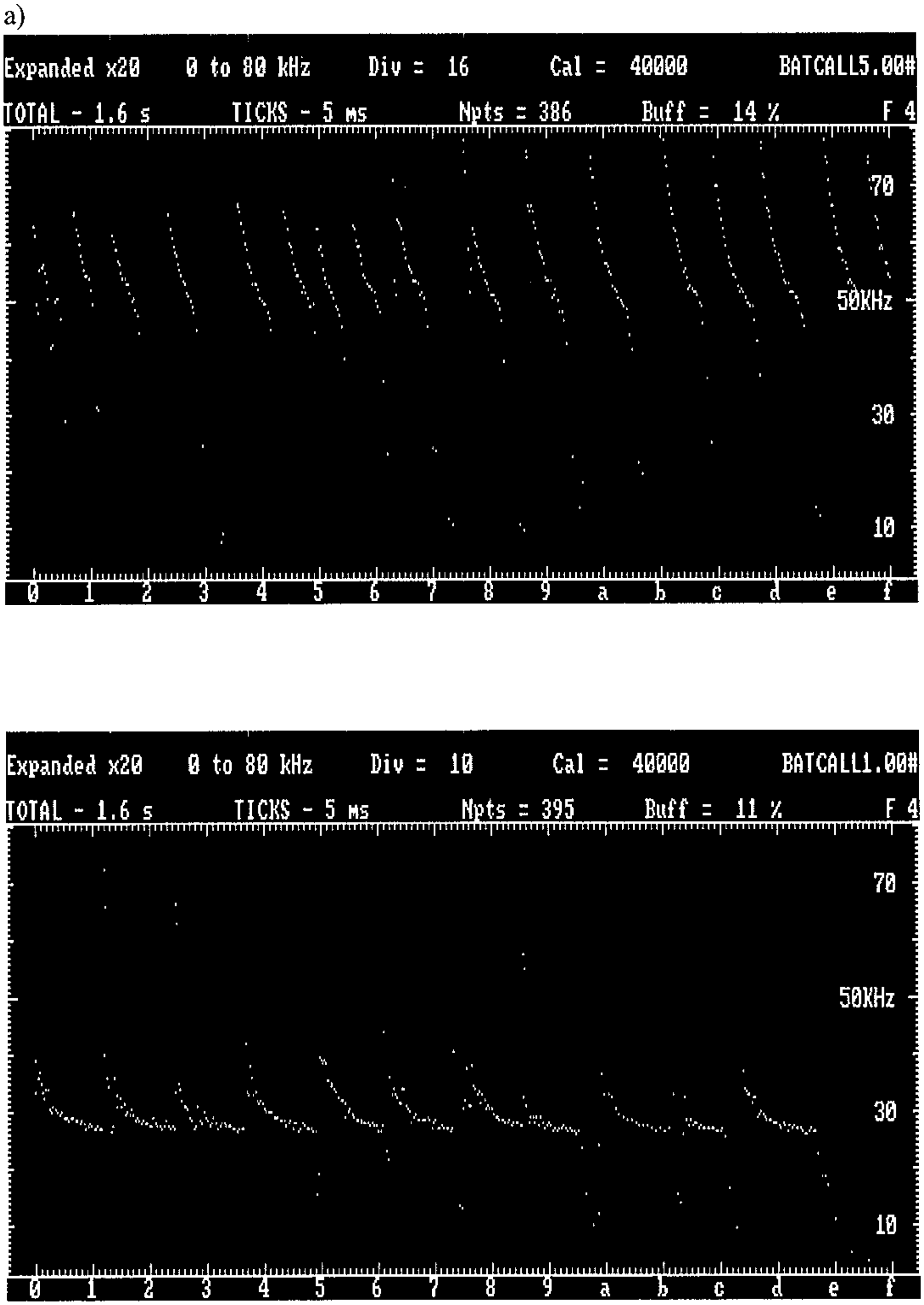
Site	1992-1993			1995-1996		
	Myotis	Non-*Myotis*	All bats	*Myotis*	Non-*Myotis*	All bats
Control						
Abernathy	54.6 (15.1)	0.1 (0.1)	55.4 (15.1)	28.1 (7.8)	0	28.2 (7.8)
Elbe Hills	6.3 (2.4)	0	6.3 (2.5)	41.7 (i2.9)	0.4 (0.4)	22.5 (11.5)
Hotel Creek*	1.0 (0.5)	0	1.8 (0.5)	40.8 (7.6)	42.4 (13.0)	0.3 (0.3)
Porter Creek*	25.9 (9.5)	0	26.4 (9.5)	20.1 (8.4)	0.1 (0.1)	10.7 (2.6)
Taylor Creek*	9.8 (2.3)	0.9 (0.9)	11.1 (2.8)	22.1 (11.5)	40.9 (7.6)	3.3 (1.5)
Vail	11.0 (3.7)	0.2 (0.2)	15.5 (5.5)	10.3 (2.6)	0	14.5 (3.4)
Modified						
Blue Tick	1.8 (1.3)	0	1.8 (1.3)	il.3 (2.5)	20.1 (8.4)	0.2 (0.2)
Eleven Creek 31*	1.9 (0.7)	0	2.4 (0.9)	7.6 (4.2)	0.4 (0.3)	7.8 (4.3)
Griffin Creek*	3.8 (1.4)	0	3.9 (1.8)	47.3 (14.5)	0.1 (0.1)	47.3 (14.6)
Ms. Black*	37.3 (9.7)	0	37.7 (9.6)	32.4 (6.7)	0.4 (0.3)	32.9 (6.8)
Ryderwood 860*	12.1 (2.9)	0	12.3 (2.9)	4.8 (1.2)	0.2 (0.1)	5.1 (1.2)
Side Rod	15.3 (3.7)	0.1 (0.1)	15.8 (3.9)	37.1 (19.2)	I.8 (1.2)	39.0 (20.2)
State						
Eleven Creek 32*	4.6 (2.7)	0.0	4.7 (2.7)	38.3 (13.3)	0.4 (0.2)	38.7 (13.3)
Kapowsin	19.4 (11.9)	0.0	19.6 (12.0)	41.5 (11.3)	0.7 (0.3)	42.2 (11.2)
Night Dancer	8.3 (3.1)	0.0	9.5 (3.4)	14.1 (3.1)	8.2 (3.7)	22.3 (4.7)
Pot Pourri*	2.1 (0.5)	0.0	2.1 (0.5)	1.7 (0.6)	0.2 (0.1)	2.7 (1.1)
Simmons Creek*	24.0 (11.8)	0.0	24.0 (11.8)	51.7 (7.5)	6.6 (1.9)	58.3 (9.0)
Overall Mean	14.2 (1.9)	0.1 (0.1)	14.7 (1.9)	26.5 (2.5)	1.3 (0.30)	27.9 (2.51)

Table 2. Mean number (± SE) of bat detections for *Myotis*, non-*Myotis* and all bats on Ryderwood 1557 (State site) and Abernathy (Control site) before (1993 and 1995) and after (1997-1998) harvest (means for unidentified calls are not shown).

Site	Calls per night					
	1993 and 1995			1997-1998		
	Myotis	Non-*Myotis*	All Bats	*Myotis*	Non-*Myotis*	All Bats
Ryderwood 1557	14.1 (4.6)	0.5 (0.4)	14.8 (4.9)	73.8 (13.9)	0.2 (0.1)	86.1 (16.3)
Abernathy	42.4 (10.3)	0.1 (0.1)	42.9 (10.4)	90.9 (24.9)	1.1 (0.7)	102.5 (28.5)
Overall Mean	27.3 (5.9)	0.3 (0.2)	27.9 (6.0)	82.7 (14.4)	0.7 (0.4)	94.6 (16.5)

Table 3. Mean number (± SE) of bat detections for *Myotis*, non-*Myotis* and all bats recorded at riparian and upland sampling stations (means for unidentified calls are not shown).

Site	Years	Caiis per night					
		Riparian			Upland		
Control		*Myotis*	Non-*Myotis*	Total	*Myotis*	Non-*Myotis*	Total
Abernathy	1996-1998	70.5 (18.00)	0.7 (0.44)	78.2 (20.62)	0.4 (0.17)	0	1.3 (0.31)
Elbe Hi%	1996	75.0 (16.41)	0.8 (0.83)	75.8 (16.73)	8 (2.07)	0	8 (2.07)
Hotel Creek	1996	46.8 (11.98)	0	46.8 (11.98)	4 (2.79)	0	4 (2.79)
Porter Creek	1996	32.3 (15.75)	0	32.3 (15.75)	1 (0.63)	0	1 (0.63)
Taylor Creek	1996	34.9 (20.72)	0.7 (0.57)	35.6 (20.54)	2.6 (1.17)	0	2.6 (1.17)
Vail	1996	7.8 (3.59)	0.7 (0.49)	8.5 (3.45)	0.7 (0.67)	0	0.7 (0.67)
Modified							
Blue Tick	1996	13.3 (3.28)	6.5 (2.35)	19.8 (5.08)	2.0 (0.45)	3.0 (1.06)	5.0 (0.97)
Eleven Creek 3 1	1996	14.8(8.59)	0.5 (0.34)	15.3 (8.62)	2.8 (1.76)	1 .0 (0.82)	3.8 (1.62)
Griffin Creek	1996	81.8(20.86)	0.2(0.17)	82.0 (20.94)	6.0 (0.55)	9.4(4.34)	15.4 (4.15)
Ms. Black	1996	40.0(12.17)	0.8 (0.54)	40.8 (12.43)	2.3 (0.92)	5.5(1.65)	7.8 (1.78)
Ryderwood 860	1996	4.6 (2.15)	0.1 (0.14)	4.7 (2.16)	1.3 (0.57)	l(0.58)	2.3 (0.89)
Side Rod	1996	66.3(35.72)	3.7 (2.23)	70.0 (37.47)	8.7 (1.05)	3.0(0.93)	11.7 (1.52)
State							
Eleven Creek 32	1996	67.3(18.69)	0.1 (0.14)	67.4 (18.78)	4.0 (1.41)	0.6(0.24)	4.6 (1.36)
Kapowsin	1996	16.93.46	1.1(0.48)	18.0 (3.74)	1.6 (0.60)	0	1.6 (0.60)
Night Dancer	1996	15.24.84	15.5(6.18)	30.7 (6.77)	2.2 (1.05)	7.3 (3.89)	9.5 (4.89)
Pot Pourri	1996	2.21.17	0.3(0.33)	4.3 (2.26)	1.2 (0.65)	1.3 (0.61)	2.5 (1.12)
Simmons Creek	1996	38.510.38	5.0(2.21)	43.5 (12.27)	4.6 (2.04)	2.2 (1.28)	6.8 (3.12)
Ryderwood 1557	1997, 1998	73.8 (13.88)	0.2 (0.12)	86.1 (16.28)	6.0 (2.26)	1.4 (0.40)	9.1 (2.69)
Overall Mean		43.9 (4.60)	1.8 (0.43)	48 (5.02)	3.1 (0.39)	1.7 (0.34)	5.0 (0.57)

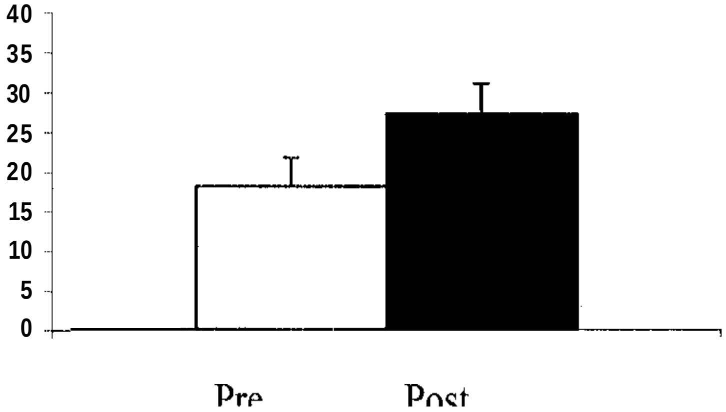

a)

```
Expanded x20    0 to 80 kHz    Div =  16    Cal =  40000    BATCALL5.00#
TOTAL - 1.6 s       TICKS - 5 ms     Npts = 386     Buff =  14 %     F 4
```

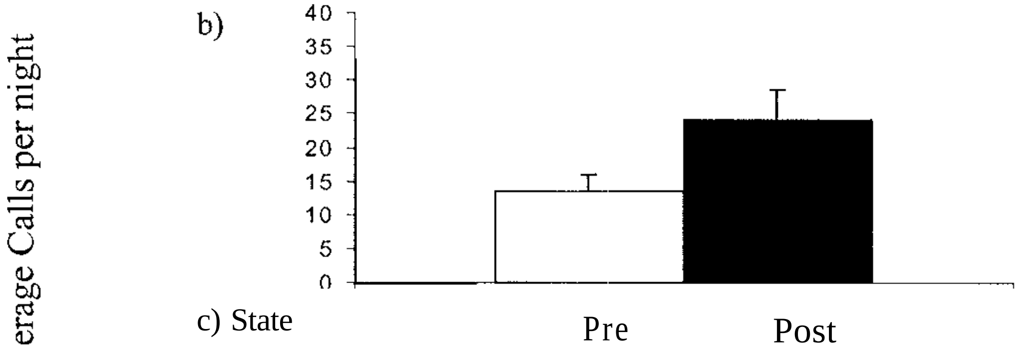
```
Expanded x20    0 to 80 kHz    Div =  10    Cal =  40000    BATCALL1.00#
TOTAL - 1.6 s       TICKS - 5 ms     Npts = 395     Buff =  11 %     F 4
```

Figure 1. Typical echolocation calls for a) *Myotis* and b) non-*Myotis* bats.

a) Control



b) Modified



c) State

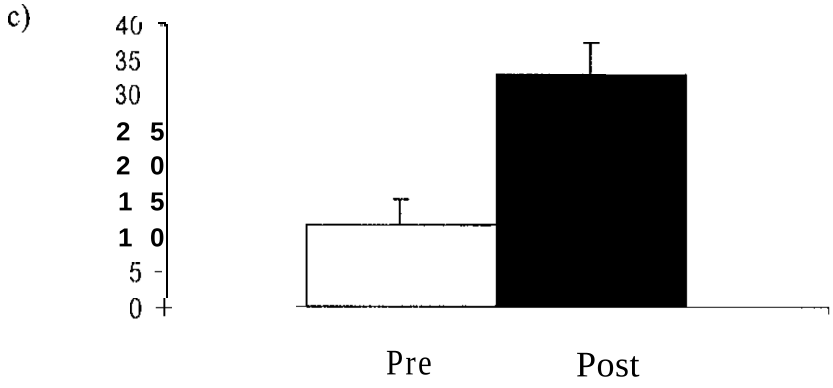


Figure 2. Mean calls per night (mean ± se) pre- (1992-1993) and post-harvest (1995-1996) on a) Control, b) modified and c) State sites.

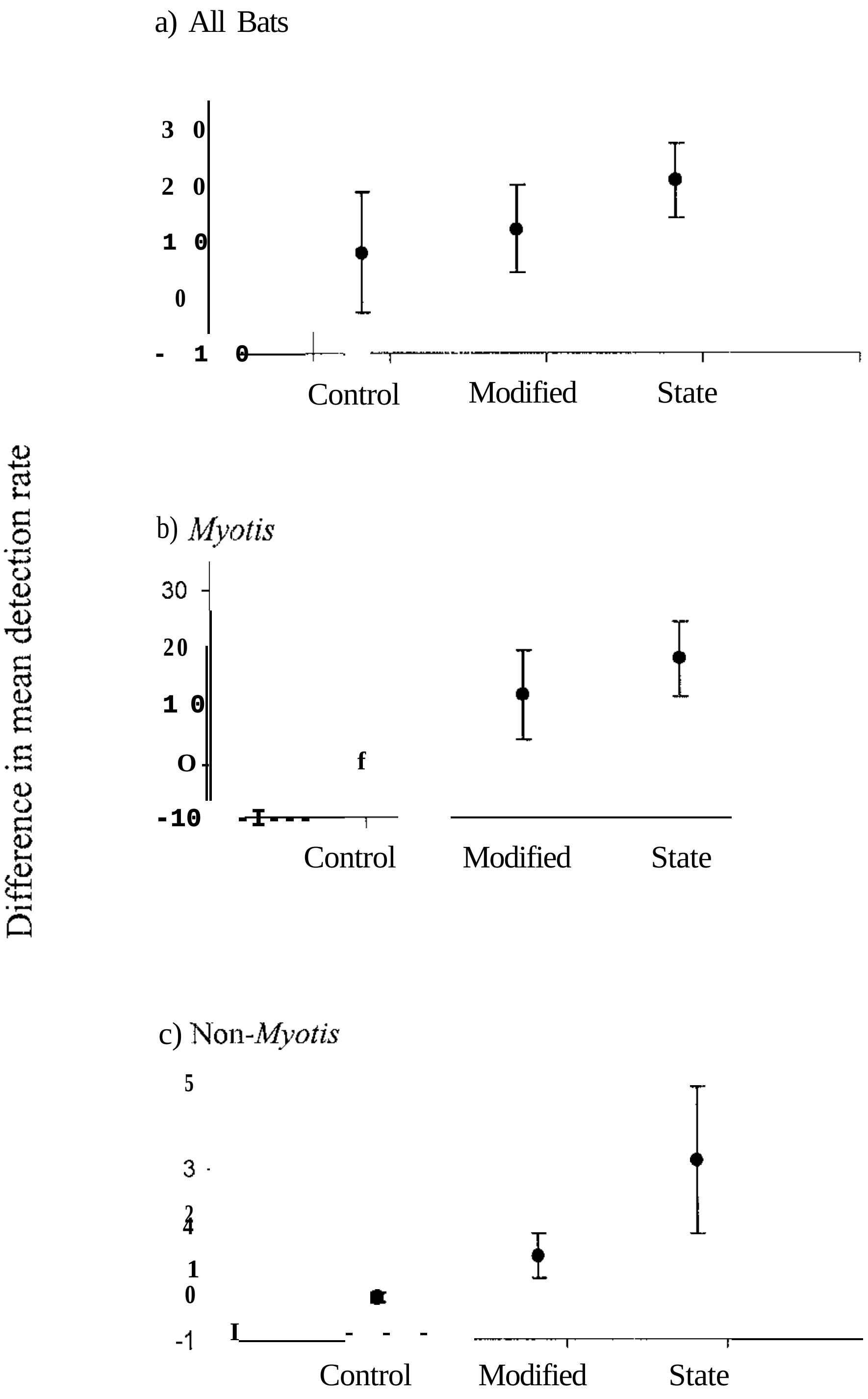
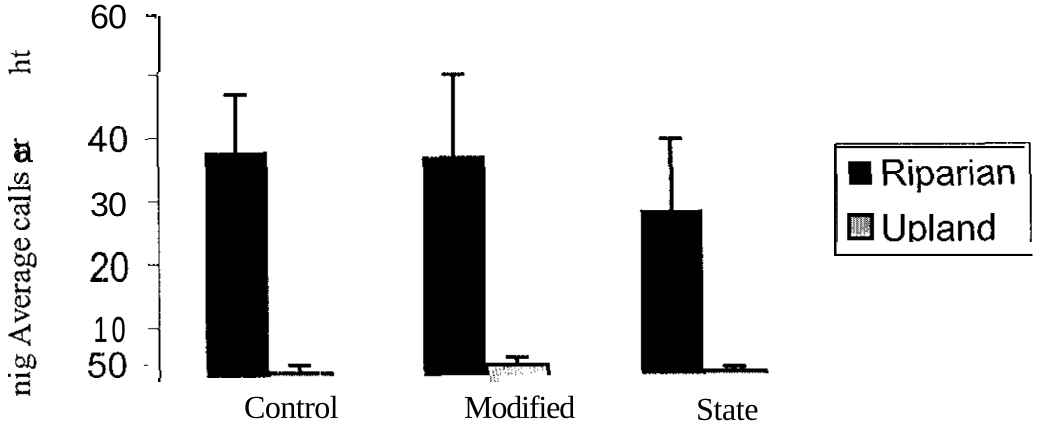


Figure 3. Differences by treatment in mean (SE) detection rates per night for a) all bats, b) *Myotis* bats, and *c)* Non-*Myotis* bats before and after harvest for the riparian sampling station. Indices are derived from mean detection rates before (1992-1993) and after (1995-1996) harvest. Charted values are $\bar{x}_{Post} - \bar{x}_{Pre}$.

a) *Myotis*



b) Non-Myotis

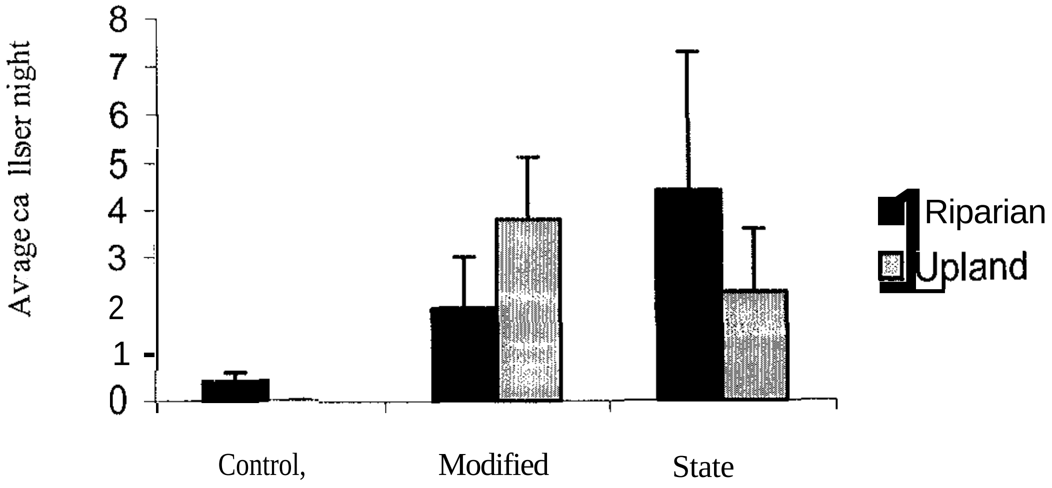


Figure 4. Mean (± SE) calls per night in riparian and upland habitat on Control, Modified, and State regulation sites for a) *Myotis* and b) non-*Myotis* bats.

Chapter 13

EAST-SIDE BAT SURVEYS

Abstract. Bat activity was studied at 18 riparian and adjacent upland managed forest sites in northeastern Washington before and after logging to examine habitat associations and response to different riparian timber harvest prescriptions. Capturing bats in mist nets or harp traps determined the composition of the bat communities. Bat activity was monitored using ultrasonic bat detectors. Eight species were captured in mist nets or harp traps and an additional species was detected with the bat detectors. Bat activity was assessed from 6,402 calls collected during 45 1 sampling nights. Activity levels of the *Myotis* species were greater in the riparian than upland habitats, but did not differ between habitats for *Eptescicus fuscus* or *Lasionycteris noctivagans*. Activity levels of the *Myotis* species in the riparian habitats increased after logging on sites which were harvested according to a Modified riparian buffer prescription designed to protect habitat features such as snags and seeps, but remained the same between years on the sites (State) which were harvested according to the Washington State Forest Practice Guidelines and on unharvested Control sites. Activity levels of E. *fuscus* decreased after logging in the riparian habitats on the State sites, but remained similar between years on the Modified and Control sites.

INTRODUCTION

Of the 14 species of bats found in Washington, 11 species use forested lands as either primary or secondary habitat, three are designated as species of concern in the state, and 9 of these 14 species were ranked as potentially sensitive to disturbance of the riparian habitat (O'Connell et al. 1993). Advances in radio-telemetry and ultrasonic detection technology combined with an increased concern for the management of these microchiropterans have resulted in recent studies addressing the ecology of bats in temperate forests (e.g., Campbell et al. 1996).

The abundance and distribution of bats in temperate forests has been tied to the effects of forest structure on movement patterns, resource distribution, and microclimate variations (Bradshaw 1996). The home ranges of bats encompass day and night roost sites, foraging areas, and water sources and, by their movements between these, bats link these habitats. Riparian habitats represent an important resource for bats as either foraging areas (especially for *Myotis* species, e.g., Lunde and Harestad 1986, Thomas and West 1991, Brigham et al. 1992) or as sources of free water (e.g., Cross 1988) and are considered a key environmental correlate for 10 bat species in the Interior Columbia Basin (Marcot 1996). However, roost sites of at least some species of bats in the Pacific Northwest tend to be removed from riparian habitats, perhaps due to microclimatic considerations (e.g., Betts 1996, Campbell et al. 1996, Frazier 1998). The juxtaposition ofriparian and forested upland habitats as well as the structure of the riparian and upland forests are important factors in management of bats in the forests of this region. Given their wide movement patterns and reliance on different habitats, it is not surprising that evidence suggests that the response of bats to timber management is variable. For example, logging resulted in increased bat activity in upland forests of western Washington (Erickson and West 1996), but decreased bat activity in riparian forests of western Oregon (Hayes and Adam 1996). Many of these studies have compared bat activity in forests that had previously been cut to unharvested forests; few have conducted pre- and post-harvest studies. In this study, we examined the response of bats to timber management in riparian habitat of northeastern Washington. Our specific objectives were to 1) identify the species inhabiting these riparian and adjacent upland forests and 2) to compare bat activity before and after different timber harvest treatments of riparian forests.

METHODS

Study area

Research was conducted in mixed-coniferous forests in the Selkirk Mountains of northeastern Washington (Stevens and Pend Oreille counties). Forest composition in this region is variable and is affected by slope, aspect, edaphic factors, fire history, and timber management practices. Dominant tree species included Douglas-fir (*Pseudotsuga menziesii*), lodgepole pine (*Pinus contorta*), western redcedar (*Thuja plicata*), western hemlock (*Tsuga heterophylla*), western larch (*Larix occidentalis)*, grand fir (*Abies grandis)*, and alders (*Alnus incana* and *Alnus sinuata).*

We selected 18 sites that met the following criteria: 1) 800-m reach of Type 3 or permanent Type 4 stream; 2) >16.2 ha previously harvested stands of harvestable age on either side of stream; 3) ≥610 m and ≤1200 m elevation; 4) mixed coniferous forest; 5) landowners in agreement to either leave sites unharvested for 10 yr (Controls) or to harvest sites within timeframe and specifications of study design (State and Modified sites). Seven sites were unharvested Control sites. The upland areas of 11 sites were selectively harvested for timber in 1993-1994 to yield a 6- to 12-m post-harvest spacing of trees. The riparian zones of 6 of the 11 cut sites were harvested according to the Washington State Forest Practices RMZ (State sites) guidelines and 5 of the cut sites were harvested according to a modified prescription (Modified sites) designed for this project. The intent of the buffers on the Modified sites was to incorporate a site-specific approach to riparian management. Within 33-m zone of the stream, habitat features, such as seeps, snags, and deciduous trees, were identified and protected. For example, one snag per 2 acres was buffered by a no-entry zone equal to 1.5X the height of the snag and all seeps were buffered by a 1 O-m no-entry zone that extended to the stream. Following timber harvest, the mean width of buffers on the State sites was 14.1 ± 3.0 m with a range from 8-22.6 m. and the mean width of the buffers on the Modified sites was 29.7 ± 17.4 m with a range from 12 to 144 m.

Bat sampling

To determine the bat species present in these forests, bats were captured using mist nets and harp traps. The mist nets were standard, 4-pocket nets, either 6 m or 12 m in length. The harp traps were constructed according to the collapsible design presented in Tidemann and Woodside (1978). The trap is designed like a harp and is strung vertically with monofilament line in two banks. Bats are captured in a protective pocket directly below the two banks of line. This technique has been shown to capture up to 10 times more bats than the traditional mist net (Tidemann and Woodside 1978). Traps were set across slow moving areas of streams and across shallow ponds (9 sites), which are potential drinking and foraging locations for bats, and across narrow roads (10 sites). Captured bats were identified to species, sex, age, and reproductive condition. Female reproductive condition was determined by abdominal palpation and examination of mammary condition. Males were judged to be reproductive if their testes were scrotal. In addition ear, forearm, and hind foot length were recorded.

Ultrasonic sampling

Bat activity at the 18 RMZ sites was measured using ultrasonic detectors based on the design by Anderson and Miller (1977) and constructed by Titley Electronics, Ballina, N.S.W., Australia. The detectors consisted of 1) a divide-by-10 circuit board to count the waves in the signal and construct a new wave at the rate of 1 for 10 that is within the range of human hearing, 2) a talking alarm clock which turned the system on at dusk and off at dawn and entered time markers, and 3) a voice-activated tape recorder to store bat calls and time announcements. These units were housed in a waterproof plastic tackle box with a hole cut out for the microphone which was protected from rain by a plastic cover. Each unit was secured on a metal pole that elevated the detector ca. 1 m above the ground and angled the microphone upward at a 40” angle.

Bat detectors were placed in the same location along the riparian transect at each site for 2 consecutive nights in August 1993, 2 consecutive nights per month during June, July, and August 1994-1995 and along the riparian and upland transects for 2 consecutive nights per month during June, July, and August 1996. If rain occurred during the sampling nights or if detectors malfunctioned, every attempt was made to resample the site during the same month.

Call analysis

Tapes of the calls and time announcements were analyzed using a Zero-Crossing Analysis Interface Module (ZCAIM, Titley Electronics, Ballina, N.S.W., Australia) and signal processing software (Anabat 5). With zero-crossing analysis the input signal is squared and the successive zero-crossings are counted to determine the instantaneous frequency at different times. The signal is digitized, sent to a computer, and displayed as a function of frequency and time (sonogram). The software incorporates an equalizer that eliminates interfering noise to provide a clearer sonogram.

A detection or bat pass was defined as a sequence of pulses recorded as a bat flies through the airspace sampled by the microphone. Feeding activity was identified as high repetition rate “feeding buzzes” occurring during prey capture. Each detection was reviewed for maximum and minimum frequency, duration, pulse shape, number of pulses, and occurrence of feeding buzzes. Calls with <3 pulses in the sonogram were not included in the analyses. Designation of calls to species or species groups was based on comparison with libraries of calls compiled by recordings made from free flying bats of known identity, the

U.S. Forest Service's Old Growth Wildlife Habitat Program, and the University of Washington. Calls for which only fragments were recorded or that were otherwise unclear were classified as "unknown."

Data analysis

Relative bat activity was measured as either the mean number of calls per site-night or the mean number of calls per 30-min interval. To examine inter-site variation, habitat associations (i.e., riparian vs. upland) of the species, and treatment effects, the mean number of bat calls per sample night were analyzed using the Kruskal-Wallis analysis of variance (SAS Institute 1989). Treatment effects on individual species were analyzed using the mean number of calls per sample night, but were converted to log scale for presentation. Habitat associations were also examined by comparing the mean number of feeding buzzes per sample night using the Kruskal-Wallis analysis of variance (SAS Institute 1989). Temporal patterns of bat activity were (examined using the mean number of calls per 30-min interval.

RESULTS

Bat sampling

Fifteen sites in the Calispell basin, along creeks and roads, were trapped during July-September 1992 (32 trap nights: 20 mist net, 12 Tuttle trap nights) and May-August 1993 (46 trap nights: 31 mist net, 15 Tuttle trap nights). We captured 114 bats representing eight species: *Myotis californicus*, *M. ciliolabrum*, *M. evotis* (a Washington state sensitive species), *M. lucifugus*, *M. yumanensis*, *Eptesicus fuscus*, *Lasionycteris noctivagans*, and *Lasiurus cinereus* (Table 1). Species composition varied with capture site (Table 2). All eight species were captured over water, but two of the *Myotis* (*M. californicus* and *M. ciliolabrum*) were more commonly captured at the road sites. All eight species were caught in mist nets, but only *Myotis* species appeared in the Tuttle traps. Fifty-four percent of bats captured were adult males; 33% percent were adult females. The proportion of males to females varied by species (Table 1). Sixty-three percent of adult females and 47% of adult males were reproductive. Fourteen juvenile bats (8 male, 6 female) were trapped, accounting for 12% of all captures. Juvenile *Myotis lucifugus* and *M. californicus* first appeared in traps on 15 July 1992 (n = 5 for this date). The first juvenile *M. yumanensis* and *Lasionycteris noctivagans* appeared on 4 August 1992 (n = 5 for this date). Juvenile bats accounted for 23% of all captures after 15 July 1992. No juvenile bats were captured in 1993. Trapping ended in August prior to the emergence ofjuvenile individuals. A late summer season and prolonged rain and cold weather probably delayed parturition (Grindal et al. 1992, Findley 1993).

Ultrasonic sampling

A total of451 sample nights (=night/site) were monitored for bat activity between August 1993 and August 1996 yielding 6,402 calls (Table 3). There were 116 nights with no bat calls detected. Equipment malfunctions or adverse weather conditions resulted in 69 nights with no sampling.

Four species and one species group were detected (Table 3). Three of these species, *Eptesicus fuscus*, *Lasionycteris noctavagans*, *Lasiurus cinereus*, had been captured by mist netting whereas *Corynorhinus townsendii* had not been captured. However, bat sampling elsewhere in the Selkirk Mountains confirmed the presence of *C. townsendii* (J. G. Hallett and M. A. O'Connell, personal observation). All *Myotis* species were grouped.

Bat activity varied signigicantly both between and within sites. At all sites there were ≥2 sample nights with no bat activity recorded. The proportion of sample nights with no calls varied from 9-60% (Fig. 1). The number of sites with a high proportion of no call sample nights (>50%) and the number of sites with a low proportion of no call sample nights (<25%) were equally distributed between the control and treatment sites (Fig. 1). The mean number of calls per sample night per site ranged from 1.5 ± 0.0 to 66 ± 23 (Kruskal-Wallis $X^2 = 50.9$; df = 17; $P = 0.0001$) (Fig. 1). The large standard errors associated with these means (Fig. 1) reflect the variation within sites. Several sites exhibited significant annual variation in mean calls per sample night and much of this variation was attributed to one or a few nights with very high bat activity.

Habitat: riparian versus upland

Habitat use of riparian versus upland habitats was based only on the 1996 data. The mean number of calls per sample night did not differ between riparian and upland habitats on the Control Sites for *Lasionycteris noctavagans* (Kruskal-Wallis $X^2 = 1.3\ 1$; df = 1; $P = 0.25$) or *Eptesicus* ***fuscus*** (Kruskal-Wallis $X^2 = 0.01$; df = 1; $P = 0.91$) (Fig. 2). In contrast, the mean number of calls per sample night was greater in the riparian than upland habitats on the Control sites for *Myoti*s (Kruskal-Wallis $X^2 = 8.05$; df = 1; $P = 0.004$) (Fig. 2). A similar pattern of habitat association, was observed with respect to feeding activity (Fig. 3). The mean number of feeding buzzes per sample night was greater for *Myotis* in riparian ($\bar{x} = 3.1 \pm 1.5$) as compared to upland ($\bar{x} = 0.3 \pm 0.2$) habitats (Kruskal-Wallis $X^2 = 6.4$; df = 1; $P = 0.01$). Although the mean number of feeding buzzes for *Eptesicus fuscus* was greater in upland ($\bar{x} = 0.23 \pm 0.02$) than riparian habitats ($\bar{x} = 0.02 \pm 0.02$), sample size was low and the differences not significant (Kruskal-Wallis $X^2 = 0.5$; df = 1; $P = 0.82$). The mean number of feeding buzzes did not differ between habitats for *Lasionycteris noctivagans* (Kruskal-Wallis $X^2 = 1.5$; df = 1; $P = 0.22$) (Fig. 3). Neither *Lasiurus cinereus* nor *Corynorhinus townsendii* were observed on the Control sites.

Similar to the Control sites, the mean number of calls per sample night did not differ between riparian and upland habitats on the State and Modified sites for *Lasionycteris noctavaguns* (Kruskal-Wallis X^2 = 2.1; df = 1; P = 0.15) and the mean number of calls per sample night was greater in the riparian than upland habitats for *Myotis* (Kruskal-Wallis X^2 = 12.1; df = 1; P = 0.0005; Fig. 2). In similar fashion, the mean number of feeding buzzes per sample night was greater for *Myotis* in riparian as compared to upland habitats on the State and Modified sites (Kruskal- Wallis X^2 = 3.9; df = 1; P = 0.048) and did not differ between habitats for *Lasionycteris noctivagans* (Kruskal-Wallis X^2 = 0.2; df = 1; P = 0.65; Fig. 3). In contrast, *Eptesicus fuscus* was detected more often in the upland habitats of the State and Modified sites (Kruskal-Wallis X^2 = 5.6; df = 1; $P = 0.02$; Fig. 2). However, the mean number of feeding buzzes per sample night did not differ between habitats for *Eptesicus fuscus* (Kruskal-Wallis $X^2 = 0.52$; df = 1; $P = 0.47$; Fig. 3). *Corynorhinus townsendii* was observed only in the riparian habitats of the cut sites. The mean number of detections per sample night of *Lasiurus cinereus* did not differ between the riparian ($\bar{x}$ = 0.01 ± 0.01) and upland ($\bar{x}$ = 0.05 ± 0.04) habitat on the cut sites (Kruskal-Wallis X^2 = 0.51; df = 1; $P = 0.47$).

Temporal patterns of total bat activity were generally similar between the riparian and upland habitats and in neither habitat were uniformly distributed throughout the night (Fig. 4). Activity in both habitats was greatest in the first part of the night (17:30-22:30). The mean calls per sample night was greater in the upland riparian habitat during only two time periods, early in the evening (18:30) and at dawn (04:00).

Treatment effects

The mean number of total bat calls per sample night did not differ between the Control, State, and Modified (Kruskal-Wallis X^2 = 6.02; df = 4; P = 0.20; Fig. 5). Although the mean number of total bat: calls per sample night decreased between the pre-harvest ($\bar{x}$ = 24.6 ± 13.4) and post-harvest ($\bar{x}$ = 19.7 ± 4.8) on the State sites and increased between the pre-harvest ($\bar{x}$ = 18.3 ± 5.8) and post-harvest ($\bar{x}$ = 28.4 ± 7.4) on the Modified sites, the variances were great and the differences not significant. However, there were significant differences observed between sites for individual species. The activity of both *Eptesicus fuscus* (Kruskal-Wallis $X^2 = 16.1$; df = 4; $P = 0.003$) and *Lasionycteris noctivagans* (Kruskal-Wallis $X^2 = 12.8$; df = 4; $P = 0.01$) was greater on the State sites as compared to the Control and Modified sites (Fig. 6). The mean number of bat calls for *Eptesicus fuscus* per sample night decreased between pre- and post-harvest on the State sites (Fig. 6). In contrast, the mean number of calls per sample night for *Myotis* (Kruskal-Wallis $X^2 = 10.5$; df = 4; $P = 0.03$) was greater after harvest on the Modified sites (Fig. 6). There were no differences between treatments in the mean number of feeding buzzes for *Myotis* in the riparian zone (Kruskal-Wallis $X^2 = 1.34$; df = 4; $P = 0.85$).

DISCUSSION

Thomas and West (1991) predicted the presence of 12 species of bats in the forests of Washington. During this study we observed all but three of those species, *Myotis thysanodes*, *M. keenii*, and *M. volans*. *M. thysanodes* is associated with drier habitats than the forests of northeastern Washington. *M keenii* is found in forests on the west side. It is likely that *M.* volans is present in the forests of northeastern Washington.

Comparison between riparian and upland habitats on Control sites

Our observation of higher detection rates for calls of the *Myotis* group in the riparian than upland habitats is consistent with most studies that have used ultrasonic sampling of bat populations in the Pacific Northwest (e.g., Lunde and Harestad 1986, Grindal 1996, Hayes and Adam 1996, Parker et al. 1996). The increased activity of *Myotis* species in riparian habitat has been attributed to their use of these habitats for feeding (e.g., Barbour and Davis

1969, Lunde and Harestad 1986). We also observed elevated rates of feeding buzzes for the *Myotis* group in the riparian habitat. Interspecific differences in habitat use and the reliance upon riparian sites for feeding have been suggested for Myotis. For example, in our review of wildlife use ofriparian habitat, we cited evidence that *M. yumanensis* appears more reliant on water than *M. ciliolabrum* (O'Connell et al. 1993). Our capture data support this; *M. ciliolabrum*, was more commonly captured at road sites and *M. yumanensis was* more common over water. *M. californicus* was also more commonly captured at the road sites, but evidence suggests that this species relies on riparian habitat for feeding (e.g., Fenton et al. 1980).

Detection rates for calls of *Eptesicusfuscus* were comparable between riparian and upland habitats on the Control sites, but were greater in the upland than riparian habitats on the cut sites, Studies of habitat use by *Eptesicusfuscus* in Pacific Northwest forests have reported that this bat is detected more commonly on sites that have been harvested as compared to mature stands (e.g., Erickson and West 1996).

The relative abundance of bat species in riparian versus upland habitats reflected both the feeding activities as well as the roosting activities of bats in these habitats. The higher detection rates in the upland than riparian habitats during the first part of the evening and at dawn most likely corresponded to movement from roosting sites. For example, *Lasionycteris noctivagans* has been reported to roost in upslope habitats in these forests (e.g., Campbell et al. 1996), *Eptesicus fuscus* traveled up to 4.1 km from roosting sites (Brigham and Fenton 1986), and *Myotis lucifigus* has been observed foraging up to 5 km from roosting sites. For species that utilize trees for roosting, the availability of suitable roost trees, especially snags, (Campbell et al. 1996, Vonhof and Barclay 1996) might dictate relative use of riparian and upland habitats.

Post-treatment effects

The site-specific protection of snags that was incorporated into the riparian harvest of the Modified sites might explain the increased activity of the *Myotis* group after harvest of the upland on these sites. At least two species of *Myotis* (*M. yumanensis* and *M. lucifigus*) are known to roost near water when suitable roosts are present (Barbour and Davis 1969). Shorter distances between suitable roost trees and low canopy closure have been shown to influence roost selection in other species (Vonhof and Barclay 1996). Therefore, retention of snags in the riparian habitats on the Modified sites after harvest might have led to increased activity. In combination with the protection of snags in the riparian habitat, the partial cut in the upland might have contributed to the increased rates of detection. In a preliminary study of the effects of partial cutting on bat activity, Perdue and Steventon (1996) observed greater or equal activity in partial cut forests as compared to no harvest or clearcut forests.

The use of ultrasonic detection to sample bat populations is based on a number of assumptions about the consistency of call structure (e.g., Erickson and West 1996) and must be approached with caution (e.g., Lance et al. 1996). As Thomas and West (1989) concluded, ultrasonic detection is best used to examine broad patterns. In this study we combined ultrasonic detection data with capture data to examine patterns of habitat use between riparian and upland habitats and between different harvest treatments in riparian habitats. The patterns of abundance of the bat species or species groups in riparian and upland habitats that we observed were similar to those observed by others in the region. In addition, we observed that the site-specific approach to riparian harvest that was incorporated into the riparian buffers on the Modified sites resulted in increased activity of the *Myotis* species group. The importance of upland habitats for the bats of this region must also be stressed and a management approach that ensures adequate roosting habitat in the uplands is an essential complement to riparian habitat management.

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Table 1. Bat species captured in July-September 1992 and May-August 1993 at 19 locations in northeastern Washington. Totals are shown for each species and each group.

	Adults		Juveniles		Total
Species	M	F	M	F	
Myotis californicus	5	14	2	2	23
Myotis ciliolabrum	13	6	--	--	19
Myotis evotis	4	2	--	--	6
Myotis lucifugus	12	1	3	2	18
Myotis yumanensis	12	1	--	1	14
Eptesicus fuscus	2	--	--	--	2
Lasionycteris noctivagans	12	14	3	1	30
Lasiurus cinereus	2	--	--	--	2
	62	38	8	6	114

Table 2. Bat species captured at 9 water and 10 road sites in July-September 1992 and May-August 1993 in the North Fork of the Calispell watershed.

Species	Water sites	Road sites
Myotis californicus	8	15
Myotis ciliolabrum	3	16
Myotis evotis	2	4
Myotis lucifugus	16	2
Myotis yumanensis	12	2
Eptesicus fuscus	2	0
Lasionycteris noctivagans	30	0
Lasiurus cinereus	2	0

Table 3. Summary of sample nights and total bat-call detections by habitat for ultrasonic sampling on 18 study sites in northeastern Washington,1993-1996.

Habitat	Sample nights	No calls	*E. fuscus*	*L. cinereus*	*L. noctivagans*	*Myotis*	*C. townsendii*	Unknown
Riparian	305	84	236	18	357	5149	7	119
Upland	77	32	108	4	108	285	0	11

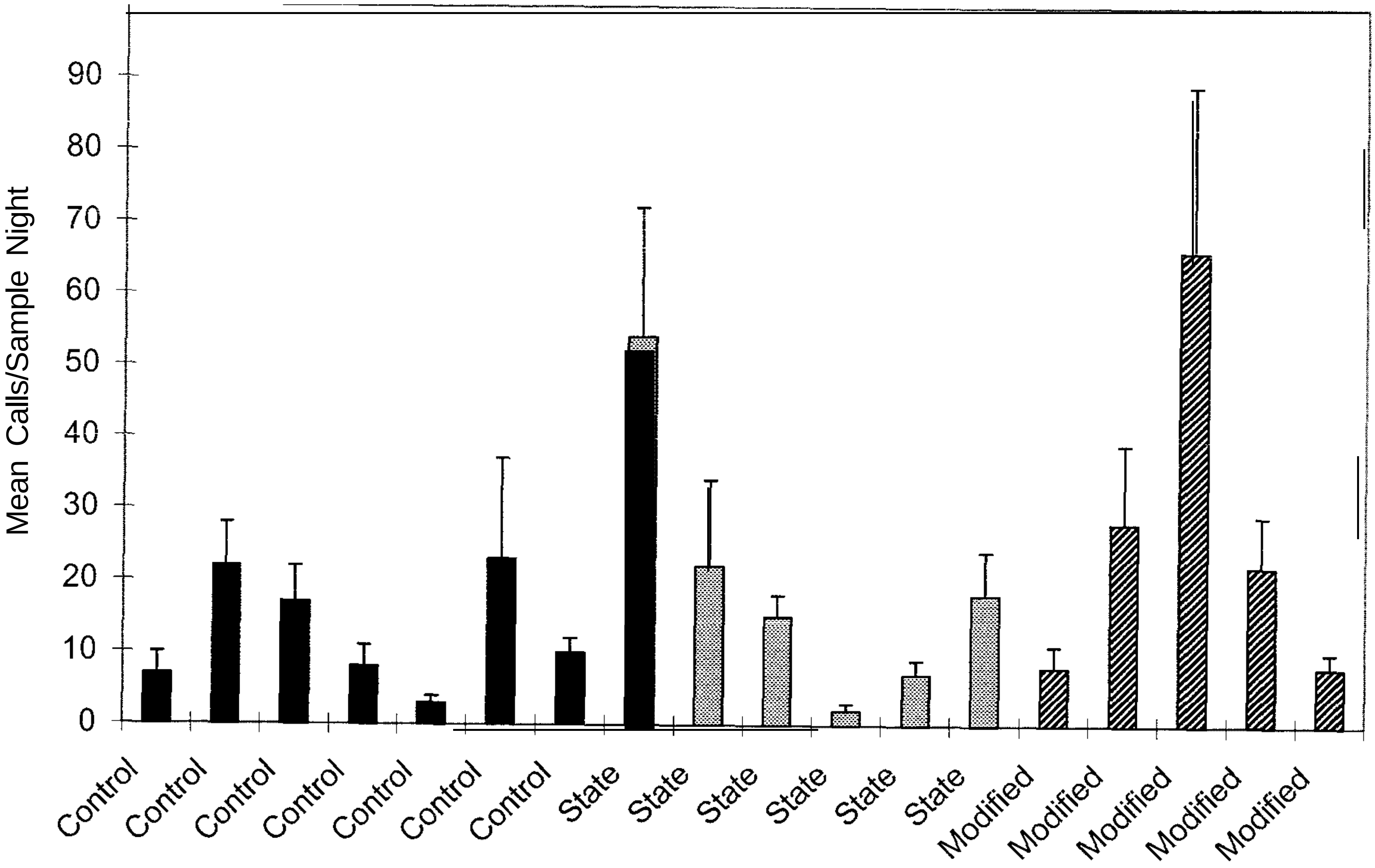

Figure 1. Mean number of bat calls (± SE) per sample night at the 18 RMZ study sites in northeastern Washington.

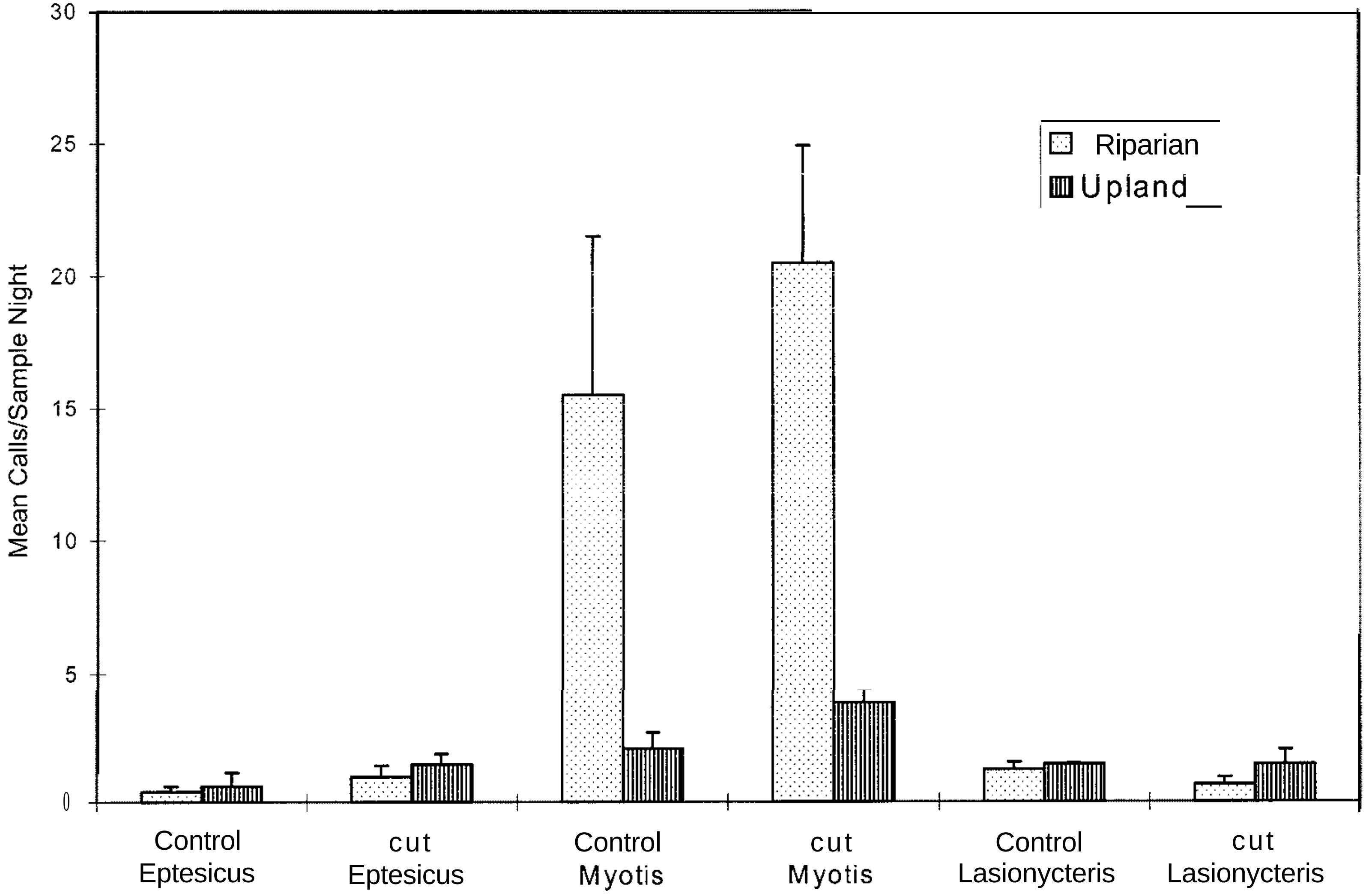

Figure 2. Mean number of calls (± SE) by species per sample night in riparian and upland habitats of the Control and cut (State and Modified) sites.

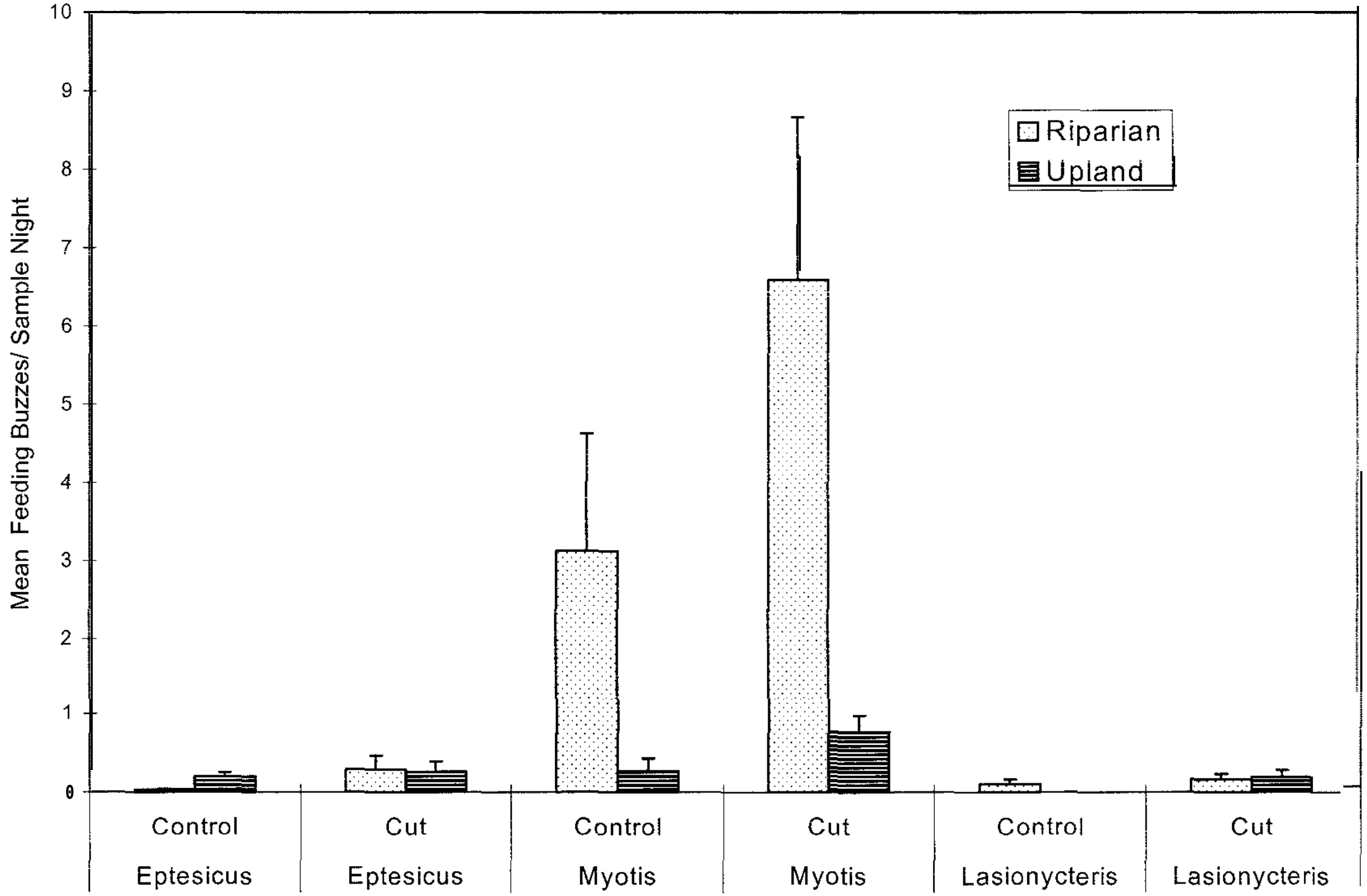

Figure 3. Mean number of feeding buzzes (± SE) per sample night in riparian and upland habitats of Control and cut (Modified and State) sites.

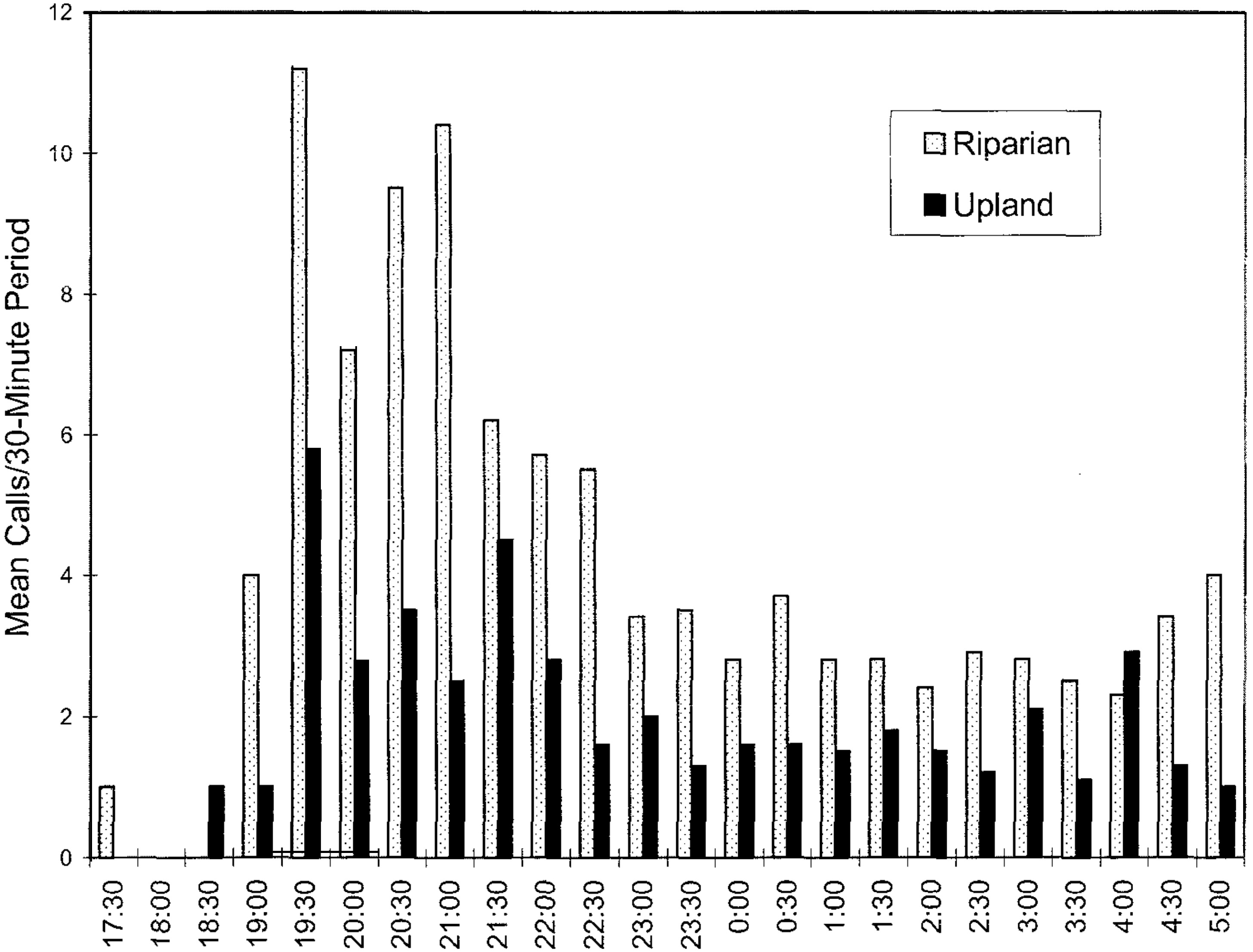

Figure 4. Mean number of calls (± SE) per 30-min sampling period across all 18 sites and 4 yr.

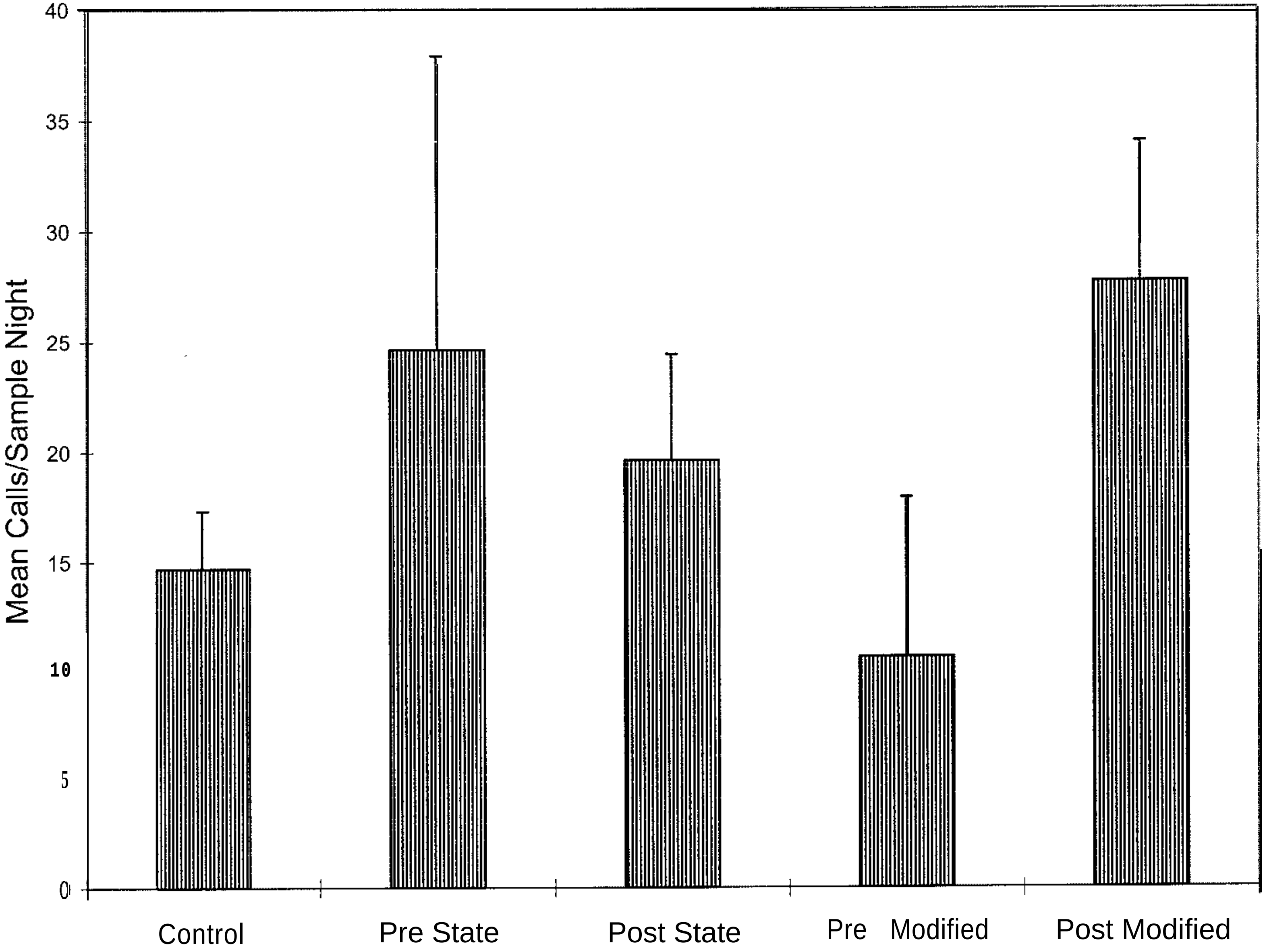

Figure 5. Mean (± SE) number of total calls per sample night on the Control, State and Modified sites,

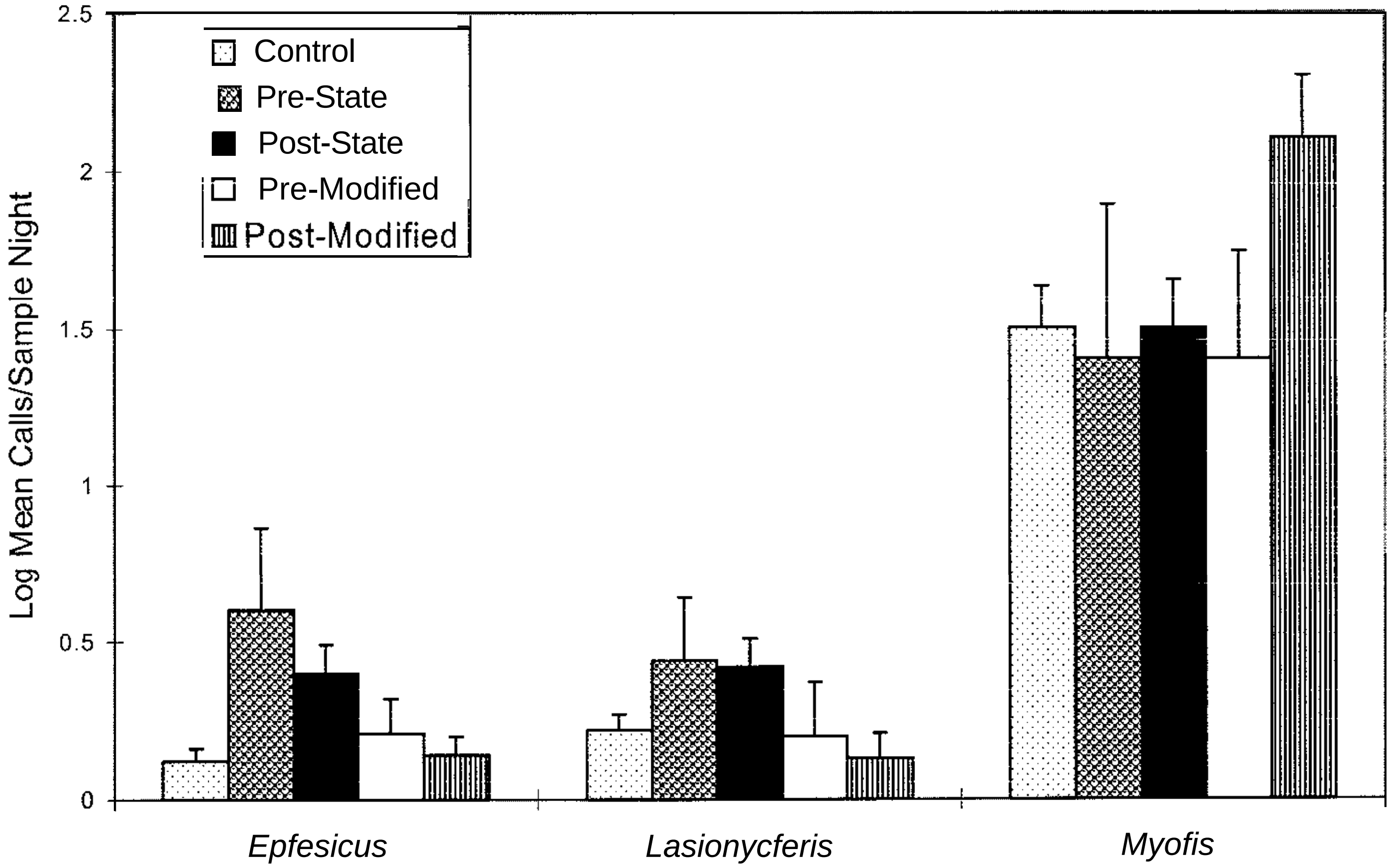

Figure 6. Log mean (± SE) calls per sample night of three bat species in riparian habitats of Control, State, and Modified sites.

Chapter 14

SUMMARY AND MANAGEMENT RECOMMENDATIONS

INTRODUCTION

In the preceding chapters, we compared habitat conditions in riparian and adjacent uplands of Pacific Northwest coniferous forests managed for timber harvest, and vertebrate abundance and diversity in these two habitats. Further, we reported the response of the vertebrate species to different riparian harvest treatments. By conducting parallel studies in the distinct forest regions of western and northeastern Washington State, we addressed these topics for forests that support different wildlife assemblages and that are managed under different harvest guidelines. In this chapter we 1) summarize our comparisons of riparian and upland habitats and wildlife associations, 2) review the results of our large-scale experiment to evaluate the efficacy of riparian buffers, and 3) provide management recommendations, In our initial review of wildlife use of riparian habitats (O'Connell et al. 1993) we presented an assessment of the sensitivity of terrestrial wildlife species inhabiting riparian habitats in Washington State. The Appendix to this chapter provides a revised assessment based on our research tindings.

RIPARIAN AND UPLAND HABITATS

Unlike studies conducted in arid regions, the overall contrast between riparian and upland habitats in this region, was not great. The density of drainage systems and the proximity to water even in upland forests of the Pacific Northwest might explain the overall similarity between riparian and upland habitats observed in this region. From the perspective of wildlife habitat, most vegetation characteristics were shared between habitats and habitat elements typically differed in degree rather than kind.

In western Washington forests, the riparian zones had more red alder trees, berry-producing and other deciduous shrubs, herbs, ferns, bare soil, and rock than upland habitats. Upland habitats had significantly greater numbers of western hemlock trees, snags, litter cover and depth, and higher canopy cover.

The vegetation of the riparian and upland forests of eastern Washington differed in few, but significant respects. The riparian zone had greater dispersion of shrubs, more deciduous vegetation, and more trees and snags in the largest size classes. Although canopy cover was more closed in the: riparian, a greater diversity of herbaceous plants was present. The riparian zone also had down wood of greater diameter and greater decay.

Wildlife abundance and richness in riparian and upland habitats

For most groups of wildlife we studied, there were few overall differences in abundance and richness between riparian and upland habitats before timber harvest (Table 1). Most differences were observed at the species level.

Birds

In the western Washington forests, avian species richness and diversity were not significantly different between riparian zones and uplands. The American Robin, Black-throated Gray Warbler, Pacific-slope Flycatcher, and Winter Wren favored the riparian zone. No species was significantly associated with the uplands, although there was a positive trend in abundance for the Brown Creeper and Golden-crowned Kinglet. Deciduous trees were an important habitat component for birds, In contrast, in eastern Washington forests, diversity was greater in the uplands and more individual species were associated with the upland than the riparian habitat. However, species richness, turnover rates, and rates of nest predation were equal between the two habitats. Although overall abundance was comparable between habitats, individual species exhibited differences. Four species were more abundant in the riparian habitat, responding to either the deciduous component or the larger trees that were present in these habitats. Nine species were more abundant in the upland habitats. Most of these species were associated with more open overstory and shrubs.

Based on our observations of bird associations in riparian and habitat associations, we added two species to our assessment of riparian-associated species: Winter Wren, Hammond’s Flycatcher (Appendix). In addition, we increased values for habitat specificity of the Black-throated Gray Warbler, Northern Waterthrush, Pacific-slope Flycatcher (formerly Western Flycatcher) to reflect our observations of their association with riparian habitat (Appendix)

Terrestrial amphibians and reptiles

There were only slight differences in richness and abundance of terrestrial amphibians between riparian and upland transects in the west side forests. About one-third of all captures were in the riparian zone with most captures of *Ensatina* in the uplands. Adult tailed frogs used the uplands extensively. Sampling during autumn rains may have lessened restriction to the riparian zone.

In contrast, although amphibian abundance was very low in east side forests, abundance and richness were greater in riparian habitats. These species require slower moving water for breeding than was common on these sites. Reptile abundance (albeit very low) was either equal to or greater in upland as compared to riparian habitats.

Terrestrial small mammals

As a group, small mammals were more consistently associated with riparian habitats of the forests of both western and eastern Washington, however associations of individual species differed. In western Washington forests, species richness before harvest was higher within the riparian zones than in the adjacent uplands. Species evenness and overall abundance were not different:. Species composition was similar between riparian zones and uplands. Four species, the montane shrew, the marsh shrew, the Pacific jumping mouse, and the long-tailed vole, were caught at greater rates on riparian transects, whereas only the southern red-backed vole was caught more often on the upland transects.

Small-mammal species diversity, evenness, and species richness were similar in riparian habitats and the adjacent uplands of eastern Washington forests prior to harvest. Overall abundance of small mammals was consistently greater in riparian than in upland habitats. During a population peak in 1994, species richness and diversity increased dramatically as infrequently encountered species moved onto the stream sites. This effect was greater in the riparian zone suggesting that some species may use these habitats as travel corridors at some times.

Based on our surveys of small-mammal species, we modified the habitat specificity values for several species in our assessment of riparian-associated species (Appendix). In west-side forests, marsh shrews, montane shrews, and Pacific jumping mice were more associated with riparian habitats than we had previously indicated (O’Connell et al. 1993).

On the east side, additional distribution data for northern bog lemmings, western jumping mice, and water voles suggested greater association of these species with riparian habitat than we had previously indicated (O'Connell et al. 1993).

Bats

Bat response to riparian and upland habitats was similar between western and eastern Washington forests. Detection rates of *Myotis* bats were greater in riparian habitat, There were no habitat differences observed for detection rates ofbig brown or silver-haired bats, Bats typically travel between. roosting and foraging sites thus linking riparian and upland habitats.

Management implications of riparisn and upland habitat associations

The vertebrate communities of these small riparian zones and adjacent uplands are largely a shared fauna. There are differences in the relative abundance of some species, with about equal numbers favoring either riparian or upland habitat. Other species use both habitats to fulfill different and critical life functions, such as some stream-breeding amphibians and several bats. Given such a close connection between these habitats, management consideration of both habitats should be a goal. A greatly simplified upland habitat, for example, would no doubt seriously degrade the habitat value of a riparian buffer. Conversely, if uplands are managed with structural diversity and attention to habitat features of importance to wildlife in mind, riparian buffers might not require extensive area.

TREATMENT EFFECTS

Habitat changes following timber harvest were largely predictable. Given the clearcut harvest prescription for the uplands of the western Washington forests and the partial harvest prescription for the uplands of the eastern Washington forests, differences were more pronounced on the west side. Changes in post harvest habitat condition between the two sides of the state also reflect differences in the state regulations for Riparian Management Zones (RMZ). West-side RMZ's were narrower that those on the east side.

Following timber harvest, riparian areas in the western Washington forests remained dominated by red alder. The width of the buffer strip was about twice as large on Modified sites than on State sites. Riparian canopy cover differed significantly among treatment types Control sites provided 90-100% canopy cover within riparian areas while state buffer sites provided <50% cover. Modified buffers ranged from 40-90% cover. Percentage cover of ferns, moss, and bare soil decreased significantly whereas litter cover and berry-producing shrubs increased within riparian areas at treatment sites.

Following timber harvest on the east side, Modified sites had wider, but considerably more variable buffers than did State sites. Changes after harvest accentuated differences between riparian and upland habitats with predictable reductions in canopy cover, shrub layer, regenerating stems, deciduous trees, and decayed down wood. State sites had greater floristic changes than Modified or Control sites including reductions in the abundance of shrub species in both upland and riparian and herbaceous species in the riparian. Several weedy species increased in abundance or appeared for the first time after harvest,

Wildlife response to treatments

For some groups of wildlife we studied, we were able to attribute changes in overall abundance and diversity to treatment effects within 2 yr of timber harvest (Table 1). For other groups, broad changes in abundance and diversity were not discemable within the 2 yr timeframe (Table 1). Given that the response of wildlife species to timber harvest will vary with each species' habitat requirements, an understanding of wildlife response to these different harvest treatments also requires evaluation at the species level.

Birds

In general avian response to timber harvest was more pronounced in western than in eastern Washington forests, which is not surprising given the differences between clearcuts and partial cuts, Overall riparian species richness and diversity were significantly greater on the Modified, compared to Control sites. State sites were intermediate, but not significantly different from either. In western Washington, upland habitats on harvested sites showed significantly lower species richness and diversity, due to the loss of closed-canopy forest species. About 50% of the common species showed significant treatment effects.

In eastern Washington forests, the general patterns of riparian and upland associations remained the same after harvest, but there were differences in the associations of individual species. The pre- vs. post harvest associations of the riparian species were more consistent than those of the upland species. Within the upland habitats, the changes in species richness and diversity were most pronounced on the State sites. Within the riparian habitats, there were no differences between pre- and post harvest with respect to species richness, turnover rates, diversity, or overall abundance. At the species level, there were declines in several riparian-associated species and increases in the upland-associated species on the State sites. Rates of predation on artificial nests were greater on the State sites. State sites maintained similar species composition as Control and Modified sites.

Although survey methods were different between the east- and west-side study regions, we can compare indices of bird species richness (BSR), bird species diversity (BSD), relative abundance, and responses of some individual bird species. There were no differences in BSR between upland and riparian habitats on either side of the Cascades before harvest (Table 1). After harvest were applied, BSR on the East side was unaffected by the treatments, however, on the West side, BSR varied significantly in the riparian areas. BSR was highest in Modified sites compared to Control or State sites. In upland habitat, BSR was higher at all sites after treatments on the East side. On the West side, highest BSR was on the Control sites, indicating that the treatments had a detrimental impact on BSR on the State and Modified sites.

BSD generally followed a pattern similar to BSR. There were no differences in BSD between uplands and riparian areas before harvest in either study region. Once the treatments were applied, BSD increased on the upland portion of State sites compared to other sites on the East side. On the West side, the opposite pattern emerged: BSD was highest in control sites and lowest on State sites. In riparian areas, BSD was not affected by the treatments. On the West side, however, BSD increased significantly on Modified sites compared to either Controls or State sites.

Differences in abundance patterns reflected feeding and nesting strategies with birds needing a well-developed canopy for nesting or feeding generally decreased with the severity of the treatment. Birds such as kinglets, chickadees and warblers, and Hammond's and Pacific-slope Flycatchers were detrimentally impacted by canopy removal. Edge species such as the Song Sparrow, Spotted Towhee, Dark-eyed Junco and American Robin increased after harvest in most areas. The differences between the East side and West side can mostly be attributed to the fact that canopy removal was less on the East-side, thus providing some additional habitat features that would attract canopy-dwelling species and those using some shrub understory.

Stream amphibians

No differences in the abundance of stream amphibians were found in stream habitat pre- and post-harvest on western Washington sites. Although not significant, larvae of Pacific giant salamanders tended to increase after harvest. Tailed frog tadpoles showed no change. Irregular distributions of the different species among the 18 study sites made statistical tests weak.

Terrestrial amphibians and reptiles

In western Washington, more species of terrestrial amphibians were encountered in the riparian habitat of the Modified sites following timber harvest, but differences were not significant. Upland abundance of these amphibians was lower on the two harvested sites compared to the Control sites. Captures of ensatina were lower in both riparian and upland habitats of Modified sites as compared to State and Control sites. Upland captures of tailed frogs, red-legged frogs, and northwestern salamanders showed a decreasing trend following timber harvest on both State and Modified sites. Low captures of all species except for Ensatina and the western red-backed salamander resulted in low statistical power.

The diversity and abundance of amphibians on the east side forest sites were markedly lower than on the west side. Nonetheless, almost all species known from the region were observed on the east side sites and treatment effects were discerned. Amphibian species richness and abundance declined in the riparian habitat on the State sites as compared to the Modified and Control sites. Based on our observations of the habitat associations and declines in response to logging, we modified the habitat specificity values for the western toad and Columbia spotted frog in our assessment of riparian-associated species (Appendix).

Reptile abundance declined in the uplands of the State sites as compared to the Modified and Control sites.

Small mammals

In general, small-mammal response to the different treatments was more pronounced on the eastern Washington sites as compared to the west side sites. This was rather surprising given that the upland harvest was more intense and the riparian buffers narrower on the western Washington sites. However, on both the western and eastern Washington sites, the Modified buffer design appeared to provide the better chance for persistence for small mammal populations. Abundance is the key to the long-term persistence of these mammals, and the Modified buffers supported a greater relative abundance of more species.

In western Washington forests, species richness and evenness in riparian habitat did not differ significantly among treatments. Species composition of the riparian transects between harvest treatments was very similar. No species showed a statistically significant change in capture rate with respect to treatment on the riparian transects. On upland transects species richness and evenness did not differ significantly among treatments. A change in species composition reflected losses of Insectivores and gains by the deer mouse and the creeping vole. Capture rates on the uplands declined significantly for the marsh shrew, the shrew-mole, and the forest deer mouse. Capture rates increased for the creeping vole.

Small-mammal habitat associations remained generally constant before and after treatments on the eastern Washington sites. After harvest, overall abundance and species richness of small mammals was consistently greater in riparian than in upland habitats. Species that had their highest abundances in the riparian continued to do so following harvest. Rare species were more likely to be found in the riparian zone and may use these areas as corridors. Modified sites had greater abundances of small mammals in both the riparian and upland habitats, suggesting that the buffers at these sites may increase population persistence. Reflecting the increase in some colonizer species, species richness on State sites increased temporarily in the upland after harvest and was greater than on Modified or Control sites, but then declined.

Bats

After harvest on the western Washington sites, detection rates of the non-*Myotis* species increased on the State sites compared to the Control and Modified sites. There were no differences between treatments for the detection rates of the *Myotis* bats.

On the east-side sites, the mean detection rates of all bats did not differ between Control, State, or Modified sites. Detection rates of big brown bats and silver-haired bats were greater on the State and Modified sites than Control sites, Detection rates of the big brown bat decreased between pre- and post-harvest on the State sites. In contrast, detection rates for *Myotis* was greater after harvest on Modified sites as compared to the Control or State sites.

Effectiveness of State and Modified Buffers

The primary management objective of riparian buffers has been protection of the integrity of the aquatic environment. Additionally, riparian buffers can provide habitat for terrestrial wildlife. For the terrestrial wildlife, buffers are initially stopgap devices to provide habitat during the post-harvest years before the young forest on the adjacent uplands develops a closed canopy. At canopy closure, the riparian zone is once more buffered by the surrounding forest and at lower risk from weather extremes and the negative biotic effects associated with high-contrast edges. The basic strategy is to design a riparian buffer that will maintain the biota of the riparian zone through these early post-harvest years. An added benefit, if the buffer is to remain largely unharvested through forest rotations, is the addition of much needed structure to the riparian zone and to managed forests generally in the form of large trees, snags, and down wood.

One measure of success for a particular buffer design is whether riparian obligate species and forest-associated fauna will persist within the buffer between the time of harvest and canopy closure. It may not be necessary for population abundance to remain at pre-harvest levels, but at least a consistent presence by these species within the buffer would allow populations to recover quickly once the canopy of the surrounding forest closed. Species of generalized habitat requirements or those associated with early seral conditions should find ample habitat in the lands adjacent to the buffer. Judged from this perspective, a buffer for a riparian obligate or a species associated with closed-canopy forest represents a habitat of intermediate quality-somewhere between well-developed and recently logged forest. The design challenge is to provide sufficient structure to allow persistence and yet provide economic return from the harvest. In the following, we summarize our management recommendations about the importance if riparian buffers, differences in effectiveness of the State RMZ buffers and the Modified buffers, and the importance of site-specific management approaches.

First, riparian buffers are important because riparian habitat is the foundation for much of the region’s biodiversity. Although we did not observe the pronounced differences in riparian versus upland faunas that have been documented in other regions, many species were clearly associated with the riparian habitat. The main factor contributing to the shared riparian and upland faunas is most likely the spatial proximity of the drainages in this region.

Although the maritime climate of west-side forests undoubtedly contributes to the riparian and upland fauna1 similarities, this does not hold for east-side forests. Despite their spatial proximity, riparian habitats represent a relatively restricted area as compared to upland habitats, Protecting the integrity of these riparian habitats is therefore important. Two years after harvest both the State RMZ buffers and the Modified buffers retained a large proportion of species associated with riparian zones and closed-canopy forest.

However, the buffer on the Modified sites holds greater promise of species persistence during the early post-harvest years than the State RMZ buffer. First, considering habitat condition, the riparian habitat on the Modified sites was more similar to that of the Control sites for west-side forests and the number of post-harvest structural and floristic changes in the riparian zone were less on the Modified sites for east-side forests. Second, trends in abundance and habitat associations of individual vertebrate taxa indicate the potential for greater persistence on the Modified sites. Riparian and closed-canopy species tended to be more abundant and exhibit positive associations with the Modified buffer. Conversely, open habitat species tended to be more abundant and exhibit positive associations with the State RMZ buffer. Most likely this was due to both the greater area and structural diversity of the Modified buffer.

The intent of our Modified buffer was to incorporate a more site-specific approach to riparian management by identifying and protecting habitat features of importance to wildlife. The largely shared vertebrate fauna between riparian and upland habitats in these Pacific Northwest forests argues for incorporation of such a site-specific approach to both riparian and upland habitats. Identification and protection of upland habitat features of importance to wildlife would reduce the threat of a greatly simplified upland habitat degrading the habitat value of a riparian buffer. If both uplands and riparian habitats are managed with structural diversity and attention to habitat features of importance to wildlife in mind, protection of the region's terrestrial vertebrate diversity can be enhanced.

By design, our results focus on the years immediately following harvest. We have provided a baseline from which future changes within the buffers and adjacent uplands can be compared. Studies of wildlife response to different buffer harvests in other regions have indicated changes in composition and abundance between the immediate post-harvest years and later years. From some trends in this study and our experience with the habitat patterns shown by vertebrates in the TFW Landscape Study, we expect several such changes in the next few years.

To document these changes these sites must be resurveyed at regular intervals. We suggest returning about five years post-harvest and again at about 10 yr post-harvest. The first decade should encompass the most active period for decline in species associated with riparian and closed canopy forest. Without additional sampling the effectiveness of these RMZ designs cannot be assessed.

LITERATURE CITED

O'Connell, M. A., J. G. Hallett, and S. D. West. 1993. Wildlife use of riparian habitats: a literature review. Washington Department of Natural Resources **TFW-WLl-93-001.**

Table 1. Overall comparisons of vertebrate response to riparian versus upland habitats before timber harvest, to different riparian buffer treatments, and to timber harvest in uplands in northeastern and western Washington.

Taxon/ Attribute	East side	West side
Stream amphibians		
Treatment effects: Riparian		
Abundance	NA	No differences
Terrestrial amphibians		
Pre-treatment: Upland vs Riparian		
Species richness	Higher in riparian	No difference
Abundance	Higher in riparian	Higher in upland
Treatment effects: Riparian		
Species richness	Declined on State	Higher on Modified sites
Abundance	Declined on State	No overall differences, variable response by species
Treatment effects: Upland*		
Species richness	No differences	Higher on Control
Abundance	No differences	No overall differences; variable response by species
Reptiles		
Pre-treatment: Upland vs Riparian		
Species richness	Higher in upland	NA
Abundance	Higher in upland	NA

Table 1. Continued

Taxon/ Attribute	East side	West side
Reptiles		
Treatment effects: Riparian		
Species richness	No differences	NA
Abundance	No differences	NA
Treatment effects: Upland*		
Species richness	No differences	NA
Abundance	Decreased on State	NA
Birds		
Pre-treatment: Upland vs Riparian		
Species richness	No differences	No differences
Species diversity	No differences	No differences
Abundance	No overall differences; Riparian dependent species decreased on State	No differences, variable response by species
Treatment effects: Riparian		
Species richness	No differences	Highest in Modified sites
Species diversity	No differences	Highest in Modified sites
Abundance	No overall differences; variable response by species	No overall differences, variable response by species
Treatment effects: Upland*		
Species richness	Higher post-treatment	Highest on Control; lowest on State
Species diversity	Increased on State	Highest in Control; lowest in State
Abundance	No overall differences; variable response by species	No overall differences; variable response by species

Table 1. Continued

Taxon/ Attribute	East side	West side
Terrestrial small mammals		
Pre-treatment: Upland vs Riparian		
Species richness	No differences	Higher in riparian
Species diversity	No differences	No differences in species evenness
Abundance	Higher in riparian	No overall differences, variable response by species
Treatment effects: Riparian		
Species richness	Lowest on Control	No differences
Species diversity	No differences	No differences in species evenness
Abundance	Highest on Modified	No overall differences, variable response by species
Treatment effects: Upland*		
Species richness	Initial increase on State then decline	No differences
Species diversity	Initial increase on State then decline	No differences in species evenness
Abundance	Higher on Modified	No overall differences; variable response by species
Bats		
Pre-treatment: Upland vs Riparian		
Detection rate	No difference for non-*Myotis* bats; *Myotis* higher in riparian	No difference for non-*Myotis* bats; *Myotis* higher in riparian
Treatment Effects: Riparian		
Detection rate	*Myotis* increased on Modified; Big brown bats decreased on State	No difference for *Myotis* Non-*Myotis* bats increased on State

* Harvest units on the West side were clearcuts; East side partial cuts,

APPENDIX

Taxa

The taxa included in this assessment 1) were designated as inhabitants ofriparian ecosystems in O'Connell et al. (1993) and for which we have data from the present study or 2) were not included in O'Connell et al. (1993) but were identified as riparian-associated species in the present study.

Descriptive variables

Life form descriptions for western Washington are from Brown (1985) and those for eastern Washington are from Thomas (1979).

Life form	Reproduces	Feeds
1	in water	in water
2	in water or in trees	on the ground, in bushes
3	on the ground around water	on the ground and in bushes, trees, and water
4	in cliffs, caves, rimrock, or talus	on the ground or in air
5	on the ground without specific water, cliff, rimrock, or talus	on the ground
7	in bushes	on the ground, in water, or in air
8	in bushes	in trees, bushes, or air
9	primarily in deciduous trees	in trees, bushes, or air
10	primarily in conifers	in trees, bushes, or air
11	in conifers or deciduous trees	in trees, in bushes, on the ground, or in air
12	on very thick branches	on the ground or in water
13	in own hole excavated in tree	in trees, in bushes, on the ground, or in air
14	in a hole made by another species or in a natural hole	on the ground, in water, or in air
15	in a burrow underground	on the ground or under it
16	in a burrow underground	in the air or in the water

Primary habitat is designated by a "1"; secondary, or marginal, habitat is designated by a "2". Information on use of habitats is from Brown (1985) and Thomas (1979).

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Variables used in assessing sensitivity (W & E)

Habitat specificity

Habitat specificity scores are derived from versatility scores in Brown (1985) for forests of western Washington (W), from Thomas (1979) for forests of the Blue Mountains (E), and our research results.

Versatility = Number of plant communities used for reproduction + Number of successional stages used for reproduction + Number of plant communities used for feeding + Number of successional stages used for feeding. Numbers in left column represent scores.

3 High habitat specificity (versatility score 1-16).

2 Medium habitat specificity (versatility score 17-29).

0 Low habitat specificity (versatility score > 30).

If information was available for a taxon on only one side of the state, the versatility score for that taxon on the side of the state where it occurred was doubled, to give a maximum of 6 possible points.

Population trend throughout range of taxon (PT)

6 Populations known to be or suspected of decreasing throughout all or most of range of taxon.

3 Populations formerly experienced serious declines throughout range of taxon but presently thought to be stable or increasing, or population decreasing in part of its range.

0 Populations are stable or population trends are unknown.

Geographic range (GR)

Geographic range refers to the area in the U.S. and Canada over which the taxon is distributed during the season when distribution is most restricted.

6 (< 130,000 km^2, i.e., < approximately 1/3 the area of California)

3 (130,000 1/3 km^2 - 400,000 km^2, i.e., > 1/3 the area of California-the area of California).

0 (> 400,000 km^2)

If a taxon has no wintering range in North America (e.g., black tern, solitary sandpiper), it scored 6, even if it has a large breeding range in Central or South America. The justification for this is that species wintering in Central or South America are likely vulnerable on their wintering grounds because of habitat loss or exposure to pesticides.

Reproductive potential for recovery - clutch size (CS)

The only risk factor affecting reproductive potential for recovery included in this analysis was the number of young produced per year, computed as litter size × number of clutches (litters) produced per year. As more information is obtained, information on survival and age at sexual maturity will be added to the matrix, so that reproductive potential will reflect these additional components as well as clutch size. A high risk factor for each component is worth 2 points; moderate risk factors score one point. In computing sensitivity scores using only clutch (litter) size, we multiplied scores by 3, so that reproductive potential had the same weight (6 possible points) as each of the other risk factors.

2 For amphibians: l-25 eggs/clutch.

2 For reptiles, birds, and mammals: clutch or litter size × number of clutches (litters) produced per year <3.

1 For amphibians: clutch size >25 and <76.

1 For reptiles, birds, and mammals: clutch or litter size × number of clutches (litters) produced per year >2 and <6.

0 For amphibians: >76 or clutch size unknown.

0 For reptiles, birds, and mammals: clutch or litter size × number of clutches (litters) produced per year >6 or unknown.

Population concentration (CO)

Population concentration reflects the degree to which individuals congregate or aggregate seasonally at specific locations (e.g., hibemacula, breeding sites, migration focal points) or daily at specific locations (e.g., communal roosts).

6 Majority of the Washington population concentrates at 1-5 locations within the state.
3 Individuals sometimes concentrate in colonies, communal roosts, or large flocks.
0 Individuals rarely congregate or aggregation behavior unknown.

Variables used in assessing significance

Systematic significance (SS)

This score includes total of all categories that apply:

3 Monotypic family.

2 Monotypic genus.

1 Monotypic species (i.e., no subspecies).

0 Species includes >1 subspecies.

Biogeographic significance (BS)

6	75-100% of total range occurs in Washington.
4	50-74% of total range occurs in Washington.
2	**2549%** of total range occurs in Washington.
0	<25% of total range occurs in Washington.

Scores

Sensitivity score (SEN)

The sensitivity score assesses the sensitivity to disturbance, especially loss of riparian habitat, This score is calculated as:

Sensitivity score = habitat specificity score for western Washington + habitat specificity score for eastern Washington [or 2(habitat specificity score for western **or** eastern Washington)] + population trend score + geographic range score + population concentration score + (clutch size) 3.

Significance score (SIG)

The significance score assesses contribution of taxon in Washington to biological diversity. This score is calculated as:

Significance score = systematic significance score + biogeographic significance score.

Table 2. A ranking system to asses the relative vulnerability of Washington's terrestrial riparian vertebrates to disturbance. Variables and calculation of scores are described in the text.

		Life forms		Habitat specificity		Sensitivity variables				Significance variables		Scores	
Taxa		West	East	West	East	PT	GR	CS	CO	SS	BS	SEN	SIG
AMPHIBIANS													
Caudata													
Ambystomatidae													
North-western salamander	*Ambystoma gracile*	2		2		0	3	0	3	i	0	10	i
Long-toed salamander	*Ambystoma macrodactylum*	2	2	2	2	0	0	0	3	0	0	7	0
Dicamptodontidae													
Cope's giant salamander	*Dicamptodon copei*	1		3		0	6	1	0	1	6	15	7
Pacific giant salamander	*Dicamptodon tenebrus*	2		2		0	6	0	0	1	2	10	3
Rhyacotritonidae													
Olympic salamander	*Rhyacotriton olympicus*	2		3		0	6	2	0	1	6	18	7
Salamandridae													
Roughskin newt	*Taricha granulosa*	2		2		0	3	1	3	0	0	13	0
Plethodontidae													
Ensatina	*Ensatina eschscholtzii*	5		0		0	3	2	0	2	0	9	2
Dunn's salamander	*Plethodon dunni*	5		3		0	6	2	0	1	0	18	1
West. redback salamander	*Plethodon vehiculum*	5		2		0	6	2	0	1	2	16	3
Anura													
Leiopelmatidae													
Tailed frog	*Ascaphus truei*	2	2	3	2	0	3	1	0	3	2	11	5
Bufonidae													
Western toad	*Bufo boreas*	2	2	2	2	2	0	0	3	0	0	9	0

Table 2. Continued.

Taxa		Life forms		Habitat specificity		Sensitivity variables				Significance variables		Scores	
		West	East	West	East	PT	GR	CS	CO	SS	BS	SEN	SIG
Hylidae													
Pacific tree frog	*Hyla regilla*	2	2	2	0	0	0	0	0	0	0	2	0
Ranidae													
Northern red-legged frog	*Rana aurora*	2		2		3	3	0	0	0	2	10	2
Columbia spotted frog	*Rana luteiventris*		2		3	2	0	0	0	1	0	8	1
BIRDS													
Ciconiiformes													
Ardeidae													
Great blue heron	*Ardea herodias*	12	12	3	3	0	0	1	3	0	0	12	0
Galliformes													
Phasianidae													
Ruffed grouse	*Bonasa umbellus*	5	5	2	2	0	0	0	0	0	0	4	0
Piciformes													
Picidae													
Downy woodpecker	*Picoides pubescens*	13	13	2	3	0	0	1	0	0	0	8	0
Passeriformes													
Tyrannidae													
Dusky flycatcher	*Empidonax oberholseri*	8	8	3	2	0	6	1	0	1	0	14	1
Pacific-slope flycatcher	*Empidonax difficilis*	11		3		0	6	1	0	0	0	13	0
Hammond's flycatcher	*Empidonax hammondii*	11	11	2	3	0	6	1	0	0	0	14	0
Paridae													
Black-capped chickadee	*Parus atricapillus*	14	14	2	3	0	0	0	3	0	0	8	0

Table 2. Continued

		Life forms		Habitat specificity		Sensitivity variables				Significance variables		Scores	
Taxa		West	East	West	East	PT	GR	CS	CC	SS	BS	SEN	SIG
Troglodytidae													
Winter wren	*Troglodytes troglodytes*	14	3	2	3	0	0	0	0	0	0	5	0
Cinclidae													
American dipper	*Cinclus mexicanus*	3	3	2	3	0	0	1	0	3	0	8	3
Muscicapidae													
Golden-crowned kinglet	*Regulus satrapa*	10	10	3	2	0	0	1	0	0	0	8	0
Swainson's thrush	*Catharus ustulatus*	8	8	0	1	0	6	0	0	0	0	7	0
Vireonidae													
Warbling vireo	*Vireo gilvus*	11	11	2	3	3	6	1	0	0	0	17	0
Emberizidae													
Nashville warbler	*Vermivora ruficapilla*		9		2	0	0	1	0	0	0	7	0
Yellow-rumped warbler	*Dendroica coronata*	10	10	0	2	0	0	0	3	0	0	5	0
Black-throated *gray* warbler	*Dendroica nigrescens*	10		3		0	6	1	0	0	2	13	2
Townsend's warbler	*Dendroica townsendii*	10	10	2	2	0	0	1	0	0	0	7	0
Northern waterthrush	*Seiurus novaboracensis*		3		3	0	6	1	0	1	0	15	1
MacGillivray's warbler	*Oporornis tolmiei*	8	7	2	2	0	6	1	0	0	0	13	0
Wilson's warbler	*Wilsonia pusilla*	6	5	3	0	3	6	0		0	0		0
Western *tanager*	*Piranga ludoviciana*	10	10	0	2	0	6	1	0	1	0	11	1
Black-headed grosbeak	*Pheuticus melanocephalus*	9	11	2	2	0	6	1	0	0	0	13	0
Spotted towhee	*Pipilo maculatus*	7	7	0	3	0	0	0	0	0	0	3	0
Chipping sparrow	*Spizella passerina*	11	7	0	1	2	0	1	0	0	0	7	0
Song sparrow	*Melospiza melodia*	7	7	2	3	0	0	0		0	0	5	0
White-crowned sparrow	*Zonatrichia lecophrys*	7	7	0	1	0	1	1	0	0	0	5	0
Dark-eyed junco	*Junco hymemalis*	5	5	0	2	0	0	1	0	0	0	5	0

Table 2. Continued.

		Life forms		Habitat specificity		Sensitivity variables				Significance variables		Scores	
Taxa		West	East	West	East	PT	GR	CS	CC	SS	BS	SEN	SIG
MAMMALS													
Insectivora													
Soricidae													
Marsh shrew	*Sorex bendirii*	16		3		0	3	0	0	0	2	9	2
Masked shrew	*Sorex cinereus*	15	15	2	2	0	0	0	0	0	0	4	0
Montane shrew	*Sorex monticolus*	15	15	3	2	0	0	0	0	0	0	5	0
Water shrew	*Sorex palustris*	16	16	3	3	0	0	0	0	0	0	6	0
Trowbridge's shrew	*Sorex trowbridgii*	15		0		0	3	0	0	0	2	3	2
Vagrant shrew	*Sorex vagrans*	15	15	2	2	0	0	0	0	0	0	4	0
Talpidae													
Shrew-mole	*Neurotrichus gibbsii*	15		0		0	3	0	0	2	2	3	4
Coast mole	*Scapanus orarius*	15	15	0	0	0	3	1	0	0	2	6	2
Townsend's mole	*Scapanus townsendii*	15		2		0	3	1	0	1	4	10	5
Chiroptera													
Vespertilionidae													
Big brown bat	*Eptesicus fuscus*	14	14	0	2	0	0	2	0	0	0	8	0
Silver-haired bat	*Lasionycteris noctivagans*	14	14	2	2	0	0	2	0	3	0	10	3
Hoary bat	*Lasiurus cinerea*	11	11	2	3	0	0	2	0	1	0	11	1
California myotis	*Myotis californicus*	14	14	2	2	0	0	2	3	0	0	13	0
West. small-footed myotis	*Myotis ciliolabrum*		4		3	0	0	2	3	0	0	15	0
Long-eared myotis	*Myotis evotis*	14	14	0	2	0	0	2	0	0	0	8	0
Keen's myotis	*Myotis keenii*	14	14	2		0	0	2	3	0	0	13	0
Little brown myotis	*Myotis lucifugus*	14	14	0	2	0	0	2	3	0	0	11	0
Long-legged myotis	*Myotis volans*	14	14	0	2	3	0	2	3	0	0	14	0
Yuma myotis	*Myotis yumanensis*	14	14	0	3	0	0	2	3	0	0	12	0
Townsend's big-eared bat	*Corynorhinus townsendii*	4	4	3	2	3	0	2	3	0	0	17	0

Table 2. Continued.

Taxa		Life forms		Habitat specificity		Sensitivity variables				Significance variables		Scores	
		West	East	west	East	PT	GR	cs	co	ss	BS	SEN	SIG
Rodentia													
Sciuridae													
Northern flying squid	*Glaucomys sabrinus*	14	14	2	2	0	0	1	0	0	0	7	0
Yellow-pine chipmunk	*Tamias amoenus*	15	15	0	0	0	0	0	0	0	0	0	0
Muridae													
Deer mouse	*Peromyscus maniculatus*	15	15	0	0	0	0	0	0	0	0	0	0
Forest deer mouse	*Peromyscus keeni*	15		2		0	6	0	0	0	4	10	4
Southern red-backed vole	*Clethrionomys gapperi*	15	15	2	0	0	0	0	0	0	4	5	0
Long-tailed vole	*Microtus longicaudus*	15	15	2	0	0	0	0	0	0	0	2	0
Creeping vole	*Microtus oregoni*	15		0		3	0	0	0	0	0	3	0
Meadow vole	*Microtus pennsylvanicus*		15		0	0	0	0	0	0	0	0	0
water vole	*Mictotus richardsoni*	15	16	3	3	0	0	0	0	0	0	6	0
Heather vole	*Phencomys intermedius*	15	15	3	2	0	0	0	0	0	0	5	0
Northern bog lemming	*Synaptomys borealis*	15	15	3	3	3	0	1	0	0	0	12	0
Dipodidae													
Western jumping mouse	*Zapus princeps*	3	3	2	3	0	0	1	0	0	0	8	0
Pacific jumping mouse	*Zapus trinotatus*	3		3		0	6	1	0	0	2	12	2