Anadromous Fish Habitat in California's Mediterranean-climate Watersheds: Influences of Riparian Vegetation, Instream Large Woody Debris, and Watershed-scale Land Use

by

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Due to widespread declines in anadromous fish populations, significant resources are being invested in restoration of their freshwater habitat. To improve the effectiveness of these investments, restoration strategies must be targeted to the appropriate scale and tailored to the specific natural processes that create and maintain habitat within a region. This dissertation investigates several of the basic processes, operating across multiple spatial scales, which shape anadromous fish habitat in Mediterranean-climate streams of Northern California.

Although it is well established that large woody debris (LWD) provides critically important habitat values for salmonids in conifer-dominated watersheds, the relationship between LWD and fish habitat had not been previously investigated in the hardwood forests of California's Mediterranean-climate watersheds. Loading of LWD in hardwood streams was strongly related to characteristics of site- and reach-scale vegetation and management. Although debris loading was considerably lower than values reported from conifer streams, hardwood LWD provided similar habitat features: pool frequency was positively correlated with LWD loading, and LWD-formed pools had significantly higher shelter values than pools formed by other mechanisms. Debris jams were responsible for the majority of LWD-formed pools. Almost half of the channel-spanning jams contained a key piece composed of "living LWD" – a tree that entered the channel but remained rooted and living. Jams with a living key piece were significantly

larger, more persistent, and more likely to cause a pool than jams without a living key piece. Living LWD potentially compensates for the smaller size and faster decay rate of LWD from riparian hardwoods.

I also investigated the relative influences of land use and land cover (LULC) at various spatial scales on levels of fine sediment within spawning gravels (embeddedness) in the Russian River basin. Agriculture, urban, and herbaceous LULC categories were positively correlated with embeddedness, while forest was negatively correlated. Watershed-scale variables for LULC consistently explained the most variation in embeddedness levels, while LULC values within the adjacent riparian corridor explained little variation. Thus, LWD and spawning gravels are primarily influenced by factors operating at disparate scales, indicating that effective restoration strategies for these key habitat elements must target the appropriate scale.

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CHAPTER 1

ANADROMOUS FISH IN MEDITERRANEAN-CLIMATE WATERSHEDS

Anadromous Fish in Mediterranean-climate Watersheds

Anadromous salmonids of the Pacific Ocean (genus *Oncorhynchus*) spawn in streams from Japan and the Russian Far East to southern California. Until the past century, steelhead trout (*Oncorhynchus mykiss*) spawned as far south as Baja California in Mexico. Within this expansive range, salmonids spawn in streams spanning from small first-order tributaries to large main-stem rivers and within terrestrial habitats that include temperate rain forests, tundra, and inland arid grasslands.

Stocks of anadromous fish on the Pacific coast of the U.S. have declined significantly in the past century due to overharvesting, dams, and overall habitat degradation (Nehlsen et al. 1991, National Research Council 1996, Stouder et al. 1997). Salmonid freshwater spawning and rearing habitat have been impacted by a variety of anthropogenic activities operating at multiple spatial scales. The loss of riparian vegetation and removal of instream large woody debris (LWD) degrade habitat at the site and reach scales, while widespread changes in land use and land cover can affect habitat quality across an entire watershed by increasing inputs of fine sediment or changing thermal regimes.

Coastal watersheds in California once supported prodigious runs of six species of anadromous fish: coho (*Oncorhynchus kisutch*) , chinook (*O. tshawytscha*), pink (*O. gorbuscha*), and chum salmon (*O. keta*), and steelhead (*O. mykiss*) and sea-run coastal cutthroat trout (*O. clarki clarki*) (Moyle 2002). These species were further differentiated into numerous distinct runs, generally based on the primary time of year that spawning migration occurred (e.g. winter vs. summer steelhead). Due to a variety of factors, these runs have declined dramatically. Many of the species have been extirpated from coastal drainages (e.g. pink, chum, and possibly chinook from the Russian River), while those that remain are generally listed as threatened or endangered under the Federal Endangered Species Act (Steiner Environmental Consulting 1996, Mills et al. 1997, Busby et al. 2000, Hard et al. 2000, Weitkamp et al. 2000).

To address widespread declines in salmonid populations and the quantity and quality of their freshwater habitat, resources and attention devoted to stream restoration have greatly increased, and currently millions of dollars are being spent to restore anadromous fish habitat (National Research Council 1996, Roper et al. 1997, Roni et al. 2002). To improve the effectiveness of these investments, restoration strategies must be tailored to the specific natural processes that create salmonid habitat within a region and they must address the primary limiting factors of a population (Roni et al. 2002). Thus, research efforts can improve restoration programs by illuminating region-specific key processes and limiting factors.

Anadromous fish spawn within a range of stream sizes. Most runs of chinook in California spawn in main-stem rivers while coho and steelhead generally spawn in lower-order tributaries. After emerging from the redds, juveniles spend varying amounts of time within freshwater environments, from brief periods (pink, chum, and many runs of chinook salmon) to a year or more (coho salmon and steelhead) (Moyle 2002). Due to these variations, restoration strategies differ for fish in main-stem rivers (e.g. dam management, access to floodplains, and reducing mortality from diversions) and those with long rearing times in small, forested tributaries (e.g. instream habitat features such as pools, cover and water temperature).

In the western U.S., nearly all research available to inform restoration programs for anadromous fish in forested lower-order tributaries have been conducted in the conifer-dominated watersheds of Oregon, Washington, British Columbia, Southeast Alaska, and northern coastal California. Conversely, very little research on salmonid habitat has occurred within the hardwood-dominated, Mediterranean-climate watersheds of northern and central California. These watersheds have different riparian species and hydrologic regimes and more varied land uses than the more mesic coniferous forests. Therefore, restoration strategies developed in other regions may not be able to be directly transferred to Mediterranean-climate watersheds. An understanding of the basic processes that shape habitat, and the scales at which they operate, can assist the adaptation of restoration strategies to this region. The primary limiting factors may also differ, requiring different priorities for restoration.

This dissertation investigates several of the basic processes shaping anadromous fish habitat in Mediterranean-climate streams of Northern California, including channel morphology, riparian vegetation, water temperature, large woody debris (LWD), and fine sediment. By utilizing a variety of techniques and data sources – from intensive channel surveys to GIS analyses with an extensive data set collected by the California Department of Fish and Game - I also explore the various scales of influence to which these processes respond.

California's Mediterranean climates, vegetation, and water and land use

Regions with a Mediterranean climate undergo extreme intra- and inter-annual variability in precipitation and runoff (Figure 1). Winters are wet and cool and summers are dry and warm, although coastal temperatures can be moderated by marine effects.

In California's Mediterranean-climate watersheds, nearly 90% of the annual rainfall occurs between November and April, and the wettest months are December, January, and February. Rainfall is out of phase with the growing season of most crops, and, thus, agriculture generally requires storage of winter flows for irrigation in the summer. Intense winter rainstorms can produce extreme flooding events, while summer droughts can reduce flow to a trickle. During the annual summer drought, many streams are reduced to a series of isolated pools, while others dry up completely (Figure 2). Due to the inter-annual variability, streams may flow throughout the dry season during wet years, maintain isolated pools in "average" years, and dry completely during particularly dry years. This inter-annual and intra-annual variability exerts considerable selective pressures on organisms that inhabit Mediterranean-climate streams, as they must contend with both extreme floods and extreme drought within each year (Gasith and Resh 1999).

The vegetation of California's Mediterranean-climate watersheds is a mosaic of oak woodlands, mixed evergreen forests, oak savannas, annual grasslands, and chaparral (Barbour and Billings 1988). Conifer forests of Douglas Fir (*Pseudotsuga menziesii*) and Redwood (*Sequoiadendron sempervirens*) can dominate near the coast, where the fog belt and cooler temperatures moderate summer drought conditions. Conifer forests can also be found further inland within favorable microclimates, such as north-facing slopes.

California's Mediterranean-climate watersheds include very little public land and encompass a diversity of land–use types including rangeland, agriculture, and urban, suburban, and exurban development. Several watersheds (e.g. Napa River, Sonoma Creek, Russian River) are experiencing rapid land-use changes due to the demand for high-quality wine grapes as well as suburban/exurban expansion (Merenlender et al.

1998, Merenlender 2000). Because of the variability of runoff, Mediterranean stream systems are often dammed and intensely managed to provide dependable water supplies for agricultural and municipal water. In addition to large dams and inter-basin transfers (Griffin 1998, Langridge 2002) Mediterranean-climate stream networks are also greatly affected by numerous small-scale diversions, impoundments, and groundwater pumping. Because they are dispersed widely across the landscape, the impact of these small-scale manipulations is difficult to quantify, but they likely reduce the amount of perennial habitat available during the summer baseflow period. Small-scale impoundments may also reduce the extent of the stream network that is available to adult anadromous fish for spawning - either as direct barriers or by removing peaks from the hydrograph, which adult fish may require to access spawning sites in small streams (Moyle and Kondolf 2000).

Organization of the dissertation

In Chapters 2 and 3, I investigate the influence of large woody debris (LWD) on salmonid habitat in Mediterranean-climate streams. Wood in streams has consistently been identified as one of the most important elements shaping fish habitat (National Research Council 1996), although nearly all research on LWD and fish habitat has been conducted in coniferous forests. Chapter 2 provides an overview of the relationships between riparian corridors, LWD, and fish habitat in California's hardwood-dominated streams. Chapter 3 focuses on one important mechanism – the role of "living LWD" in creating and stabilizing debris jams – that may greatly increase the effectiveness of LWD in hardwood systems. Because of the direct relationship between riparian vegetation and LWD, Chapters 4 and 5 investigate a method of restoring riparian vegetation and the consequent effects on fish habitat. Chapter 4 focuses on the influence of deer herbivory on the regeneration of woody riparian species and the efficacy of promoting riparian restoration through the exclusion of deer. Chapter 5 examines the effects of riparian restoration on channel morphology, water temperature, and the accumulation of LWD. Finally, in Chapter 6, I investigate the influence of land use and land cover at multiple scales – the reach, riparian corridor, and entire watershed – on fine sediments within spawning gravels.

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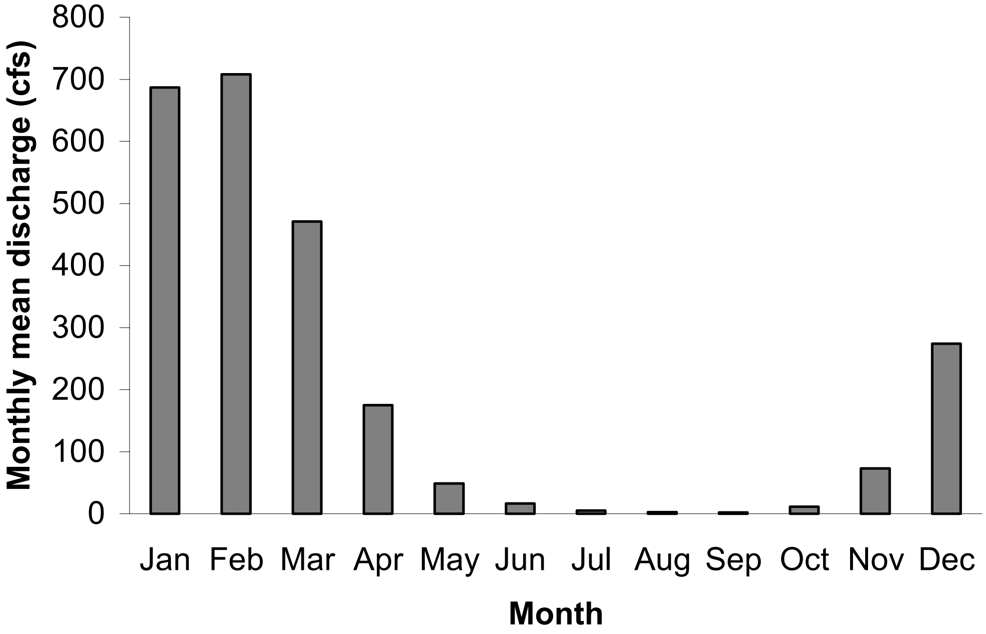
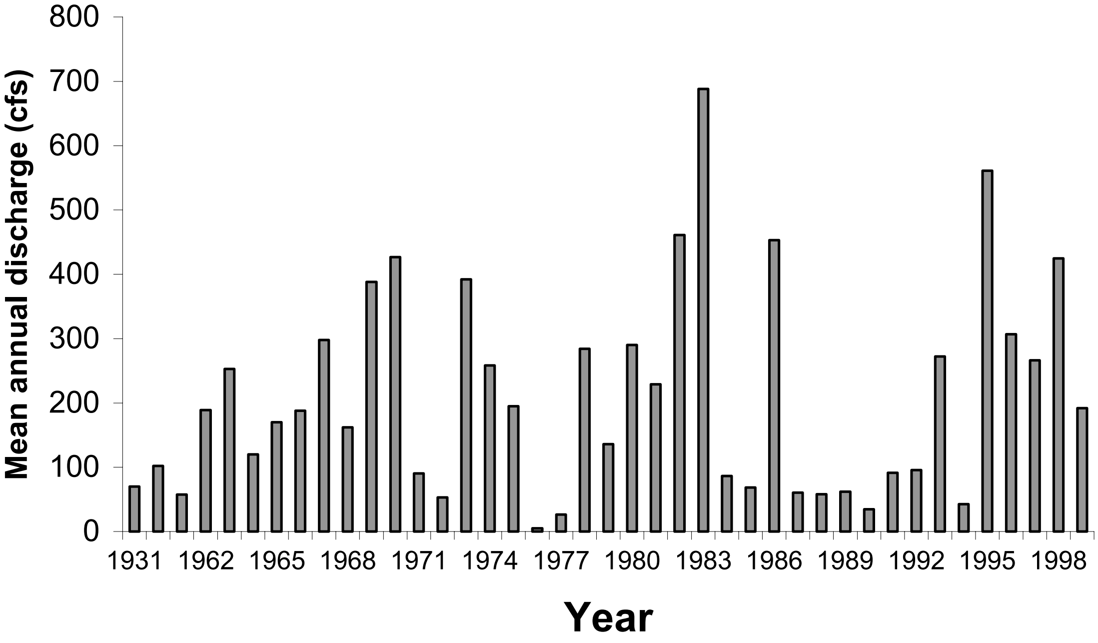
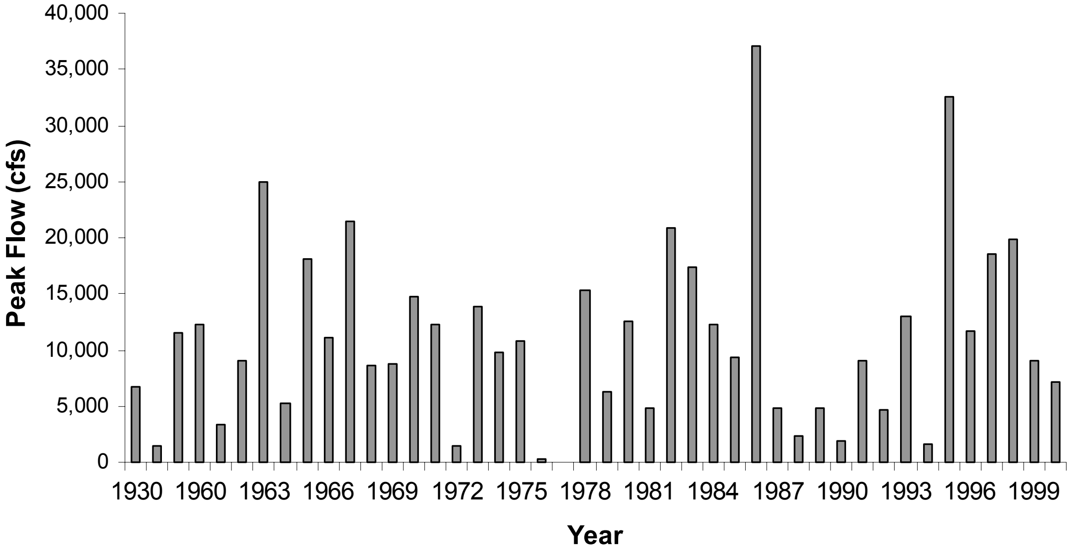
Figure 1. The Napa River watershed in northern California has a Mediterranean climate with a predictable pattern of winter rain and summer drought and considerable interannual variability in precipitation. This intra- and inter-annual variability is reflected in the patterns of discharge in the Napa River: A) monthly mean discharge; the month with the greatest average discharge (February) has 340 times as much flow as the lowest month (September); B) mean annual discharge; C) and annual peak discharge; both mean annual discharge and annual peak discharge vary by more than two orders of magnitude. (source USGS gauge # 11458000, Napa River near Napa, California; http://waterdata.usgs.gov/nwis/discharge/?site_no=11458000)



A.

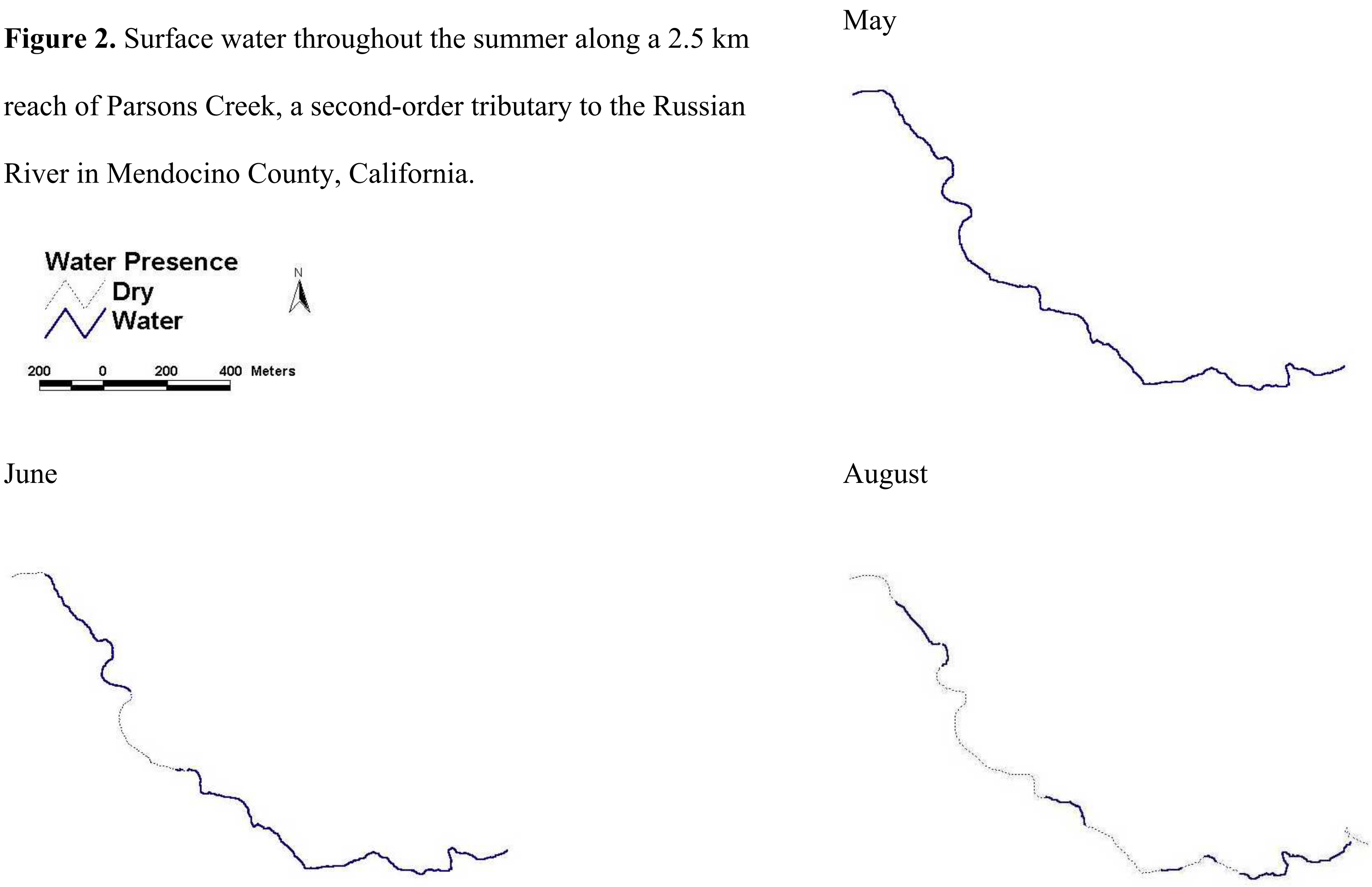

Mean annual discharge (cfs)
800
700
600
500
400
300
200
100
0
1931
1962
1965
1968
1971
1974
1977
1980
1983
1986
1989
1992
1995
1998
Year

B.


Peak Flow (cfs)
40,000
35,000
30,000
25,000
20,000
15,000
10,000
5,000
0
1930
1960
1963
1966
1969
1972
1975
1978
1981
1984
1987
1990
1993
1996
1999
Year

C.

Figure 2. Surface water throughout the summer along a 2.5 km reach of Parsons Creek, a second-order tributary to the Russian River in Mendocino County, California.



CHAPTER 2

LARGE WOODY DEBRIS AND FISH HABITAT IN CALIFORNIA'S MEDITERRANEAN-CLIMATE, HARDWOOD-DOMINATED WATERSHEDS

Large Woody Debris and Fish Habitat in California's Mediterranean-climate, Hardwood-dominated Watersheds

Abstract

Large woody debris (LWD) creates essential habitat features used by anadromous fish during their freshwater residency. Nearly all research on LWD and fish habitat has been conducted within the conifer-dominated forests of the Pacific Northwest. Consequently, data from conifer forests provide the scientific basis that underpins regulations, restoration strategies, and models focused on LWD. In California, however, anadromous fish also occupy watersheds with few or no conifers. This chapter investigates whether LWD provides fish habitat in hardwood watersheds and, if so, through which mechanisms. I used two data sets: California Department of Fish and Game habitat-typing surveys on 273 stream reaches in the Russian River basin, and more intensive field surveys of LWD and fish habitat on 30 hardwood-dominated streams in the Russian River basin and the San Francisco Bay Area. Many streams in the Russian River basin had very low LWD loading (< 25 m^3 *wood/ha channel) and displayed little influence of LWD on channel form or fish habitat. However, LWD was a major influence on fish habitat within streams with higher loading values and, in streams with the highest loading values, the majority of pools were formed by LWD. Pool frequency was positively correlated with LWD loading and pools formed by LWD had significantly greater shelter values than pools formed by other mechanisms. Because individual pieces of hardwood LWD are considerably smaller than LWD provided by mature conifers, single pieces of*

hardwood rarely influenced the channel. Instead, the majority of pools formed by LWD were formed by debris jams. In general, streams on private land had significantly less LWD than streams in protected watersheds, suggesting an important role for extension programs that educate private landowners on the relationship between wood in streams and fish habitat.

Perhaps no other structural component of the environment is as important to salmon habitat as is large woody debris.
(National Research Council 1996)

Although hardwoods, such as red alder and big-leaf maple may provide adequate shade and small woody debris to streams, they do not provide a long-term source of LWD important for creating and maintaining instream fish habitat.
(Roni et al. 2002)

Introduction

Large woody debris (LWD), defined as trees, logs, branches and rootwads within a stream, play a critical role in aquatic ecosytems (Harmon et al. 1986, Bisson et al. 1987). LWD serves as both a major geomorphic element within streams- directing flows to shape channel form and influencing sediment transport, deposition and storage - and an ecological element - providing habitat, substrate, cover, and influencing storage and processing of organic matter (Lassettre and Harris 2000). Due to these multiple influences on stream ecosystems, researchers have consistently identified LWD as a primary contributor to the quality of anadromous fish habitat (Murphy et al. 1986, Bisson et al. 1987, National Research Council 1996, Beechie and Bolton 1999). The majority of research on relationships between LWD and anadromous fish habitat has been conducted in conifer-dominated watersheds in the Pacific Northwest (PNW) (For example, see the annotated bibliography that accompanies Lassettre and Harris (2000)). In these studies, riparian hardwood trees are generally described as producing LWD that is either too small or decays too rapidly to remain stable and effectively influence channel morphology (Hyatt and Naiman 2001, Roni et al. 2002)

In California, however, anadromous fish, such as steelhead trout (*Oncorhynchus mykiss*), are not confined to conifer-dominated watersheds. They also utilize hardwood-dominated landscapes with very few or no conifers in their riparian corridor. These watersheds, and their riparian corridors, are dominated by species such as California bay laurel (*Umbellularia californica*), white alder (*Alnus rhombifolia*), and various oak (*Quercus spp.*) and willow (*Salix spp.*) species. Note that "hardwood" as a term doesn't refer specifically to the density of wood (willows, for example, have very soft wood); rather it is used as general label for both deciduous (e.g. alder, willow) and evergreen (e.g. bay laurel and live oak) angiosperms.

Hardwood-dominated watersheds generally correspond to the portion of California with a Mediterranean climate – cool, wet winters and warm, dry summers. Nearly all precipitation falls between November and April, leaving the warmest six months of the year with little or no rain. This hydrological regime produces a pronounced summer low-flow period (Gasith and Resh 1999). Thus, anadromous fish in these watersheds must find refugia of cool, well-oxygenated water in microhabitats that provide protection from predators, whose hunting effectiveness may increase when streams begin drying into isolated pools. Therefore the formation of pools with high cover values may be particularly valuable under this hydrological pattern. Anadromous fish in Mediterranean-climate, hardwood dominated watersheds have received little study, and the role of LWD in these ecosystems is generally unknown. In this study, I explore the relationship between LWD and anadromous fish habitat in California's hardwood-dominated, Mediterranean-climate watersheds.

Background

Geomorphic influences

LWD functions as a large structural element in streams. Stable pieces or aggregations of LWD can concentrate flow during periods of high discharge and direct this convergent flow against the bed or banks, inducing scour which results in pool formation. Pool types caused by LWD include plunge pools, dammed pools, and lateral scour pools (Bisson et al. 1987, Bilby and Ward 1989). Pool frequency and total pool area generally increase with LWD loading (Bilby and Ward 1989, Montgomery et al. 1995, Beechie and Sibley 1997). Reaches with high LWD loading and frequency in conifer forests of the PNW can have greater than 70% of their pools formed by LWD (Andrus et al. 1988, Robison and Beschta 1990, Montgomery et al. 1995, Wood-Smith and Buffington 1996). Pools are generally most important for larger fish (Moyle 2002), and thus the loss of LWD, and consequent simplified channel form with fewer pools, can be particularly detrimental for older age classes (National Research Council 1996).

LWD can also interact with high flows to induce deposition of sediment, thereby promoting creation and maintenance of bars and regulating the movement of bedload (Nakamura and Swanson 1993). Experimental removal of LWD from stream channels has resulted in significant increases in sediment transport due to loss of storage sites and channel roughness (Bilby 1981, Smith et al. 1993). Debris jams can also promote deposition of substrate for tree regeneration, and LWD can protect saplings from scour, promoting riparian forest regeneration. This phenomenon has been described in systems ranging from individual trees on small streams to development of floodplain forests on relatively large, braided rivers (Fetherston et al. 1995, Abbe and Montgomery 1996, Collins et al. 2002)

Due to deposition behind channel-spanning debris jams, LWD can influence the local gradient of streams. Aggradation behind jams can produce localized, low-gradient, alluvial reaches within portions of the stream network that would otherwise consist only of high-gradient, bedrock-dominated reaches (Montgomery et al. 1996). These local influences on channel morphology have important benefits for both the formation of riparian vegetation and fish habitat.

Ecological Influences

Similar to its role regulating bedload and sediment transport, LWD also influences the transport of organic matter (Bilby 1981). LWD and debris jams induce deposition of fine woody debris and fine organic matter such as twigs and leaves (Bilby and Ward 1989), and increase the retention of salmon carcasses (Cederholm and Peterson 1985, Cederholm et al. 1989) which can be an important source of nutrients for juvenile fish, terrestrial predators and the adjacent riparian forest (Helfield and Naiman 2001).

By slowing the transport of these materials out of stream systems, LWD influences the rate and effectiveness at which aquatic food webs process organic material. This retention of organic material can enhance the productivity of streams. Further, LWD itself provides an important and unique substrate for invertebrates, which can greatly increase the diversity and overall productivity of a stream system (Dudley and Anderson 1982).

LWD also plays a critical role providing cover from predators for organisms ranging from invertebrates to mature fish. Although low pool frequency has been linked to declines in salmonids, lack of cover within pools has also been found to reduce the carrying capacities of streams (Inoue et al. 1997). The value of cover for fish greatly increases during low-flow periods, when water is clear and fish (particularly larger fish) are restricted to pools (Shirvell 1990, Giannico 2000). During high flows, LWD functions as hydraulic cover, allowing fish to enter velocity shadows to avoid high-energy flood waters (McMahon and Hartman 1989, Harvey et al. 1999), increasing over-winter survival (Quinn and Peterson 1996). Thus, the loss of LWD makes fish more vulnerable to predation during the summer and more vulnerable to high-energy storms during the winter.

The ecological role of LWD varies with position within the channel network (Lassettre and Harris 2000). Within smaller tributaries, LWD may not play a role during the summer because the stream dries up (although it may be important for aquatic invertebrates or terrestrial species). However, during the winter, fish utilize these small tributaries to spawn and to avoid higher-energy mainstem channels. At this time, LWD in small streams may provide hydraulic shelter and induce deposition of spawning gravels (Brown and Hartman 1988, Rosenfeld et al. 2002).

Research questions

This study complements previous research efforts focused on conifer-dominated watersheds by exploring the influence of LWD on fish habitat in hardwood-dominated watersheds. I conducted my own field work and analyzed an extensive data set collected by the California Department of Fish and Game (CDFG) to address the following questions: 1) What is the distribution of forest types within salmonid-bearing tributary watersheds of a Mediterranean-climate river basin in northern California (the Russian River)? To answer this question I analyzed land use/land cover data that classified forests into hardwood, conifer, and mixed evergreen. 2) How does the relative influence of LWD on channel form vary between these different watershed and riparian vegetation types, and with channel characteristics? I used the CDFG data set to explore this question by analyzing the proportion of pools formed by LWD across streams with different riparian vegetation, bankfull widths, and substrate sizes. 3) How does LWD loading in hardwood watersheds compare to values from other systems? To understand loading I conducted field surveys to measure LWD within hardwood-dominated streams. 4) Does LWD provide important habitat functions for salmonids in hardwood watersheds? If so, which functions and through which mechanisms and species? Based on the field surveys I investigated to what extent LWD was influencing pool formation and providing shelter values for fish.

Methods

I analyzed two fish-habitat data sets: an extensive habitat-typing data set collected by CDFG in the Russian River basin in 1995-2000 and field data that I collected on 30 streams in the Russian River and the San Francisco Bay Area in 2000-2002, including 11 reaches previously surveyed by CDFG.

CDFG data

CDFG's habitat-typing surveys in the Russian River basin followed protocols in the California Salmonid Stream Habitat Restoration Manual (CDFG 1998) (Figure 1). During the low-flow summer season, two-person crews identified reach types based on the Rosgen classification system (Rosgen 1994). In this chapter, I used habitat-typing data from 125 streams divided into 373 reaches. The watersheds' drainage areas ranged in size from 42 to 22,000 ha (Table 1).

Survey crews further partitioned reaches into habitat units (e.g. various types of riffles, glides and pools *sensu* Bisson et al. (1982)) and recorded dimensions and shelter ratings for every pool (pool types are listed Appendix I). Shelter ratings (ranging from 0 – 300) were calculated by multiplying a qualitative assessment of fish cover quality, ranging from 0 – 3, by the percent of the pool with cover.

I integrated CDFG surveys into a Geographic Information System (GIS) using dynamic segmentation (Byrne 1996, Radko 1997, Jones and Moore 2000) and calibration within ArcView (ESRI). Using 10-m Digital Elevation Models (DEMs) and the ArcView extension Watershed Delineator, I defined a discrete watershed boundary upstream of the beginning point of each CDFG-defined reach. The watershed boundary was then used to clip land use/land cover (LULC) coverages (see Chapter 6).

Additionally, 30-m buffers were defined around each reach, and the same set of LULC variables were clipped to these buffers. Thus for every reach I could calculate the proportion of a given forest type for both the entire upstream watershed and within a 30-m buffer surrounding the reach. For the purposes of this study, I classified watersheds and buffers into categories based on the LULC that dominated the unit (>50%). Forest types included hardwood, conifer, and mixed (corresponding to the 'mixed evergreen' forest type in Barbour and Billings (1988)).

Field surveys

I surveyed 30 streams (total survey length = 21.7 km) in the Russian River basin and San Francisco Bay Area between 2000 and 2002 (Figure 2). The Bay Area contains several regional, state, and national parks as well as watersheds protected for municipal water reservoirs. Study sites were selected within hardwood-dominated forests of these protected watersheds. Because the Russian River basin is composed almost entirely of privately owned land, selection of study sites within the basin was biased by the willingness of landowners to allow field work to be conducted on their property. Initial selection of study streams was based on recommendations from CDFG of landowners who would be likely to grant access for field work. Although these streams were not selected through a random stratification process, the streams in the field data set provide a fairly representative sample of stream sizes (e.g. bankfull widths and drainage areas), compared to the larger CDFG sample (Table 1). Because LWD was often very rare within the initial set of streams surveyed in the Russian River, and I wanted to sample streams with a broad range of LWD loading, I used the CDFG database to identify stream reaches with a high proportion of pools caused by LWD (> 40%, the top quartile). I then specifically contacted the landowners of these reaches to seek access. Six reaches were identified and surveyed through this process. Because these streams had been specifically selected for their high LWD influence, I placed them within a separate group for an analysis of the relationship between land ownership and LWD loading.

Within each reach, bankfull width was measured every 100 m using indications of bankfull channel dimensions as described in Harrelson et al. (1994) and Platts et al. (1983). Reach locations were recorded with a Global Positioning System (GPS) to allow subsequent calculation of drainage area within a GIS. Because field indications of bankfull width in Mediterranean-climate channels can be influenced by the time since episodic major floods (Wolman and Gerson 1978, Hecht 1994, Kondolf et al. 2001), I analyzed the relationship between bankfull width and drainage area to confirm the utility of using bankfull width to stratify stream sizes.

LWD and debris jams

I sampled all pieces of LWD (pieces of wood > 1 m in length and 10 cm in diameter) within the bankfull dimensions of the channel. For each piece of LWD, I recorded the following information: length and diameter, species (when possible; if not possible to determine species it was classified as either conifer or hardwood), decay, channel position, function, input mechanism (when possible to determine), and input distance. (Further information on the categories that I used can be found in Appendix II). Debris jams were defined as accumulations of three or more pieces of LWD and were classified according to their channel position (on the bank, partially spanning the channel, or spanning the channel). I also noted if the jam caused a step in the channel profile, caused or contributed to the formation of a pool, and whether or not the jam was stabilized by standing trees along the channel margin.

Pools

I defined pools as portions of the stream characterized by low velocities and greater depths (> 12 inches in residual depth). Because most of the streams were surveyed during summer baseflow, pools were generally visually obvious. I measured the average width, total length, and depth of the pool. To control for varying discharge (e.g. between a stream measured in late May and a stream measured in August), I calculated the residual depth of pools by subtracting the depth of the pool tail crest from all pool depth measurements (Lisle 1987). I classified each pool according to the factor responsible for its formation, using a modified version of that found in the CDFG Salmonid Stream Restoration Manual (1998) to facilitate comparisons with the broader data set collected by CDFG (Appendix I). Pools were further classified according to the level of LWD influence: Primary (LWD is the only apparent factor causing the pool), Contributing (LWD and another factor both contributing significantly to the formation of the pool), Enhanced (LWD added some scour to the pool but the pool was primarily caused by another factor), or None. Thus, a pool classified as a Meander pool (a non-LWD type of pool) could still be categorized as having a major LWD influence if wood was playing a significant role shaping the pool (e.g. it would be classified as a Meander-Contributing). I also visually estimated the surface area of the pool that was influenced by various sources of cover, including large woody debris, fine woody debris, boulders, terrestrial vegetation, aquatic vegetation, undercut banks, and roots/rootwad. This technique differed from the CDFG shelter rating in that I did not use a qualitative estimate of shelter quality and, thus cover values from the LWD field surveys could range from 0 – 100.

Riparian structure

Within every 100-m segment of channel, I sampled the riparian forest using two randomly selected riparian plots (one on each side of the channel). Each plot extended 10 m along the channel and a variable distance perpendicular to the channel. The variable distance was based on the ability of a tree within that plot to fall and enter the stream and was therefore dependent on tree heights and the slope of the plot. Within these plots the diameter-at-breast-height (dbh) of every living tree and snag was measured along with its slope-distance from the stream, and the landform where it grew (bank, bar, floodplain, terrace, hillside) was classified.

Results

CDFG data

Hardwoods were the most common cover type in the 125 watersheds surveyed by CDFG crews. Within the Russian River basin, 33% of surveyed watersheds were dominated by hardwoods, 9% were dominated by mixed evergreen forest, while only 5% were dominated by conifers. Thirty-five percent of watersheds surveyed were primarily forest without a single dominant (>50%) forest type. In only 11% of these heterogeneous watersheds was there more conifer than hardwood. Within a 30-meter buffer of CDFG surveyed streams, 49% were dominated by hardwoods, 15% by mixed evergreen, and 10% by conifers.

Across all CDFG-surveyed reaches, the most common pool types were mid-channel (27%) and lateral-scour rootwad (22%). None of the other categories of LWD-formed pools constituted more than 5% of the total. In sum, 28% of all pools were formed by LWD, with most of these classified as lateral-scour rootwad. The CDFG survey methods did not distinguish between pools formed by rootwads of living trees on the bank and those formed by rootwads that had entered the channel and, thus, it is not clear what proportion of later-scour-rootwad pools were formed by wood that had entered the stream versus trees along the bank. A very small proportion of pools were formed by logs or debris jams in the channel (6% total from logs and dammed pools). Subsequently, I will provide analyses of the proportion of pools formed by logs and debris jams ("in-channel LWD") as well as "LWD overall" which includes lateral-scour rootwad pools. Thus, the "LWD overall" category for CDFG data includes pools formed by in-channel LWD as well as from rootwads of trees rooted at the channel margin.

Considering reaches with at least 10 pools ($n = 250$), an average of 6% of pools were formed by in-channel LWD. Including rootwad pools in this analysis raised the average proportion of LWD pools to 18%. Half of all reaches had less than 3% of their pools formed by in-channel LWD, and half of all reaches had less than 13% of their pools formed by LWD overall. The proportion of pools formed by LWD was strongly influenced by the substrate of the channel bed; reaches with cobble/gravel or fine substrate had significantly more LWD pools than reaches with beds dominated by bedrock and boulders (ANOVA, $F_{2, 247} = 22.2$; $p < 0.0001$) (Figure 3). Within channels dominated by cobble, gravel, or fine substrate (where LWD pools were more common), the proportion of pools formed by in-channel LWD was strongly influenced by bankfull width, while the proportion of pools formed by rootwads was not related to channel width This trend was consistent within both conifer and hardwood reaches (Figure 4).

Controlling for substrate size and bankfull width, conifer-dominated reaches had a higher proportion of pools formed by LWD than hardwood-dominated reaches (Figure 5). Across all reaches, pools formed by logs and jams (i.e. instream-LWD) had significantly higher shelter ratings (52.5 ± 1.8), than pools formed by rootwads (44.6 ± 0.9) and non-LWD pool types (31.3 ± 0.7) (ANOVA, $F_{3,\ 9118} = 101.8$, $p = < 0.0001$).

LWD field surveys

Large woody debris and riparian trees

Within the hardwood reaches that I surveyed, *Alnus rhombifolia* (white alder) was the most common species of LWD (29% of all pieces), followed by unidentified hardwood (27%), *U. californica* (California bay; 17%), *Salix spp.* (willow species; 14%), conifer species (primarily *Psedotsuga menziesii* (Douglas fir) and *Sequoiadendron sempervirens* (redwood); 5%), and *Quercus spp.* (oak species 5%). Because a few streams were sampled more extensively than others and would thus contribute disproportionately to an analysis of LWD species distributions across all wood sampled, the average proportional contribution of each species was also analyzed (Figure 6). Alder, bay, and oak LWD were found at nearly all sites, while wood from willow, redwood, and Douglas fir were found at less than half of the sites (Figure 7). LWD pieces from conifer species had greater diameters, lengths, and volumes than LWD from hardwood species. Among hardwood species, California bay provided the largest pieces of wood (Table 2).

Bay, willow and alder were the three most common trees in riparian corridors by average frequency; while bay, alder and oak provided the greatest average basal area within hardwood reaches (Figure 8). Alder, oak, and bay were found in the riparian corridors of nearly all the sampled reaches; while willow, Douglas fir, and redwood were less commonly present among the reaches sampled (Figure 7).

Loading of LWD within hardwood streams varied from 0.0 to 173 m^3 wood/ha of channel, with a mean of 61 ± 9 (mean ± standard error). Loading averaged 115 ± 13 m^3 wood/ha within streams in protected watersheds. For comparison, two streams within redwood forests were sampled, and their loading was 962 m^3 wood/ha (McChristian Creek with mature redwoods) and 303 m^3 wood/ha (Sheephouse Creek with second growth forest). A second reach on McChristian was located within a section where Douglas firs were common within the riparian corridor and among the LWD, and its loading was 243 m^3 wood/ha.

Ninety per cent of all hardwood LWD pieces whose source could be identified were recruited from within 10 m of the channel (Figure 9). I was able to determine the source for 708 out of the 4391 (16%) measured pieces of LWD. The most common input mechanism was bank erosion (34%), followed by tree fall (27%), grow (17%), branch breakage (12%), and mass wasting (8%).

Pools

The most common pool types were lateral-scour-bedrock (21%), meander (14%), lateral-scour-rootwad (13%), and lateral-scour-boulder and lateral-scour-jam (each 12%) (Figure 10). Overall, 42% of pools were formed by LWD (Figure 11) and 27% were formed by in-channel LWD (i.e. excluding LWD formed by the rootwads of living trees on the banks). From in-channel LWD, most pools (65%) were formed by jams (Figure 12). The proportion of pools formed by in-channel LWD was positively correlated with the loading of LWD within the reach ($r^2 = 0.52$; $p = 0.0002$) (Figure 13). Pool spacing (the inverse of pool frequency, expressed as the number of bankfull channel widths per pool) was negatively correlated with LWD frequency (Figure 14), based on a multiple regression model with pool spacing as the dependent variable and model terms LWD frequency (LWD_{freq}) and bankfull width (BF_{w}) ($r^2 = 0.70$; $p < 0.0001$; $n = 20$; LWD frequency leverage, $p = 0.0008$). The resulting regression equation was:

$$\text{Pool Spacing} = 8.8 - 6.1 * LWD_{freq} - 0.4 * BF_w$$

Considering only reaches with at least five pools ($n = 24$), the proportion of pools within a reach formed by in-channel LWD ranged from 0 – 100%, with a mean of 30%. After including rootwads of living, upright trees, the mean increased to 46%. Including all pools with some LWD influence (i.e. including non-LWD type of pools with some LWD contribution e.g a meander pool whose morphology is influenced by LWD), the mean increased to 57% (Table 3).

Within hardwood streams, pools formed by LWD had significantly higher cover values (32.8 ± 1.0) than non-LWD formed pools (18.0 ± 1.0) ($F_{1, 524} = 101.7$, $p < 0.0001$). This trend was consistent when comparing pool cover values at the level of individual reaches. Within every reach, LWD pools had higher cover values than non-LWD pools (paired $t = 6.8$, 22 *df*, $p < 0.0001$). The average cover value of all pools within a reach was positively related to the frequency of LWD within the reach ($r^2 = 0.66$; $p < 0.0001$) (Figure 15). Among the pool types influenced by wood, pools formed by the rootwads of trees rooted at the channel margin had the lowest cover values, while pools formed by debris jams had the highest (Figure 16).

Alders were responsible for the majority (71%) of pools formed by the rootwads of living trees rooted along the bank, followed by bays and willows (9% each). The average dbh of a an alder forming a rootwad-pool was 43 cm (standard deviation = 15.5 cm; standard error = 2.0, n = 58).

In the eleven streams surveyed by both me and CDFG, the proportion of pools classified as formed by LWD were highly correlated ($r^2 = 0.86$, $n = 11$, $p < 0.0001$; Figure 17). Of all pools that I classified as being influenced by rootwads (those that would have been classified as lateral-scour-rootwad by CDFG), nearly 80% were from rootwads of living trees growing at the channel margin, and approximately 20% were from rootwads and logs with rootwads within the channel.

Debris jams

I found a total of 306 jams in the 30 stream reaches: 152 spanning, 65 partially spanning, and 89 bank jams. The proportion of jams with key pieces varied by channel position, with most bank jams having no key piece and almost 2/3 of all spanning jams having a key piece or pieces (Figure 18). Twenty-two percent of partially spanning jams and 45% of all spanning jams had a key piece that was still living. Considering only those jams with a key piece, 41% of partially spanning and 61% of spanning jams had a live key.

Controlling for drainage area, channel spanning jams were significantly larger than those on the bank or partially spanning the channel (ANCOVA, $F_{3, 279} = 8.0$; $p < 0.0001$) (Figure 19). Channel-spanning jams were significantly more likely to cause pools than other jam types (Figure 20; $X^2_{2, 297} = 56.5$, $p < 0.001$). Overall, 66% of all LWD pieces and 59% of all LWD volume was located within debris jams. Of the volume stored within debris jams, 61% was found within channel-spanning jams. Bay, willow, and alder were the most common key pieces for all jams as well as for channel-spanning jams (Figure 21). Bays provided half of all keys for channel-spanning jams in reaches with drainage areas greater than 1000 ha.

Jam frequency is expressed in terms of number of jams per 10 bankfull widths (10BFw). These units allow the spacing of jams to be compared across channel sizes by scaling the frequency to the bankfull width of the stream. Jam frequency ranged from 0 to 2.7 jams/10BFw, with a mean of 0.7 ± 0.1 jams/10BFw, while spanning-jam frequency ranged from 0 to 1.8 jams/10BFw, with a mean of 0.3 ± 0.1 jams/BFw. The frequency of spanning jams was significantly negatively correlated to bankfull width ($F_{36, 1} = 13.5$; p = 0.0008; $r^2 = 0.28$), and no spanning jams were found on streams with bankfull widths greater than 10 m.

Land ownership

After removing all streams with bankfull widths greater than 10 m, there was no longer a relationship between spanning jam frequency and bankfull width. The frequency of spanning jams was significantly higher within public land than on privately owned streams ($t = 2.3$, 21 *df*, $p = 0.03$); overall jam frequency was higher on public land, although the difference was not statistically significant ($t = 1.0$, 29 *df*, $p = 0.32$) (Figure 22). Streams on private land had significantly lower loading values (37 ± 10 m^3/ha) than streams on public land (100 ± 13 m^3/ha) ($t = 3.9$, 30 *df*, $p = 0.0005$)**.** Similarly, LWD frequency was lower in private (0.09 ± 0.02 pieces LWD/m) than in public lands (0.21 ± 0.02 pieces LWD/m) ($t = 4.0$, 30 *df*, $p = 0.0004$). Riparian corridors on public and private streams had similar basal area within the first 3 m from the channel, with riparian corridors on private land having significantly lower basal area at greater distances from the channel (Figure 23). The initial sample of hardwood reaches from private lands suggest that there are fewer LWD pools within these streams, and this is supported by the much larger CDFG sample of streams from the Russian River basin, which is almost entirely privately owned (Table 4).

Discussion

Loading of LWD within hardwood-dominated streams in northern California was considerably lower than values reported from conifer-dominated streams of the Pacific Northwest. Loading values from 84 conifer-forest streams reported in the literature, with a variety of stand ages, ranged from 10 – 4500 m^3 LWD/ha and averaged 642 ± 80 m^3 LWD/ha (Keller and Tally 1979, Harmon et al. 1986, Robison and Beschta 1990, O'Connor Environmental 2000). The median value was 450 m^3 LWD/ha with an interquartile range of 236 – 757 m^3 LWD/ha. Thus, the median value for these conifer-dominated streams was more than twice the highest value for a hardwood stream found in this study (Olema Creek with 173 m^3 wood/ha).

These differences in loading can be attributed to several factors. The maximum dimensions for LWD from hardwood species were considerably smaller than the maximum dimensions for conifer LWD (although my survey design was not intended to extensively sample conifer LWD to determine maximum values, and thus maximum values for conifers are no doubt greater). The largest conifer piece, in terms of volume, was six times larger than the biggest bay piece and eight times larger than the biggest alder piece. The largest hardwood debris jam, in terms of volume, was a bank jam on Olema Creek consisting of 19 pieces of LWD with a total volume of 36 m^3 of wood. In volume, this jam was slightly smaller than the largest single redwood piece that I found.

These examples illustrate that streams with conifer LWD will have higher loading values simply due to the massive volume of individual pieces of conifer LWD.

Further, piece stability is a function of piece size, so conifer debris will tend to be more stable within channels (Braudrick et al. 1997, Braudrick and Grant 2000). Conifer debris also decays more slowly, due to both size and wood properties (Scheffer and Cowling 1966, Harmon et al. 1986). These factors – stability and decay resistance – allow conifer debris to persist longer within channels, leading to greater loading values. Finally, due to the great height of conifer species wood from trees that are farther from the channel can more easily enter the stream. While oaks, alders, and bays can be recruited to a channel through treefall from distances of 20 – 30 m, redwoods and Douglas firs can be recruited from distances of up to 80 – 120 m (although mass wasting events can allow hardwood debris to enter from greater distances).

This smaller recruiting area is reflected in the observed source distance: 90% of debris whose source could be identified was recruited from within 10 m of the channel. This value likely underestimates recruitment distances, as it is more difficult to identify sources that are more distant from the channel. However, even if the source distance required to capture 90% of recruited debris was actually greater by 50% or 100% (i.e. 15 – 20 m), it would still be approximately half that of a mature conifer forest (McDade et al 1990).

Although the loading values were considerably lower than in mature conifer forests, hardwood LWD provided many of the same important habitat functions for fish. Pool spacing declined with increasing LWD frequency, and LWD-formed pools had significantly higher shelter values. The total cover available for fish within a reach was also positively correlated with LWD frequency. Thus, hardwood LWD provides many of the same essential habitat features for anadromous fish that conifer LWD provides in PNW streams. However, important differences exist in the ways that hardwood debris influences channel morphology, and there are differences in the scale across which these interactions occur.

While single conifer logs frequently function as major geomorphic elements, single hardwood logs were less likely to create pools or otherwise influence channel morphology. Of all pools influenced by wood, only 22% were associated with single logs or rootwads. The remainder were caused by debris jams (47%) and rootwads of living trees rooted at the channel margin (31%). A large proportion of the debris jams that cause pools are stabilized by living key pieces. When combined with the number of pools caused by the rootwads of living trees, this indicates that a high proportion of all wood-influenced pools within hardwood streams are associated with living trees rooted at the channel margin, including those still standing and those that have fallen into the channel but remain alive. The importance of these rooted trees to the overall influence of wood on pools is consistent with the fact that hardwood LWD pieces are less stable and persistent in the channel. Thus, in hardwood streams, living trees are responsible for much of the beneficial influences of wood on stream morphology and fish habitat.

Bay and willow provided the greatest number of key pieces, with both species contributing to keys in greater proportion than to LWD overall. Nearly all willows that formed key pieces were *Salix laevigata* (red willow), and the disproportionate representation of willows among key pieces was due almost entirely to the ability of willows to function as living key pieces (Chapter 3). Bay also provided numerous living key pieces and was the most important contributor of dead key pieces (Chapter 3). Their prevalence among dead keys is likely due to the fact that bays provided the largest single pieces of LWD and are composed of stronger, more decay-resistant wood than the other common riparian trees – alders and willows. Because bay and red willow are the most important providers of key pieces, these two species may be particularly important to promote during riparian restoration in hardwood streams. Willows are commonly planted because they can establish quickly and have a high success rate growing from cuttings, while bays are planted much less frequently. However, Narrow-leaved willow (*Salix exigua*) is one of the most commonly planted willows in the Russian River basin because it is widely available and has rapid vegetative growth. This species has a shrubby growth form and I did not find any *S. exigua* acting as a key piece or otherwise providing LWD. Bays are considered a late successional species capable of reproducing and growing under an established canopy (McBride 1974). Thus, both the potential for promoting bays directly during restoration and the ability of bays to eventually grow within restored riparian corridors merit further study.

The relative influence of LWD on channel morphology, including pool formation, declines with increasing channel size (Lassettre and Harris 2000) and the potential for wood to influence channel morphology is a function of the size of the piece relative to the size of the channel (Bilby and Ward 1989, Keim et al. 2000). Due to its smaller dimensions, the "zone of influence" of hardwood LWD on channel morphology is consequently limited to a smaller range of channel sizes than conifer LWD. In the hardwood streams in this study, channel-spanning jams provided the greatest contribution to pool formation. While channel-spanning jams were not found in streams with bankfull widths greater than 10 m, Frissell (1992) found numerous channel-spanning jams on reaches with bankfull widths up to 22 m within mature conifer forests. However, none of the streams within protected watersheds, which had the highest loading and most frequent debris jams, had bankfull widths greater than 10, and thus the sample within this study may not effectively capture the upper limit of channel-spanning jams.

The CDFG data indicate that there is currently very little influence of in-channel LWD on pool formation in the Russian River basin, which is supported by the sub-sample of Russian River basin tributaries that I surveyed in the field. (Based on my surveys which found that 80% of pools associated with rootwads were from the rootwads of living, standing trees on the channel margin, it is likely that most of the rootwad pools in the CDFG data set are from standing trees rather than in-channel rootwads.) As an example, the two reaches of Olema Creek within Pt. Reyes National Seashore, with bankfull widths of 7.6 and 9.3 m, had 47% and 33% respectively of their pools formed by in-channel LWD. Streams in the Russian River with similar bankfull widths and similar substrate had less than 10% of their pools formed by in-channel LWD. Although rootwad pools were more common among the streams surveyed by CDFG, in-channel LWD provides different habitat features and pool types than do the rootwads of trees along the bank. Rootwad pools form during high flows, and during summer low flows the water level of the pool is frequently below the level of the rootwad. Thus the roots themselves may not provide direct cover during the summer. Pools formed by logs, rootwads, or jams that extend into the low-flow channel continue to provide the interface of woody elements and water into the dry season, improving the cover value of pools from in-channel wood. Both the LWD survey and CDFG habitat-typing data indicate that rootwad pools had lower shelter values than log or jam-formed pools.

This lack of in-channel LWD pools in many reaches of the Russian River could be a primary cause of decline of salmonid populations. A simplified stream channel lacking LWD is particularly poor habitat for older age classes (National Research Council 1996). A review of limiting factors in the Napa River basin (which has a similar mix of conifer, hardwood, and mixed evergreen vegetation as the Russian River) concluded that low levels of LWD were a primary factor contributing to the decline of salmonids (Stillwater Sciences and Dietrich 2002).

This and other studies (Frissell 1992, Hilderbrand et al. 1997) have found that LWD pools are less common in streams with large substrate, and this study suggests that the influence of LWD declines in channels > 10 m in width. Therefore, identification of stream reaches with insufficient LWD should focus on reaches less than 10 m in width with a substrate of cobbles or smaller. Although log structures have the potential to provide much of the same functions as natural LWD, the most sustainable way to restore LWD levels is to promote restoration of natural riparian corridors (Chapter 5), allow fluvial processes to occur (e.g. because most LWD is recruited through bank erosion), and encourage landowners to leave existing LWD within channels.

During my field work I found ample evidence, both through observations in the field and in discussions with landowners, that removal of LWD within channels is still very common. This is motivated by concern that LWD and debris jams can lead to bank erosion and potentially threaten downstream bridges or culverts. Cutting for firewood may also play a role. Finally, there is lingering uncertainty over the importance of LWD, and many landowners may feel that debris jams are barriers to upstream migration because at low flows jams may appear or actually function as barriers. However, at higher flows, when most fish movememt occurs, debris jams rarely block fish migration. For years, Fish and Game agencies across the West promoted the removal of LWD "barriers" as a way to improve fish habitat, contributing to current landowner perceptions of LWD in stream channels. Extension programs that promote the importance of LWD can potentially dispel misconceptions and encourage landowners to leave wood in streams.

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Table 1. Descriptive statistics for drainage area and bankfull width for the streams sampled by CDFG and during the LWD field surveys.

	Drainage area (ha)		**Bankfull width (m)**	
	Median (interquartile range)	Mean (s.d.)	Median (interquartile range)	Mean (s.d.)
CDFG (n=373)	752 (492, 2142)	1580 (2060)	5.6 (4.5, 9.4)	6.9 (3.5)
LWD field surveys (n=30)	728 (307, 2388)	2241 (3582)	6.2 (4.5, 9.1)	7.1 (3.9)

Table 2. Dimensions of LWD from common hardwood and conifer species found within the hardwood-dominated study streams.

Species	Count	Max	Mean (SE)	Median	Interquartile
		Diameter (cm)			
S. sempervirens	50	250	33.3 (3.6)	25	16 – 45
P. menziesii	60	125	39.7 (3.5)	35	15 – 64
U. californica	737	120	23.7 (0.5)	20	14 – 30
A. rhombifolia	1242	100	20.9 (0.4)	16	12 – 26
Quercus spp.	217	80	22.4 (0.8)	19	14 – 28
Salix spp.	593	60	16.4 (0.3)	14	11 – 20
		Length (m)			
S. sempervirens	50	15.25	4.00 (0.40)	3.1	2.1 - 5.0
P. menziesii	60	27.50	7.30 (0.80)	5.2	3.1 - 10.1
U. californica	737	16.80	5.10 (0.12)	4.6	2.4 - 7.6
A. rhombifolia	1242	19.20	3.90 (0.08)	3.1	1.8 - 4.9
Quercus spp.	217	12.20	3.60 (0.16)	3.1	2.0 - 4.6
Salix spp.	593	13.70	4.20 (0.10)	3.7	2.4 - 6.1
		Volume (m^3)			
S. sempervirens	50	37.4	0.83 (0.27)	0.16	0.05 - 0.43
P. menziesii	60	28.1	2.07 (0.55)	0.44	0.06 - 2.90
U. californica	737	6.1	0.38 (0.03)	0.12	0.05 - 0.41
A. rhombifolia	1242	4.7	0.24 (0.01)	0.06	0.03 - 0.20
Quercus spp.	217	4.7	0.23 (0.03)	0.09	0.04 - 0.27
Salix spp.	593	2.2	0.12 (0.01)	0.06	0.03 - 0.13

Table 3. Proportion of pools within a reach formed or influenced by LWD. Pools formed by debris jams, logs, and downed rootwads sum to form "in-channel LWD." Adding pools formed by the rootwads of trees rooted on the bank results in the proportion of pools formed by "LWD." The columns for "LWD influence" provide additional information: the degree to which LWD influences pools which are not classified as LWD pools and the degree to which LWD pools are influenced by other features. The column "LWD is primary" indicates pools classified as LWD pools that had no other feature (e.g. a boulder) contributing to the pool. The column "some influence LWD" adds to the "LWD primary" column those pools which are formed by other features, in addition to LWD.

					Proportion of All Pools Formed By:					**LWD Influence**:	
Creek Name	Survey Length (m)	Stream Order	Drainage Area (ha)	BF Width (m)	Debris Jam	Log and Downed Rootwad	**In-channel LWD**	Rootwad	**LWD**	LWD is primary	Some influence LWD
McChristian (hardwoods)	300	1	312	4.6	0.60	0.40	**1.00**	0.00	**1.00**	0.80	1.00
Wine	100	1	327	3.5	0.33	0.33	**0.67**	0.33	**1.00**	1.00	1.00
Grape	100	1	341	3.5	0.71	0.14	**0.86**	0.00	**0.86**	0.86	0.86
Lower San Leandro Creek	400	2	903	4.8	0.07	0.43	**0.50**	0.21	**0.71**	0.71	0.86
McChristian (Mixed evergreen)	300	1	526	5..5	0.27	0.45	**0.73**	0.09	**0.82**	0.73	0.82

Creek Name	Survey Length (m)	Stream Order	Drainage Area (ha)	BF Width (m)	Debris Jam	Log and Downed Rootwad	**In-channel LWD**	Rootwad	**LWD**	LWD is primary	Some influence LWD
Upper Olema	2800	2	2400	7.6	0.26	0.21	**0.47**	0.14	**0.61**	0.60	0.72
Devil's Gulch	400	1	629	5.6	0.15	0.15	**0.31**	0.23	**0.54**	0.54	0.69
Porter	200	1	2128	10	0.00	0.00	**0.00**	0.67	**0.67**	0.67	0.67
San Leandro	500	1	505	4	0.39	0.06	**0.44**	0.11	**0.56**	0.50	0.67
Lower Olema	1200	2	4200	9.3	0.27	0.06	**0.33**	0.24	**0.57**	0.57	0.63
Morrison	400	1	212	3.4	0.00	0.00	**0.00**	0.13	**0.13**	0.13	0.63
Mark West (upper)	300	2	4667	15	0.13	0.00	**0.13**	0.50	**0.63**	0.63	0.63
Wildcat Canyon 1	3300	2	800	4.6	0.22	0.01	**0.23**	0.21	**0.45**	0.44	0.61
Upper Wildcat	1500	1	494	4.2	0.14	0.16	**0.30**	0.14	**0.45**	0.34	0.59
Crane	300	2	422	8	0.14	0.00	**0.14**	0.14	**0.29**	0.14	0.57
Redwood Creek	700	1	461	3	0.27	0.08	**0.35**	0.06	**0.41**	0.41	0.55
Frank's Canyon	400	1	582	5	0.00	0.20	**0.20**	0.00	**0.20**	0.17	0.50
Wildcat Canyon 2	3550	2	1780	5.7	0.07	0.05	**0.12**	0.07	**0.19**	0.17	0.42

Creek Name	Survey Length (m)	Stream Order	Drainage Area (ha)	BF Width (m)	Debris Jam	Log and Downed Rootwad	**In-channel LWD**	Rootwad	**LWD**	LWD is primary	Some influence LWD
McNab	300	2	3136	6	0.00	0.17	**0.17**	0.17	**0.33**	0.33	0.42
Parsons	900	2	2028	7	0.00	0.00	**0.00**	0.13	**0.13**	0.17	0.42
Crane	200	1	395	6.5	0.10	0.10	**0.20**	0.10	**0.30**	0.30	0.30
Rockwell	400	1	722	5	0.00	0.07	**0.07**	0.07	**0.14**	0.07	0.14
Ingalls	200	1	591	4.9	0.00	0.00	**0.00**	0.00	**0.00**	0.00	0.00
Feliz Middle	200	2	1137	7.5	0.00	0.00	**0.00**	0.00	**0.00**	0.00	0.00

Table 4. Proportion of pools formed by LWD. The CDFG samples exclude reaches with bedrock and boulder-dominated channel beds. "LWD overall" includes pools formed by the rootwads of living trees rooted at the bank. The High LWD private reaches were selected from among CDFG hardwood reaches that had a high proportion of their pools (>40%) formed by LWD.

		In-channel LWD (logs and jams)			**LWD overall**		
	n	**Mean (SE)**	**Median**	**Interquartile**	**Mean (SE)**	**Median**	**Interquartile**
Public	10	0.31 (0.05)	0.32	0.20 - 0.45	0.46 (0.06)	0.49	0.33 - 0.58
Private	5	0.07 (0.05)	0	0 - 0.18	0.13 (0.06)	0.13	0 - 0.26
High LWD private	6	0.48 (0.17)	0.43	0.10 - 0.89	0.68 (0.13)	0.76	0.29 - 1.0
CDFG Hardwood	98	0.08 (0.01)	0.04	0 - 0.11	0.25 (0.02)	0.20	0.09 - 0.40
CDFG Conifer	20	0.13 (0.02)	0.11	0.06 - 0.11	0.38 (0.05)	0.34	0.15 - 0.63

Figure 1. Streams in the Russian River basin with habitat-typing surveys.

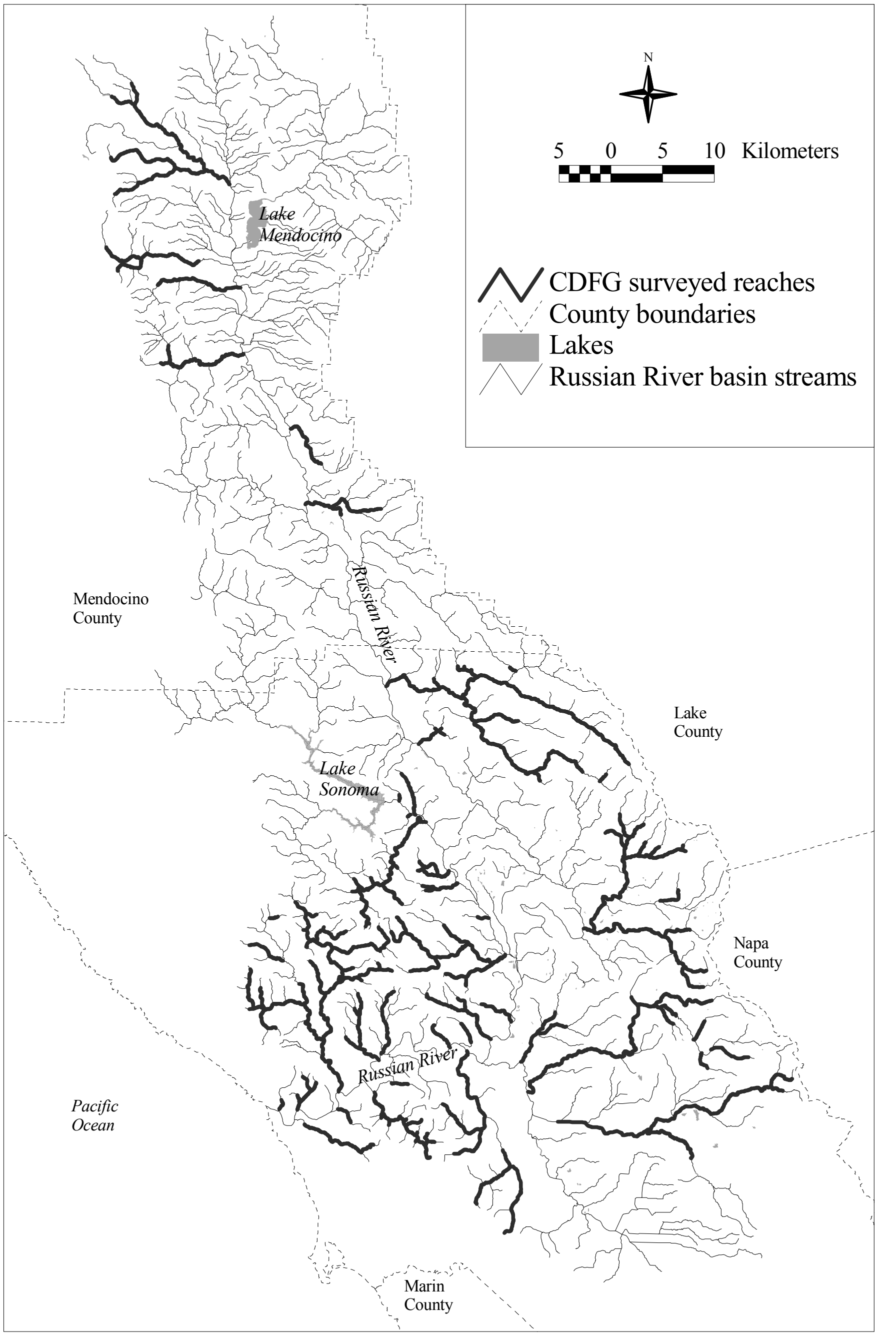


Figure 2a. Streams in the East Bay, California that were surveyed for LWD.

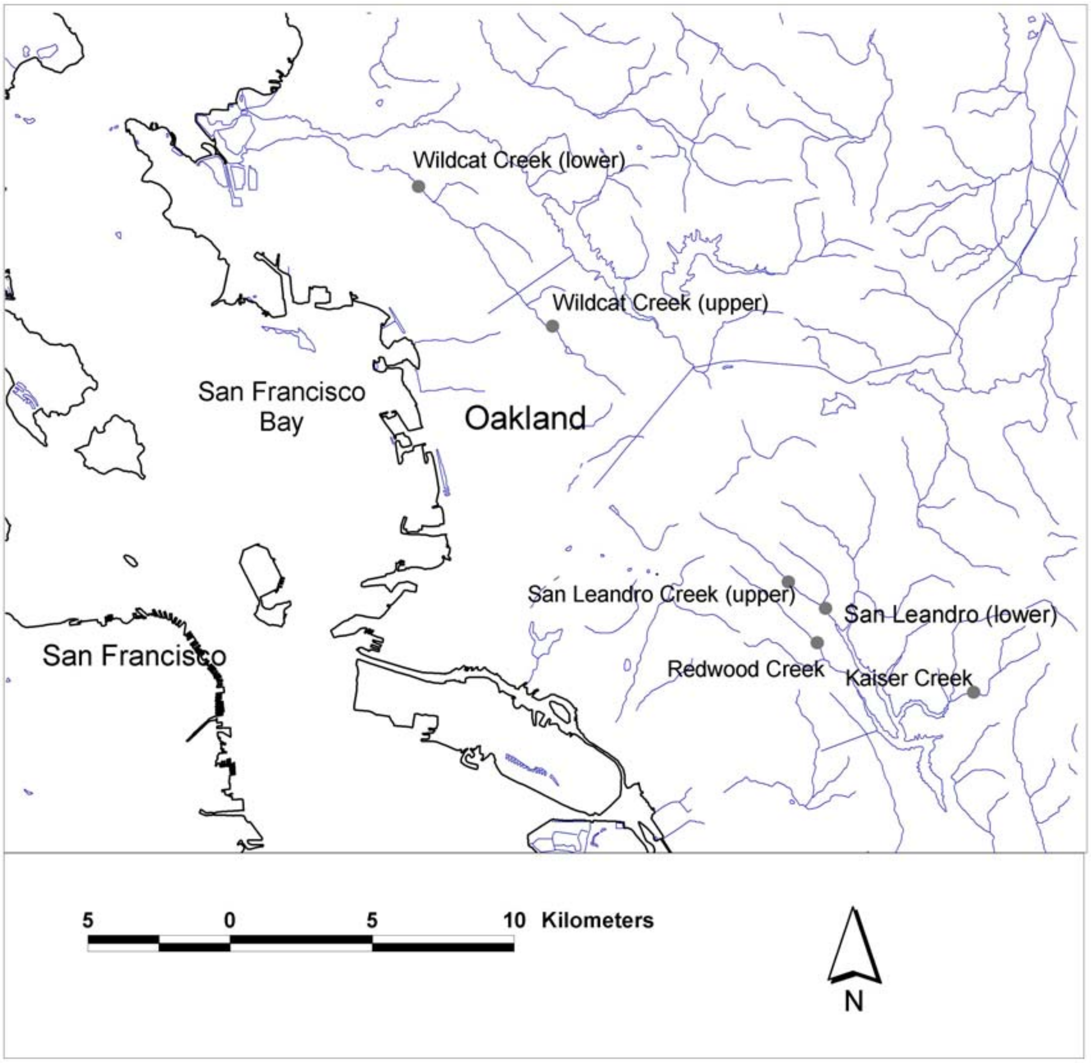


Figure 2b. Streams in the Marin County, California that were surveyed for LWD.

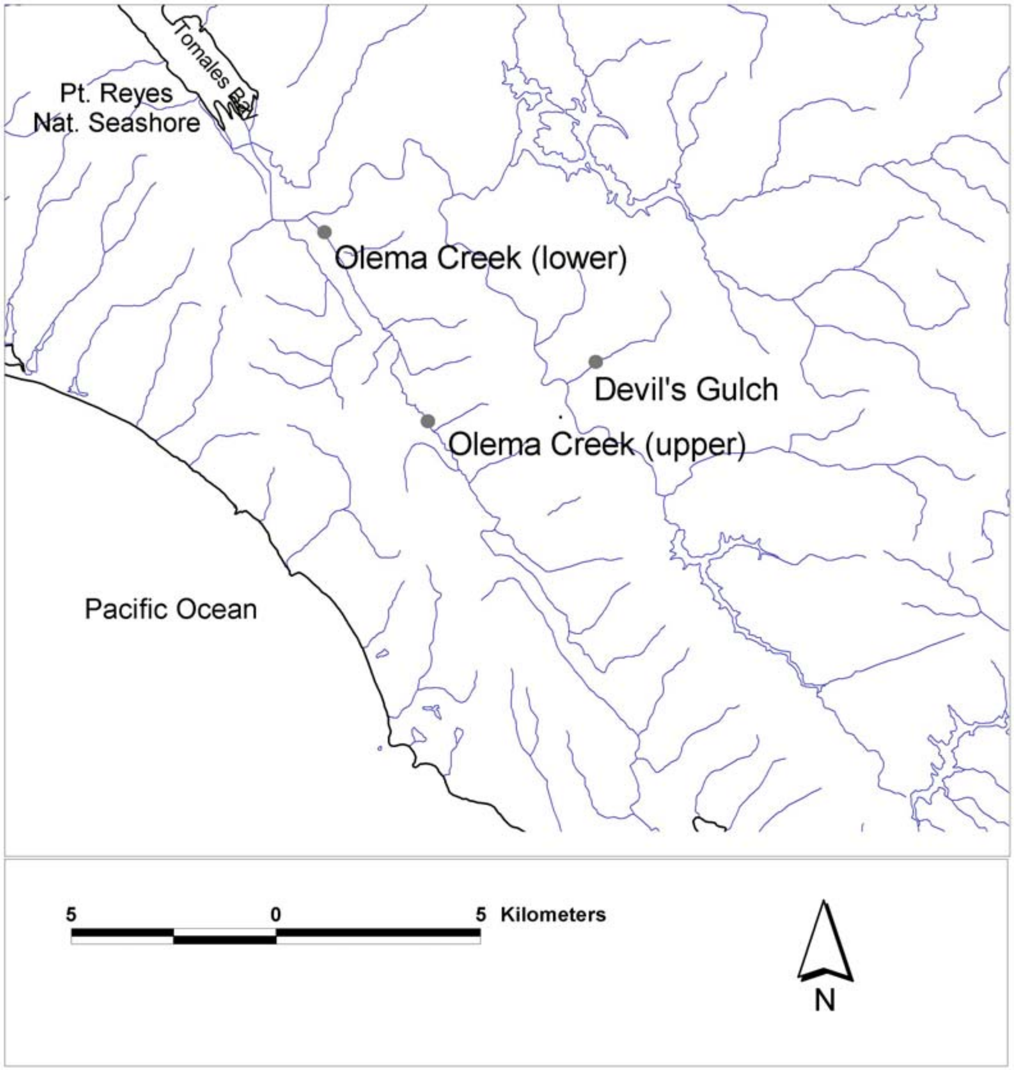


Figure 2c. Streams in the Russian River basin that were surveyed for LWD.

Figure 2d. Streams in the Sonoma County, California that were surveyed for LWD.

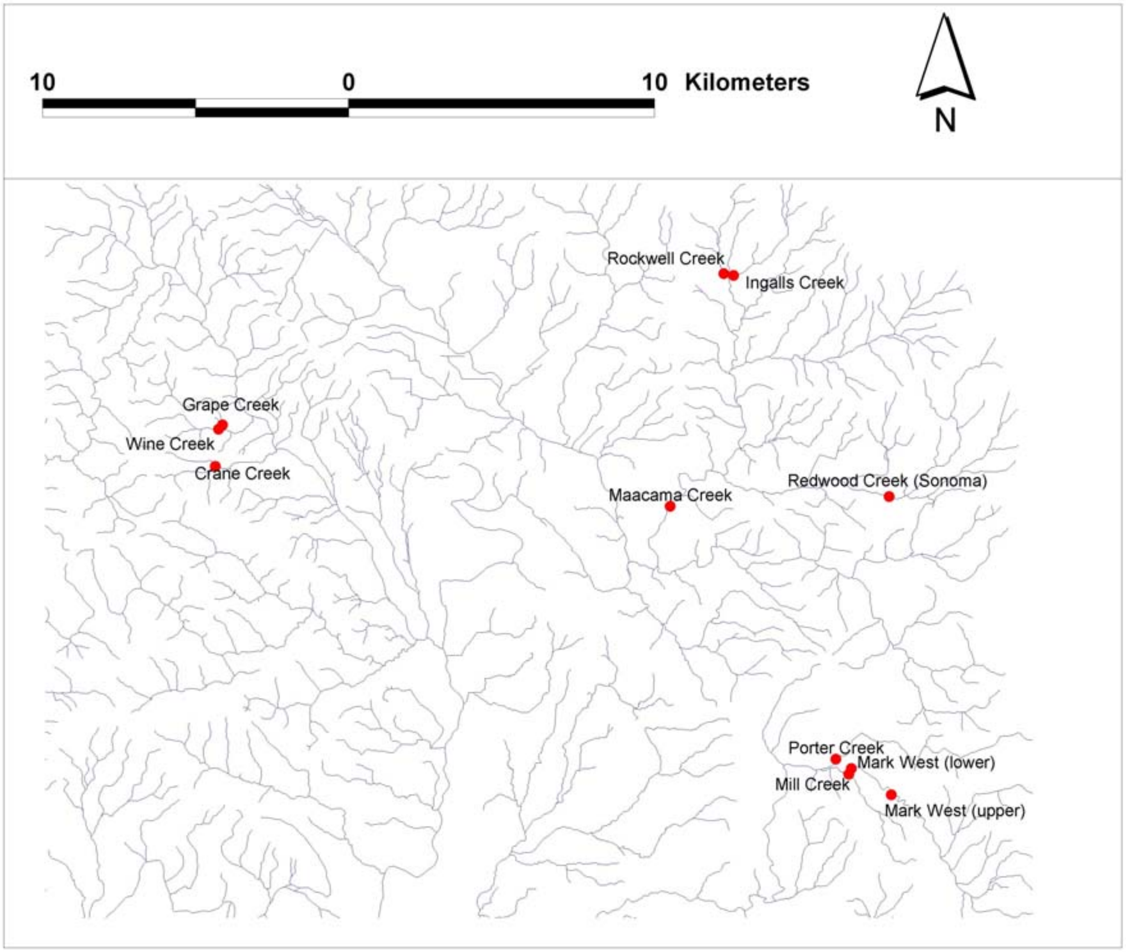


Figure 2e. Streams in the Mendocino County, California that were surveyed for LWD.

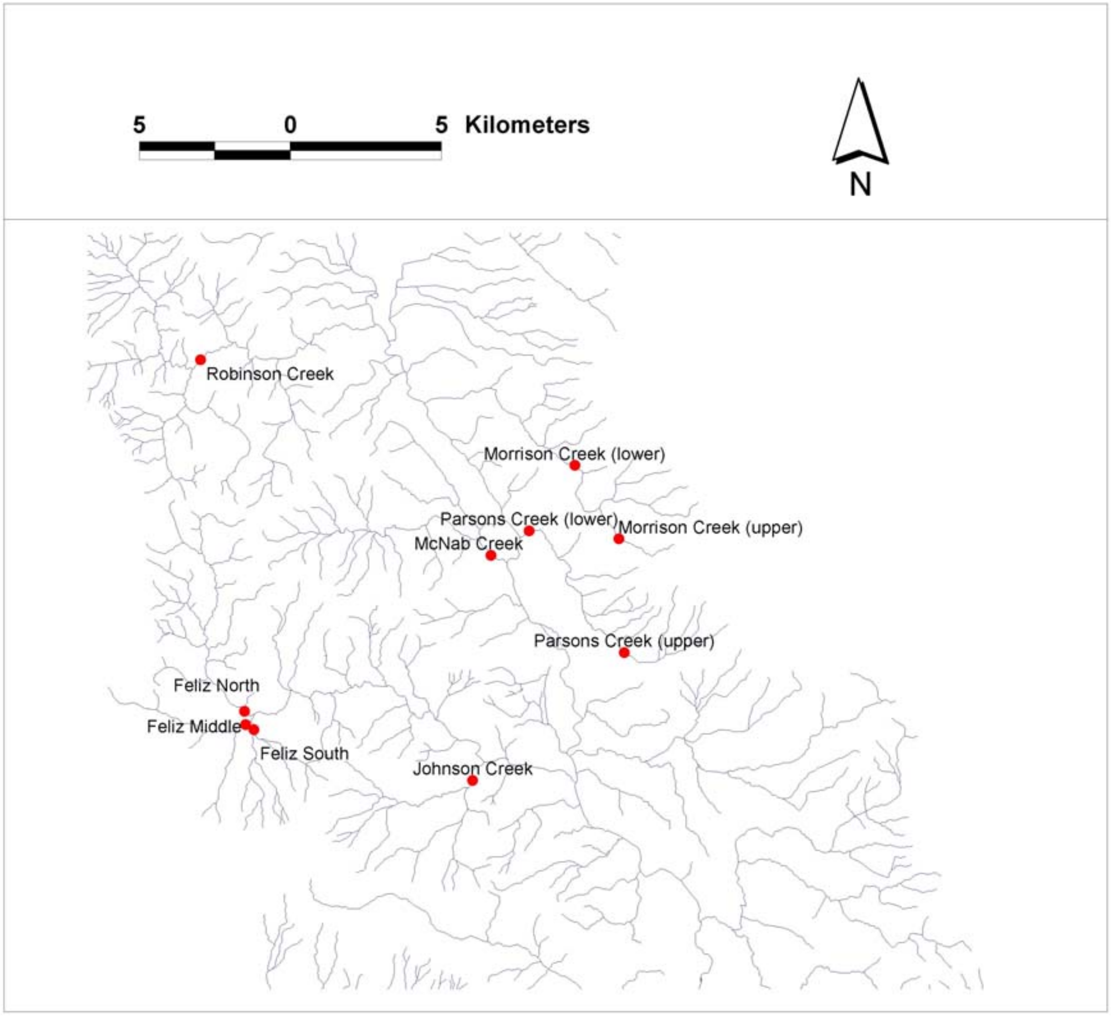
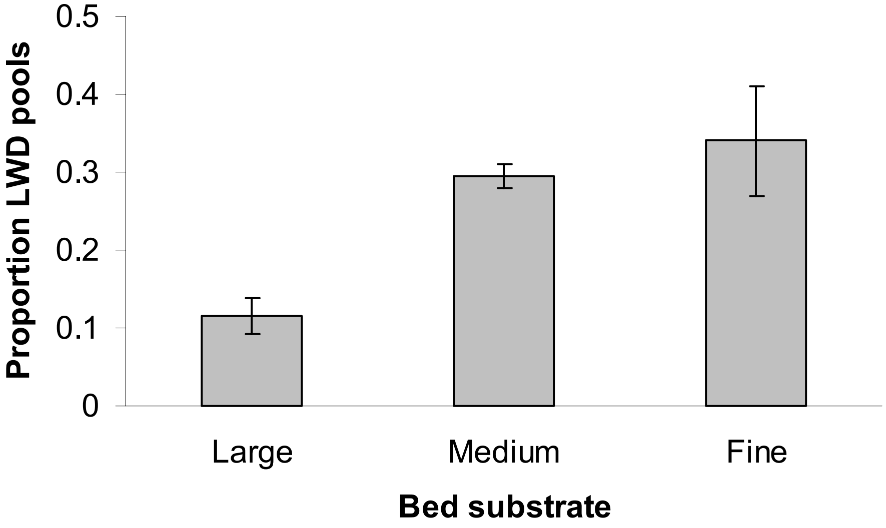


Figure 3. Proportion of pools caused by instream LWD or rootwads from living trees based on the Rosgen channel-type numeric value (Large = Rosgen numeric 1 and 2; Medium = Rosgen numeric 3 and 4; Fine = Rosgen numeric 5 and 6; n = 250 reaches from CDFG with at least 10 pools; $F_{2, 247} = 22.2$, $p < 0.0001$).



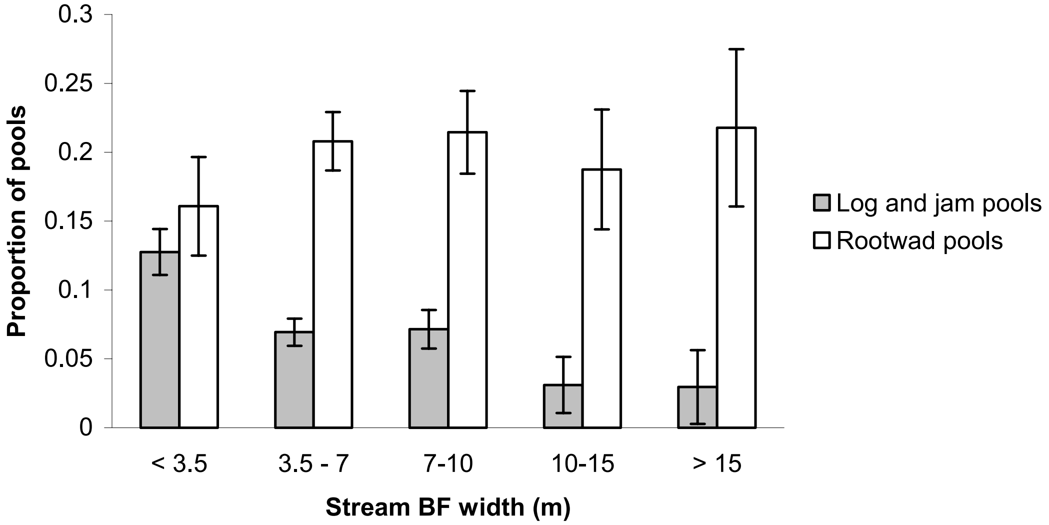


Figure 4a. Average proportion of pools within a reach formed by logs and jams (i.e. in-channel LWD; shaded bars) and by rootwads (open bars) (mean ± standard error) based on the bankfull width of the reach. **Figure 4b.** Average proportion of pools formed by in-channel LWD within reaches with riparian corridors dominated by conifers and hardwoods, based on bankfull width (mean ± standard error). The two largest classes of bankfull width were omitted because of low sample sizes.

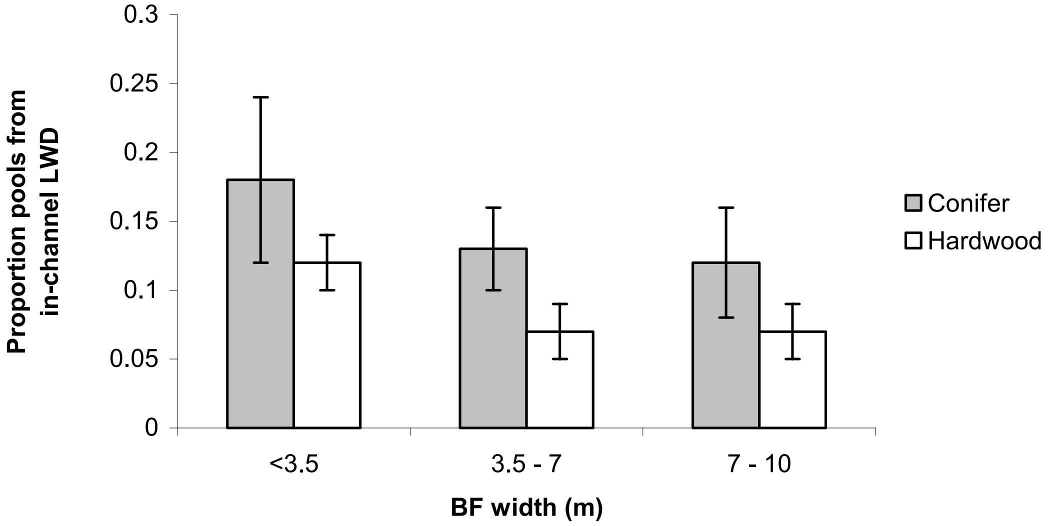


Figure 5. Proportion of pools formed by downed LWD (logs and jams) and by all LWD combined (logs, jams, and rootwads) within reaches with riparian corridors dominated by conifers, hardwoods, and mixed evergreen (controlling for substrate size of bank and bed, and bankfull width; n = 250 reaches from CDFG with at least 10 pools; log and jam model $r^2 = 0.22$; $p < 0.0001$; LWD model $r^2 = 0.28$; $p < 0.0001$).

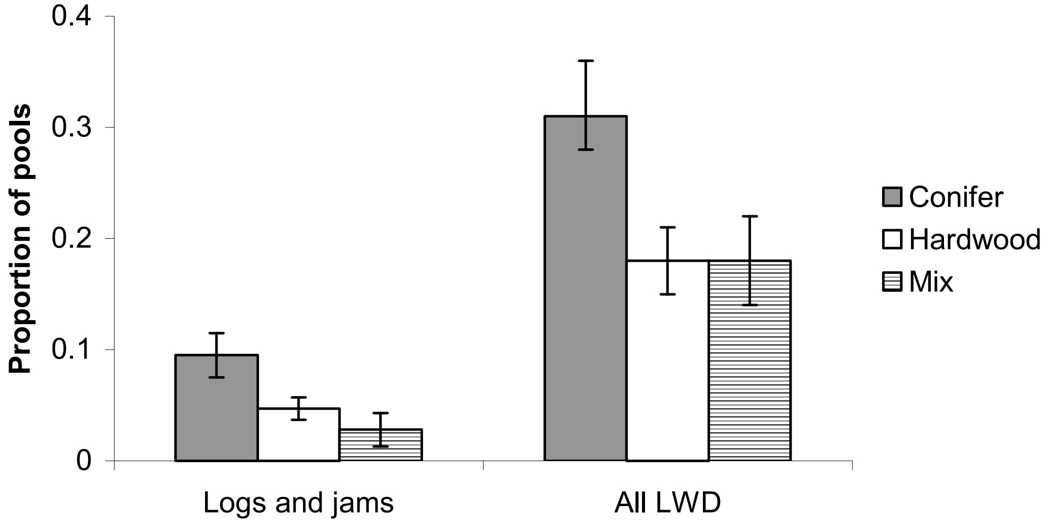


Figure 6. Average proportion of tree species' frequency and volume within the LWD of hardwood reaches.

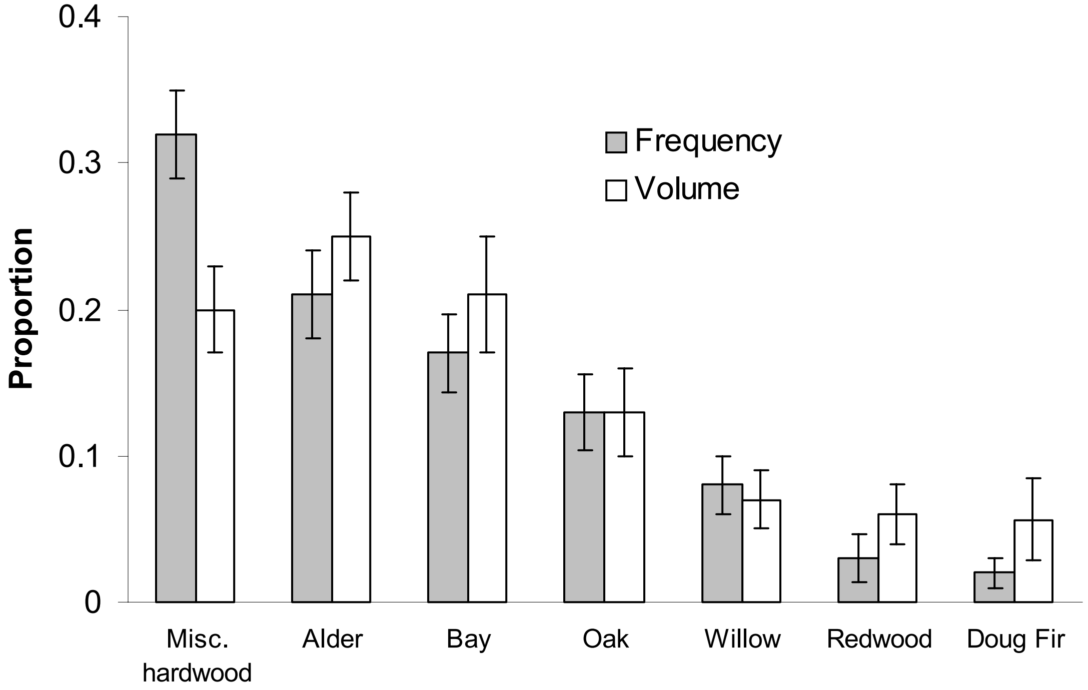


Figure 7. Proportion of streams in which tree species were present within the riparian corridor (shaded bars) and LWD (open bars).

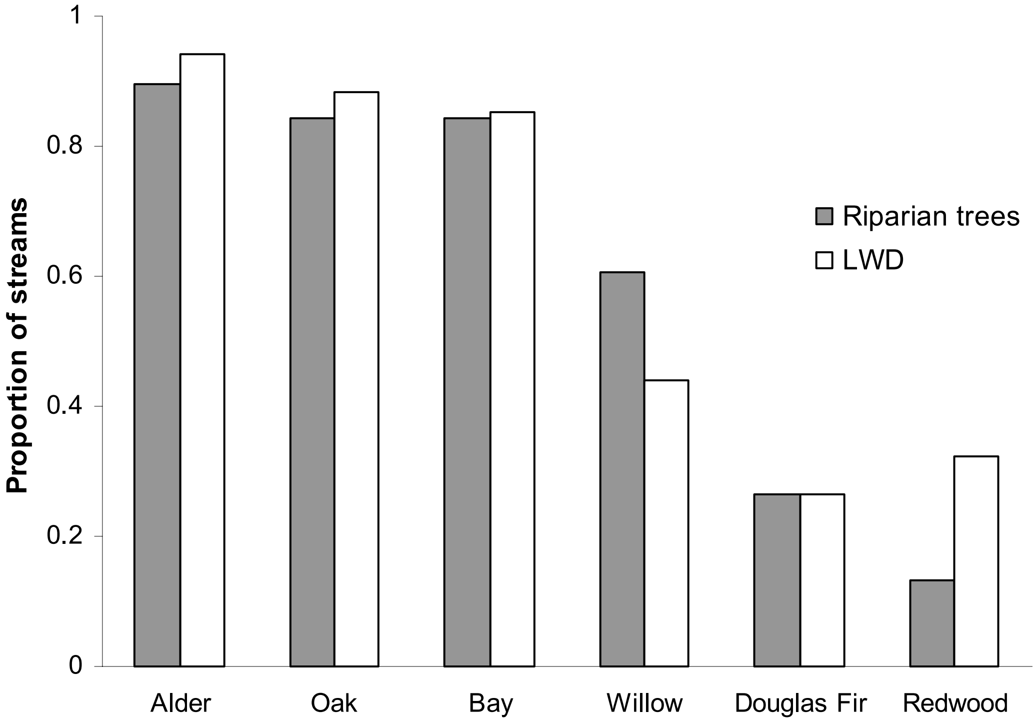


Figure 8. Average proportion of tree species' frequency (shaded bars) and basal area (open bars) within the riparian corridors of hardwood reaches.

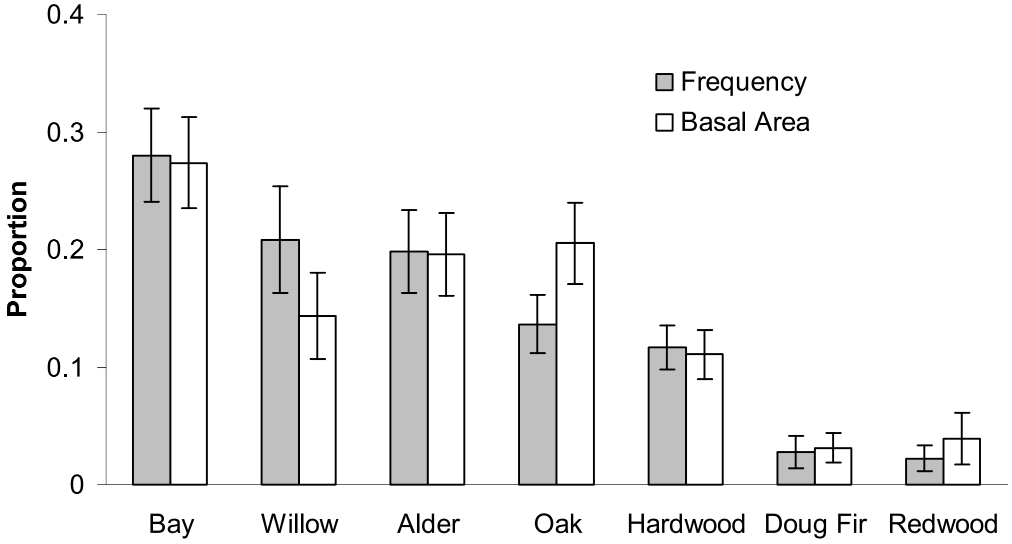


Figure 9. Distribution of recruitment distances for hardwood and conifer LWD that could be identified.

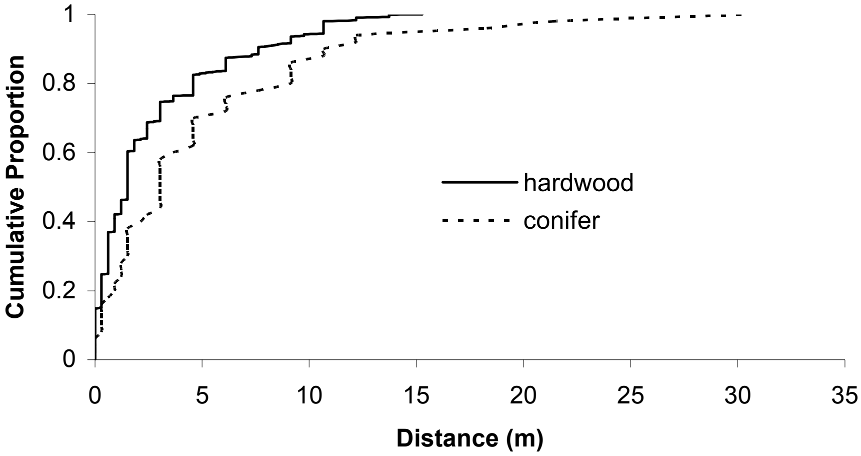


Figure 10. Distribution of pool types. LSBR = lateral-scour bedrock; MEA = meander; LSB = lateral-scour boulder; MCP = mid-channel pool; PPB = plunge-pool boulder; LSR = lateral-scour rootwad (living tree on bank); LSJ = lateral-scour jam; PPJ = plunge-pool jam; LSL = lateral-scour log; LSDR = lateral-scour downed rootwad; PPL = plunge-pool log. (only shows pool types with frequency > 2%).

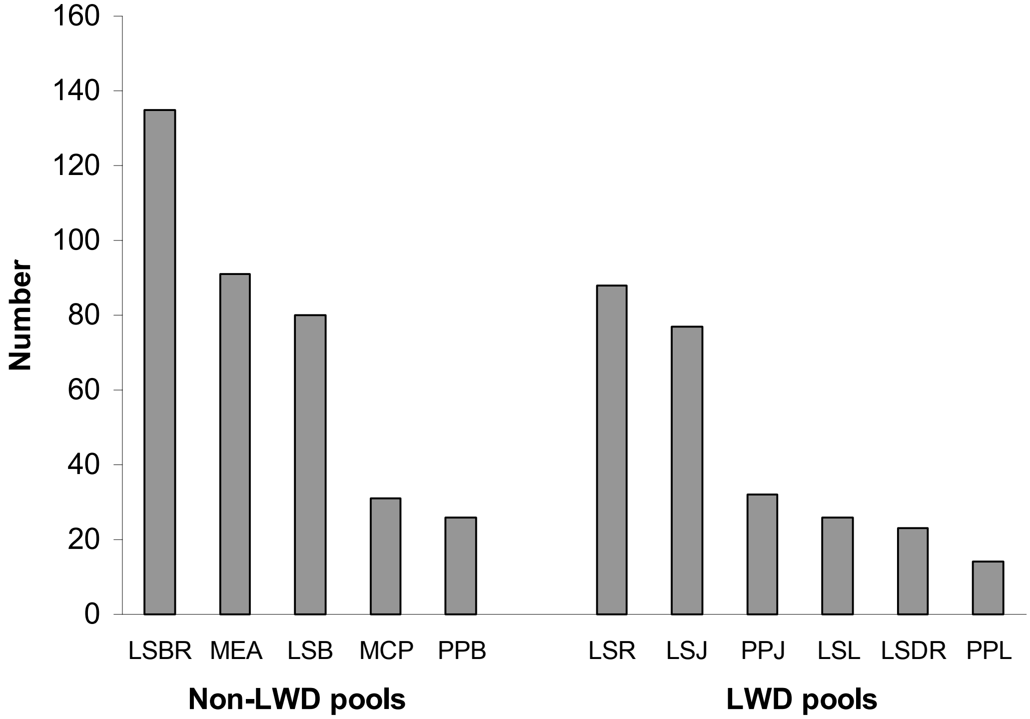


Figure 11. Among the pools formed by wood, pools were formed by debris jams, the rootwads of trees on the bank, and from single logs and in-channel rootwads.

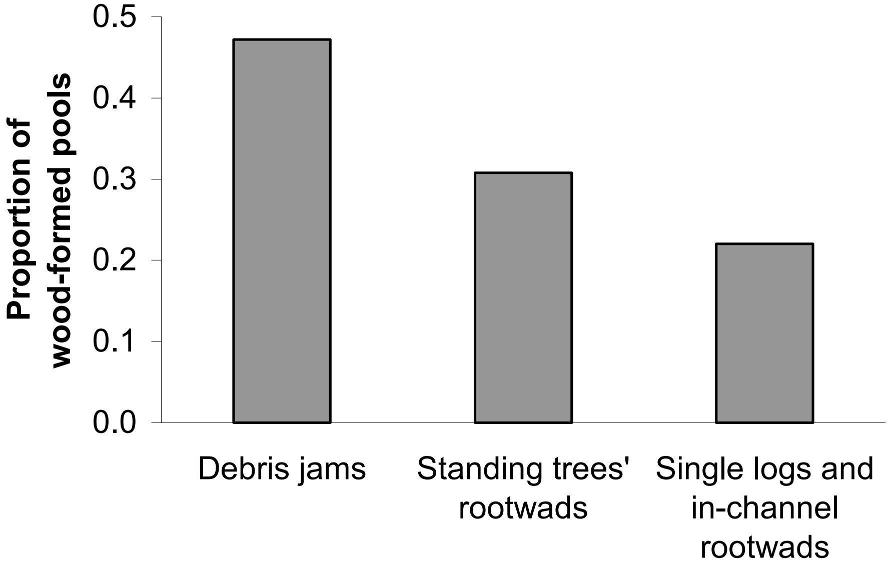


Figure 12. Distribution of pools formed by in-channel LWD. Pools formed by debris jams are in solid shades, while pools formed by single pieces of wood are represented by patterns. LSJ = lateral-scour jam; PPJ = plunge-pool jam; DAM = dammed pool; LSL = lateral-scour rootwad; LSDR = lateral-scour downed rootwad; PPL = plunge-pool log.

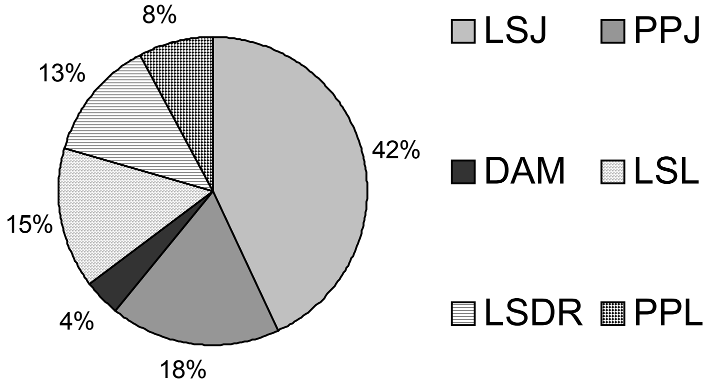


Figure 13. The proportion of pools within a reach formed by in-channel LWD and loading of LWD.

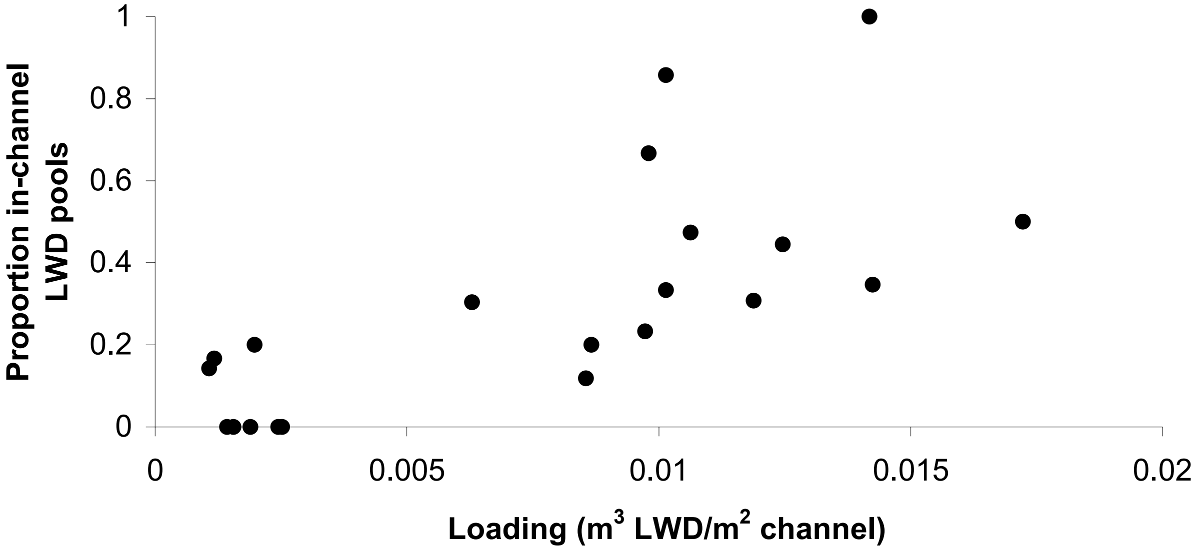


Figure 14. Pool spacing (number of bankfull channel widths per pool) was negatively correlated with LWD frequency, based on multiple regression model with pool spacing as the dependent variable with model terms LWD frequency (LWD_{freq}) and bankfull width ($BF_{w)}$ ($r^2 = 0.70$; $p < 0.0001$; n = 20). Regression equation: Pool Spacing = 8.8 – 6.1* LWD_{freq} – 0.4 *BF_w

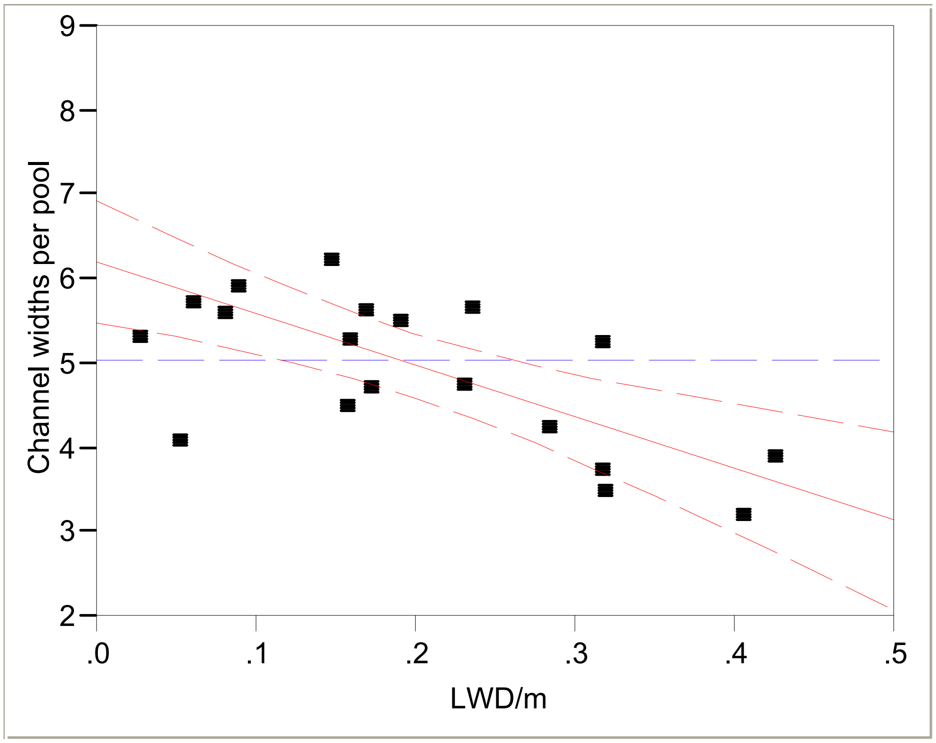


Figure 15. The average cover value of pools within a reach based on the frequency of LWD within the reach.

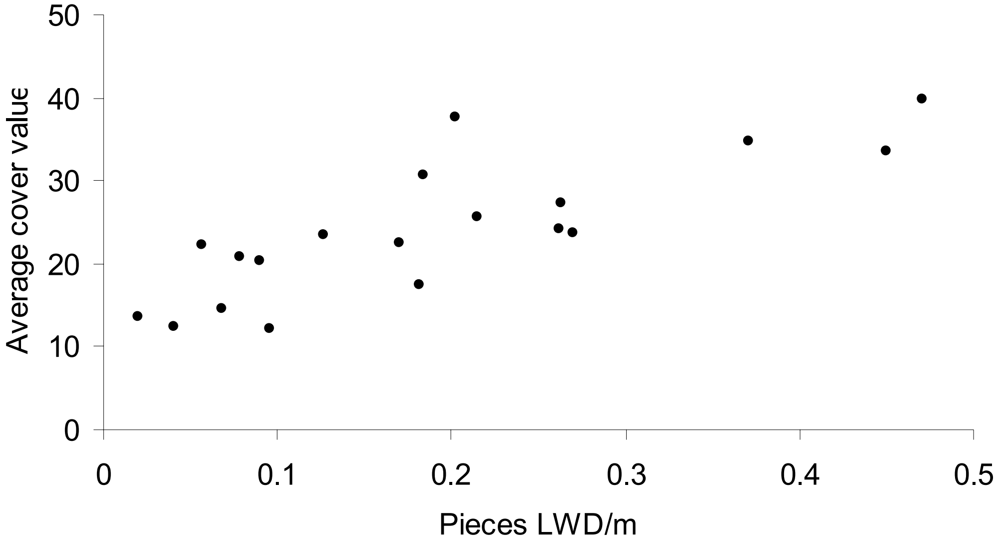


Figure 16. Average cover value of pools based on formative factor. 'Debris jam' includes plunge-pool-jam, lateral-scour-jam, and dammed pools; 'log/in-channel rootwad' includes lateral-scour-log, plunge-pool-log, and lateral-scour down rootwad; 'fluvial' include meander pools, confluence pools, and mid-channel pools; "boulder-bedrock" includes lateral-scour-boulder, lateral-scour-bedrock, and plunge-pool-boulder.

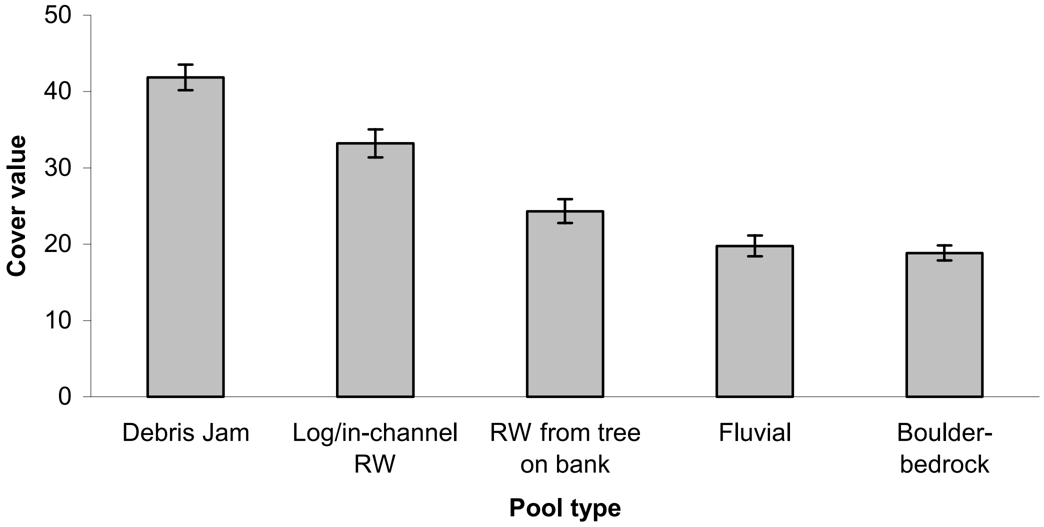


Figure 17. Comparing the proportion of pools caused by LWD as determined by CDFG and by follow-up field surveys, for 11 overlapping reaches.

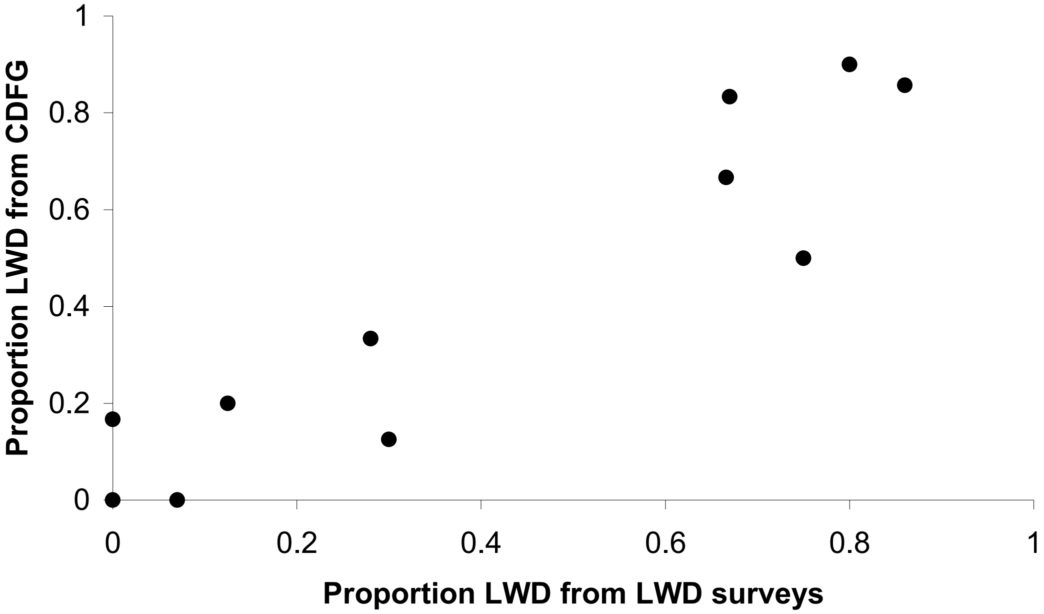


Figure 18. Type of key piece (none, live key, or dead key) for jams by channel position.

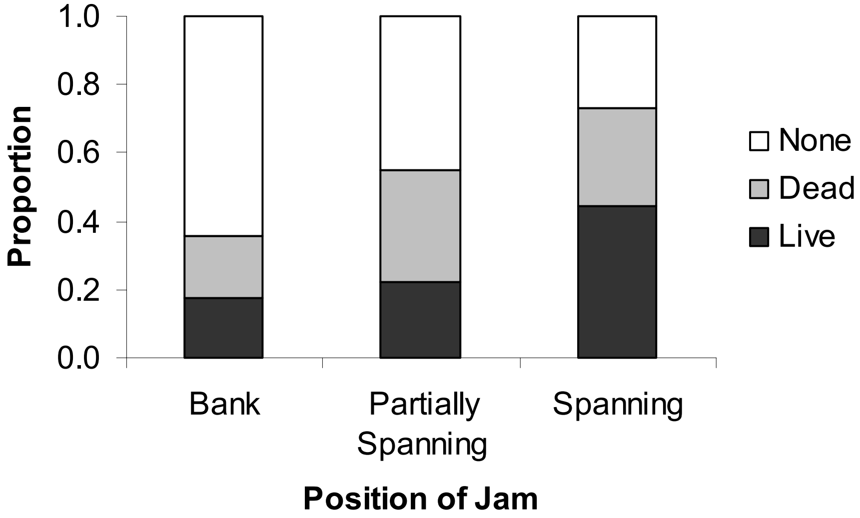


Figure 19. Size of debris jams (m^3 of LWD) based on channel position.

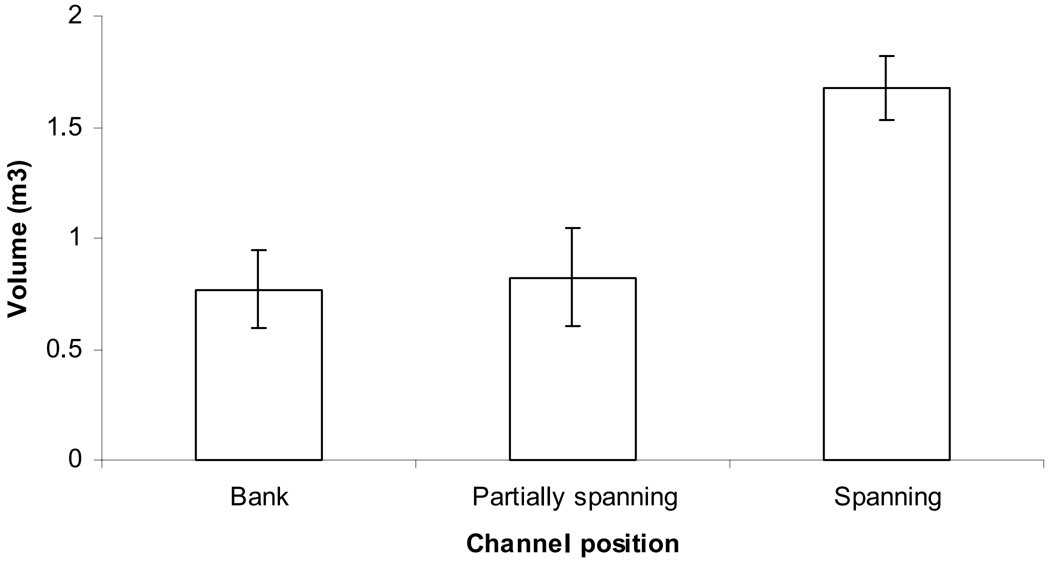


Figure 20. Proportion of debris jams forming a pool, based on channel position.

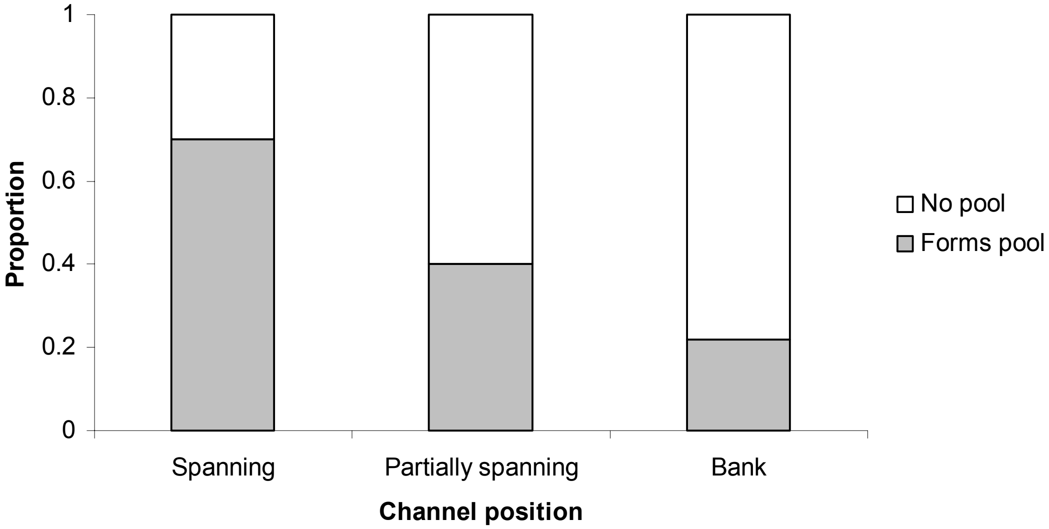


Figure 21. Species of LWD forming key pieces for debris jams and channel-spanning debris jams.

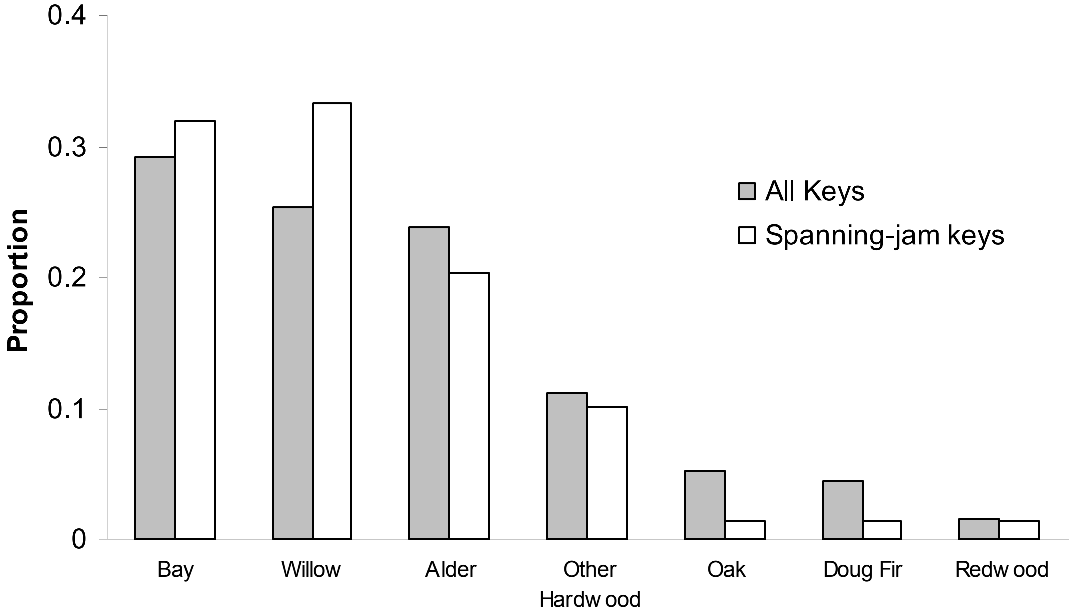


Figure 22. The frequency of all debris jams and spanning debris jams on streams flowing through private and public land.

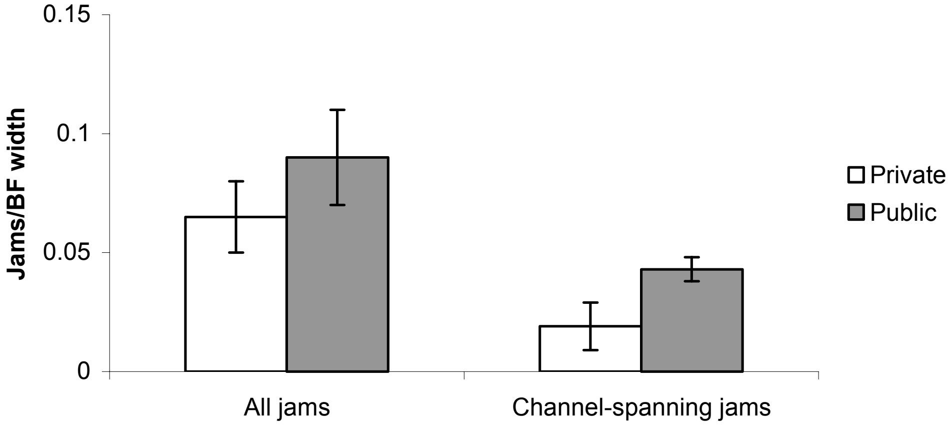
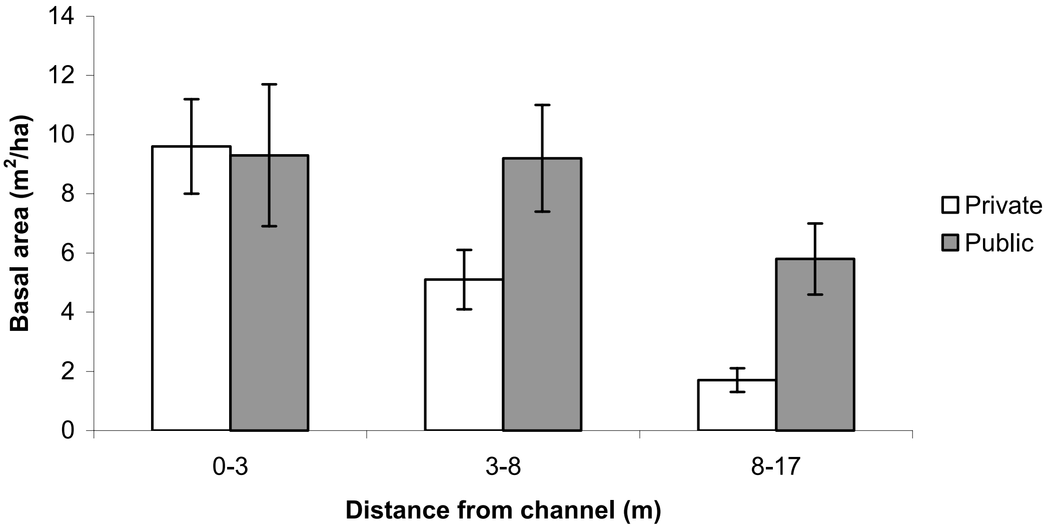


Figure 23. The basal area (m^2/ha) of trees at increasing distance from the channel within riparian corridors of streams on public and private land.



CHAPTER 3

LIVING TREES AND WOODY DEBRIS JAMS IN HARDWOOD WATERSHEDS

Living Trees and Woody Debris Jams in Hardwood Watersheds

Abstract

Large woody debris (LWD) provides critical habitat features for anadromous fish (e.g. coho salmon and steelhead trout) during their freshwater stream residency. Large woody debris can scour pools, provide cover, and store spawning gravels. Nearly all research on LWD has been conducted in the Pacific Northwest where conifers provide the primary source of LWD. In these studies, LWD from hardwood trees has generally been described as inferior to conifer LWD because of its smaller dimensions and more rapid decay rates. In parts of California, however, anadromous fish occupy watersheds that have few or no conifers. This study investigates a mechanism – "living LWD" - that may increase the effectiveness of hardwood LWD in these watersheds. Field surveys on 25 hardwood-dominated streams in Northern California (the Russian River basin and the Bay Area) found that debris jams were the primary mechanism by which LWD influenced channel morphology and provided fish habitat. A high proportion of debris jams (40 – 80%) had key pieces that were still living. I hypothesize that "living LWD" – trees that enter the channel through bank erosion but remain rooted and living – can compensate for the smaller size and faster decay rate of hardwood debris. Living LWD was capable of serving as a key piece at a smaller size than dead LWD indicating that riparian hardwoods can provide functional LWD at smaller dimensions. Jams formed behind living key pieces were significantly larger and more likely to cause a pool than jams that lacked a living key piece. By repeating these surveys 1-2 years later I found that debris jams with living key pieces were significantly more persistent than debris jams without a

living key piece: 98% of jams with a living key piece had persisted compared to 74% of debris jams without a living key piece.

Introduction

Large woody debris (LWD) can exert significant influences on stream channel morphology and aquatic ecosystems (Harmon et al. 1986). Numerous studies have documented the mechanisms through which LWD improves anadromous fish habitat, including pool formation (Montgomery et al. 1995, Beechie and Sibley 1997), cover from predators (Shirvell 1990, Harvey et al. 1999, Giannico 2000), refuge during high flows (Tschaplinkski and Hartman 1983, Murphy et al. 1986), deposition and storage of bed load and other sediments (Nakamura and Swanson 1993, Smith et al. 1993, Montgomery et al. 1996), spawning gravels (Crispin et al. 1993), organic matter (Bilby and Likens 1980, Muotka and Laasonen 2002), substrate for invertebrates (Dudley and Anderson 1982), and promotion of riparian regeneration (Abbe and Montgomery 1996).

Nearly all studies focused on the relationship between LWD and anadromous fish habitat have been conducted in the conifer-dominated forests of the Pacific Northwest. These studies have generally concluded that debris provided by riparian hardwood trees is ineffective in influencing channel morphology (see Hyde 2001) (Swanson and Lienkamper 1978, Roni et al. 2002), and conversion from hardwoods to conifer is frequently recommended as a restoration strategy in PNW forests (Bilby and Ward 1991, Beechie et al. 2000). The ineffectiveness of hardwood LWD is generally attributed to its smaller size (both length and diameter) and more rapid decay rates (Swanson and Lienkamper 1978, Cederholm et al. 1997). These factors contribute to a smaller total loading of debris and individual pieces that are less stable during high flows and, therefore, less effective in influencing channel morphology.

In California, anadromous fish utilize both coastal coniferous watersheds and hardwood-dominated watersheds, which generally are found within the Mediterranean-climate zone. Steelhead trout (*Oncorhynchus mykiss*) are the most common anadromous fish found in hardwood-dominated watersheds. Because steelhead require 1-3 years of freshwater rearing (Moyle 2002) and because of the pronounced summer drought in Mediterranean climates (Gasith and Resh 1999), deep pools with high cover values are critically important as refugia. Although LWD provides this habitat feature in other ecosystems, very little is known about the contribution of LWD to anadromous fish habitat in hardwood-dominated watersheds.

In Chapter 2, I reported that debris jams are the primary mechanism by which LWD influences channel morphology in hardwood-dominated streams. In this chapter I examine factors that influence the effectiveness and stability of debris jams and, specifically, explore the role of "Living LWD" (LLWD). I define LLWD as trees or branches that enter the stream channel, generally through bank erosion, and remain partially or fully rooted and continue living (Figure 1). Living LWD may be particularly important due to its ability to provide key pieces for debris jams (Chapter 2). I hypothesize that, because these woody elements are still living, they will have greater stability and longevity than typical dead hardwood LWD. Stability should be enhanced because the tree retains a living root system anchoring the piece to the bank. Longevity should be increased because the piece remains alive – rooted and photosynthesizing – and thus will not decay.

These properties of LLWD lead to the following specific hypotheses: 1) because of their increased stability in the channel, the minimum size for a living key piece can be smaller than that for a dead key piece, within given channel dimensions; 2) due to the increased stability of LLWD, jams with living key pieces will be more stable through time, and have greater influence on channel morphology than jams without living key pieces; 3) due to both longer persistence and the branching structure of living key pieces, which may be more effective in trapping other pieces of wood, jams with living key pieces will tend to be larger than jams without living key pieces.

Methods

I surveyed debris jams and pools in 30 stream reaches within hardwood-dominated watersheds in the San Francisco Bay Area and the Russian River basin (Sonoma and Mendocino counties) (see Figure 2 in Chapter 2). All the streams were historically utilized by anadromous fish, primarily steelhead trout (*Oncorhynchus mykiss*) with some streams supporting coho salmon (*Oncorhynchus kisutch*). Most of the streams continue to support at least remnant populations of anadromous fish while six of the streams are blocked by dams or other impassable barriers and only support resident rainbow trout.

Length of surveys varied, depending primarily on landowner access; long surveys (> 1 km) were possible only within publicly owned watersheds. Debris jams were defined as aggregations of wood containing at least three pieces of LWD, although most debris jams were larger (median = 9 pieces, mean = 12.0 ± 0.6, interquartile range was 6-15). LWD was defined as any piece of wood > 1 m in length and 10 cm in diameter within the bankfull dimensions of the channel. For all pieces within a jam I measured the length and diameter, identified the species where possible (otherwise I classified the wood as either hardwood or conifer), and characterized the state of decay from a scale of 0 (still living) to 5 (highly decayed) (see Appendix II). A living tree was only categorized as LWD if it had actually entered the channel through undercutting, bank erosion, or hillslope failures, with the primary trunk oriented parallel to the channel bed and within the bankfull dimensions of the channel. Thus the trunks and roots of living, standing trees rooted at bankfull or within the channel were not categorized as LWD and their contributions to channel form or debris jam stability were noted separately.

I assessed whether a jam had a key piece or pieces and, if so, measured dimensions, categorized channel position (Appendix II) and, where possible, recorded its source and recruitment distance. I defined the key as the piece or pieces primarily responsible for trapping the other wood and/or providing the greatest stability to the jam. I recorded whether or not the key piece had major branching, defined as the primary piece having any branch with dimensions such that if it was independent it would meet the criteria for LWD.

Jams were further categorized according to channel position: 1) *bank* jams were confined to the channel margins; 2) *partially spanning* jams extended at least half-way across the wetted channel; and 3) *spanning* jams extended across the entire wetted channel with pieces touching both banks (Figure 2). Additionally, I recorded the following binary (yes/no) information: 1) creates a step in the profile (i.e. at low flow the water surface was stepped); 2) creates a pool; and 3) is stabilized by standing trees. I recorded the species and diameter-at-breast-height (dbh) of all standing trees that contributed to the stability of the jam.

I investigated persistence of debris jams by resurveying three reaches: I initially surveyed Redwood Creek (0.7 km) and upper Wildcat Creek (1.5 km) in November 2000 and resurveyed them in December 2002; lower Wildcat Creek (6.6 km) was initially surveyed in July and August 2001 and resurveyed in August 2002. Upper and lower reaches of Wildcat Creek are separated by a dam and reservoir. To facilitate additional monitoring of lower Wildcat Creek, survey endpoints and several intermediate locations were recorded using a GPS, red flagging was hung every 100 m, and several intermediate distances were marked with large numbers in orange paint on riparian trees.

Results

Within individual streams, the proportion of spanning jams with living keys averaged 44%, ranging from 0 - 100%, but this result includes several streams with only a single spanning jam. For the five streams that had at least 8 spanning jams, the percentage with live keys averaged 53%, ranging from 41 – 67% (Table 1). Several subsequent analyses focus on these five streams with a high sample size of spanning jams.

Willow species, primarily Red willow (*Salix laevigata*), were the most common living key species and provided 50% of all live keys. However, only 9% of dead keys were from willow (Figure 3). This disproportionate representation of willows within the live keys was true for 4 of the 5 streams with multiple spanning jams: Wildcat Creek (willow representing 19% of dead keys, 66% of living), San Leandro (0% dead, 67% living), Redwood (0%, 43%), Grape (0%, 100%). Olema Creek was the exception with willow composing 13% of dead keys and 15% of living keys; in Olema 54% of the living keys were alder and 23% were California bay. Across all streams, California Bay

(*Ubellularia californica*; 25%) and White alder (*Alnus rhombifolia*; 20%) were the most common living keys after willow. White alder and California bay were the most common species providing dead keys (Figure 3).

For jams with a single key, the average diameter of live keys for spanning jams (27.0 ± 3.7 cm) was significantly smaller than that for non-living keys (39.0 ± 3.8 cm; F 3, 62 = 4.0 p = 0.02), controlling for drainage area and whether or not the jam was stabilized by standing trees. To ascertain the minimum diameter for a channel-spanning jam's key piece I considered the five streams with multiple channel-spanning jams of each type (at least three living and three dead channel-spanning keys). In all five streams the minimum diameter for a living key was smaller than the minimum diameter for a dead key (paired t = 5.2, df = 4, p = 0.007). Additionally, the mean diameter of live keys was significantly smaller than dead keys in all five of these streams (paired t = 3.8, df = 4, p = 0.02) (Figure 4).

In all five of the streams with multiple spanning jams, the largest spanning jam (by volume of wood) had a living key (paired t = 2.0, df = 4, p = 0.11). Additionally, in all five streams the mean size of spanning jams was significantly greater for jams with live keys than without (paired t = 4.6, df = 4, p = 0.01; Figure 5).

Partially and fully spanning jams with live and dead keys were equally likely to cause a step in the profile. However, among spanning and partially spanning jams, those with a live key were more likely to cause a pool (71%) than those without a live key (55%) (X^2 = 4.9; df = 1, 195, p value = 0.03).

Live keys were also more likely to have major branching (61%) than dead keys (32%) (X^2 = 4.9; df = 1, 113, p value = 0.002), and, controlling for drainage area, jams with branched keys were significantly larger (i.e. stored volume) than jams with non-branched keys: (2.0 ± 0.2 compared to 0.9 ± 0.3 m^3; F 2, 112 = 8.5; p = 0.004). Jams with a live key were on average three times as large if the key piece had major branching (2.4 ± 0.4 to 0.8 ± 0.5 m^3; $F_{2, 58} = 7.7$; p = 0.05). For jams with dead keys there was no statistical difference in stored volume between branched and non-branched keys.

In all three reaches that were resurveyed, a higher proportion of spanning or partially spanning jams with living keys were still present compared to jams without living keys. In lower Wildcat Creek, 35 of the 46 (76%) jams without a live key found in 2001 were still present in 2002, while 23 out of 24 (96%) of spanning or partially spanning jams with a live key were still present in 2002 ($X^2 = 5.2$; $df = 1, 68$; $p = 0.02$). In upper Wildcat Creek all 11 jams with a live key found in 2000 were still present in 2002, while 6 out of 9 jams without a live key were still present ($X^2 = 5.5$; $df = 1, 18$; $p = 0.02$). Similarly, in Redwood Creek all 9 jams with a live key found in 2000 were still present in 2002, while 7 out of 10 jams without a live key were still present ($X^2 = 4.4$; $df = 1, 17$; $p = 0.04$); one jam without a live key that I considered still present had moved downstream 15 m. Overall, 43 out of 44 (98%) jams with a living key had persisted while 48 out of 65 (74%) jams without a live key were still present after one year (lower Wildcat) or two years (upper Wildcat and Redwood).

Discussion

Living trees strongly influence the dynamics of LWD in hardwood-dominated streams of northern California, including trees that remain living after falling into the stream and function as key pieces for debris jams. These living key pieces play a significant role creating and maintaining debris jams that influence channel morphology and provide fish habitat. Living LWD potentially increases the ability of hardwoods to influence channels by compensating for the characteristics of hardwood LWD that generally reduce its effectiveness relative to that of conifers – smaller size, diminished stability, and more rapid decay.

The ability of LWD to remain stable in a channel and, hence, to influence channel morphology, is based on a piece of wood's ability to resist two processes: transport and decay. Resistance to transport is a function of piece length relative to channel length (Lienkamper and Swanson 1987, Nakamura and Swanson 1994), diameter (Braudrick et al. 1997, Braudrick and Grant 2000), channel orientation, the presence of a rootwad (Braudrick and Grant 2000), and whether the piece is partially buried in the bed or bank (Young 1994) or stabilized by structural elements such as boulders or standing trees. Piece size relative to channel width also influences the ability of wood to induce pool formation (Beechie and Sibley 1997).

Along the Pacific coast, hardwood species are generally much smaller than conifers at maturity. For example, white alder and California bay, the dominant riparian species in the study streams, typically attain maximum heights of 35 m and 45m, respectively, whereas Douglas fir (*Pseudotsuga menziesii*) and Redwood (*Sequoiadendron sempervirens*), the primary species in California's conifer-dominated coastal streams reach maximum heights of 70 and 115 m, respectively. Further, hardwood trees generally are not composed of a single long trunk, having multiple major branches and, thus, the boles they produce are even shorter than these numbers would indicate (Hickman 1993, Stuart and Sawyer 2001). The largest alders and bays observed in the riparian corridors of these study streams had diameters between 80 and 120 cm (dbh) (Chapter 2), whereas conifer species can attain diameters of several meters. Thus, the debris produced by mature hardwoods will tend to be much smaller than that produced by mature conifers.

Resistance to decay is a function of piece size and properties of the wood itself. All else being equal, larger pieces decay more slowly than smaller ones due to their decreased surface-area-ratio and greater proportion of heartwood. Thus, simply by having greater diameter, mature conifer debris will decay less quickly than mature hardwood debris (Harmon et al. 1986). Further, the heartwoods of hardwoods contain lower concentrations of decay-resistant chemicals than do conifers (Scheffer and Cowling 1966), and several studies have reported that riparian hardwood debris decays much faster than conifer debris (Swanson and Lienkamper 1978, Cederholm et al. 1997).

Thus, across stream widths, individual pieces of wood produced by mature hardwoods will tend to be much less stable, persistent, and influential on channel form than debris produced by mature conifers. Additionally, total loading of LWD is much lower in hardwood streams (Chapter 2) because individual pieces are smaller and the total potential contributing area is narrower due to the reduced height of riparian hardwoods compared to conifers (Van Sickle and Gregory 1990). These factors – smaller, less stable individual pieces and, in aggregate, significantly lower loading – explain why riparian hardwoods have generally been dismissed as providing LWD that is inferior to conifer LWD for fish-habitat functions.

However, living LWD (LLWD) may greatly increase the functional effectiveness of LWD in California's hardwood-dominated watersheds. Living LWD produces individual pieces with greater stability and influence on channel morphology, primarily in their role as key pieces for debris jams. Because spanning debris jams store a large proportion of total LWD volume within hardwood reaches (Chapter 2), any process that promotes jam formation will tend to increase total loading by capturing a higher proportion of debris in transport. Finally, if living keys result in more persistent debris jams (as suggested by the repeat surveys) they will contribute to larger LWD loadings through time. Through these processes, LLWD may increase both the effectiveness and total loading of debris in a hardwood channel beyond that which would be expected from the average dimensions of hardwood debris pieces and the size of the potential recruitment area.

Several characteristics that have been found to enhance the stability of dead LWD give insight to the mechanisms that may promote the stability of LLWD. In general, stability of a piece of LWD is increased when one end of the wood is buried in the bed or bank, and/or the piece is attached to a rootwad (Keim et al. 2000)(Braudrick) (Young 1994). Saplings can utilize LWD as a nurse log and the root systems of these dependent saplings and trees can also increase the stability of a piece of LWD (Swanson et al. 1976). Living LWD essentially functions as a piece attached to a rootwad with one end buried in the bank. Because of the living root system (and its ability to bind soils and resist erosion), these pieces may be even more stable than a partially buried dead log.

Conclusions

Living LWD plays a significant role in creating and maintaining debris jams in hardwood-dominated watersheds. Living LWD appears to be capable of functioning as a key piece for a spanning jam at a smaller size than dead LWD. Spanning and partially spanning debris jams with a living key are more persistent through time and more likely to induce pool formation. The complex branching structure of many living keys leads to more efficient trapping of wood and, thus, larger debris jams. The presence of branches and leaves directly over the channel may also increase the input of fine organic matter and terrestrial invertebrates to the stream. These allochthonous contributions to the food web may be particularly important in isolated pools during the summer drought in Mediterranean-climate systems.

The ability of hardwood riparian species to function as living key pieces greatly enhances their influence on stream channels, beyond that which would be expected based on the size of debris that they can contribute. For example, standard models of wood stability and potential for pool formation place strong emphasis on piece diameter (Beechie et al. 2000, Braudrick and Grant 2000). In this study, I found that the minimum diameter required for a piece to function as a key for a channel-spanning debris jam was smaller for living LWD than dead LWD. Additionally, across a wide range of stream sizes, the average living key was significantly smaller than the average dead key. Living and dead key diameters had similar values for maximum sizes; the lower average of living keys was due to an extended lower distribution. Through the process of living LWD, riparian hardwoods can provide effective LWD at smaller sizes than would be predicted by conventional models.

Riparian hardwoods' flexible growth form and ability to function as living LWD greatly enhances their ability to influence stream channels. This phenomenon needs to be considered in any model predicting LWD characteristics in hardwood-dominated streams in northern California and, perhaps, elsewhere. For example, traditional models focus on the recruitment of dead material to the stream and utilize as inputs the range of sizes of wood that can be contributed from riparian trees. Equations for transport and decay are then applied to the pool of wood provided to the stream. Because living key pieces can function effectively at smaller sizes, and because they can potentially remain in the stream much longer than a decaying dead piece, models that only estimate the rate of input and persistence of dead material will underestimate the volume and effectiveness of LWD in hardwood systems.

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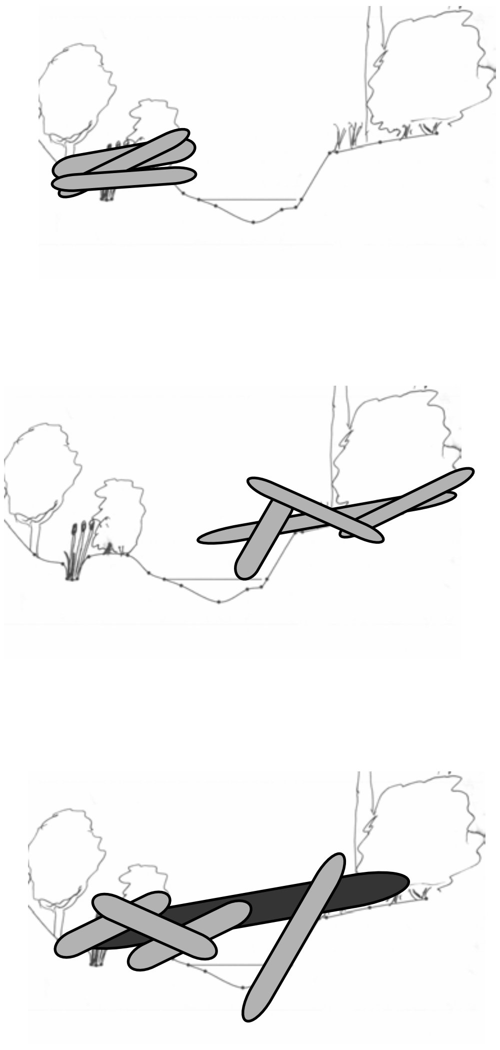
Table 1. Proportion of spanning debris jams with living key pieces.

Stream	# spanning jams	# spanning jams with living key piece	Proportion with living key piece
Wildcat	64	27	0.42
Olema	29	12	0.41
Redwood	13	7	0.54
San Leandro	9	6	0.67
Grape	8	5	0.63

Figure 1. A red willow (*Salix laevigata*) that has fallen into Wildcat Creek (Alameda County, California) but remains rooted and living (the arrow points to the lower branch of the willow which is sprouting branches with leaves). The photo is taken looking upstream. A channel-spanning debris jam has accumulated behind this willow which has contributed to the formation of the pool in the lower left portion of the picture.

Figure 2. Conceptual diagrams of debris jams' channel positions: a) a bank jam; b) a partially-spanning jam, and c) a channel-spanning jam. The black piece illustrates a *key piece*.



a.

b.

c.

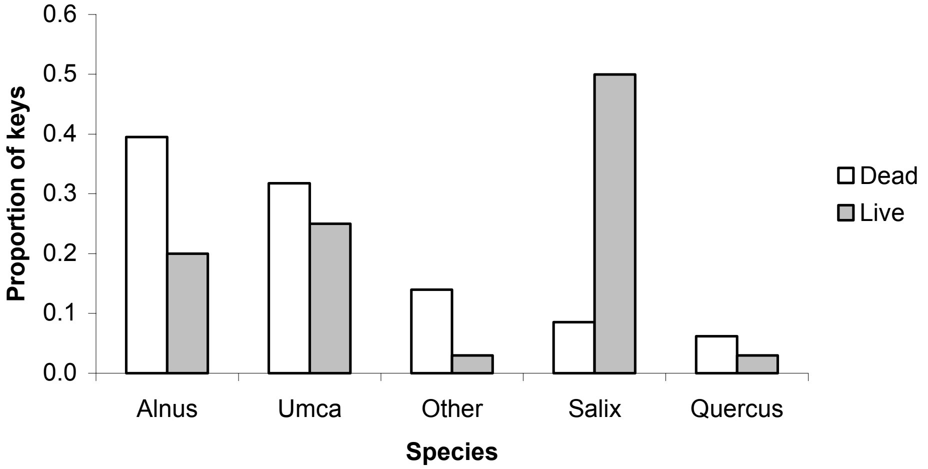


Figure 3. Species of tree providing key pieces for debris jams, open bars represent keys that were dead and shaded bars represent keys that were living.

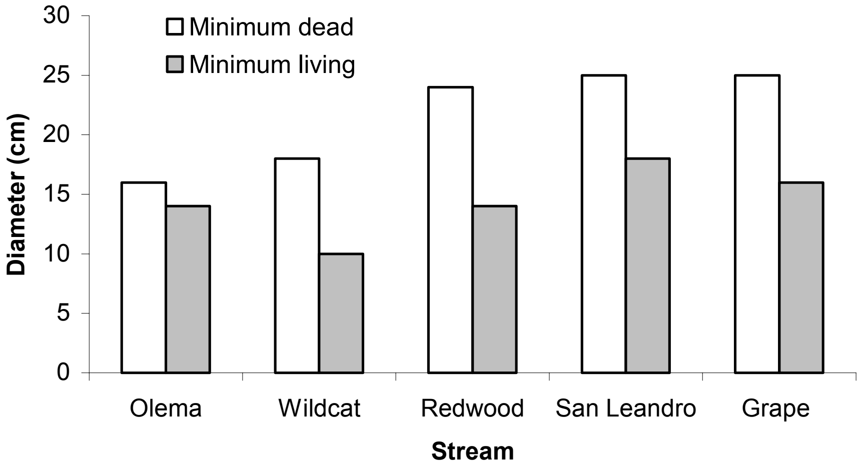
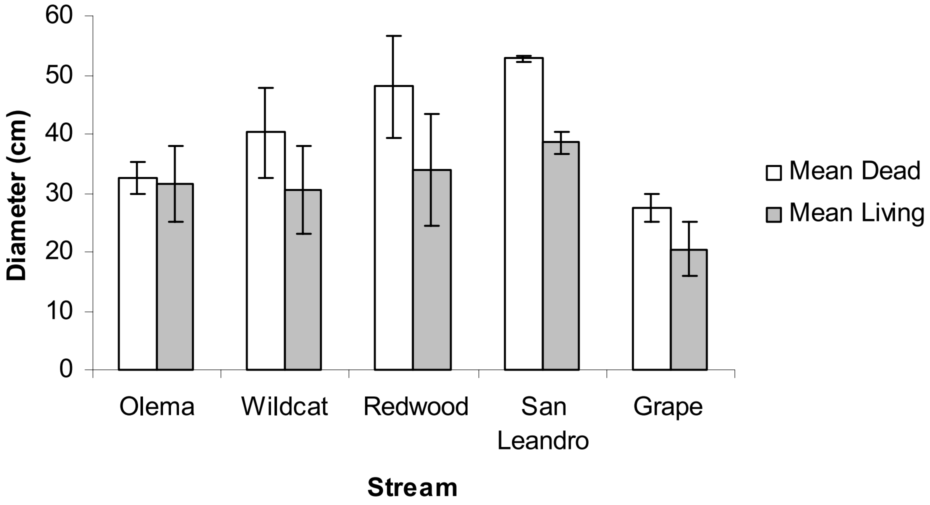


Figure 4a. Minimum diameter of key piece for single-key channel-spanning jams, with living and non-living keys, in five streams with at least 3 jams of each type.

Figure 4b. Mean diameter of key piece for single-key channel-spanning jams, with living and non-living keys, in five streams with at least 3 jams of each type.



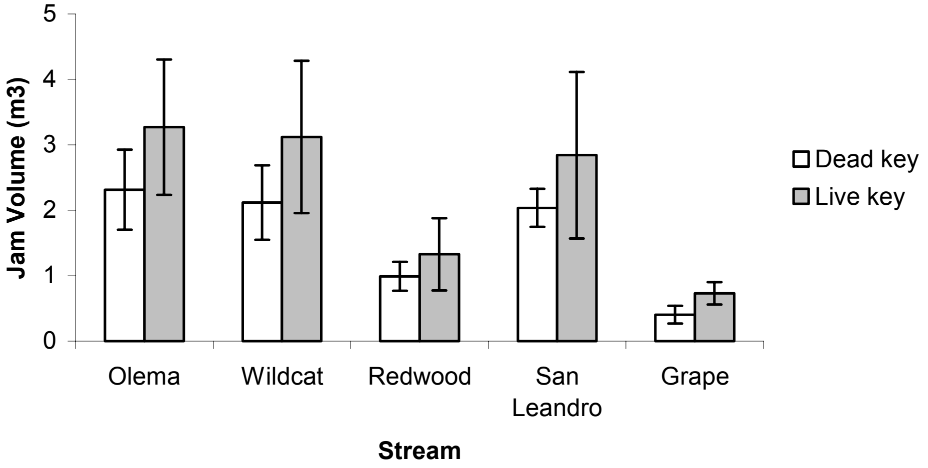
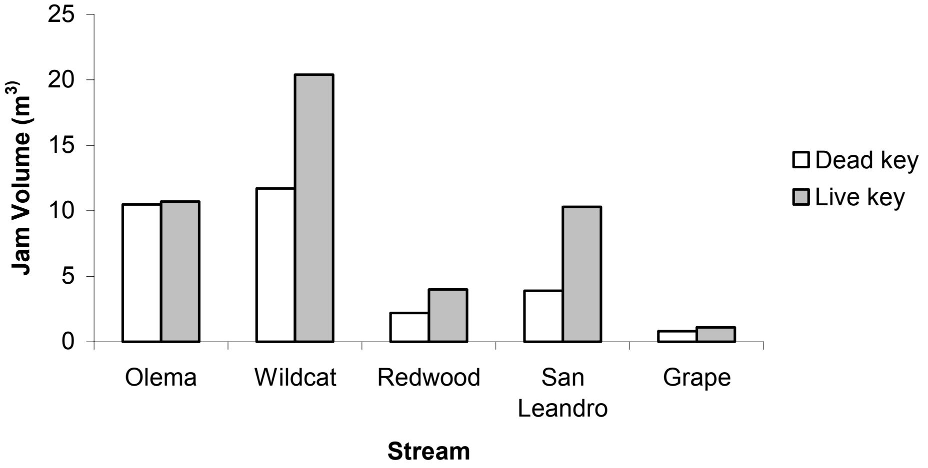


Figure 5a. Mean volume of channel-spanning jams with living and non-living key pieces. Based on streams with at least 3 spanning jams of each type.

Figure 5b. Maximum volume of channel-spanning jams with living key piece and without living key piece. Based on streams with at least 3 spanning jams of each type.



CHAPTER 4

DEER HERBIVORY AS AN ECOLOGICAL CONSTRAINT TO THE RESTORATION OF DEGRADED RIPARIAN CORRIDORS

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Abstract

Ungulate herbivory can impact riparian vegetation in several ways, such as by reducing vigor or reproductive output of mature plants, and through increased mortality of seedlings and saplings. Much work has focused on the effects of livestock grazing within riparian corridors, while few studies have addressed the influence of native ungulate herbivory on riparian vegetation. This study investigated the effect of deer herbivory on riparian regeneration along three streams with degraded riparian corridors in Mendocino County, California. I utilized existing stream restoration efforts by private landowners and natural resource agencies to compare six deer exclosures with six upstream control plots. Livestock were excluded from both exclosure and control plots. Three of the deer exclosures have been in place for 15 years, one for six years, and two for four years. The abundance and size distribution of woody riparian plant species such as Salix exigua, S. laevigata, S. lasiolepis, Alnus rhombifolia*, and* Fraxinus latifolia *were quantified for each exclosure and control plot. The mean density of saplings in deer exclosures was* $0.49 \pm 0.15/m^2$*, while the mean density of saplings in control plots was* $0.05 \pm .02/m^2$*. Within exclosures, 35% of saplings were <1 m and 65% were >1 m; within control plots, 97% of saplings were <1 m in height. The fact that little recruitment had occurred in control plots suggests that deer herbivory can substantially reduce the rate of recovery of woody riparian species within degraded riparian*

corridors. Exclusionary fencing has demonstrated promising results for riparian restoration in a region with intense deer herbivory.

Introduction

Riparian corridors are systems of high biotic, structural, and functional diversity (Gregory et al. 1991). Along smaller streams, riparian vegetation contributes much of the energy and nutrients for aquatic food webs through allocthonous inputs of leaf litter and branches. Overhanging vegetation provides shade that maintains the lower water temperatures necessary for survival of cold water fish species (Meehan et al. 1977; Vannote et al. 1980; Barton 1985). The input of larger branches and trunks creates instream structure and habitat, and this large woody debris plays a major role in channel forming processes such as pool formation and gravel bar stabilization (Abbe & Montgomery 1996). Riparian vegetation contributes to bank stability through root systems that anchor soil, and by increasing roughness to slow the velocity of high flows (Kondolf & Curry 1986).

In the western United States, the extent of riparian ecosystems has been considerably reduced, and remaining habitats are often highly degraded or fragmented by a variety of human activities (NRC 1992; Kondolf et al. 1996). A degraded riparian zone can be defined as one that lacks the capacity to provide ecosystem functions such as bank stability, maintenance of water temperatures and stream flows, and habitat features (U.S. Department of the Interior 1993; Kauffman et al. 1997). Grazing by livestock has been implicated in the decline of riparian forests (Keller & Burnham 1982; Platts & Wagstaff 1984; Knapp & Mathews 1996). Livestock can compact soil, exacerbate bank erosion, and consume seedlings and saplings of woody riparian species (Platts 1991; Fleischner

1994). Armour et al. (1994) estimate that 50% of western riparian corridors are degraded due to livestock grazing. In the western U.S., the realization that riparian degradation has contributed to the decline of anadromous fisheries has prompted much interest in the protection and restoration of these systems (Meehan et al. 1977; NRC 1992).

Techniques for riparian restoration include planting of riparian woody species, irrigation, and channel modification (Briggs et al. 1994; Kauffman et al. 1995). A key component of any successful restoration is the identification of stressors that are contributing to the decline of the system or preventing system recovery. The failure to address such stressors will often render other restoration efforts ineffective (Briggs et al. 1994; Kauffman et al. 1997). Grazing pressure is an example of a stressor that can prevent recovery of a riparian corridor. Many studies have documented vigorous growth of riparian vegetation following the elimination of livestock grazing (Briggs et al.1994; Green & Kauffman 1995; Kauffman et al. 1995).

In Mendocino County, California, riparian corridors on several streams that had been degraded due to livestock overgrazing did not recover following the removal of livestock. Biologists from state agencies believed that herbivory from *Odocoileus hemionus columbianus* (black-tailed deer) may have been responsible for the slow response. This hypothesis was based on direct observations of the impact of deer herbivory on regenerating riparian vegetation, as well as observation of greater growth of riparian vegetation in areas that were not accessible to deer (Jack Booth, California Department of Fish and Game (CDFG), personal communication). To address this perceived impact, deer exclosures were erected by landowners and resource agencies on three streams in the upper Russian River watershed.

This study assessed the response of the woody vegetation within these exclosures to the elimination of deer herbivory. The riparian corridors of the study sites have not been grazed by livestock while exclosures have been in place, and thus these projects allow the quantification of the effect of wild ungulate herbivory. The goal of this research was to assess the results of existing restoration projects implemented by private landowners and state resource agencies. Although restoration projects are generally not designed as experiments (e.g. they do not have true replication or random assignment of treatments) they provide a source for opportunistic study of ecological processes and the efficacy of restoration strategies; every attempt should be made to learn from both successes and failures. This approach ensures that science is addressing practical applications of restoration and providing insight to improve project effectiveness (Kondolf 1995).

Study Sites

This study utilized exclosures on Parsons, Robinson, and Feliz Creeks, located within the upper Russian River watershed, in the North Coast Range of Mendocino County, California (Fig. 1). The region has a Mediterranean climate, with cool, wet winters, and hot, dry summers. Yearly precipitation averages 95 cm with the majority falling as rain in late autumn and winter. Major land uses in the region include logging, sheep and cattle grazing, orchards, and vineyards. Many riparian corridors in the area have been cleared for agriculture, or have become degraded due to livestock overgrazing.

Six deer exclosure sites were compared with control sites on Parsons, Robinson, and Feliz Creeks. Since the fenced creek exclosures were placed by individual landowners the sites were not selected according to random experimental design protocol. Parsons Creek is a second-order stream east of the Russian River. In the winter of 1992-93, the Hopland Research and Extension Center (HREC) of the University of California implemented a demonstration restoration project on the riparian corridor of Parsons Creek. Treatments included sheep exclosures, deer exclosures, and planting of riparian woody species. Deer exclosures have fences 2 m tall and also exclude sheep. Fencing for sheep exclusion does not exclude deer. For this study, two deer exclosures and two control plots excluding sheep only were compared. Thus the only ungulate herbivores within control plots were deer. Neither deer nor sheep fencing excludes rodent or invertebrate herbivores. Plots were rectangular in shape, parallel to the channel, and the average plot size was 1000 m^2.

Robinson Creek is a third-order stream on the western side of the Russian River valley. Fencing to exclude deer was constructed on a 300 m long section of the creek's riparian corridor in 1991, which encompassed roughly 2 ha of the riparian corridor. Although this property has not been grazed by livestock for many years, little natural regeneration of woody riparian species had occurred prior to the construction of the deer exclosures. Vegetation within this fenced area was sampled and compared to an unfenced control area upstream of approximately equal size. Both the fenced and control areas are under the same ownership and management.

Feliz Creek is a third-order stream also on the west side of the valley. The North, Middle, and South Forks (second-order streams) of Feliz Creek converge on the property of a single landowner. He erected deer exclosures on each of the three forks, as well as a portion of the main stem. The fenced area of the riparian corridor on the South Fork was 54 m long by 10 m wide (5 m on each side of the stream). The fenced area of the North

Fork was 300 m long by 24 m wide (12 m on each side of the stream), and the fenced area of the Middle Fork was 225 m long by 24 m wide (12 m on each side of the stream). These fences were constructed between 1980 and 1982. This property has not been grazed by livestock for the past 18 years.

Methods

Plots on the three streams were surveyed for woody vegetation during June and July, 1997. Regeneration at the sites consisted primarily of saplings of *Salix exigua* (narrow-leaved willow), *S. laevigata* (red willow), *S. lasiolepis* (arroyo willow), *Alnus rhombifolia* (white alder), and *Fraxinus latifolia* (Oregon ash). Saplings were placed into two size classes, based on height and number of stems (U.S. Department of Agriculture 1992). Size class 1 included saplings less than 1 m tall, with branching, woody stems, or obvious new growth. Saplings greater than 1 m tall or with five or more stems were placed in size class 2. The density of saplings within a plot was determined either by a complete survey, or estimated by sampling (described below). I elected to use density rather than percent cover as a measurement of recruitment; tree density is a more precise measurement and is easier to quantify in sparsely vegetated stream reaches with few or heavily browsed plants, as was the case in several of my plots.

Within the Parsons Creek plots all woody vegetation was surveyed. The area of suitable substrate for riparian regeneration (i.e. non-rock ground cover) within each treatment was measured with a tape in order to calculate sapling density. Some of the plots at Parsons Creek had been planted with willow and alder in 1994. Therefore, a planted deer exclosure plot was paired with a planted control plot upstream of the fence, and an unplanted deer exclosure plot was paired with an unplanted control. To avoid counting planted individuals as natural regeneration, size class 2 saplings were recorded, but not included in the data for either of the planted plots. This was a conservative choice because few planted individuals (< 6%) survived into 1997 (J. Opperman, unpublished data), and it is therefore likely that many of the size class 2 saplings were from natural regeneration. High mortality of planted trees was attributed to scour during floods and drought stress (R. Keiffer, HREC, personal communication).

The exclosure on the South Fork of Feliz Creek was relatively small which permitted a complete survey of woody vegetation using the same methods described for the Parsons Creek plots. The two other deer exclosures on Feliz Creek, and the exclosure on Robinson Creek were much larger and, consequently, woody regeneration within these plots was sampled along transects. Four m wide belt transects were extended from the wetted channel to the beginning of upland vegetation, perpendicular to the stream. Plots at Feliz Creek were sampled every 15 m of channel length, and plots at Robinson Creek were sampled every 20 m of channel length. The variable lengths were chosen such that approximately 30 transects could be placed within each plot, 15 on each side of the channel. Transect length varied due to differences in topography, with a mean length of 13.2 m. Control plots were selected at Feliz and Robinson creeks; unfenced reaches with similar channel type and geomorphology to that found in the exclosure were sampled upstream of the exclosures using the same methods. The control plots were located within the same property boundaries as the exclosures, and were under the same land management prior to, and following, the construction of exclosures.

Treatment and control sites were not randomly assigned. Treatment sites had already been determined by the landowners who implemented the restoration projects.

Controls were placed adjacent, or nearly adjacent, to the treatment sites in order to minimize differences between confounding variables such as flow regime, seed source, geomorphology, and land ownership and management. Controls were placed upstream of treatments in order to avoid areas potentially influenced by the treatment. For example, vigorous vegetation within exclosures may have served as an abundant seed source for downstream reaches.

The amount and type of vegetation existing prior to fencing was not documented in either the exclosures or control plots. However, the property owners and others involved with the projects (e.g. consultants, agency personnel) described the exclosures and control reaches as being quite similar, with little or no riparian vegetation prior to erecting exclosures. Additionally, photographs of the Robinson Creek project during its first year showed both exclosures and control reaches with a sparsely vegetated riparian corridor. The effect of fencing on riparian regeneration was tested using a paired sample t-test, with the six exclosure/control pairs as replicates.

Results

Density of regenerated saplings was greater in fenced plots than in control plots ($p = 0.015$, $df = 5$, Fig. 2). Mean density within the six exclosures was 0.49 ± 0.15 saplings/m^2, compared to mean density within control plots of 0.05 ± 0.02 saplings/m^2. Comparing regeneration within exclosures, density was lowest in the Parsons Creek plots, highest in the Robinson Creek plot, and intermediate in the Feliz Creek plots (Fig. 2).

The species and age classes of sapling regeneration varied between streams (Figs. 2 and 3). Regeneration in the four Parsons Creek plots was composed almost entirely of

S. exigua and *S. laevigata*. Regeneration within the Robinson Creek exclosure was primarily *S. exigua*, *S. laevigata, S. lasiolepis*, and *Alnus rhombifolia.* Primarily size class 1 saplings of the same three *Salix* spp. were recorded in the Robinson Creek control plot, with a smaller component of size class 2 saplings of *S. lasiolepis*. The Middle and South Fork exclosures of Feliz Creek were composed of a near uniform canopy of size class 2 *A. rhombifolia*, with a small understory component of *Fraxinus latifolia*. Regeneration within the North Fork was composed of *S. exigua*, *S. laevigata, S. lasiolepis*, *A. rhombifolia*, *F. latifolia*, as well as a small component of other species (including *Populus fremontii* (Fremont cottonwood) and *Acer* macrophyllum (big leaf maple)). Regeneration within the three Feliz Creek control plots was composed primarily of size class 1 *A. rhombifolia*.

High densities of *A. rhombifolia* and *Salix* spp. seedlings were found within all control plots, indicating that seedling establishment is occurring. Very few saplings were found in any of the controls, and those that were found were primarily of the smaller size class: 97% of saplings found in control plots were size class 1, and 3% were size class 2. Within exclosures, 35% of saplings were size class 1, and 65% were size class 2. Many of the saplings found in the control plots displayed leaf and stem damage characteristic of deer browsing. For example, nearly every size class 2 *Salix* sapling within the Robinson Creek control plot displayed signs of heavy browsing. Within the Feliz Creek control plots, *A. rhombifolia* saplings were found only in a prostrate, bushy form, indicative of repeated browsing.

Discussion

In Mendocino County, several creeks with sparsely vegetated riparian corridors responded with limited recruitment of woody vegetation following the removal of livestock. The lack of recovery did not meet expectations based on results from other regions, where considerable regeneration of riparian vegetation has occurred following the cessation of livestock grazing (Briggs et al. 1994; Green & Kauffman 1995; Kauffman et al. 1995). Six deer exclosures were constructed on three streams in an attempt to facilitate riparian recovery by eliminating deer herbivory. Density of sapling regeneration was greater within all six deer exclosures than paired upstream control plots. These results indicate that herbivory by black-tailed deer may be significantly retarding or preventing the regeneration of riparian vegetation within this region.

The recovery of vegetation within deer exclosures on Robinson and Feliz Creeks was quite rapid with considerable growth observed after two years (Dahinda Meda, landowner, personal communication; Roscoe Morris, landowner, personal communication). Currently, the streams within the exclosures have a continuous riparian corridor. The density of regeneration recorded in this study was apparently higher at Robinson Creek than Feliz Creek due to the differences in tree age and size. The exclosure at Robinson Creek has been in place for six years and, therefore, the regeneration consists of dense stands of young saplings (Fig. 4). Exclosures on Feliz Creek have been in place for 15 years, and the trees are much larger (e.g. greater average height and dbh), and more widely spaced. Control plots on Feliz and Robinson Creeks had little growth of riparian vegetation, and very few saplings in the larger size class were found at either stream (Fig. 5).

The density of regeneration within the Parsons Creek exclosures was much lower compared to the other streams. This may be attributed to more harsh abiotic conditions for establishment and survival within the Parsons Creek riparian corridor. The plot reaches on Parsons Creek become dry in May or June of most years, whereas the plot reaches on Feliz and Robinson Creeks generally flow year round. Seedlings of many riparian species (e.g. *Salix* spp.) require contact with the water table during the growing season (McBride & Strahan 1984; Braatne et al. 1996). Seedling mortality due to drought stress may result if the water table at Parsons Creek declines too quickly for seedlings' root systems to maintain contact with water. McBride and Strahan (1984) found that 63% of seedlings along reaches with surface water survived through the summer, while 16.5% of seedlings survived along reaches that dried out by the end of the summer.

Much work on riparian restoration has focused on the effects of livestock fencing on regeneration (Briggs et al. 1994; Green & Kauffman 1995; Kauffman et al. 1995). However, few studies have considered the role that native ungulate herbivory plays in these systems, and to what extent this herbivory may be limiting recovery of degraded systems. Case and Kauffman (1997) compared growth of riparian woody species within deer and elk exclosures to growth outside of the exclosures along a stream in northeast Oregon. Livestock had been removed from the riparian corridor prior to the study. They observed marked differences in crown volume, height, and willow catkin production. During their two year study, the mean height growth of existing woody riparian plants within the exclosures was 47 ± 6 cm, compared to 16 ± 4 cm in the controls. Within the exclosures, 34% of willows produced catkins, while only 2% within the controls did so.

Crown volume of willows within the exclosures increased 550%, compared to 195% outside. Kay (1995, 1997) and Kay and Chadde (1992) reported significant effects of elk herbivory on willows in Yellowstone National Park's northern range. Mean height of willows within long-term exclosures was 274 cm, compared to 34 cm in controls. They found an average of 307,000 seeds/m^2 of willow canopy within exclosures, while no seed production was observed outside the exclosures. Also working in Yellowstone, Keigley (1997) hypothesized that herbivory by native ungulates may eliminate cottonwoods in the park's northern range.

While based on a relatively small sample size, the results of this study indicate that herbivory by deer is severely limiting to natural riparian regeneration on these streams; regeneration density of woody riparian species within deer exclosures was approximately ten times greater than regeneration density within controls. Further, 97% of saplings found within control plots were of the smaller size class, and many displayed signs of heavy browsing. Possible explanations for such strong effects of deer on riparian regeneration in Mendocino County include:

1) *Deer densities*. Although quantitative data are not available, deer densities are estimated to be quite high in Mendocino County (Jack Booth, CDFG, personal communication). Higher deer population densities have been implicated in significant changes in vegetation composition in the eastern United States (Alverson et al. 1988; Tilghman 1989) and may amplify the effects that deer have on riparian regeneration.

2.) *Mediterranean Climate – dry season pressure*. In the eastern United States, the greatest impacts due to deer herbivory occur during the winter when little other browse is available and deer preferentially feed on certain woody species (Alverson et al. 1988;

Tilghman 1989). The Mediterranean climate of Mendocino County results in a comparable season of low food availability during the dry months. Generally, little rain falls between May and October, and during the late summer and early fall riparian corridors are one of the few sources of green vegetation. The annual drought may thus result in a seasonal bottleneck on riparian seedling and sapling survival due to herbivory.

3.) *Threshold effects*. Systems that have been damaged due to a stressor may recover after that stressor has been removed. However, recovery may not be possible if the degree of degradation exceeds a certain threshold (Hobbs & Norton 1996). The riparian corridors of the streams in this study may have crossed below such a threshold for recovery, due to the near complete removal of riparian vegetation. In other words, deer herbivory may not be sufficient to significantly influence a riparian corridor that has been lightly disturbed, but the same level of deer herbivory may be able to prevent a degraded riparian corridor from recovering toward its original condition. The streams in this study had almost no riparian vegetation prior to the construction of deer exclosures. The lack of other riparian vegetation would have increased the browsing pressure on any seedlings and saplings that did establish outside the exclosures.

Conclusions

The results of this study emphasize the importance of monitoring and documenting restoration projects in order to learn from the results. In the upper Russian River watershed, removal of livestock from the riparian corridors of three streams was not sufficient to promote regeneration of woody vegetation. Landowners and agency biologists believed that deer herbivory might have been responsible for the lack of recovery and they implemented fencing projects to address this possible stressor.

Following the elimination of deer herbivory, riparian corridors on Feliz and Robinson Creek responded with vigorous regeneration. The regeneration at Parsons Creek has not been as dramatic, but provides further evidence that deer herbivory may act as a stressor to the recovery of degraded riparian systems. Although the six exclosures were not originally established as a scientific experiment, I believe that much can be gained from documenting the results of these ongoing stream restoration projects; lessons learned can be disseminated to other landowners, agencies, and ecologists. The influence of deer herbivory should be considered when planning a riparian restoration project in this region, and potentially in other regions with similar patterns of degradation, deer density, and/or seasonal dynamics. Preliminary fencing projects would be recommended to determine if deer herbivory is limiting riparian regeneration at a specific site.

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Figure 1. Map of Feliz, Parsons, and Robinson Creeks, within the upper Russian River watershed, Mendocino County, California.

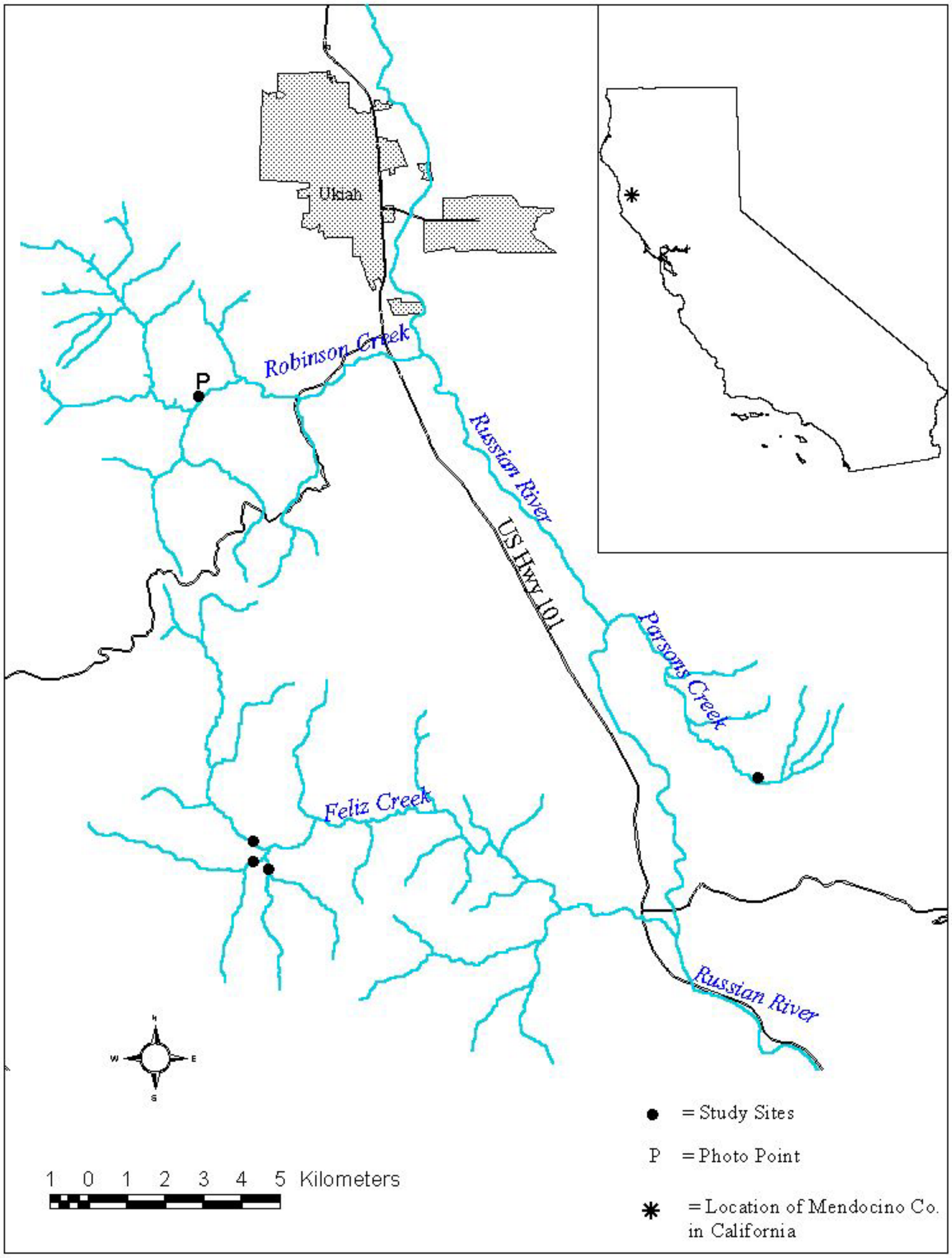

Figure 2. Density and size class distribution of riparian woody sapling regeneration within exclosure (Ex) plots and control plots (C). Size class 1 saplings are represented in the black portion of each bar and size class 2 saplings are in the white portion of each bar. Ex = exclosure, C = control; Rob = Robinson Creek; FS = South Fork of Feliz Creek; FM = Middle Fork of Feliz Creek; FN = North Fork of Feliz Creek; PU = Parsons Creek unplanted plots; PP = Parsons Creek planted plots. Feliz North, Feliz Middle, and Robinson plots were sampled (standard error bars shown); all vegetation was recorded in Feliz South, Parsons unplanted, and Parsons planted plots.

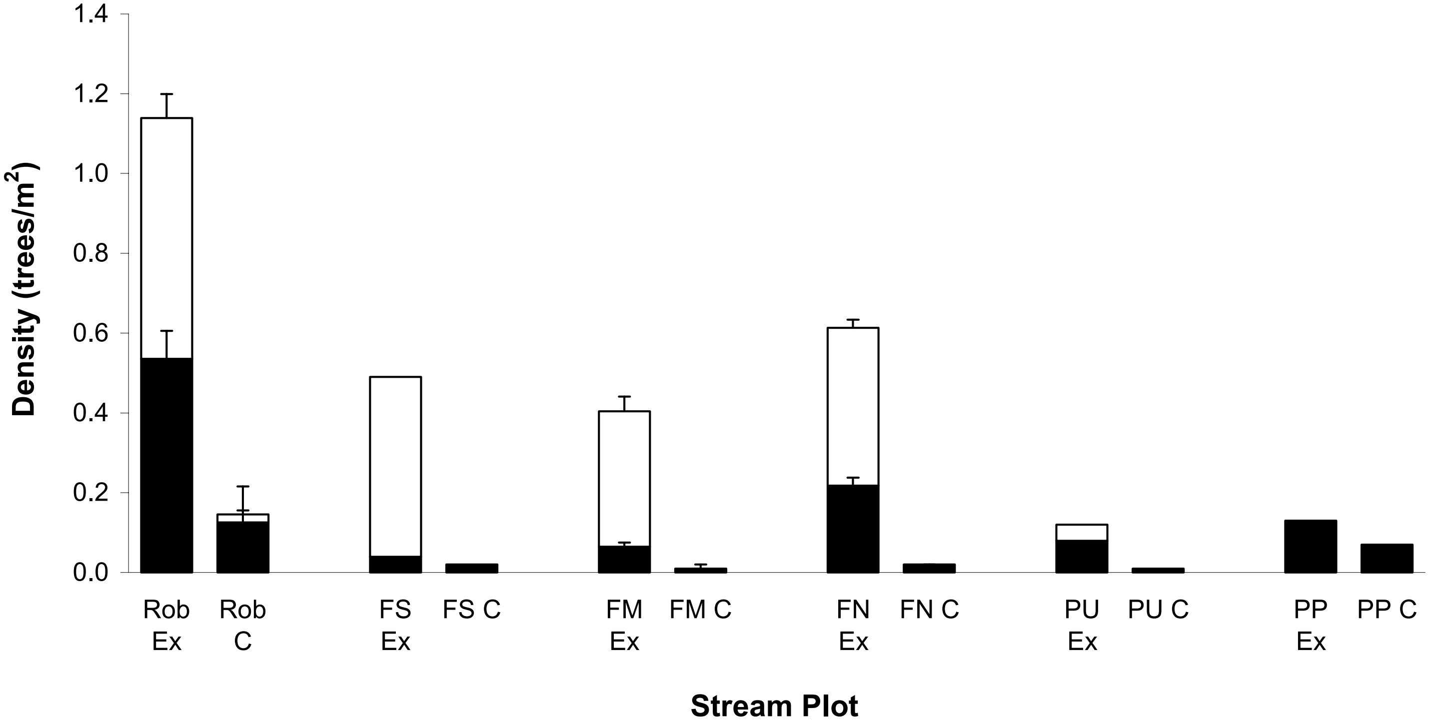

Figure 3. Comparisons between the species composition of woody riparian regeneration within six deer exclosures. Salix includes *S. exigua*, *S. laevigata,* and *S. lasiolepis*

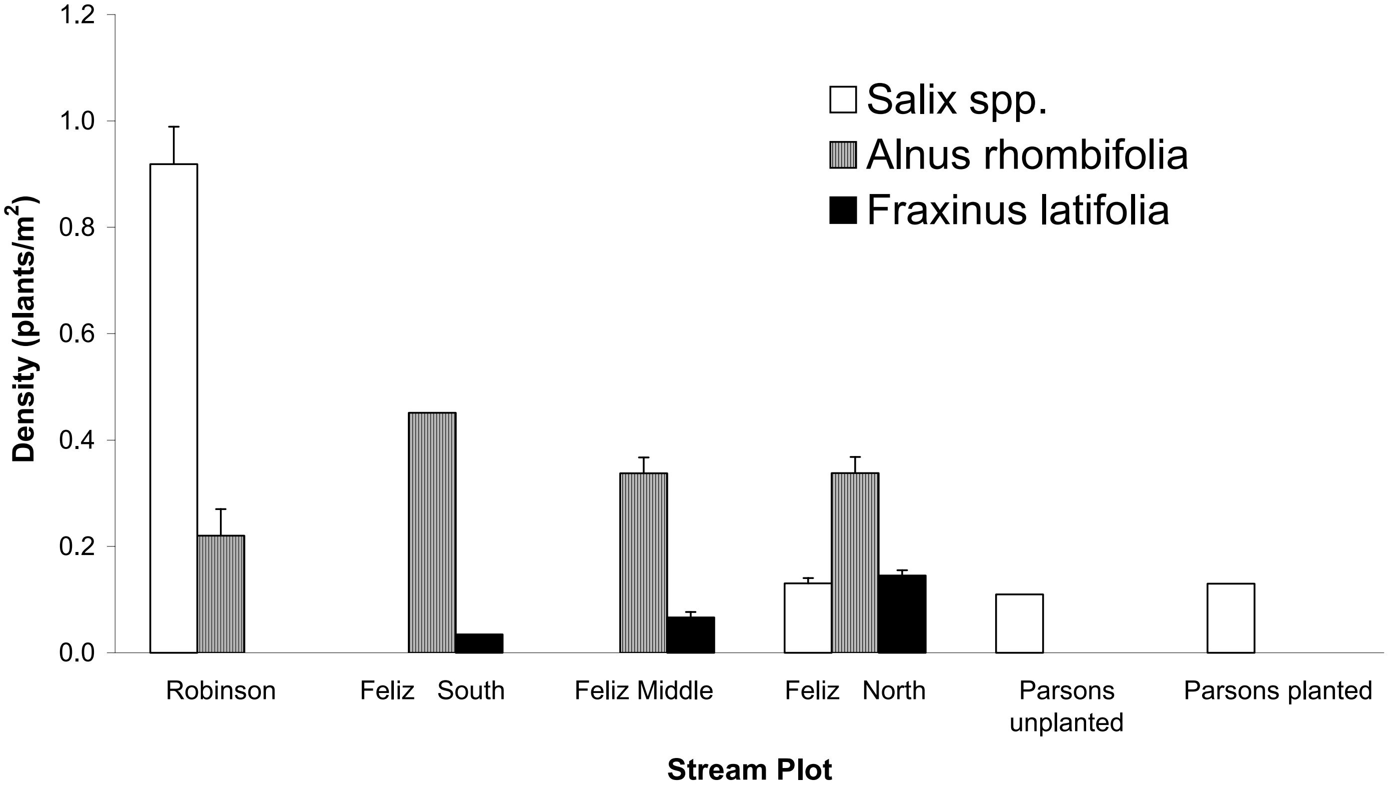

Figure 4. Fence line of the Robinson Creek deer exclosure (the exclosure is to the left). The photograph shows natural regeneration of riparian woody species nine years after construction of the exclosure. This vegetation is primarily composed of *Alnus rhombiforlia, Salix laevigata*, and *Salix exigua*.

Figure 5. Robinson Creek control site upstream of the deer exclosure.

CHAPTER 5

THE EFFECTS OF RIPARIAN RESTORATION ON CHANNEL MORPHOLOGY AND LARGE WOODY DEBRIS

The Effects of Riparian Restoration on Channel Morphology and Large Woody Debris

Abstract

Currently significant resources are invested in restoration of salmonid habitat and recent reviews have suggested that restoration strategies should focus on restoring natural processes rather than instream structures. Riparian restoration is often implemented to improve channel morphology and fish habitat, although studies report equivocal evidence that restored vegetation can effect changes in channel form over a period of years to decades. In this study, I examine the effectiveness of riparian restoration for improving channel morphology and fish habitat in four streams in Mendocino County, California. Deer fencing was used to restore riparian corridors on Robinson Creek 10 years ago and three branches of Feliz Creek 20 years ago (livestock had already been excluded from these properties). I compared morphology, large woody debris (LWD), and late-summer water temperature between the exclosure reaches and control reaches that were selected upstream. Control reaches had similar gradient, substrate, drainage area, valley morphology and management as the restored reaches. Channels within exclosures were narrower and more heterogeneous. The frequency of LWD and debris jams was significantly greater in exclosure reaches than in controls, and comparable to values from similar streams with mature forests. However, the size of the LWD pieces and, consequently, the loading (m^3 of wood/ha of channel) was considerably lower than values from mature-forest streams. Inside exclosures late-summer water temperature was within the acceptable range for steelhead trout, while water temperatures in controls were significantly warmer and potentially deleterious to trout.

In these streams, riparian restoration has resulted in significantly improved habitat characteristics and qualitatively different channels. In this region it is likely that riparian restoration can produce benefits for salmonids that are more comprehensive, sustainable, and cost-effective than attempting to produce similar gains through instream structures.

Introduction

Stocks of anadromous fish on the Pacific coast have declined significantly in the past century due to overharvesting, dams, and degradation of freshwater habitats (Nehlsen et al. 1991, National Research Council 1996, Stouder et al. 1997). To reverse declines in freshwater habitat quantity and quality, resources devoted to stream restoration have greatly increased and currently millions of dollars are being spent on restoration projects (National Research Council 1996, Roper et al. 1997, Roni et al. 2002). Restoration strategies range from attempts to replace instream habitat functionality by placing spawning gravels or physical structures in rivers, to broader strategies focused on repairing riparian and watershed processes. Reviews of strategies to restore instream habitat through emplacement of structures have generally found little evidence that these techniques are effective or sustainable over a period of decades or even years (Frissell and Nawa 1992, Beschta et al. 1994, Roper et al. 1994, Lassetre 1997). Consequently, recent reviews have suggested that restoration efforts adopt a watershed-scale approach addressing linkages between hillslopes, roads, and the channel network, and promoting interactions between healthy riparian corridors and stream channels (Kauffman et al. 1997, Roper et al. 1997, Beechie and Bolton 1999, Kondolf 2000).

Intact riparian vegetation provides numerous benefits to instream fish habitat, including shading, bank stabilization, and inputs of fine organic matter and large woody debris (LWD) (Gregory et al. 1991). Consequently, the loss or conversion of riparian vegetation is associated with decline in the suitability of fish habitat, resulting in warmer stream temperatures, lack of LWD and other organic inputs, and wider, shallower channels. Because of the widespread losses of riparian corridors (Armour et al. 1994, Kondolf et al. 1996) and the multiple benefits they provide, riparian restoration has been promoted as a key strategy to restore the critical processes that create and maintain fish habitat (Roper et al. 1997, Beechie and Bolton 1999, Roni et al. 2002).

Riparian restoration can frequently be achieved by the removal of an ongoing stressor that limits natural regeneration, a process called passive restoration (Briggs et al. 1994, Kauffman et al. 1995, Briggs 1997). Numerous studies and projects have demonstrated that fencing to reduce grazing and browsing pressure can result in rapid and impressive recovery of riparian vegetation (Platts 1981, Knapp and Mathews 1996, Kauffman et al. 1997).

Both theoretical (Ikeda and Izumi 1990) and empirical (Andrews 1984) studies of geomorphology have concluded that channels with vegetated banks are narrower and deeper than channels lacking bank vegetation. Similarly, numerous studies have reported that removal of riparian vegetation results in wider, more shallow and less complex channels (Gunderson 1968, Platts 1981). Thus, one of the common objectives for riparian restoration is the promotion of narrower and deeper channels.

Although the ability of fencing to promote vegetative recovery has been clearly established, evidence that this recovery is followed by improvements in channel form is more equivocal (Sarr 2002). While many restoration projects report that channels have narrowed following riparian restoration, Kondolf (1993) cautions that vigorous vegetative growth can give the appearance of a narrower channel, even if channel form remains unchanged. Therefore, qualitative assessments of channel form may produce inaccurate conclusions of recovery, and Kondolf (1993) suggests that monitoring for channel recovery should directly measure channel form through cross-sections and long profiles.

Studies that have directly measured channel dimensions have reported diverse results. Keller (1978) and Duff (1977) asserted that restored vegetation resulted in narrower channels after three and six years, respectively, while Hubert et al. (1985) and Clifton (1989) found similar results in exclosures of >25 and 50 years, respectively. Conversely, Hubert et al. (1985) found little change in a 4-year old exclosure and Myers and Swanson (1995) and Kondolf (1993) found no narrowing of channels after removal of grazing for 12 and 24 years, respectively, although vegetation had recovered.

This paper explores the effectiveness of recreating and improving instream habitat through restoration of riparian corridors. I examine the ability of riparian restoration to effect three primary improvements to anadromous fish habitat that are commonly used to justify this restoration strategy: 1) increased LWD; 2) narrower, deeper, and more complex channel morphology; and 3) cooler water temperatures.

This study focuses on streams in the hardwood-dominated, Mediterranean-climate streams of Northern California, which provide habitat for anadromous or resident *Oncorhynchus mykiss* (steelhead trout/rainbow trout). I compared riparian vegetation, channel morphology, LWD, and water temperature within reaches on four streams with restored riparian corridors with similar upstream "controls" on the same properties.

Although not initially established as experiments, this study utilizes existing restoration projects as opportunities for learning through long-term monitoring (Kondolf 1995, 1998).

Study sites

The four restored streams (and their upstream controls) are in the Russian River basin, north of San Francisco, in Sonoma and Mendocino counties (see Figure 1 of Chapter 4). The streams are within California's Mediterranean-climate zone, with cool, wet winters, and warm, dry summers (Gasith and Resh 1999). Throughout this region, the majority of annual precipitation falls as rain between November and April. The riparian corridors of these streams contain few or no conifer species, and are dominated by *Alnus rhombifolia* (white alder), *Salix spp.* (willow species), *Umbellularia californica* (California bay laurel), *Quercus spp.* (oak species), *Acer macrophyllum* (big-leaf maple), and *Fraxinus latifolia* (Oregon ash).

In 1982, a landowner fenced the riparian corridor of three forks of Feliz Creek (hereafter referred to as Feliz North, Middle and South) that flow through his property and converge to form the main stem of Feliz Creek (Figure 1). Originally he erected deer fencing around a small reach to protect oak seedlings on the adjacent terrace. He later noticed that the alders along the creek, also protected by the deer fencing, began to grow much more vigorously. He then decided to fence portions of the riparian corridors of all three streams on this property (D. Meda, pers. comm.).

In 1991 a landowner having heard of the successes on Feliz Creek also erected deer fencing along a reach of Robinson Creek. The riparian corridors on Robinson Creek and the three forks of Feliz Creek had almost no riparian vegetation prior to restoration.

Both landowners informed me that the riparian corridors had been heavily browsed by the livestock of previous owners, although neither landowner grazed livestock since purchasing the properties (R. Morris, D. Meda, pers. comm.) Pre-project photos at Robinson Creek confirmed the landowner's assessment, showing sparse riparian vegetation and a uniform channel throughout the control and exclosure reaches.

Methods

Paired exclosures and controls

I selected control reaches upstream of each exclosure. Controls were located on the same property and thus have been under the same management, with the exception of the fencing, for several years preceding and since the restoration projects. Control reaches were also located within reaches with similar channel and valley forms (*sensu* Frisell (1992). The importance of consistency for ownership and channel morphology required that exclosure and control reaches be adjacent to each other. I tested for the similarity of exclosure and control reaches through analysis of topographical maps and during channel surveys. The composition of substrate was compared in Feliz South and Feliz North through pebble counts using the technique describe in Kondolf (1997).

Large Woody Debris

I sampled all pieces of LWD (defined as wood > 1 m in length and >10 cm in diameter) within the bankfull dimensions of the channel. I recorded the length, diameter, species, and channel position of each piece of wood. Debris jams were defined as accumulations of three or more pieces of LWD and were classified according to their channel position (on the bank, partially spanning the channel, or spanning the channel). I also noted if the jam caused a step in the channel profile, caused or contributed to the formation of a pool, and whether or not the jam was stabilized by standing trees along the channel margin.

Riparian structure

For every 100 meters of channel surveyed I sampled two riparian plots (one on each side of the channel) that extended 10 meters along the channel and a variable distance perpendicular to the channel. The variable distance was determined based on the ability of a tree within that plot to fall and enter the stream. Within these plots, I measured the diameter at breast height (dbh) of every living tree and snag and identified its species, estimated its slope-distance from the stream, and classified the landform where it grew (bank, bar, floodplain, terrace, hillside).

Channel morphology

I compared the channel morphology of the reaches within exclosures and their paired controls. I surveyed a long profile for the length of the study area using a surveyor's level, recording the thalweg and water surface, and surveyed 4-6 cross-sections within each treatment (i.e. fenced and control). Additionally, wetted channel width and thalweg depth were recorded every 10 m during the long profile. I surveyed the branches of Feliz Creek in April, 2001 and Robinson Creek in June, 2001.

Temperature

I recorded water temperature in pools within exclosure and control reaches for the three branches of Feliz Creek on two days during late summer when water temperatures are likely to be highest and most problematic for fish: September 7, 2000 (air temperature = 100° F) and August 30, 2001 (air temperature = 86° F). All temperatures were measured between 14:30 and 16:30. Pool temperature was recorded by placing a thermometer on the pool bed in the deepest part of the pool. I also recorded pool depth.

Analyses

I used paired t-tests (n = 4) to compare riparian basal area, LWD values, wetted and bankfull widths, and channel depth. Differences in pool temperature between control and exclosure pools were compared using ANCOVA with model terms for treatment (exclosure or control) and pool depth as a covariate.

Results

Control and exclosure reaches

Control and exclosure reaches had similar gradients, measured from topographical maps and in the field (Table 1), although debris jams within the exclosure of Feliz North resulted in more local heterogeneity in gradient (Figure 2). Because the reaches were adjacent with no tributary inputs, drainage area for controls and exclosures were essentially equal. Valley types were similar between the control and exclosure reaches of Robinson, Feliz North, and Feliz South, while the control reach for Feliz Middle was located within a more confined valley than the control (Figure 1). Substrate size distributions were similar in Feliz Middle and Feliz North (Figure 3), and appeared to be similar on Feliz South and Robinson.

Riparian structure and LWD

Basal area of riparian trees within 5 meters of the channel was much greater in the restored reaches (185 ± 35 m^2/ha) than in the control reaches (10 ± 9 m^2/ha) ($t = 6.8$, 3 *df*, $p = 0.007$). The number of pieces of LWD per hectare of channel area was also significantly greater in exclosures (314 ± 107 pieces/ha) than in controls (70 ± 40 pieces/ha) (*t* = 3.7, 3 *df*, *p* = 0.03), with a similar trend for pieces of LWD/10 m of channel (2.2 ± 0.7 compared to 0.5 ± 0.2 pieces/m; *t* = 3.0, 3 *df*, *p* = 0.05). (Figure 4a).

Average loading of LWD was nearly identical within exclosures and control reaches (approximately 25 m^3/ha) (Figure 4b). The three branches of Feliz Creek had much higher loading values than Robinson Creek, the more recent exclosure. The greatest loading among the Feliz branches was within the exclosure in Feliz North (52.0 m^3/ha), which contained two spanning debris jams that trapped a large volume of wood, and the control reach for Feliz South (58.5 m^3/ha), which included a large spanning jam caused by a hillslope mass-wasting event into the channel.

Based on the species composition and size distribution of the LWD and riparian trees, I determined that all the LWD in Robinson Creek had been transported to the site. LWD within the Feliz Creek reaches was a combination of transported wood and wood derived from the adjacent riparian corridor. Due to the mass-wasting event into the control reach of the South Fork, much of the LWD in that reach was derived from the adjacent hillslope.

Debris jams were three times more frequent within exclosures: 1.5 ± 0.5 compared to 0.3 ± 0.2 jams/10 bankfull widths (*t* = 3.1, 3 *df*, *p* = 0.05) (Figure 5). However, spanning jams were less frequent and only 2 out of the 8 restoration and control reaches had any spanning jams. Both the exclosure on Feliz North and the control on Feliz South had a frequency of spanning jams of 0.3 jams/10 bankfull widths.

Channel morphology

Channels in the Feliz control reaches had wider bankfull widths (*t* = 8.0, 2 *df*, *p* = 0.02) and wetted widths (*t* = 4.8, 2 *df*, *p* = 0.04) than channels in the exclosures (Figures 6 and 7). Debris jams in Feliz North resulted in locally heterogeneous gradients. A large debris jam split the high-flow channel in two, separated by a well-vegetated island (Figure 2).

On Robinson Creek, comparisons between wetted and bankfull widths in the control and exclosure were difficult for two reasons. While channel surveys on Feliz Creek were conducted during spring baseflows, surveys on Robinson Creek were conducted later in the summer as the creek was beginning to dry, and the control reach was nearly dry. Further, it was very difficult to determine a bankfull width on the control reach because the channel was broad and lacked defining features. Thus, only a qualitative description of Robinson Creek is presented. Cross-sections in Robinson Creek showed that the channel in the well-vegetated exclosure had more defined low-flow channel and banks, and the long profile showed more variety in channel form and depths (Figures 8 and 9).

Temperature

Pools were significantly cooler in the fenced reaches than in control reaches on both days. In September 2000, pools in exclosures ($n = 10$) averaged 63.8 ± 0.9 ° F, while pools in controls ($n = 8$) averaged 68.4 ± 0.7 ° F (ANCOVA, $F_{2,15} = 4.9$, $p = 0.0002$). In August 2001, pools in exclosures ($n = 8$) averaged 64.7 ± 1.0 ° F while pools in controls ($n = 7$) averaged 72.8 ± 1.0 ° F (ANCOVA, $F_{2,12} = 5.5$; $p = 0.0001$).

Discussion

Post-project monitoring of riparian restoration on four streams in Mendocino County provided strong support for the ability of riparian restoration to effect improved stream habitat for fish. Reaches with restored riparian corridors generally had narrower, deeper and more complex channels, and cooler water temperatures than similar upstream control reaches. Further, the restored reaches had significantly more debris jams and pieces of LWD than control reaches.

Channel morphology

Previous research has found equivocal support for the ability of riparian restoration to effect channel change within a period or years to decades. The range of results from these studies suggest that practitioners cannot be certain that channel recovery will follow vegetative recovery within a reasonable time period (as determined by the project's objectives). Inconsistent channel responses across studies are likely due to the fact that channel morphology is a function of a number of variables, in addition to vegetation, such as substrate size distribution of the bed and banks, sediment load, flood history, and disturbances throughout the watershed (Frissell 1992, Kondolf 1993, Myers and Swanson 1997, Sarr 2002). For example, Kondolf (1993) hypothesized that a stream channel in an exclosure in California's White Mountains had not narrowed because of low sediment loads and infrequent overbank flooding to build up banks. The four Mendocino County streams examined here are within a region with very high sediment loads (Kelsey 1980, Mount 1995), promoting the process of overbank deposition and channel narrowing. Thus, the rate and form of channel response to restored vegetation is likely a function of a site's geomorphic and hydrologic context. The ability of revegetation to improve channel form in a given region can probably be predicted through an analysis of these conditions; predictions can be tested and refined by monitoring past projects (Sarr 2002).

Post-project monitoring comparing exclosures and selected "controls" must be viewed with some caution (Sarr 2002) and, clearly, the existence of pre-project surveys would strengthen inferences about channel responses to restored riparian vegetation (Downs and Kondolf 2002). However, I selected controls that were geomorphically similar and under the same long-term management as exclosures. The fact that changes in channel form were coincident with the fence line in all four streams provides strong evidence that differences in channel form can be attributed to the differences in riparian vegetation as opposed to other influences.

Large Woody Debris

Most studies of LWD have been conducted in the Pacific Northwest in conifer-dominated landscape. Studies of recovery of LWD following removal of riparian vegetation have found that input of conifer debris > 10 cm in diameter may not begin until 50 years following recovery (Andrus et al. 1988) and that full recovery of LWD levels may take centuries (Murphy and Koski 1989). Hardwood-dominated streams in Northern California have considerably lower loading than old-growth conifer forests (Chapter 2) and the most prevalent source of woody debris in these channels, *Alnus rhombifolia,* reaches maturity in less than a century (Stuart and Sawyer 2001). Therefore it is likely that loading within hardwood streams can more quickly reach levels found within mature forests than can streams in conifer-dominated forests. To better understand the trajectory of LWD recovery in these reaches, I will compare the characteristics of LWD and debris jams in restored and control reaches with values for hardwood streams with mature forests, reported in Chapter 2. While comparing restored sites with controls allows an understanding of changes that have occurred since restoration, expanding the analysis to include more mature sites allows an understanding of whether characteristics of the restored sites fall within the range of natural variability for hardwood-dominated streams (Hobbs and Norton 1996).

The frequency of LWD pieces within restored reaches was much more similar to values from mature-forest streams than was loading. For example, LWD frequency within Feliz North and Feliz South exceeded the average value from mature-forest streams (2.1 ± 0.2 LWD/10 m). However, loading values were much lower than those from streams with mature riparian corridors, which averaged 100 ± 10 m^3/ha with a maximum value of 173 m^3/ha (Chapter 2).

The riparian trees along Robinson Creek were less than ten years old, and frequency of LWD was approximately 1/3 that of mature-forest streams, while loading was an order of magnitude lower. Trees within the riparian corridor were too small to provide LWD and the only LWD within the reaches had floated in from upstream. The streams of Feliz Creek, with the more established riparian corridors (> 20 years old), had a LWD frequency comparable or greater than mature-forest streams. LWD loading, however, was on average 1/3 of the values from the mature-forest streams.

Within the three Feliz creeks, LWD was a combination of wood derived from the reach and wood that had floated in and been deposited (or, in the case of the control reach on the South Fork, entered via mass wasting). Although the larger trees within the restored riparian corridors of the Feliz Creeks were reaching a sufficient size to provide LWD (20-30 cm in diameter), mortality was generally occurring among smaller trees, and the largest pieces of LWD in these reaches appeared to have been transported to the site. Nearly 80% of the LWD found in the four restored reaches were deposited against trees rooted near bankfull, either as debris jams or individual pieces. Thus, through this trapping function, riparian restoration can result in relatively rapid increases in the amount of LWD within a reach, even before the trees within the riparian corridor itself are large enough to provide LWD directly. Although bank jams were very common in the restored reaches (comparable or greater than frequencies for mature-forest streams), channel-spanning jams were rare. These channel-spanning jams have been found to be the most important mechanism by which LWD influences channel morphology and can potentially trap and store large quantities of wood (Chapter 2).

The North Fork had the greatest loading among the restored reaches with approximately half the level from mature-forest streams. (Loading was also high in the control reach of the South Fork due to a single large landslide that brought several mature trees from the hillslope into the channel. This jam may have trapped most of the wood coming from upstream and, as a result, the South Fork had the lowest loading values of the three Feliz exclosure reaches). The North Fork's drainage area is twice that of the Middle Fork and three times that of the South Fork and, thus, it has the greatest transport capacity for LWD, based on its greater width, depth, and stream power (Braudrick et al. 1997, Braudrick and Grant 2000). The increased transport capacity may have increased total loading due to two related processes. First, the North Fork contained the largest transported single pieces of LWD, which increase loading simply due to their size. Second, these large pieces are capable of functioning as key pieces for channel-spanning jams, which can store large volumes of wood. This increases loading by increasing the trapping efficiency of the reach.

Due to the transport of larger pieces into the reach and, consequently, the formation of channel-spanning jams, the North Fork provides the best example of channel responses to restored LWD loadings. These include the creation of pools, formation of secondary channels, and a heterogeneous long profile, including a low-gradient, depositional reach upstream of the jam (Montgomery et al. 1996).

These observations suggest a three-stage process of LWD recovery following riparian restoration:

1) Young riparian trees (e.g. < 10 years old) cannot provide local source of LWD but begin to trap LWD in transport, creating bank jams (illustrated by Robinson Creek);

2) As trees within the corridor grow larger (10-20 years) they are more effective in trapping LWD in transport (because they no longer bend with high flows) and begin to provide some local contribution of LWD, increasing loading levels. Local LWD is still too small to provide key pieces for channel-spanning jams (illustrated by the Middle and South Forks of Feliz Creek). However, channel-spanning jams can occur if a sufficiently large log is transported to and deposited within the reach. Channels with greater transport capacities may therefore begin to form channel-spanning jams sooner. Due to both the contribution of larger individual pieces and the effectiveness of channel-spanning jams for trapping wood, loading in the reach will increase further (illustrated by North Fork Feliz Creek);

3) As the riparian trees become mature they provide larger pieces of LWD and key pieces for channel-spanning jams. The reach's trapping efficiency increases with increased frequency of channel-spanning jams, leading to the highest loading values and greatest influence of wood on channel morphology. Loading may be increased still further as later successional species, such as canyon live oak and California Bay begin to replace or augment the LWD provided by willows and alders (illustrated by the streams with mature riparian corridors described in Chapter 2).

Water temperature

Within the Feliz Creek exclosures, characteristics of the vegetation and channel are apparently able to maintain significantly cooler water temperatures. Pool temperatures within exclosures were within or just over the optimum range of 59 – 64 ° F (Moyle 2002) for steelhead fry and juvenile growth, while temperatures within the control reaches, particularly in 2001, approached levels whereby steelhead begin to experience oxygen deprivation (70 ° F) (McEwan and Jackson 1996) or death (73.4 ° F) (Moyle 2002). However, Erman and Leidy (1975) have reported that trout fry can persist in isolated pools with temperatures up to 72 ° F. Although these data were taken only on two days, they do provide a snapshot of pool temperatures during particularly hot periods in August and early September, when water temperatures are at their warmest.

Cooler water temperatures within exclosures may be due to both the shading provided by the riparian vegetation and by differences in channel morphology (Poole and Berman 2001). Narrower channels have less surface area to assimilate atmospheric heat energy, and deeper channels may allow more infiltration of hyporheic groundwater (Poole and Berman 2001). These effects of riparian shading and channel form may become more pronounced during the late summer when surface flow levels drop to near zero and, in places, isolated pools are separated by dry riffles. The low flows and/or isolation of pools reduces the mixing of surface waters from upstream and, therefore, accentuates the local influence of shading and channel form.

Conclusion

Monitoring is essential to the long-term success of a restoration program (Kondolf 1995, 1998, Bash and Ryan 2002, Downs and Kondolf 2002). Ultimately, the results of restoration investments must be demonstrated by meaningful outcomes such as habitat created or populations established, over sufficient time scales. These metrics are ultimately more useful measures of success than documenting successful implementation through the number of trees planted, instream projects constructed, or miles of stream fenced. Because of the time scale over which some of the more important changes occur, long-term monitoring may be required to demonstrate these outcomes, and the ultimate sustainability of a project can only be proven over longer time scales. This is particularly true in stream restoration, where sustainable success may require that a project be tested during relatively infrequent flooding events (Schmetterling and Pierce 1999, Downs and Kondolf 2002)

In addition to quantifying and demonstrating the sustainable and meaningful benefits of restoration, monitoring is also essential for learning and improving restoration programs (Downs and Kondolf 2002). Through monitoring, specific techniques and strategies can be evaluated and modified, thus improving the toolkit of restoration practitioners and furthering their understanding of common constraints. More broadly, monitoring programs can allow funders and policy makers to compare the relative costs and benefits of various techniques, improving regional strategies for prioritizing and implementing cost-effective restoration.

Post-project monitoring on these four creeks provided valuable information regarding the success of deer fencing for riparian restoration and subsequent improvements in habitat for anadromous fish. I have demonstrated that, following riparian restoration, several key habitat values were quantitatively improved in the restored reaches and, overall, the restoration projects have resulted in qualitatively different stream habitats. Several of the documented changes, such as improved channel morphology and increased levels of LWD, replicate the objectives of restoration through the use of instream structures. Passive restoration of riparian corridors, as demonstrated in these projects, provides three primary advantages over restoration using instream structures: 1) multiple benefits; 2) reliability and longevity; 3) low cost and technical knowledge required.

Riparian restoration provides numerous direct benefits, including inputs of leaf litter and LWD, shading, bank stability, terrestrial habitat, and aesthetics. Subsequently, restored vegetation can promote narrower, deeper channels and, ultimately, LWD inputs over time will increase pool formation, cover, and channel heterogeneity. In contrast, instream structures generally provide single, or few benefits, such as creating a scour pool or providing cover. Generally, these benefits are limited to the stream channel itself, whereas restored riparian corridors can provide benefits that can extend from the reach to landscape scales (e.g. as wildlife corridors) (Malanson 1993, Naiman et al. 1993, Hilty 2001).

Second, passive restoration through fencing is generally a low-risk restoration project in areas with high potential for natural regeneration (although restoration that requires planting entails greater risk; (Opperman and Merenlender 2002). If fences are maintained, dense vegetation can generally establish within a decade; riparian corridors should be self-sustaining thereafter provided that they are protected from major anthropogenic disturbances (e.g. overgrazing). The apparent longevity and resiliency of these projects is demonstrated by the fact that the restored reaches in this study have undergone several high flow events (in the Russian River basin, January 1995 was a 63-year flood, February 1986 was a 21-year flood, and January 1997 was a 13-year flood). Rather than acting as a disturbance that can destroy a restoration project, these high flows instead are essential to the process of restoration. They transport LWD that can form debris jams and sediment that can rebuild banks during overbank flow. Elsewhere, high flows may erode banks and cause mature trees to topple. This adds more LWD and opens space for new regeneration, resulting in a riparian corridor with more diverse age and size classes. Conversely, instream projects are vulnerable to failure during high flows and often do not last a decade (Frissell and Nawa 1992, Lassetre 1997). Frissell and Nawa (1992) found that more than half of the 161 structures they assessed on 15 streams in Oregon and Washington had failed before five years; the structures had experienced floods with recurrence intervals of 2-10 years. Further, structures that are stable often do not function through time as intended (Thompson 2002).

Finally, instream projects tend to be more expensive than fencing, often require heavy equipment and have greater requirements for technical knowledge. Unfortunately, many instream projects are installed by people without sufficient training, and some studies have even demonstrated that these instream projects can have negative effects over the long term (Thompson 2002). Thus, instream projects necessitate that the practitioner or landowner have significant expertise, without which they risk implementing projects that fail to meet objectives or possibly have negative impacts on the stream. On the other hand, passive riparian restoration does not require the same level of technical knowledge and there is little risk that improper placement of fences will cause harm.

A comparison of the costs for riparian fencing and instream structures provides insight into the relative costs and benefits of these strategies. Based on the dimensions of the fenced areas and a cost estimate for materials and labor gleaned from current restoration projects ($2.50/foot), fencing the three forks of Feliz would cost $12,300 and fencing Robinson Creek would cost $6,400. I randomly sampled the budgets for 7 California Department of Fish and Game (CDFG) restoration projects that built a total of 118 instream structures (most projects were combinations of log and boulder projects designed to create pools and/or provide cover) and found that the average cost per structure was $1400 (standard deviation = $223). Thus, for the same amount of funding spent to fence the three forks of Feliz Creek and Robinson Creek 9 and 4.5 structures, respectively, could have been placed in the streams. For the three branches of Feliz Creek this equates to approximately one structure per 100 m and, for Robinson Creek, one structure per 65 m. Thus, for similar costs, a strategy based on instream structures would have influenced channel morphology at the scale of the habitat unit over a proportionally small length of stream and with an unknown level of sustainability. Conversely, the restoration of the riparian corridors has apparently contributed to fundamentally different channels across the entire reach – thick riparian vegetation, narrower, deeper and more complex channels, cooler water temperatures, and significantly more LWD. These benefits can likely be maintained for decades or centuries.

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Table 1. Drainage area and gradients of control and exclosure reaches.

Stream	Drainage Area (ha)	Exclosure gradient	Control gradient
Feliz North	2300	0.015	0.017
Feliz Middle	1140	0.02	0.02
Feliz South	780	.03	0.025
Robinson	4130	0.004	0.006

Figure 1a. Deer exclosure on three forks of Feliz Creek, Mendocino County, California.

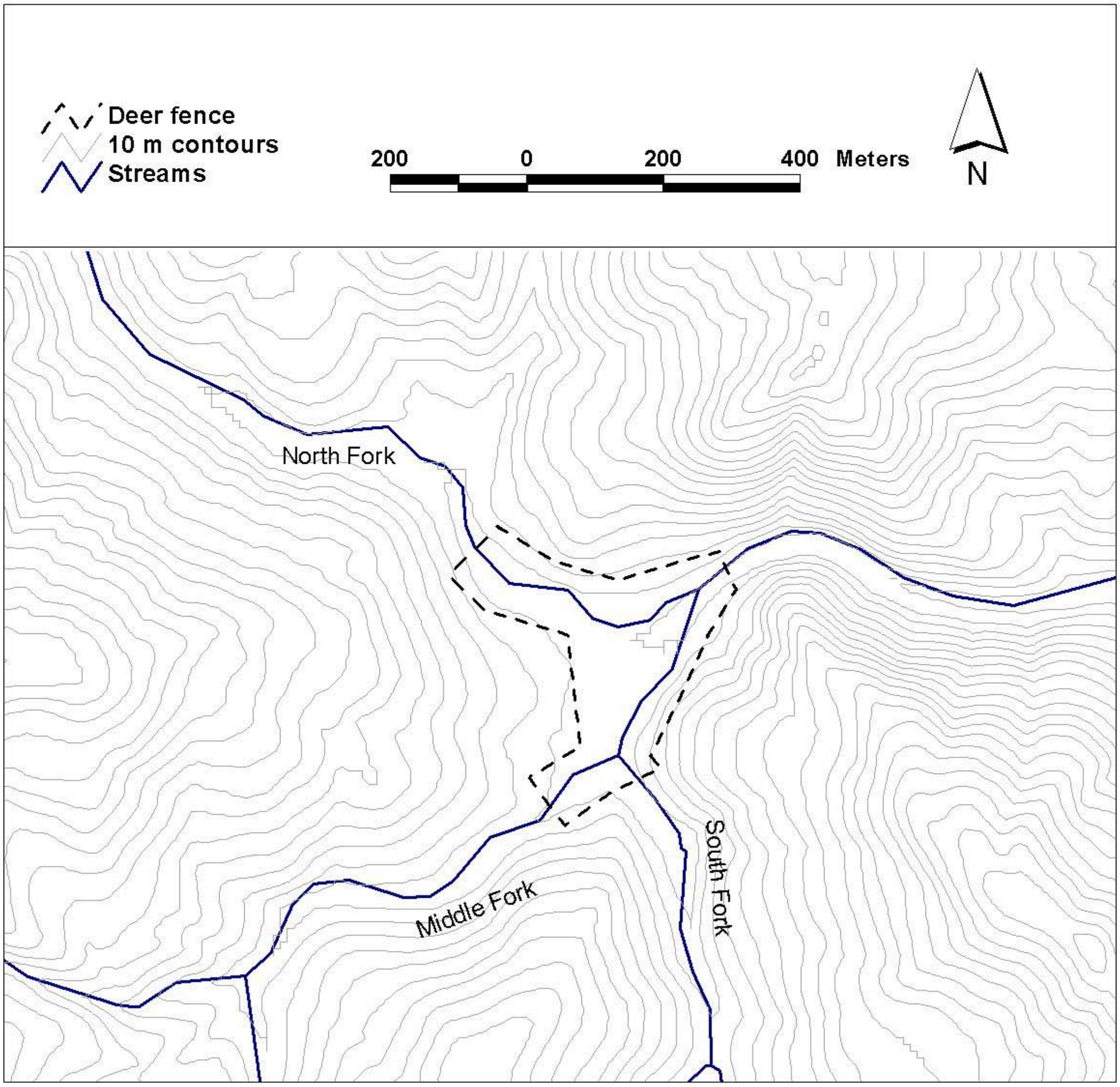


Figure 1b. Control and exclosure reaches on Robinson Creek, Mendocino County, California.

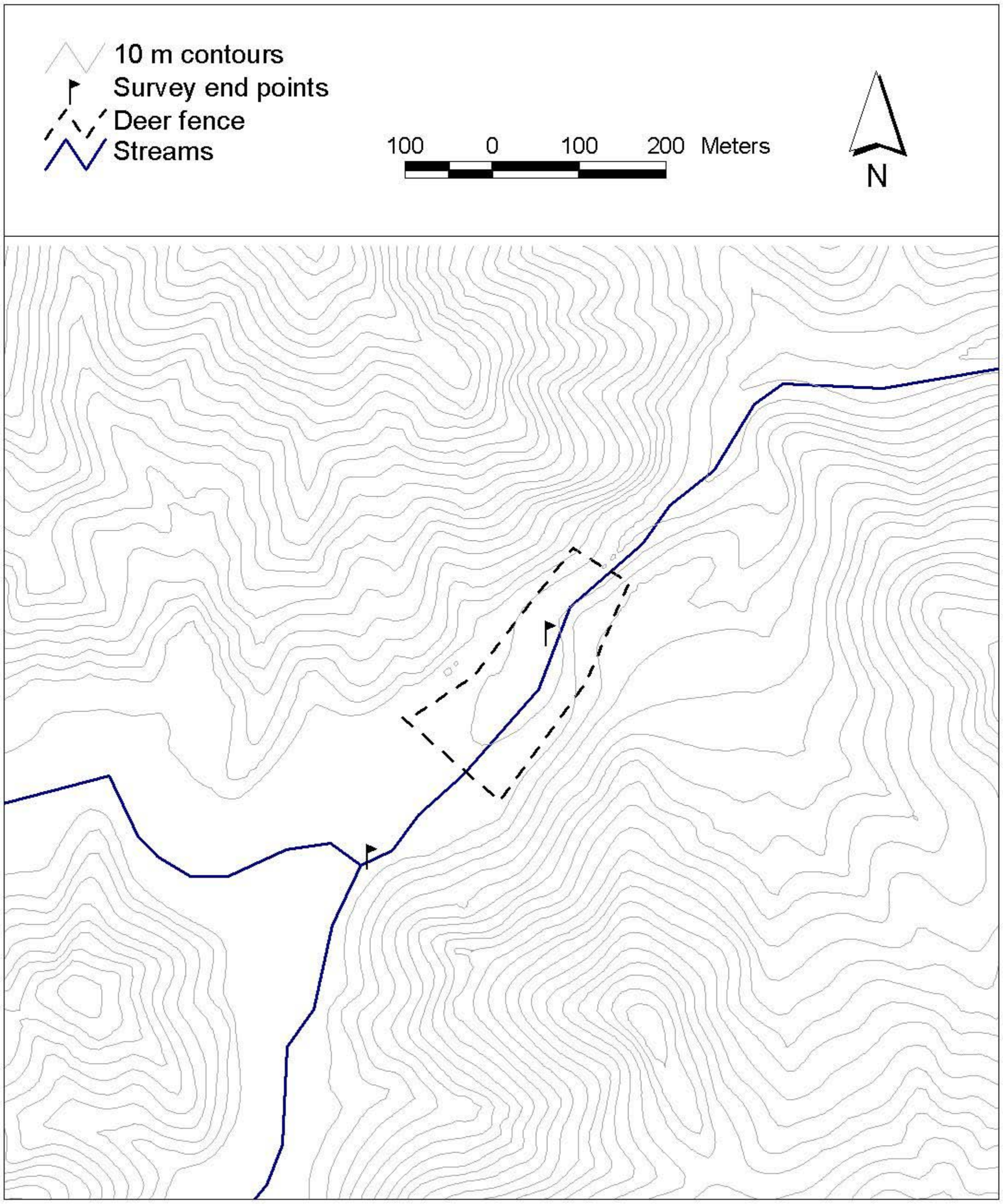


Figure 1c. Control and exclosure reaches on the North Fork of Feliz Creek, Mendocino County, California..

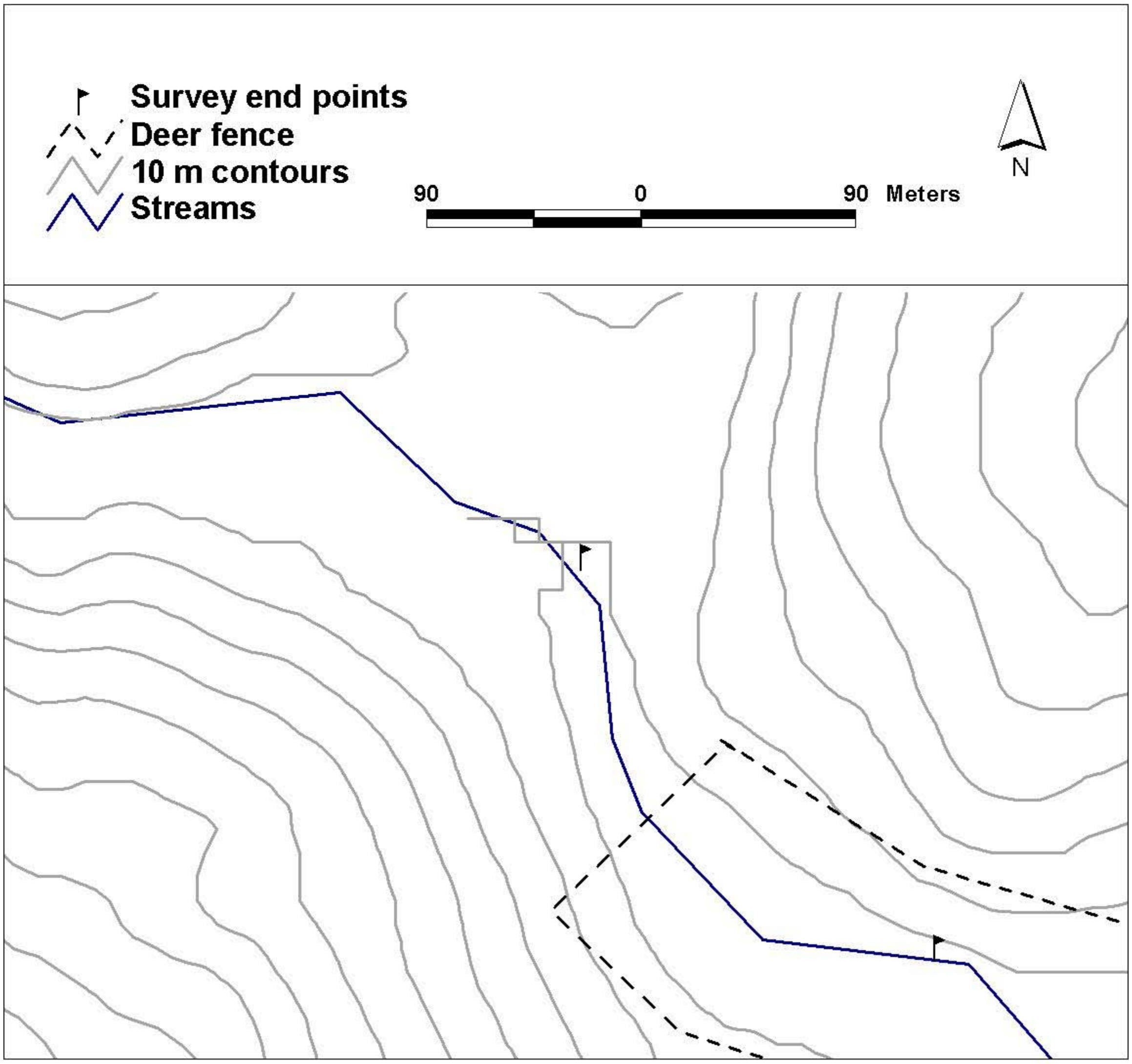


Figure 1d. Control and exclosure reaches on the Middle and South Forks of Feliz Creek, Mendocino County, California.

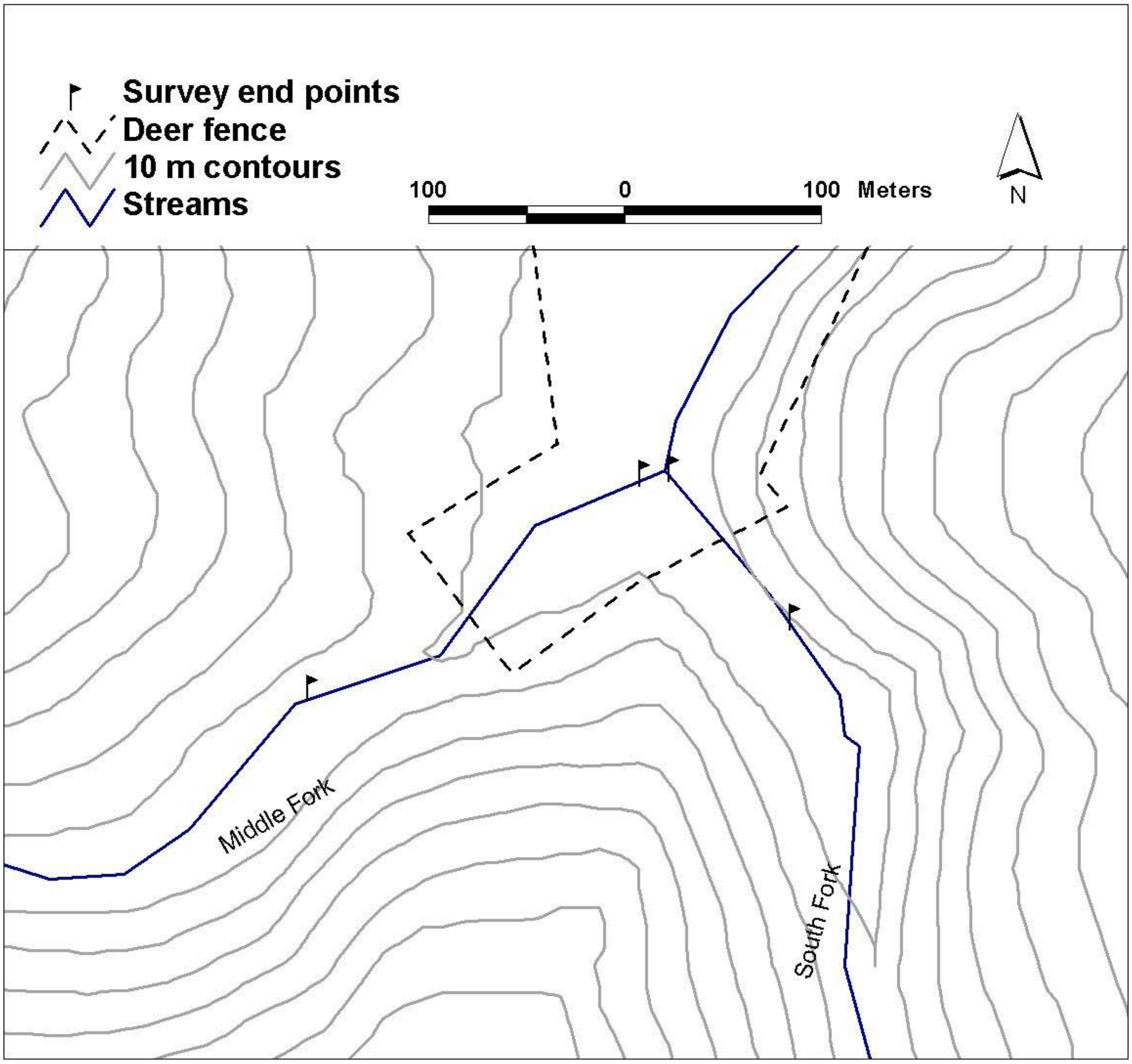


Figure 2. Long profile on North Fork of Feliz Creek. Squares represent channel-spanning debris jams. The larger square shows the large debris jam that splits the channel.

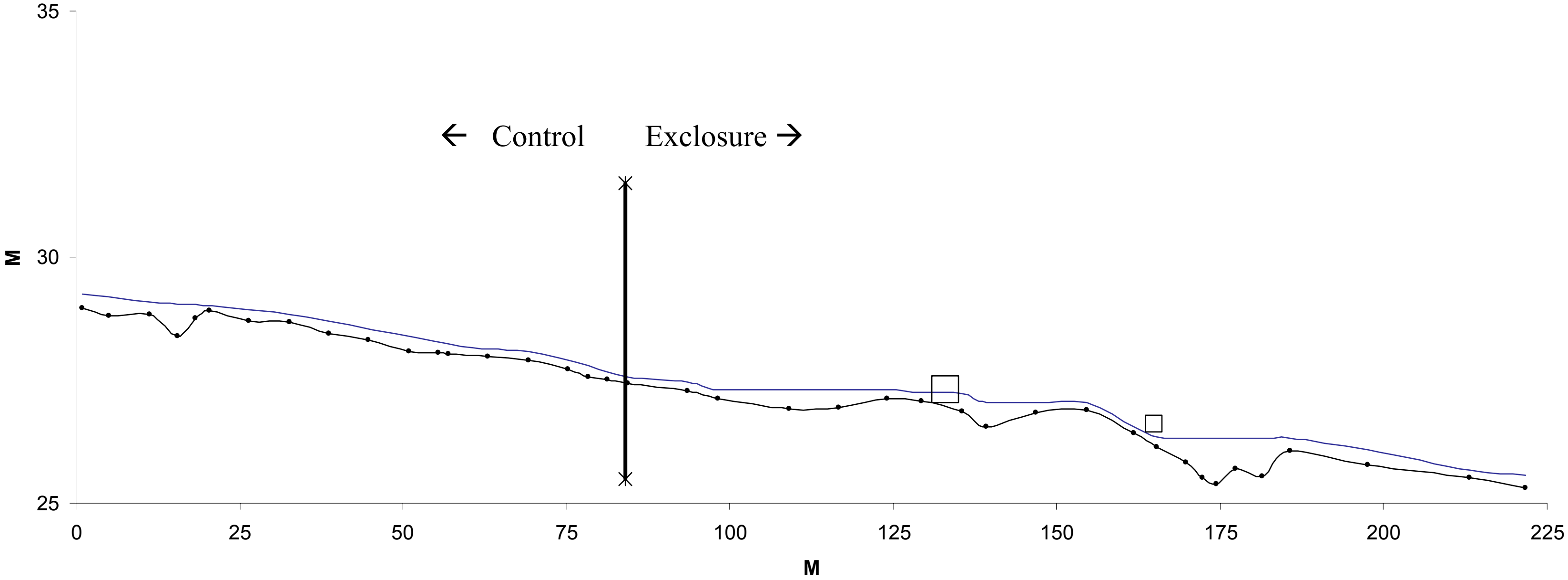
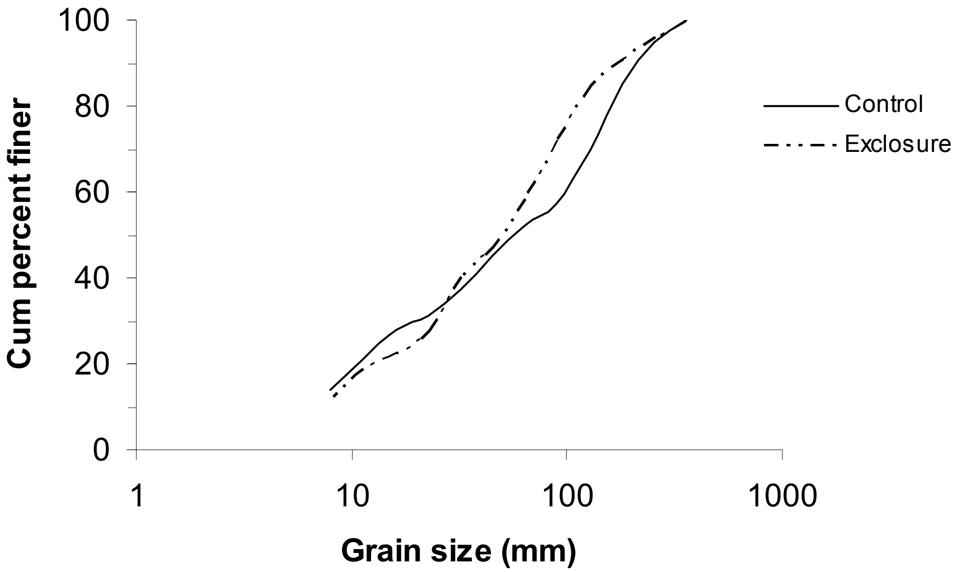
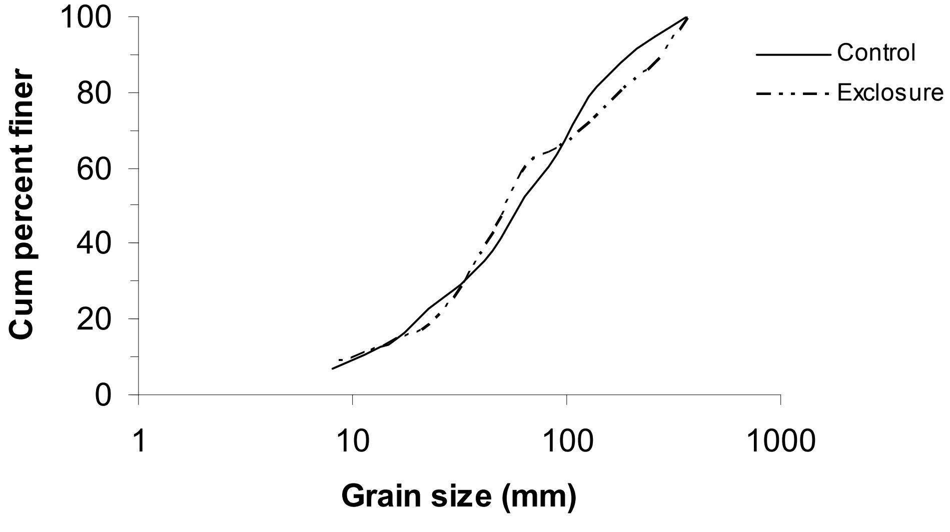


Figure 3. Cumulative size distributions of sediment in control and exclosure reaches of Feliz North (a) and Feliz Middle (b).



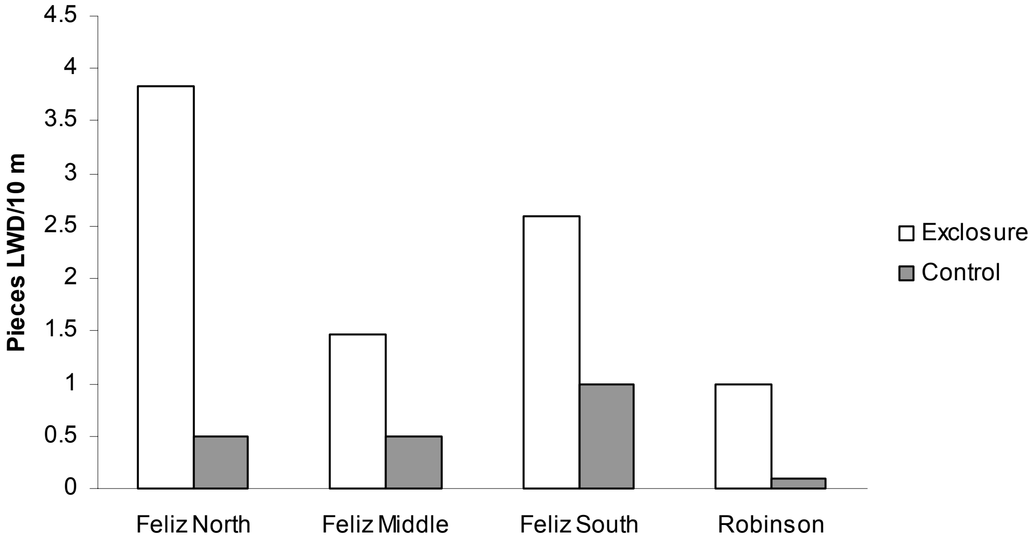
a.



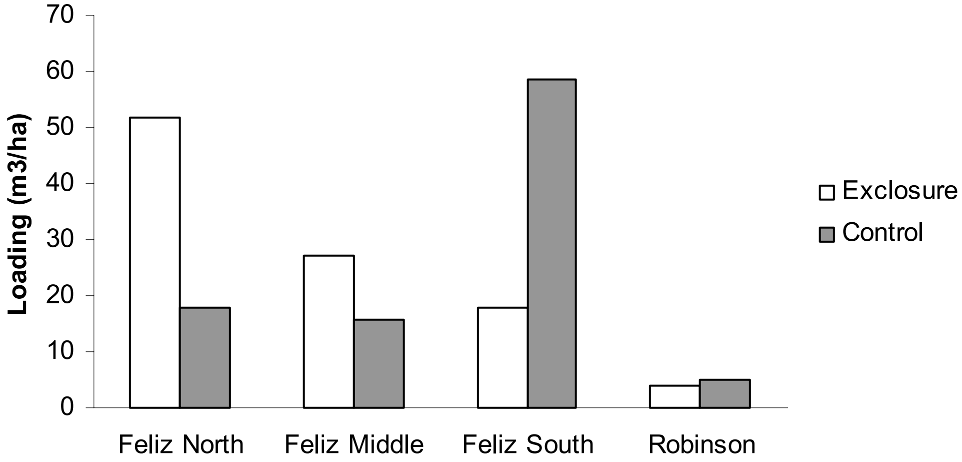
b.

Figure 4a. Pieces of LWD per 10 m of channel, comparing matched control (gray) and restored (white) reaches.

Figure 4b. Loading of LWD (m^3/ha), comparing matched control (gray) and restored (white) reaches.



4a.



4b.

Figure 5. Frequency of debris jams, comparing matched control (gray) and restored (white) reaches.

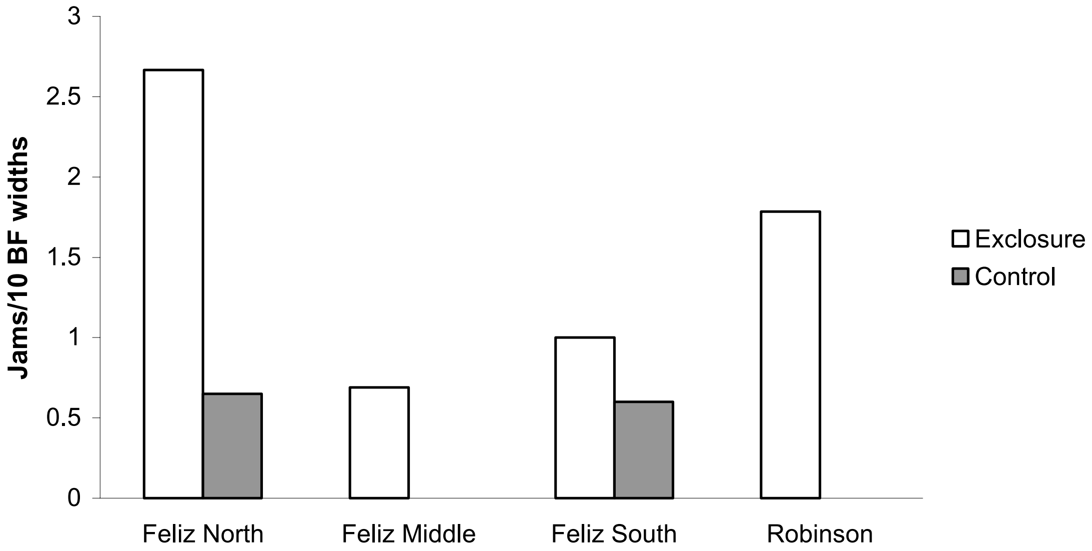
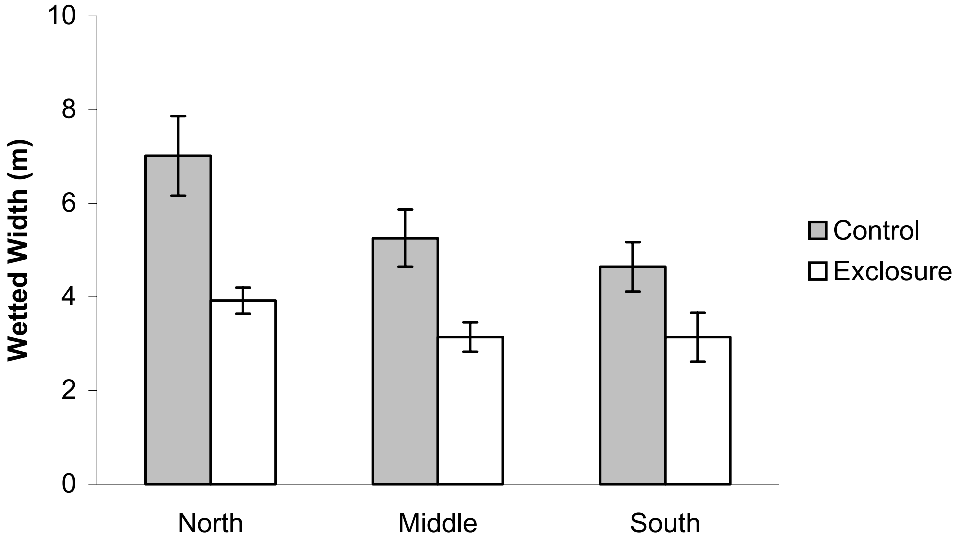
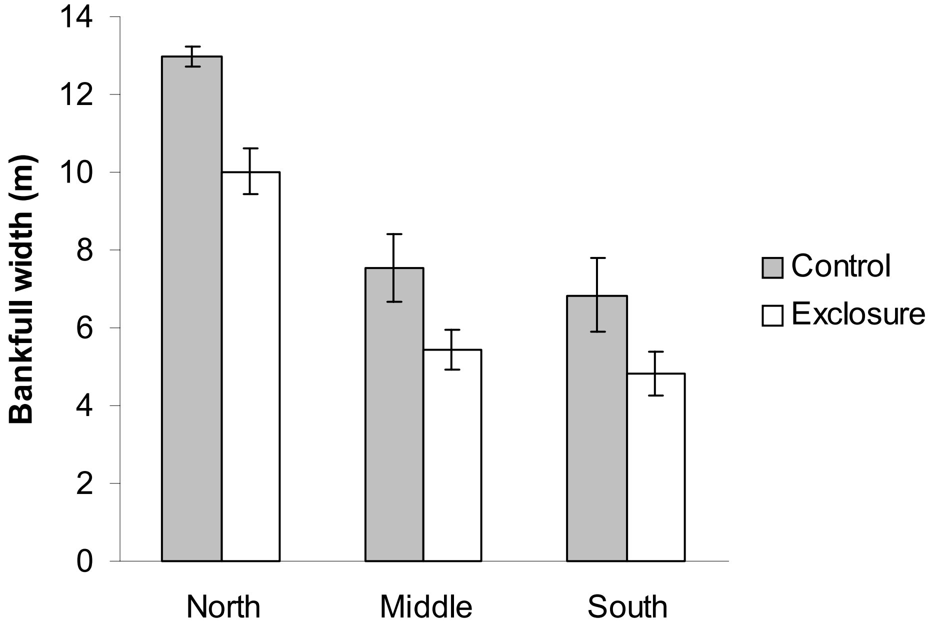


Figure 6a. Wetted widths between control (gray) and exclosure (white) reaches on the three Forks of Feliz Creek.

Figure 6b. Bankfull widths between control (gray) and exclosure (white) reaches on the three Forks of Feliz Creek.

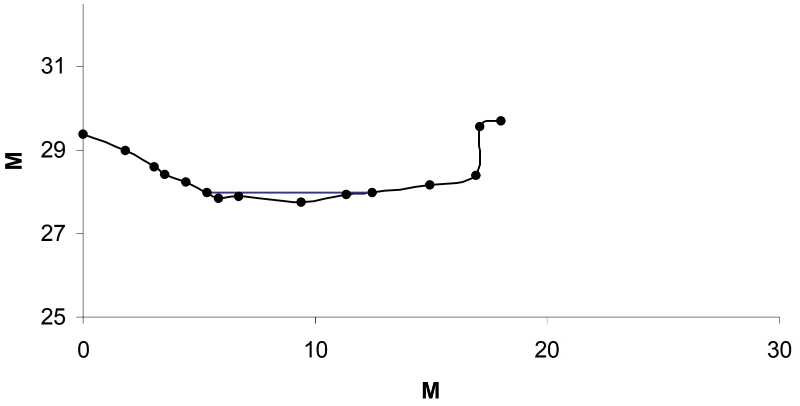


6a.

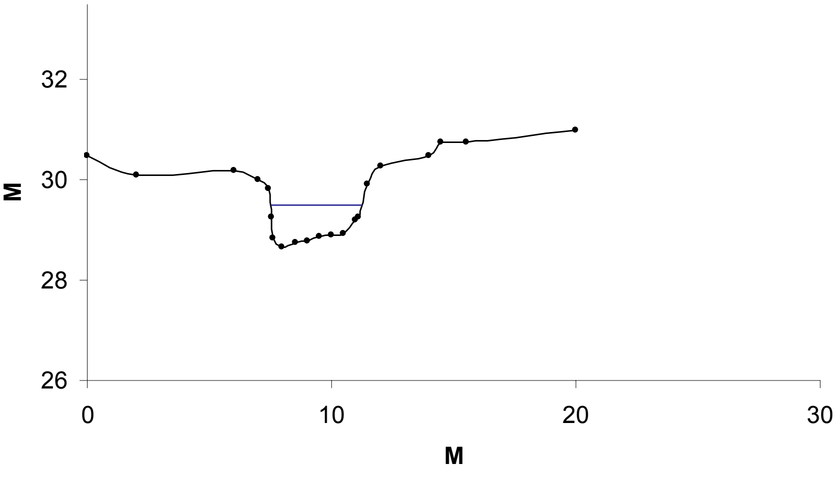


6b.

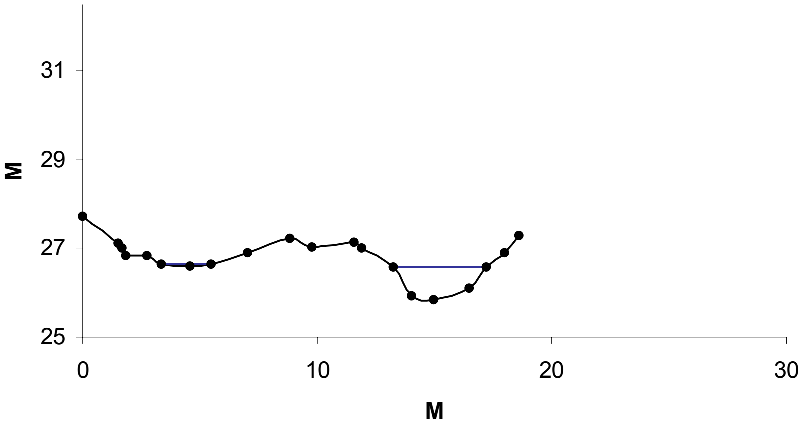
Figure 7. Cross sections in the North Fork of Feliz Creek. A. Typical cross section above the fence line. B. Typical cross section within the exclosure showing a narrow, single channel. C. Part of the reach within the exclosure is separated into two channels separated by a vegetated island. The smaller channel on the left functions as a high flow channel, and had very little flow at the time of the survey.



A.

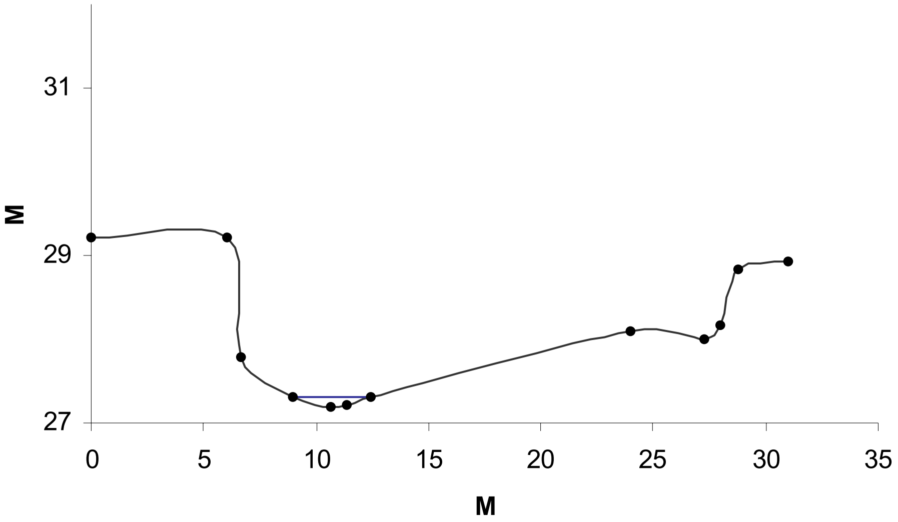


B.

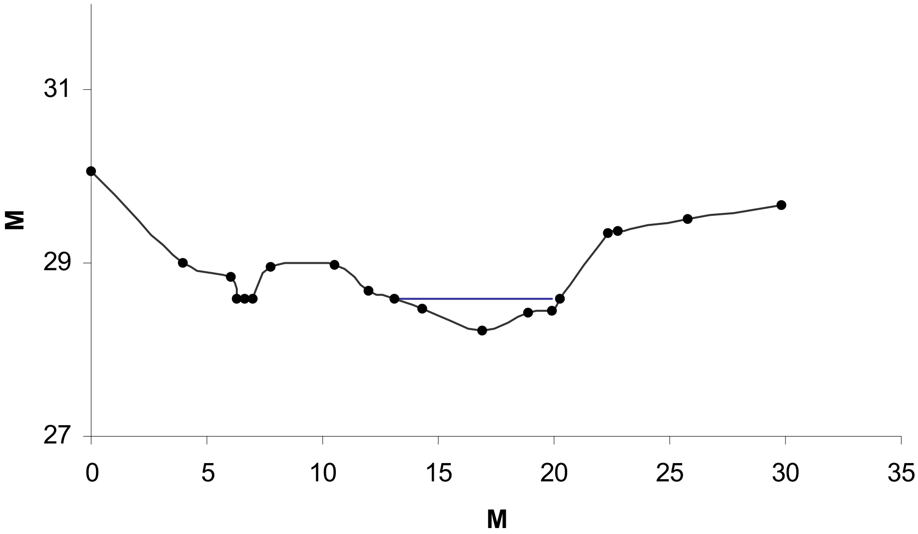


C.

Figure 8. A. typical cross section in the control section, Robinson Creek. B. Typical cross section in the exclosure section.

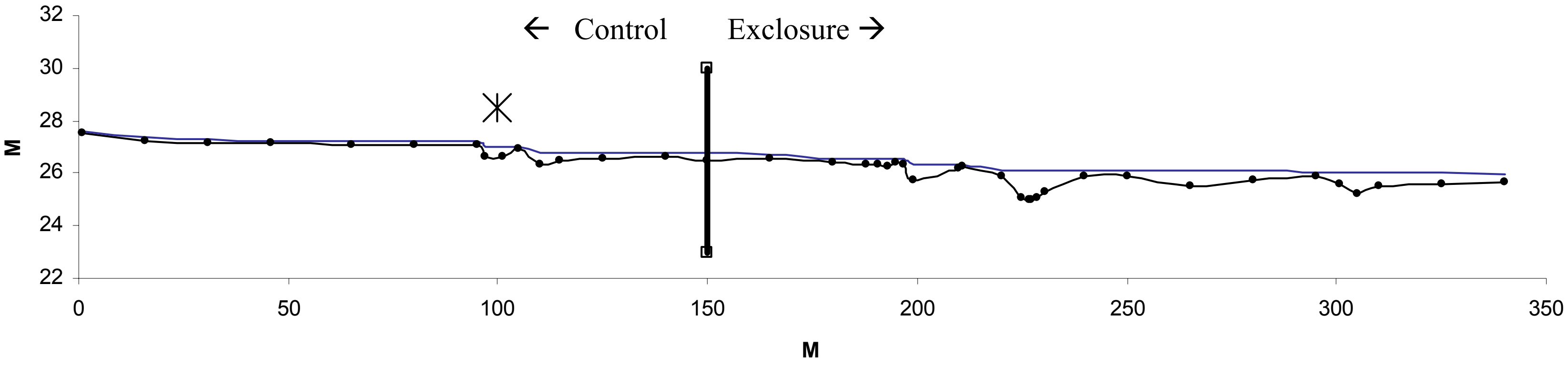


A.



B.

Figure 9. Long profile of Robinson Creek. The * indicates the location of rip-rap to prevent bank erosion that was contributing to the formation of pools.



CHAPTER 6

LANDSCAPE-SCALE INFLUENCES ON SALMONID SPAWNING HABITAT IN RUSSIAN RIVER TRIBUTARIES

Landscape-scale Influences on Salmonid Spawning Habitat in Russian River Tributaries

Abstract

Aquatic habitat responds to land use at various spatial and temporal scales. Understanding which scales influence specific characteristics of instream habitat can direct restoration and management efforts to focus on the appropriate scales. In this chapter I examine the relative influences of land use and land cover (LULC) at various scales on levels of fine sediment within spawning gravels (embeddedness) in 54 stream reaches in the Russian River basin. Eight different scales of analysis were explored, including the entire watershed, the local riparian corridor (30 m buffer), and the riparian corridor of the stream network upstream of the focal reach. Watershed-scale variables for LULC consistently explained the most variation in embeddedness levels. Agriculture, urban, and herbaceous LULC categories had significantly positive relationships with embeddedness, indicating these LULC categories were associated with higher levels of fine sediment. Forest and shrub LULC categories had significantly negative relationships with embeddedness. The explanatory power of models generally increased when the analysis was restricted to larger watersheds. A model using watersheds greater than 2500 ha had the greatest explanatory power. Within these largest watersheds, models for LULC within the whole watershed and for the riparian corridor of the entire upstream stream network explained equivalent levels of variation in embeddedness.

Introduction

Streams integrate influences across multiple spatial and temporal scales (Allan et al. 1997, Allan and Johnson 1997, Harding et al. 1998). Therefore, aquatic habitat at any given site is a function of variables that act, and have acted historically, at the local, reach, and watershed scales. Attempts to explain and predict fish population dynamics based only on site or reach-specific habitat features are likely to miss important processes operating at larger scales (Dunham and Vinyard 1997, Roper et al. 1997) and, thus, scientists and managers are increasingly focused on understanding how aquatic habitat is affected by land use both immediately adjacent to streams (i.e. riparian corridors; Lowrance et al. 1984, Peterjohn and Correll 1984, Lowrance et al. 1985, Jones et al. 1999) and throughout the contributing watershed (Karr and Schlosser 1978, Frissell 1992, Charbonneau and Kondolf 1993, Reeves et al. 1993).

Riparian buffers are frequently recommended as the most effective means of protecting streams from the negative impacts of land use (Erman et al. 1977, Erman and Mahoney 1983, Barton et al. 1985, Castelle et al. 1994, Davies and Nelson 1994, Tang and Montgomery 1995) and many regulations designed to protect streams focus on riparian buffers (Hall 1997, Hairston-Strang and Adams 1998, Ligon et al. 1999). However, there is some concern that an emphasis on riparian buffers to protect streams may overlook important influences on stream habitat that operate at larger scales (Frissell 1992, Roth et al. 1996) .

With the evolution of GIS analysis capabilities and the emergence of landscape ecology, a number of studies have tried to understand the relative influence of land use at various sales on instream habitat (reviewed in Allan and Johnson 1997, Johnson and

Gage 1997). These studies have alternatively concluded that land use within the immediate riparian corridor has the greatest influence on stream habitat (Jones et al. 1999, Lammert and Allan 1999, Stauffer et al. 2000, Sponseller et al. 2001), and, conversely, that the land use over an entire watershed has the greatest influence on stream habitat (Omernik et al. 1981, Osborne and Wiley 1988, Hunsaker and Levine 1995, Roth et al. 1996, Wang et al. 1997). However, these studies have utilized diverse response variables, sampling schemes and spatial scales. As a result, even studies conducted in the same watershed (e.g. Roth et al. 1996, Lammert and Allan 1999) have reached contradictory conclusions (possible explanations of these different conclusions are discussed in Allan et al. (1997)).

Studies comparing the relative influence of land use within the riparian corridor to land use in the overall watershed have often had small sample sizes (e.g. fewer than 10 streams; Lammert and Allan 1999, Wente 2000, Sponseller et al. 2001) or have used land-use data sources with a coarse resolution unable to distinguish the riparian corridor from the larger watershed (e.g. a minimum mapping unit of 4 ha; Richards and Host 1994, Wang et al. 1997) and, thus, can only test for watershed-scale relationships. Nearly all of the studies that have explored the relative influence of land use at various scales have been conducted in the eastern and Midwestern U.S.

General conclusions about the relative influence of riparian and watershed-scale land use on stream habitats are difficult to draw due to the narrow geographic focus of the studies, the considerable variety of response variables (including nutrients, fine sediment, channel morphology, invertebrates, fish, and aggregate habitat scores), and the differences in methods, scales, and data resolution. Further, studies that have considered historical land uses (Knopp 1993, Harding et al. 1998) conclude that past (50 – 100 years ago) land use can have greater explanatory power for channel morphology and invertebrate biota than current land use. These studies question the utility of studying current patterns of land use to predict or explain instream conditions.

Although studies of landscape-scale influences on stream habitats have been tasked with the considerable challenges described above, they address a pressing research question for land managers and decision makers: Can stream habitat be protected by a strategy that primarily focuses on the regulation of land use adjacent to streams (i.e. protection of riparian buffers and stream set-back ordinances)? This question is of particular importance for the conservation and restoration of anadromous fish, whose populations have declined dramatically along the Pacific Coast of the United States (Nehlsen et al. 1991, National Research Council 1996). Riparian buffers are the primary strategy for protecting salmonid freshwater habitat and have demonstrated the potential to do so (Erman et al. 1977, Davies and Nelson 1994). However, recent research has also shown that salmonid habitat and population dynamics can be influenced by land use at the watershed scale (Bradford and Irvine 2000, Thompson and Lee 2000, Paulsen and Fisher 2001, Sharma and Hilborn 2001).

The conflicting results from the studies comparing the influences of watershed and riparian scales may indicate that a question of which scale exerts the most influence is overly simplistic. Different components of aquatic ecosystems likely respond to processes operating at different spatial (Richards et al. 1996, Dunham and Vinyard 1997, Jones et al. 2001) and temporal (Harding et al. 1998) scales. Instead, the important challenge may be to elucidate which habitat components respond to which scales.

Answering these questions can help direct restoration activities toward the appropriate scales.

This study investigates the relationship of land use and land cover (LULC), across various scales, on patterns of fine sediment in streams within the Russian River basin, California. The production of fine sediment across a watershed is influenced by current land-use practices (Dietrich et al. 1989, Platts et al. 1989, Kinerson and Dietrich 1990, Lisle and Hilton 1999, Dunne et al. 2001). High levels of fine sediment negatively affect anadromous fish habitat, with elevated sediment in spawning gravels correlated to low survival of salmonid eggs and alevins (Everest et al. 1987, Reiser and White 1988, ASCE 1992). Excessive fine sediments within spawning gravels can lower gravel permeability, reducing the exchange of dissolved oxygen and metabolic wastes through the gravels and can also physically impede the emergence of alevins (Kondolf 2000).

The number of watersheds included in this investigation (54) is among the largest for studies of landscape-scale effects on streams and this is the first study that I'm aware of that investigates these relationships in a Mediterranean-climate system. Further, this region of California has significantly greater relief than the regions considered in previous studies of landscape-scale influences on streams (the Midwest and Eastern U.S.).

Study Area

The Russian River basin is located in northern coastal California, north of San Francisco Bay (Figure 1). The basin has a Mediterranean climate with cool, wet winters and hot, dry summers. The majority of precipitation falls between December and March, with little or no rain between May and October. Conifer-dominated forests occur near the coast and intermittently throughout the basin on north-facing slopes. However, the majority of the basin is dominated by mixed-hardwood forests, oak savannas, and grasslands (Chapter 2). Land use is varied and includes vineyards, orchards and other agriculture, sheep and cattle grazing, timber harvest, and urban, suburban, and exurban communities. Currently, there are high rates of land-use change with conversions from natural vegetation to vineyard and suburban and exurban development (Merenlender et al. 1998, Merenlender 2000). The basin provides habitat for several species of anadromous fish, including chinook (*Oncorhynchus tshawytscha*) and coho salmon (*O. kisutch*) and steelhead trout (*O. mykiss*). All three species are currently listed under the Federal Endangered Species Act as being threatened or endangered. The Russian River basin is listed as impaired due to sediment under Section 303(d) of the Federal Clean Water Act.

Methods

Instream data and the Embeddedness Index

Instream data were collected during CDFG habitat-typing surveys in the Russian River basin between 1997 and 2000. Streams were divided into reaches (*sensu* Rosgen 1994) and assigned sequential numerical reach numbers. Field crews estimated the embeddedness of the substrate of the “pool tail out” at each pool encountered. Embeddedness is defined as the degree to which spawning-sized substrate (e.g. gravel and small cobble) is surrounded by fine sediment (sand and silt). Embeddedness was measured on a 4-point scale, ranging from 1 (very low) to 4 (high). Reaches were then assigned embeddedness scores based on the proportion of units within the reach having a given embeddedness value. For example, consider a reach with 20 pools with two pools having a score of '1,' three having a score of '2,' five having a score of '3' and 10 having a score of '4.' These scores would be aggregated at the reach level in the following manner: 1: 10%; 2: 15%; 3; 25%; 4: 50%. Thus, higher percentages within the '4'category indicates a stream with high embeddedness levels. CDFG descriptions of reaches often combine the proportion of pools with a '3' or '4' to characterize the proportion of sampled units within a reach with "high embeddedness". The hypothetical reach described above would have a "high embeddedness" score of 75.

The embeddedness score is essentially a qualitative method made by multiple observers. Different observers using qualitative scales are generally able to agree on extreme values but qualitative scales have much reduced repeatability when they include intermediate scores (F. Kearns, UC Berkeley, unpublished data). Therefore, I only considered the endpoints of the embeddedness spectrum: '1' and '4.' I defined an 'embeddedness index' for each reach by subtracting the '1' value (proportion of units with a score of '1') from the '4' value (proportion of units with a score of '4'). Thus the embeddedness index can range from negative 100 (all units have very low embeddedness) to positive 100 (all units have high embeddedness). The hypothetical reach described above would have an embeddedness index of 50 – 10 = 40. To compare this embeddedness index to other uses of the data, I conducted a linear regression analysis of the index values versus the following other methods of describing the data: 1) proportion of units with a score of '4;' 2) high embeddedness as described by CDFG; and 3) a composite score, based on a weighted average, that reduces to a single number the proportions of units in each embeddedness category (1-4). The composite score was calculated with the following formula:

$$[(\% \text{‘1’} * 1) + (\% \text{‘2’} * 2) + (\% \text{‘3’} * 3) + (\% \text{‘4’} * 4)]/100$$

The hypothetical reach described above would have a composite score of 3.15. A reach with the following proportions - (1: 75%; 2: 10%; 3: 15%; 4: 0%) - would have a composite score of 1.4.

For each stream the first reach defined by field survey crews was designated Reach 1. Subsequent reaches were designated Reach 2, Reach 3, etc (Figure 2). On each stream, therefore, the watershed contributing to a Reach 3 is nested within the watershed contributing to Reach 2, and both are nested within the watershed contributing to Reach 1. To increase the number of independent samples, I initially only considered the sub-set of Reach 1's (n = 122). However, because tributaries to major streams were considered separate streams, this sample of Reach 1's does contain a limited number of watersheds that are nested within larger watersheds.

The analysis was restricted to reaches with a gradient less than 0.03 (i.e. "response" reaches that are most likely to be influenced by elevated levels of fine sediment (Lunetta et al. 1997); n = 67). Because the embeddedness index essentially derives an average score for a reach, I excluded reaches with fewer than ten sampled pools to avoid averaging a small sample size (n = 54 Reach 1 reaches with at least 10 pools and a gradient less than 0.03; Figure 1). Because the embeddedness index was calculated using the extremes (embeddedness values of '1' and '4') I also wanted to avoid reaches with no values in the extremes. Removing the reaches with less than 10 pools also removed several of the reaches with no units with a score of '1' or '4.' Within the sample of reaches with less than 10 pools, 46% had no units with an extreme value (i.e. composed of only units with an embeddedness value of ‘2’ or ‘3’), while only 11% of the reaches with more than 10 pools had no units with extreme values.

GIS data and methods

Through dynamic segmentation and calibration reach-scale data were entered into a GIS (Byrne 1996, Radko 1997). The GIS was used to calculate stream gradient, designate upstream watersheds, buffer reaches, and quantify land uses within the buffers and across the watersheds.

I created a merged 10-m Digital Elevation Model (DEM) (i.e. a single Arc/Info grid) for the entire Russian River basin. I used the Arc/Info DEMLATTICE command to convert 30 of the individual DEMs into GRID format. I used the ArcView SDTS Grid import extension version 2.0a (2001) by Craig Goodwin & David Tarboton to convert the other 9, which were in “SDTS raster” format (Spatial Data Transfer Standard). I used the LATTICEMERGE command in Arc/Info to merge the DEMs.

To estimate reach gradient, I wrote an Avenue program in ESRI’s ArcView 3.2a to place a point at the beginning and end of each CDFG-defined reach. The program sampled the elevation value in meters from the merged 10m DEM at these points. To ensure that the sampled cells were within a local low point (i.e. a channel) as defined by the DEM, I manually moved the beginning and ends of each reach into the DEM-defined stream channel, as indicated by the local high values of the flow accumulation grid. This avoided inadvertent sampling of the elevation of an adjacent hillslope cell (which could have considerable influence on the calculated gradient) due small differences in the spatial location of the DEM-defined channel and the hydrography layer, to which the dynamically segmented reach was linked. I used the FlowZones 1.2 ArcView 3.2

extension by Eugene Martin of CommEnSpace to automate the movement of reach end points into the DEM-defined channel (Martin 1999). However, this program did not always move the points into the correct channel, and would not move points that were within 1.5 meters of the local high flow accumulation cell. I found it necessary to review every reach end point point to check that it was within the flow accumulation channel. Once completed, I sampled the elevation values underneath the moved beginning and end points. I converted the field-measured length of the reach into meters, and calculated the gradient as the ending elevation minus the starting elevation, divided by the field-surveyed length of the reach.

Land use/land cover (LULC) variables were derived from the "LCMMP, Vegetation Data, 1994-1997," or Calveg, classified from 1994 Landsat TM imagery with a 2.5 acre (1ha) minimum mapping unit. The data were originally produced by the USDA Forest Service, but are now distributed by the California Department of Forestry and Fire Protection (CDF). This layer is particularly valuable for this work because it differentiates between agriculture and grassland, which most vegetation layers do not do. I used aggregated LULC categories of Agriculture (row crops, vineyards, and orchards); Herbaceous (annual grasslands); Forest (including hardwood, conifer, and mixed evergreen forests); Shrubs (generally chaparral); and Urban.

To estimate the influence of roads on streams I intersected two sets of road and stream layers to find the number of road and stream crossings in the Russian River basin. First I intersected the 1:100,000 scale U.S. Census Bureau TIGER/line roads (1998 version) with the USGS 1:100,000 scale Digital Line Graph (DLG) hydrology layer. Then I intersected the 1:24,000 scale USGS DLG hydrology layer (i.e. the rivers and streams that appear on the 7.5-minute USGS quadrangle maps, and date from the 1960s to the early 1990s) with the 1:24,000 scale DLG roads (with the same update date as the quad sheets). The advantage of the 1:100,000 scale roads is that they have been updated much more recently than the USGS 1:24,000 scale roads, but the 1:24,000 hydrology is considerably more detailed than the 1:100,000 scale.

I used two methods to represent the influences of development. In areas with high-density development (e.g. incorporated cities) the LULC class for urban can represent developed areas. However, the unincorporated parts of the county have very few areas with development that is dense enough to be classified as urban in the LULC data. Within these generally low-density areas housing density can still vary greatly between watersheds that have no land classified as urban. For example, similarly sized watersheds containing either a single ranch or a rural subdivision (e.g. 40-acre parcels) would both be classified from the Calveg data as being composed primarily of herbaceous, forest and shrub. The watershed with the rural subdivision, however, could have 50 houses compared to the single house and a few outbuildings on the ranch. Thus, in the unincorporated areas of the county I used parcel-level data to estimate housing density. The Sonoma County Assessor's Office provided data on the number of residential structures in 2001 for each parcel in unincorporated parts of Sonoma County, as of approximately 1997. I then joined the Assessor's data to the parcel data, which had previously been digitized.

I used the Wentworth (1997) database of geologic materials for the San Francisco Bay Area (which included Sonoma County) to represent the geology within the Russian River basin. I reclassified the primary geologic types into 14 general categories based on the “General PTYPE categories” described by Wentworth (1997) and Ellen and Wentworth (1995). The most common categories for the study area were Volcanic rock, Franciscan sandstone, and Franciscan mélange. I selected only the reach watersheds located entirely or mostly within Sonoma County. Because most of the watersheds were within Sonoma County only a few Mendocino County watersheds did not have geology data. These few Mendocino County watersheds were excluded from models which included terms for geology.

Scales of Analysis

I used eight methods of analysis, within three spatial scales, to investigate the influence of land use/land cover (LULC) on instream variables. The three spatial scales of analysis were the entire watershed upstream of the focal reach (*watershed*), the riparian corridor of the focal reach and/or immediately upstream of the focal reach (*riparian*), and the riparian corridor of the entire stream network upstream of the focal reach (*network*). Figure 3 illustrates these various methods and scales of analysis.

Watershed (*watershed*)

Whole watershed (WSHED). For this scale of analysis I calculated the proportion of various LULC classes, as well as geology, road and housing density, within the entire watershed upstream of the downstream end point of the survey. I used the Flowzones extension in ArcView and the merged DEM to derive a unique watershed above each reach beginning point. Some reach points were not correctly located at the beginning of the reach, so I had to manually place those. Each reach was uniquely identified with the GIS reachid (PNMCD + reach number in 01, 02, etc. format) and a Flow Zone index number (FLOZINDX).

The merged 10m DEM had to be processed before a watershed could be delineated. First, I used Flowzones to fill DEM "sinks" to allow for the correct calculation of flow direction. Then I calculated flow direction so that each cell would show the direction of the "steepest drop" to adjacent cells (Greenlee 1987, Jenson and Domingue 1988). From this, I used Flowzones to calculate the flow accumulation value (Jenson and Domingue 1988, Tarboton et al. 1991) where each cell had the cumulative total of the number of cells that flowed into it. Once I had the flow accumulation grid, I could then use the Flowzones "Delineate Flowzones" tool to create the watersheds.

To calculate the amount of landcover types for every watershed, I first converted the Calveg LULC layer into a grid using the ArcView 3.2 "Convert to Grid" function, with a 5x5m (25 square meters) cell size. Using this small cell size, I was able to get precise values of Calveg landcover types for each reach watershed using the "Tabulate Areas" function in ArcView 3.2, which is available with the Spatial Analyst extension. This function finds the areas of each item type in one layer that intersect with the boundaries of a polygon layer or grid in another layer. I used the same process to clip and quantify geology for each watershed. I calculated the density of road-stream crossings within each watershed using the "Calculate Points" extension written by Yingming Zhou (2000, http://arcscripts.esri.com/details.asp?dbid=10715).

To calculate the housing density within a watershed, I intersected the parcel data with the watershed for each reach. I calculated the proportion of each parcel that was within each watershed, and multiplied the number of residential structures in the parcel by that proportion. I compared this residential structure proportion to the number of residential structures that would be in the watershed if I assumed that any part of a parcel being in a watershed meant all of that parcel's residences were in the watershed. This gave a likely versus maximum number of parcels per watershed. I calculated the square kilometer area of each watershed, and then calculated the likely and maximum number of residential structures per square kilometer for each watershed. Of the watersheds analyzed, four of them had significant portions within incorporated areas, for which I did not have parcel data, and I tagged these for possible removal from further analyses.

Hydrologic Proximity Model (HPM). This method of analysis used the same watershed boundaries as the first method, but weighted the LULC class in a given cell by the proportion of drainage that flowed through that cell (described in Wente 2000). Because of the imprecision of the available precipitation data, I decided to use a single average precipitation value for the Russian River basin (of 112 cm), which also simplified the calculation process. First I calculated the volume of water traveling through each cell (VOLi) by multiplying the flow accumulation grid plus one, the average precipitation value, and the area of each cell (10x10m, 100 square meters). Then I calculated the total volume (TVOL) by performing a flow accumulation calculation with the VOLi values as the weight grid, but still using the flow direction grid as the other input layer. I then wrote a script in Avenue for ArcView 3.2 to calculate the HPM value of each LULC type by reach watershed, using the Avenue command ZonalStatsTable. These HPM-weighted values could then be compared to the unweighted areas of each LULC type for the watershed.

Slope-weighted model (SLOPE). This method of analysis also used the same watershed boundaries as WSHED but weighted cells by their slope. To weight the LULC types by their percent slope for each reach watershed, I modified the HPM Avenue script to allow for the use the slope grid instead of the total volume HPM grid. This enabled the script to sum the total percent slope values for each cell of each landcover type within a reach, resulting in a proportion of slope totals that could be compared to the unweighted areas of each landcover type for the watershed.

Riparian buffer (*riparian*)

Local buffer (LOCAL). I buffered each reach to a distance of either 30 m or 60 m and clipped the LULC within these buffers. Using the line layer of DFG habitat reaches, I buffered each reach by 30 m and 60 m using the ArcView 3.2 "Buffer Wizard" along with the "Buffer with attribute" script written by Thad Tilton of ESRI (2000, http://arcscripts.esri.com/details.asp?dbid=11281). The standard ArcView 3.2 buffer tool does not allow for an identifying attribute to be copied over from the layer being buffered, so Tilton's script was necessary. I then used the tabulate areas function again to calculate the amount of LULC types for each reach for the 30-m and 60-m buffer polygons. Subsequent analysis found that 30-m and 60-m buffers were nearly identical in their LULC proportions (e.g. the r^2 for agriculture, herbaceous, and forest in 30 and 60 m buffers = 0.99) so in this chapter I only consider the 30-m buffer.

Upstream buffer (UPSTRM). This method of analysis considered a 30-m buffer of the stream (and any tributaries) for 1 km upstream of the end of the reach (the 'focal reach'). I worked with Tracy Love of CDFG Region 3, who had written code that would do this. (Tracy's code, written in 2002, is available at http://arcscripts.esri.com/details.asp?dbid=10857). The reaches were divided into those derived from 1:100,000 scale dynamic segmentation and those derived from 1:24,000 dynamic segmentation so that the identified reach endpoint would be near the correct source hydrology layer. When a tributary stream was encountered that flowed into a selected segment, part, or all, of that stream was also selected so that the total distance from the reach end was 1km. For example, if a tributary was encountered 700 m upstream of the focal reach endpoint, the program would travel along both the main stem and the tributary for an additional 300 m. Once the point (or points) 1 km upstream from the focal reach endpoint was selected, I used the buffer-with-attribute script and the Buffer Wizard to buffer this line to a distance of 30 m. I then calculated the LULC variables within this buffer.

Local-upstream buffer (LOCUP). This scale of analysis combined the buffers from the previous two methods, LOCAL and UPSTRM, to provide a comprehensive representation of the riparian corridor.

Stream network buffer (*network*)

Blue-line stream network buffer (BLUE). I derived a stream network upstream of each focal reach using the 10-m DEM. I decided to derive a stream network that was approximately as dense as networks found on recent USGS maps for the region. This was preferable to simply buffering USGS blue-line streams (i.e. those streams that are depicted on USGS 1:24,000 quad sheets) because the density of USGS blue-line stream networks is strongly dependent on the year in which the map was made or updated and, thus, not uniform across the study area (Figure 4). To remove this bias, I derived new stream networks using the merged 10m DEM, using the “Stream Definition” function of the Watershed Delineator ArcView 3 extension written by Zichuan Ye and Dean Djokic of ESRI in 1997 for the Texas Natural Resource Conservation Commission. Through visual inspection I found that a minimum flow accumulation value of 1000 cells (10 ha) as the threshold for channel initiation approximated the denser stream network of more recently mapped USGS quads. I used the Watershed Delineator's "Line GRID to Shape" function to convert the linear stream grids into line shapefiles. Because some surveyed streams were tributaries to other surveyed streams, some of these watersheds overlapped. As I was using the AV3.2 Geoprocessing Wizard's intersect function, I could not have overlapping watersheds when doing the intersection. I labeled each watershed with its watershed "level", where the first watershed on a stream network was level one, the next one further upstream in the watershed's stream network was level two. The maximum watershed level encountered was four. The four levels of watersheds were separated into four different shapefiles. Each of these was intersected with the 10ha stream network to select just the stream segments that were within the individual watersheds. They were then recombined into a single layer, and buffered using the Geoprocessing Wizard and the buffer-with-attributes script. I then used the tabulate areas function by the watershed's reachid value and the Calveg LULC types.

Dense network buffer (DENSE). Because blue-line streams do not include all the channels that exist on the landscape (Hansen 2001), which can have important implications for modeling land-use effects on streams (Kelly et al. 1999), I also created a densified stream network using the same method as for BLUE but with a minimum area to generate a channel of 2 ha (Figure 4).

Statistical analysis

I used multiple linear regression to explore relationships between LULC at various scales and reaches' embeddedness index. First, I combined LULC variables from multiple scales and methods into a single model and used backwards stepwise regression to test for significant terms. To avoid collinearity I used variables from methods of analysis that were not highly correlated with each other (e.g. correlations < 0.50; Table 1). Thus, this combined model included LULC variables from WSHED, LOCAL, and UPSTRM, along with variables for geology and housing and road density. Variables from the other *watershed* and the *network* methods were highly correlated with WSHED and were not used. The approach of entering variables for LULC at multiple scales into a single model replicates that of other recent studies (Roth et al. 1996, Sponseller et al. 2001).

To compare the explanatory power of the eight different methods of analysis, I entered the LULC classes for each scale, both with and without geology variables, into backward stepwise regression to develop the best fit model for each scale. These analyses were conducted on all watersheds as well as sub-groups of watersheds of various sizes: < 1500 ha, > 1500 ha, and > 2500 ha. The sample was split into watersheds greater and less than 1500 ha to separate "smaller" and "larger" watersheds into groups with approximately similar sample sizes.

I also explored these relationships with two additional data sets: all reach 2's and all reach 3's with at least 10 pools and less than 0.03 gradient. Each of these data sets is composed of watersheds that are spatially nested within the watersheds for the main data set of reach 1's. I did not derive the stream network methods BLUE and DENSE methods for these reaches, and thus these analyses only compare *watershed* and *riparian* scales of analysis.

Results

Forest was the dominant land cover within both watersheds and riparian buffers (Table 2). At the watershed scale, agriculture and forest were negatively correlated

(Table 3). Among the scales of analysis, WSHED and SLOPE provided nearly identical information, while HPM was somewhat correlated with WSHED. Both BLUE and DENSE were highly correlated with WSHED, while the riparian scales were less correlated with WSHED (Table 1).

The proportion of the total watershed area included within the BLUE buffer averaged 0.09 ± 0.003 and the proportion of watershed area included within the DENSE buffer average 0.23 ± 0.008. These average proportions did not vary with watershed size.

The embeddedness index was highly correlated with other methods of utilizing the embeddedness data (Figure 5). The closest relationship ($r^2 = 0.95$, $F_{1, 52} = 946.8$, $p = < 0.0001$) was with the composite score, which utilized the additional information of the proportion of units with embeddedness scores of 2 and 3. This indicates that the embeddedness index was effective at representing the full range of embeddedness, from very low to very high, while only utilizing the end points of the qualitative scale.

Best-fit combined model with watershed and riparian terms. A backward stepwise regression using all LULC classes from WSHED, LOCAL, and UPSTRM, and geology, and housing and road-stream crossing density indicated that the best-fit model included only WSHED terms for Agriculture, Herbaceous, and Urban ($r^2 = 0.50$, $F_{3, 50} = 16.3$, $p = < 0.0001$; Table 4). Repeating the backward stepwise regression with the same variables and stratifying by watersheds greater than 1500 ha and less than 1500 ha also indicated that the best-fit model only included WSHED LULC variables (Table 4). The amount of variation explained by the model was greater for larger watersheds ($r^2 = 0.57$, $F_{4,26} = 8.8$, $p = 0.0001$) than smaller watersheds ($r^2 = 0.25$, $F_{1, 21} = 6.9$, $p = 0.02$). The amount of variation explained by the model was increased still further by limiting the analysis to watersheds greater than 2500 ha ($r^2 = 0.79$, $F_{2, 19} = 35.6$, $p = < 0.0001$). Throughout these models, coefficients for urban, agriculture and herbaceous LULC classes had positive slopes, indicating they were positively correlated with embeddedness, while model terms for forest and shrub LULC classes had negative slopes.

Comparison of the eight methods of analysis. The WSHED scale was the most effective in explaining variation in embeddedness followed by DENSE and BLUE (Table 5). *Riparian* models with geology variables generally explained more variation than *riparian* models lacking geology terms. *Watershed* and *network* models showed little difference in r^2 whether or not geology terms were included (Table 5), and the best-fit *watershed* and *network* models generally did not include terms for geology. Geology terms may have improved *riparian* models because of positive correlations with land use; LULC classes and geology were not distributed independently (Table 6). The proportion of agriculture within a watershed was negatively correlated with the proportion of Franciscan mélange and Franciscan sandstone. Herbaceous was also negatively correlated with Franciscan sandstone and somewhat positively correlated with Franciscan mélange. The proportion of forest was positively correlated with the proportion of Franciscan sandstone and mélange. Without the geology variables *riparian* models explained very little of the variation in embeddedness.

Models that only included watersheds greater than 1500 ha consistently explained more variation than models that included only watersheds less than 1500 ha; models restricted to watersheds greater than 2500 ha explained the most variation in embeddedness (Table 5; Figure 6). Within these largest watersheds, models for WSHED, BLUE and DENSE were of similar strength.

Reaches 2 and 3. The best-fit models for reaches 2 and 3 also only included WSHED LULC terms with similar coefficients as models using the primary data set. To examine the consistency of these models I compared models from the three data sets using the same parameters (WSHED forest and shrub). Models from reaches 2 and 3 had similar intercepts and slopes for forest as did the model from reach 1. The slope of the term for shrub was more variable (Table 7; Figure 7).

Discussion

Land use and land cover variables were effective predictors of the levels of embeddedness of spawning substrate in streams within the Russian River basin. The LULC categories in the entire upstream watershed explained the most variation in embeddedness. Land use/land cover variables within buffers at the reach scale and 1 km upstream had very low predictive power, while LULC variables for buffers of the blue-line stream network and a densified stream network had intermediate explanatory power. Explanatory power was increased for *watershed* and *network* models when using only larger watersheds (>1500 ha). The largest watersheds' (>2500 ha) LULC variables explained a high proportion of the variability of embeddedness, with the buffers of the blue-line stream and a densified stream network having nearly identical explanatory power as the overall watershed.

Within all models, LULC categories for agriculture, herbaceous, and urban had positive coefficients, indicating a positive association with embeddedness. Land covered in native vegetation generally has the lowest levels of sediment production (Leopold et al.

1964, Dunne and Leopold 1978). In this study, forest and shrubland had negative coefficients, indicating that these LULC categories were associated with lower embeddedness. Conversely, urban areas contribute large amounts of fine sediment during periods of construction (Dunne and Leopold 1978) and can continue to contribute sediment to channels through bank erosion as channels adjust to increased peak runoff (Trimble 1997, Pizzuto et al. 2000). All herbaceous land in the Russian River was historically influenced by grazing and most continues to be grazed, although generally less intensively than in the past. Grazing can increase sediment yield from hillslopes (Dunne and Leopold 1978, Myers et al. 1985, Charbonneau and Kondolf 1993) . A study of a Northern California coastal watershed with mixed land cover (including forest and grassland) reported that a disproportionate amount of sediment production within the Van Duzen basin came from gullying and slumping in grazed grasslands (Kelsey 1980). Kelsey (1980) hypothesized that historic grazing had increased sediment production over baseline levels primarily through replacement of native bunchgrasses with shallower-rooted annual grasses, which reduce the ability of the vegetative mat to resist erosive processes. Variables for agriculture generally had the highest sum-of-squares in the multiple regression models, indicating that the proportion of land within agriculture explains the most variability of embeddedness levels. Agriculture can lead to significantly higher rates of sediment production, even on moderate slopes, due to the increased amount of bare soil exposed to the erosive power of raindrops and sheet wash (Dunne and Leopold 1978). The export of non-point source pollution, including fine sediment, can be increased in agricultural areas where drainage management techniques promote runoff to directly bypass riparian buffers (Osborne and Kovacic 1993).

Bradford and Irvine (2000) reported that rates of decline in coho salmon populations were positively correlated with agricultural land use within 40 tributary watersheds of the Thompson River, British Columbia.

Models based on larger watersheds invariably had greater predictive power than models restricted to smaller watersheds. The results of this study demonstrate relationships between spatial patterns and instream variables but cannot resolve the direct mechanisms - such as bank erosion, gullying, and landslides - that contribute fine sediment. These processes are likely unevenly dispersed across the landscape (Kelsey 1980). Thus, a larger area may be more effective at integrating the various processes that contribute to sediment loading, leading to greater predictive power with larger watersheds.

For example, within the Van Duzen River basin, Kelsey (1980) found that sediment production from forested lands came mainly from episodic land slides and debris flows. Across a landscape, some small forested watersheds may have high embeddedness levels due to recent landslides, while other small forested watersheds may not have had recent landslide activity. Thus, reaches within small forested watersheds may have a wide range of embeddedness levels based on spatially variable landslide activity. This would reduce the predictive power of forested land cover for embeddedness within smaller watersheds. Smaller watersheds may also be more influenced by other factors not effectively included in my models due to the limited availability of high-resolution spatial data. This problem is likely reduced within larger watersheds where the scale of the data better matches the scale of analysis.

As the models progressively considered sub-sets of watersheds with greater acreage, the effects of spatially dispersed landslides and site-specific road conditions would tend to be assimilated within larger-scale processes within the overall watershed. Thus, at larger spatial scales, the predictive power of LULC increases and the influence of smaller-scale factors, such as landslides and site-specific road conditions, diminishes.

The Hydrological Proximity Model (HPM), which weighted LULC categories by the amount of water moving through a given cell, had less predictive power than models considering simple proportions of LULC categories, which is contradictory to the findings of Wente (2000). However, it is difficult to evaluate the results of Wente (2000) as his study included only 4 sites, sequentially nested within a single basin (i.e. there was only one independent sample; each subsequent downstream sample included the upstream sample's watershed within its watershed). Further, within his model, the date of sample explained 67% of the variance in the response variable (e.g. $r^2 = 0.67$). Including proportions of land use in the model raised the r^2 to 0.83, and including the HPM in the model raised the r^2 to 0.93. The present study indicates that the HPM approach does not improve predictability for embeddedness over the simple proportion of LULC within a watershed. Different conclusions regarding the utility of the HPM between this study and Wente (2000) may be the results of different topographies, and hence different dominant runoff mechanisms between the studies. Wente (2000) worked within a low-relief agricultural basin in Indiana, while my study considered relatively high-relief mountainous basins. Areas of subdued relief tend to have lower channel densities (Montgomery and Dietrich 1988, Montgomery and Dietrich 1992) and, therefore, a model that assumes runoff occurs through overland flow, as the HPM does, will more closely approximate natural processes. Overland flow is less important in areas of steeper terrain, which are more finely dissected by the channel network (Montgomery and Dietrich 1988, Montgomery and Dietrich 1992).

Road and housing density did not improve models to predict embeddedness. This was somewhat counterintuitive as roads in steep terrain can be a primary mechanism by which land use influences sediment dynamics (Swanson and Dyrness 1975, Montgomery 1994, Weaver and Hagans 1999, Jones et al. 2000), and previous studies have found that survival of juvenile salmonids is inversely related to road density (Paulsen and Fisher 2001). However data on road density did not include information on the topgraphic position, maintenance condition, or surface materials of the road, which can have a strong influence on sediment production (Swanson and Dyrness 1975, Reid and Dunne 1984, Rice and Lewis 1991, Montgomery 1994, Weaver and Hagans 1999, Jones et al. 2000). Further, readily available data sources on roads generally do not include ranch and timber roads, and other roads not built by a public entity (e.g. rural subdivision roads maintained by road associations). These roads tend to be built on steeper slopes and are generally not maintained as well as county roads. Thus the roads most likely to contribute to sediment problems within a basin are the least likely to be included in readily available data sources on roads. A study from a neighboring coastal watershed (the Navarro River basin) found that true road densities, as determined by analysis of large-scale aerial photography, approximately three times greater than values provided by available digital data sources (Viers et al. *in press*). Although digitizing the true road network may greatly improve the ability to understand relationships between land use and embeddedness, this task is beyond the scope of this analysis given the extent of the study area (54 watersheds totaling over 213,000 ha)

Paulsen and Fisher (2001) were working within timberland - public land (US Forest Service) or industrial forests - where the road data were generally more representative of the true road network. Their study area included harvested areas with high road density and wilderness areas with no roads. Even with these conditions, the inclusion of road density only raised the r^2 of a model including terms for time of year and drought severity from 0.51 to 0.54, indicating that, although significant, road density had only minor explanatory power.

Housing density within the unincorporated areas of Sonoma County also had little predictive power. Because housing densities were generally low (mean = 5.1 ± 1.4 residences/km^2; median = 1.4 residences/km^2), the direct impact of structures on runoff is probably very minimal. The road networks serving rural subdivisions may have a much greater influence on sediment dynamics than the housing development itself. However, for this basin, I do not know how rural subdivision road networks compare to ranching and timber road networks, in terms of extent, condition, and maintenance. It is possible that road networks have similar effects on streams within areas with very low density (e.g. ranch or timberland) or moderate density (e.g. a rural subdivision). Although housing density in the low-density unincorporated area, in which land use is dominated by extensive or intensive agriculture, had low explanatory power, the LULC variable for urban had a significant positive relationship with embeddedness. Thus, development with sufficient density to be classified as urban does appear to be contributing to embeddedness in stream channels.

The results of this study provide support for the idea that embeddedness is shaped by processes that operate at large scales, and that the local riparian corridor has little influence on embeddedness levels. This suggests that riparian restoration will have little ability to influence levels of fine sediment within the local reach if conditions throughout the watershed remain unchanged. However, this study does not provide a clear answer to the question of which has greater influence on a stream: land use directly adjacent to the stream network or the overall proportion of land uses throughout a watershed.

For watersheds greater than 1500 ha, the WSHED method of analysis explained somewhat more variability in embeddedness than did BLUE or DENSE. For watersheds greater than 2500 ha, the three methods of analysis explained essentially the same amount of variability.

However, these methods are subsets of one another: all the area contained with BLUE is also contained within DENSE and all the area contained within DENSE is also contained with WSHED. The density of the stream network in the DENSE scale resulted in a buffer that contained on average a quarter (and, for some watersheds, up to 40%) of the total area of the watershed. The proportions of LULC within these three spatial extents were highly correlated. Thus, the explanatory power of the whole watershed scale may be the result of it containing the land surrounding the stream network. Future studies could explore the amount of variability in embeddedness explained by the total proportion of LULC *outside* of a buffer around the densified stream network. This would compare the explanatory power of LULC within the 25% (on average) of the landscape within a 30-m buffer of a drainage network with the 75% (on average) of the landscape outside of this buffer.

To develop the dense stream network, I used a constant critical support area (i.e. threshold of drainage area for channel initiation) of 2 ha. However, field research has indicated that channel initiation is a function of both support area and local gradient (Montgomery and Dietrich 1988, Montgomery and Dietrich 1992). Thus the most accurate representation of a densified stream network should include the influence of local slope on channel initiation. However, obtaining a truly accurate representation of a channel network using a DEM is complicated by the local effects of land use on the density of the channel network: both roads (Montgomery 1994) and grazing (SFEI 2001) can result in headward extension of the channel network.

Finally, this study can only draw conclusions about the relative influences of different scales of LULC patterns on embeddedness, not other components of salmonid habitat. Salmonid populations can be limited by several factors in addition to spawning rates, including pools, cover, and temperature. These factors may be influenced by processes operating at different scales. For example, large woody debris (LWD) likely responds to conditions within the immediate riparian corridor (Chapters 2 and 3) including the size and species of potential LWD sources and the influences of management at the parcel scale.

Although fine sediment has been shown to negatively influence egg and alevin survival in experimental settings, extrapolation of this effect to salmonid population dynamics within a basin is difficult (Everest et al. 1987). Availability of suitable spawning sites is unlikely to limit salmonid populations as a relatively low rate of egg survival can generally produce sufficient numbers of juveniles for the available habitat (Elliot 1984). This may be particularly true in Mediterranean-climate streams where the annual summer drought greatly reduces the physical extent of habitat, indicating that over-summer juvenile survival may act as a population bottleneck. Magee et al. (1996) reported that, although spawning success was limited in watersheds with high levels of fine sediment, sedimentation did not limit recruitment. High survival rate of eggs within the few high-quality redds and increased survival of juveniles appeared to compensate for reduced availability of suitable spawning sites. A study of limiting factors in the Napa River basin, which was declared impaired for sediment under Section 303 (d) of the federal Clean Water Act, revealed that, although loading of fine sediment within spawning gravels may result in elevated mortality, this did not appear to be a primary limiting factor for salmonids (Stillwater Sciences and Dietrich 2002). Tributaries of the Napa River were well seeded with juvenile fish, and, instead, salmonid populations within this Mediterranean-climate basin are likely limited by insufficient deep-water refugia, LWD, cover, and flow during the annual summer drought. Further research may be necessary to determine whether anadromous fish in the Russian River are limited by suitable spawning sites and/or other factors. Recent research in the Russian River basin (Chapter 2) suggests that tributary streams have very low levels of LWD, which may indicate that, similar to the Napa basin, insufficient levels of LWD and cover are negatively impacting salmonid populations.

Although reduced spawning success due to fine sediment may not be the primary factor limiting salmonids in the Russian River basin, sediment from land use can negatively impact fish populations through other mechanisms. Power et al. (2002) reported that experimental increases in fine sediment within stream enclosures resulted in decreased growth and survival of juvenile steelhead. Elevated levels of fine sediment resulted in changes in the macroinvertebrate community, decreased food availability, and increased activity and intraspecific aggression as the fine sediment smoothed the bed topography and reduced the availability of hiding spaces within large substrate. Elevated levels of sediment produced by land use (e.g. roads, grazing, and timber harvest) can, over time, result in simplified channel morphologies and reduce pool depths as they fill with fine sediment (McIntosh et al. 2000). Pulses of sediment (including gravel and cobble) from increased rates of landsliding can overwhelm stream's sediment transport capacities and aggrade channels, converting formerly perennial streams to intermittent streams during the summer (Frissell 1992). Finally, the interaction of land use with rare storm events (e.g. the 1964 flood in Northern California and Southern Oregon) can result in long-term changes in channel morphology (Lisle 1982, Lyons and Beschta 1983). Salmonid habitat suitability may be greatly impaired as waves of sediment move through a drainage network. Further research should focus on identifying the primary limiting factors to salmonids in the Russian River basin and how patterns of human land use, at various scales, influence these limiting factors.

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Table 1. Correlations of proportion of area in agriculture for 8 methods of analysis.

	WSHED	**HPM**	**SLOPE**	**LOCAL**	**UPSTRM**	**LOCUP**	**BLUE**	**DENSE**
WSHED	1.00	0.57	0.95	0.33	0.45	0.46	0.79	0.83
HPM	0.57	1.00	0.43	0.44	0.42	0.54	0.47	0.41
SLOPE	0.95	0.43	1.00	0.31	0.38	0.41	0.73	0.80
LOCAL	0.33	0.44	0.31	1.00	0.32	0.87	0.31	0.37
UPSTRM	0.45	0.42	0.38	0.32	1.00	0.69	0.64	0.61
LOCUP	0.46	0.54	0.41	0.87	0.69	1.00	0.52	0.55
BLUE	0.79	0.47	0.73	0.31	0.64	0.52	1.00	0.96
DENSE	0.83	0.41	0.80	0.37	0.61	0.55	0.96	1.00

Table 2. Proportions of land use/land cover classes within 54 watersheds in the Russian River basin.

	Watershed				
	Mean (se)	**S.D.**	**Median**	**Minimum**	**Maximum**
Agriculture	0.07 (0.02)	0.13	0.01	0	0.53
Herbaceous	0.13 (0.01)	0.09	0.12	0	0.41
Urban	0.01 (0.0005)	0.04	0	0	0.22
Shrub	0.10 (0.02)	0.14	0.05	0	0.74
Forest	0.68 (0.03)	0.19	0.73	0.25	0.93

	Riparian buffer				
	Mean (se)	**S.D.**	**Median**	**Minimum**	**Maximum**
Agriculture	0.07 (0.03)	0.19	0	0	0.74
Herbaceous	0.09 (0.02)	0.17	0.03	0	0.80
Urban	0.004 (0.002)	0.02	0	0	0.08
Shrub	0.03 (0.02)	0.15	0	0	0.83
Forest	0.79 (0.04)	0.28	0.91	0.09	1.00

Table 3. Correlations between land use/land cover categories for 54 watersheds in the Russian River basin.

	Agriculture	**Herbaceous**	**Shrub**	**Urban**	**Forest**
Agriculture	1.00	-0.12	-0.06	0.23	-0.63
Herbaceous	-0.12	1.00	-0.28	0.30	-0.25
Shrub	-0.06	-0.28	1.00	-0.16	-0.52
Urban	0.23	0.30	-0.16	1.00	-0.40
Forest	-0.63	-0.25	-0.52	-0.40	1.00

Table 4. Best-fit models using backward stepwise regression and all LULC terms from WSHED, LOCAL, and UPSTRM scales of analysis, along with geology, housing and road density.

All watersheds (n = 54)

r^2	0.49	***F***	16.3
df	3, 50	***P***	< 0.0001

terms	**Estimate**	**s.d.**	**SumSquares**	**t ratio**	**p**
intercept	-24.5	8.2		-2.97	0.005
Wshed Prop Ag	167	34	21,253	4.91	< 0.0001
Wshed Prop Herb	144	51	7187	2.85	0.006
Wshed Prop Urban	249	109	4589	2.28	0.03

Watersheds less than 1500 ha (n = 23)

r^2	0.25	***F***	6.9
df	1, 21	***p***	0.015

terms	**Estimate**	**s.d.**	**SumSquares**	**t ratio**	**p**
intercept	-17	6		-2.81	0.01
Wshed Prop Ag	126	48	4735	2.63	0.015

Watersheds greater than 1500 ha (n = 31)

r^2	0.57	***F***	8.8
df	4, 26	***p***	0.0001

terms	**Estimate**	**s.d.**	**SumSquares**	**t ratio**	**p**
intercept	-10	17		-0.61	0.5
Wshed Prop Ag	169	44	12898	3.84	0.0007
Wshed Prop Herb	157	79	3441	1.98	0.06
Wshed Prop Shrub	-89	54	2392	-1.65	0.11
Wshed Prop Urban	146	116	1394	1.26	0.22

Watersheds greater than 2500 ha (n = 22)

r^2	0.79	***F***	35.6
df	2, 19	***p***	< 0.0001

terms	**Estimate**	**s.d.**	**SumSquares**	**t ratio**	**P**
intercept	161	18		9.2	< 0.0001
Wshed Prop Shrub	-191	42	9040	-4.58	0.0002
Wshed Prop Forest	-202	25	27,145	-7.94	< 0.0001

Table 5. A comparison of 8 models of the relationship between land use/land cover at various scales and an instream embeddedness index.

All watersheds (n = 54)

	With Geology			**Without geology**		
Model	**r^2**	***F***	***p***	**r^2**	***F***	***p***
WSHED	0.50	16.3	< 0.0001	0.50	16.3	< 0.0001
HPM	0.28	10.0	0.0002	0.28	10.0	0.0002
LOCAL	0.29	5.6	0.003	0.08	4.6	0.04
UPSTRM	0.30	5.8	0.002	0.07	1.9	0.16
LOCUP	0.34	5.0	0.002	0.10	5.8	0.02
BLUE	0.39	16.0	< 0.0001	0.39	16.0	< 0.0001
DENSE	0.41	9.4	< 0.0001	0.38	15.9	< 0.0001

	Watersheds < 1500 ha (without geology; n = 23)			**Watersheds > 1500 ha (without geology; n = 31)**		
Model	**r^2**	***F***	***p***	**r^2**	***F***	***p***
WSHED	0.25	6.9	0.02	0.57	8.8	0.0001
HPM	0.10	2.4	0.13	0.31	6.3	0.005
LOCAL	0.14	3.5	0.07	0.05	1.4	0.24
UPSTRM	0.06	0.6	0.55	0.06	2.0	0.17
LOCUP	0.08	1.7	0.2	0.07	2.2	0.15
BLUE	0.09	2.2	0.16	0.43	10.7	0.0004
DENSE	0.16	1.9	0.16	0.47	12.6	0.0001

Watersheds > 2500 ha
(without geology; n = 22)

Model	**r^2**	***F***	***p***
WSHED	0.79	35.6	< 0.0001
HPM	0.36	5.3	0.01
BLUE	0.77	19.9	< 0.0001
DENSE	0.79	22.4	< 0.0001

Table 6. Correlations of watershed land use/land cover and geology.

	Wshed Prop Ag	Wshed Prop Herb	Wshed Prop Forest	Prop Volcanic Rock	Prop Franciscan Melange	Prop Franciscan Sandstone
Wshed Prop Ag	1.00	-0.07	-0.62	0.07	-0.32	-0.42
Wshed Prop Herb	-0.07	1.00	-0.31	0.14	0.22	-0.41
Wshed Prop Forest	-0.62	-0.31	1.00	-0.29	0.39	0.42
Prop Volcanic Rock	0.07	0.14	-0.29	1.00	-0.40	-0.50
Prop Franciscan Melange	-0.32	0.22	0.39	-0.40	1.00	-0.15
Prop Franciscan Sandstone	-0.42	-0.41	0.42	-0.50	-0.15	1.00

Table 7. Comparison of model statistics and coefficients for three data sets (reaches 1, 2, and 3). The dependant variable is embeddedness index with model terms for proportion of watershed in forest and proportion of watershed in shrub. Watersheds for reaches 2 and 3 are spatially nested within the watersheds for reach 1. The models include only watersheds > 1500 ha.

Reach	**n**	**r^2**	***F***	***p***	**intercept**	**forest coefficient**	**shrub coefficient**
1	31	0.57	18.5	<0.0001	148	-160	-249
2	17	0.50	7.1	0.01	130	-149	-189
3	12	0.65	8.3	0.01	146	-178	-72

Figure 1. Distribution of the 54 reaches used in this chapter and surveyed by CDFG for embeddedness in the Russian River basin.

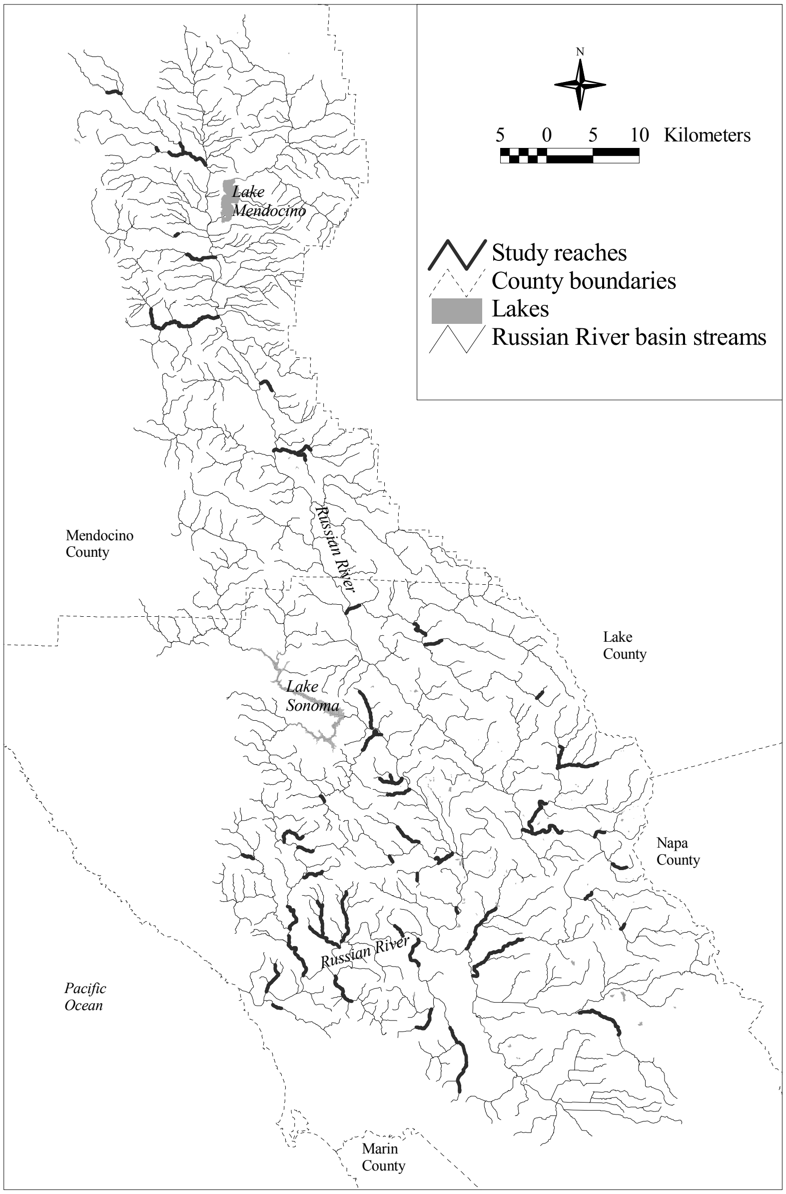


Figure 2. Example of sequential CDFG reaches (*sensu* Rosgen 1994) on Orrs Creek, Mendocino County, California.

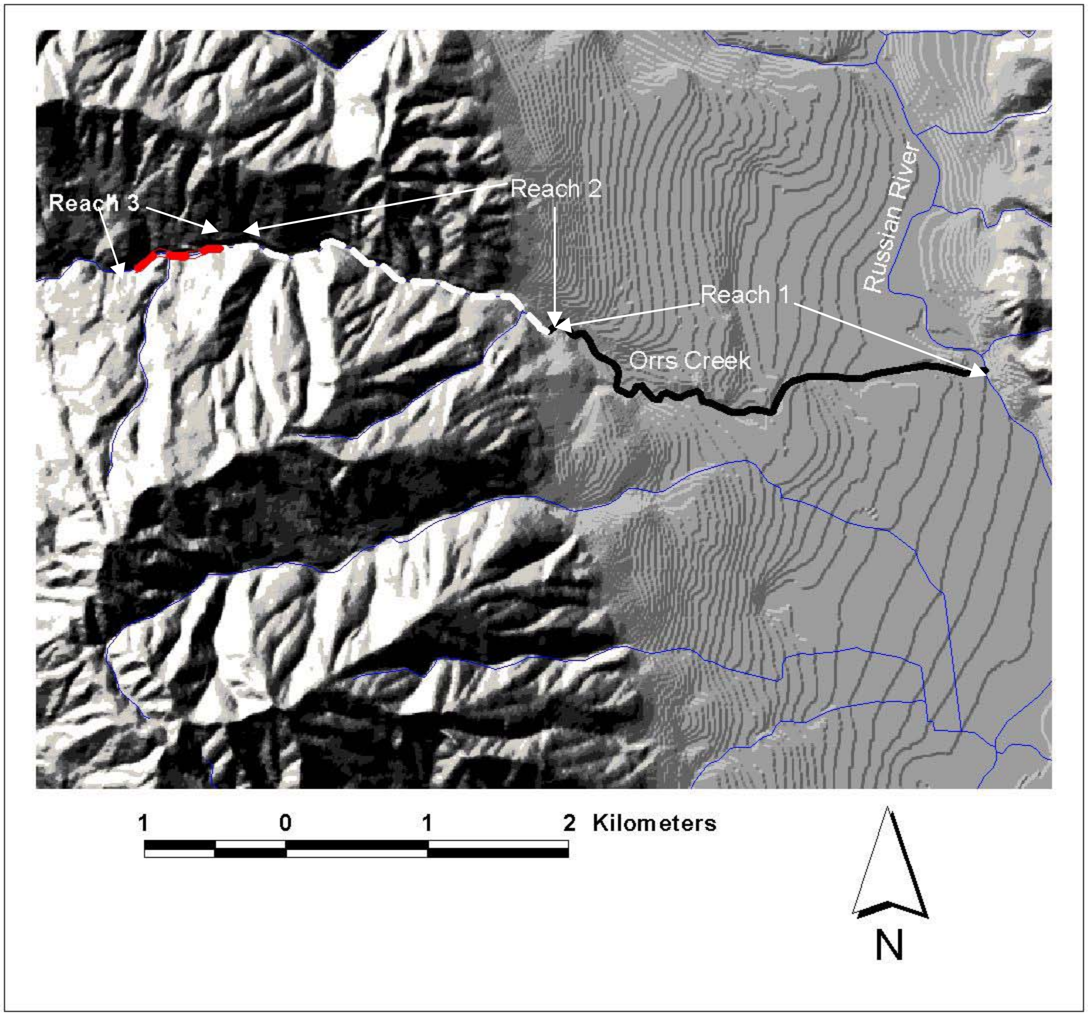


Figure 3a. Reach 1 of Lover's Gulch watershed: scale of analysis is LULC in the entire watershed upstream of the focal reach (model: WSHED).

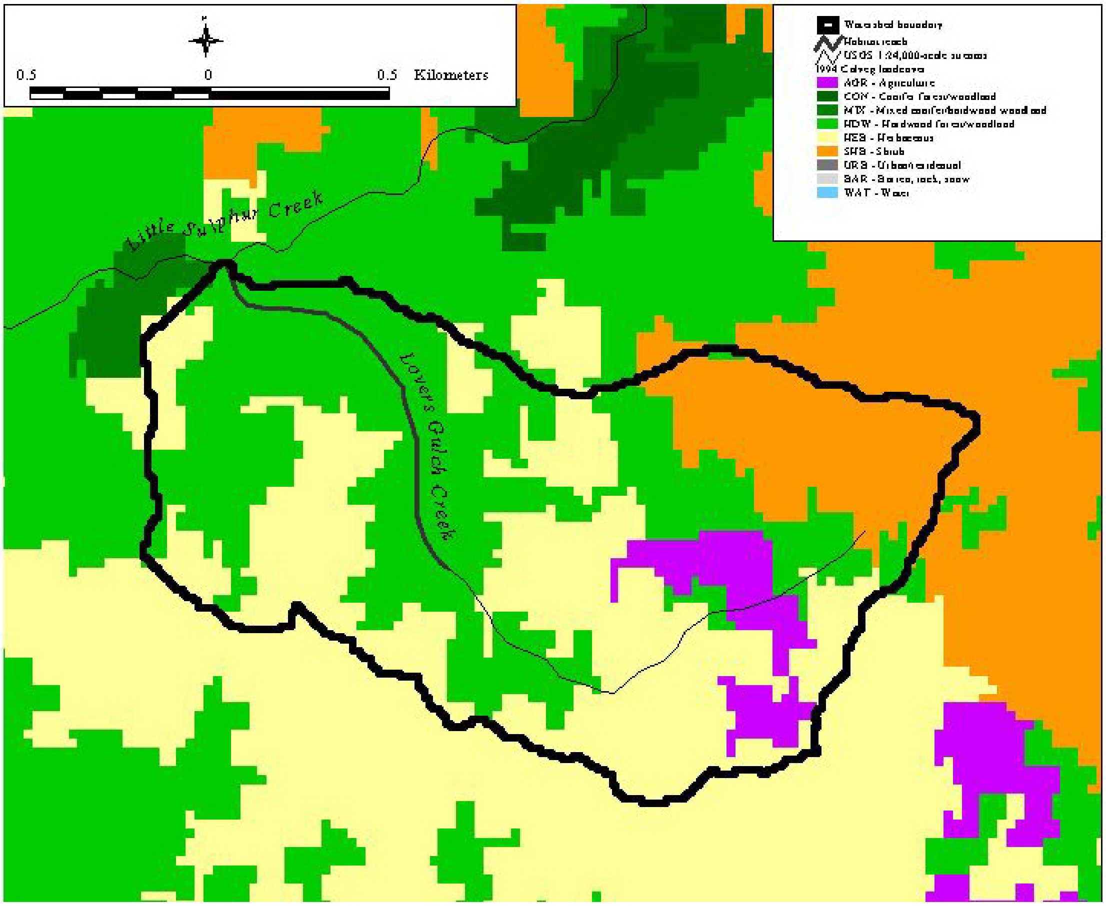


Figure 3b. Reach 1 of Lover's Gulch watershed: scale of analysis is the riparian corridor of the focal reach (model: LOCAL).

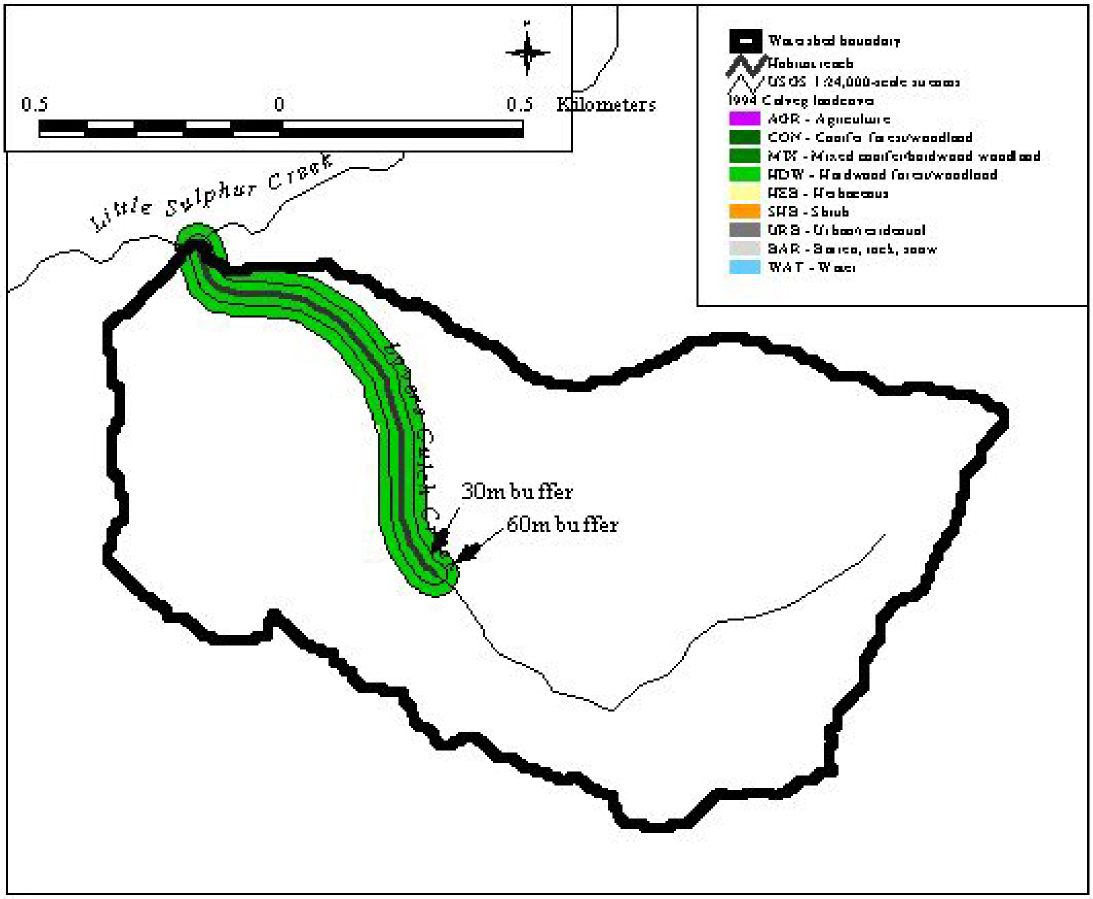


Figure 3c. Reach 1 of Lover's Gulch watershed: scale of analysis is the riparian corridor for a distance of 1 km upstream of the focal reach (model: UPSTRM).

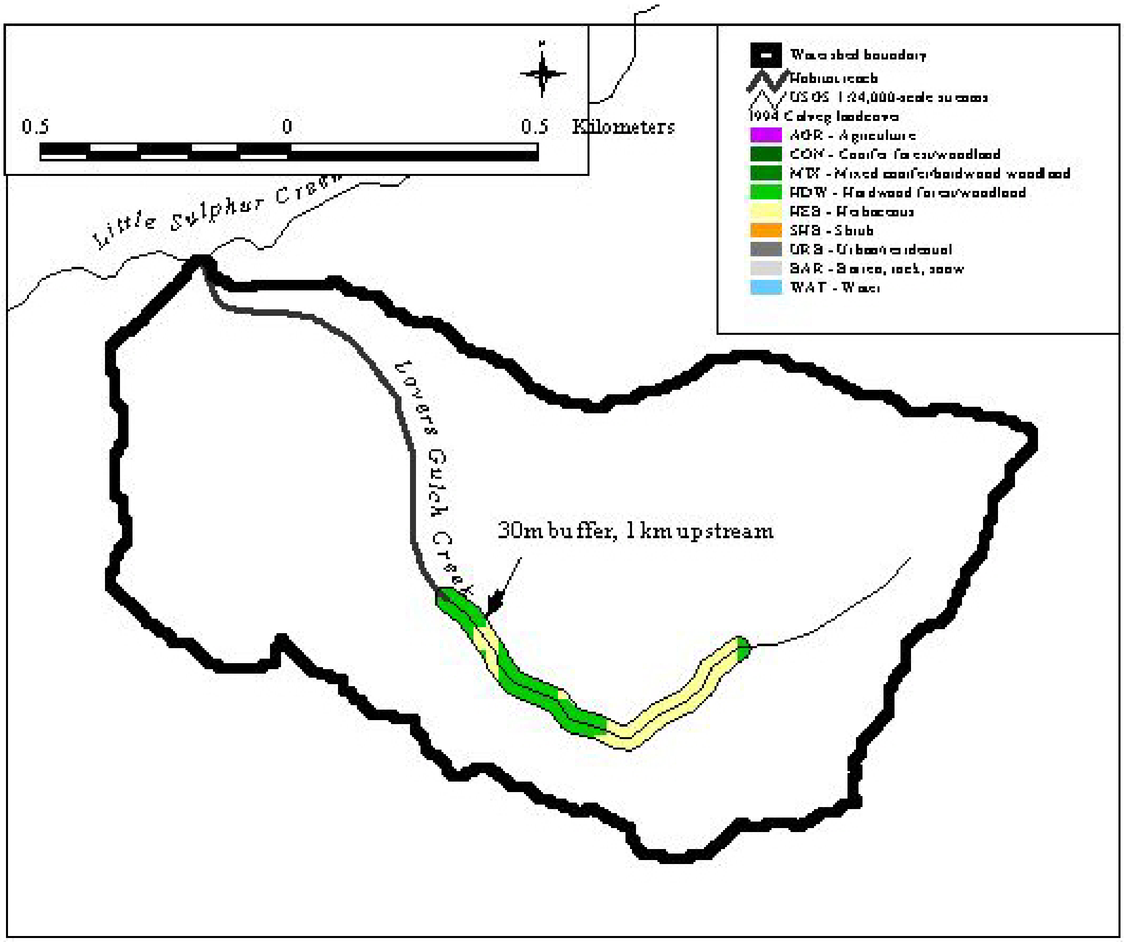


Figure 3d. Reach 1 of Lover's Gulch watershed: scale of analysis is the riparian corridor of the focal reach and 1 km above the focal reach (model: LOCUP).

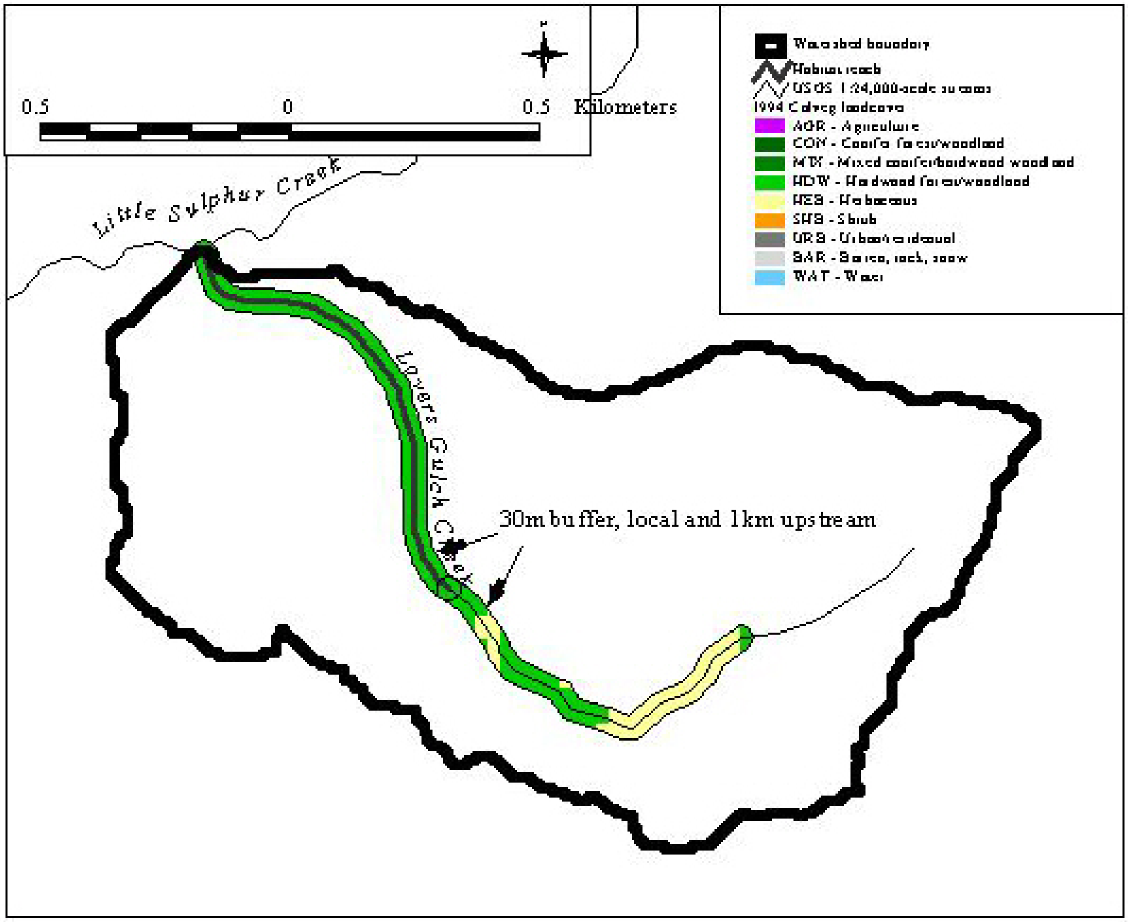


Figure 3e. Reach 1 of Lover's Gulch watershed: scale of analysis is the riparian corridor of a derived stream network upstream of the focal reach, minimum area for generating a channel is 10 ha (model: BLUE).

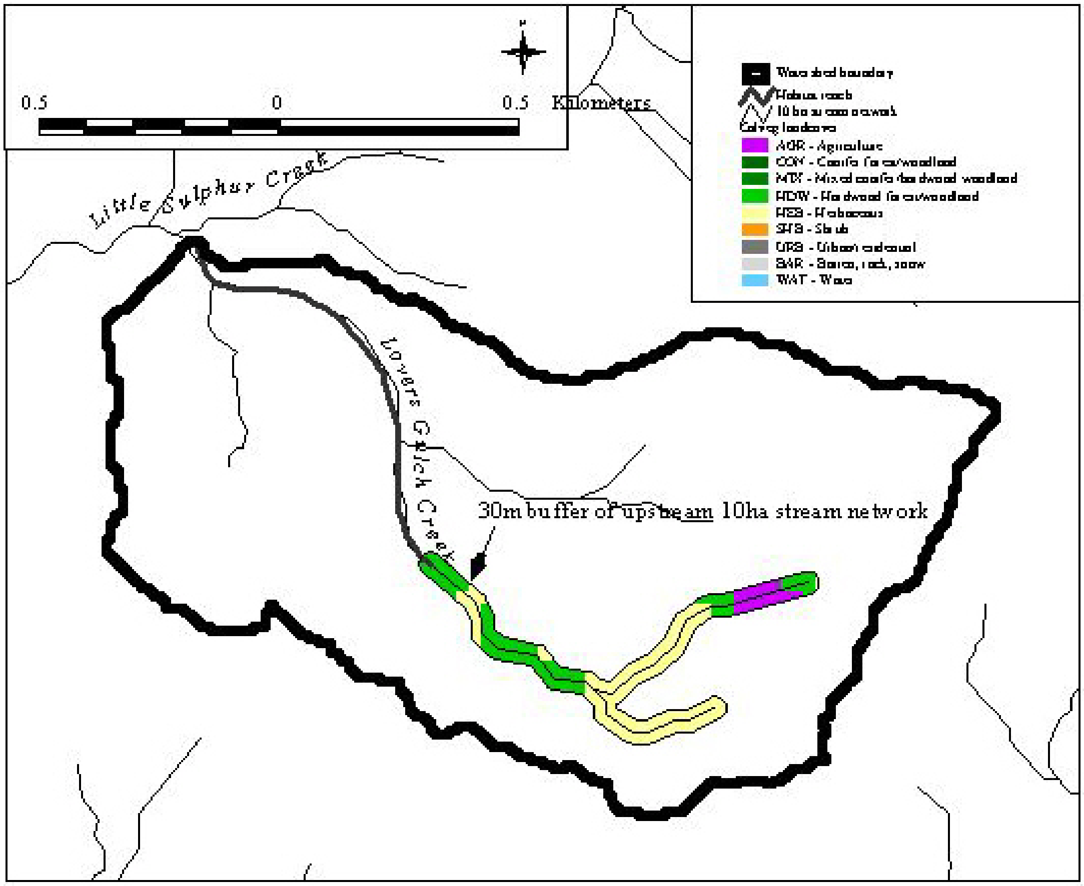


Figure 3f. Reach 1 of Lover's Gulch watershed: scale of analysis is the riparian corridor of a derived stream network upstream of the focal reach, minimum area for generating a channel is 2 ha (model: DENSE).

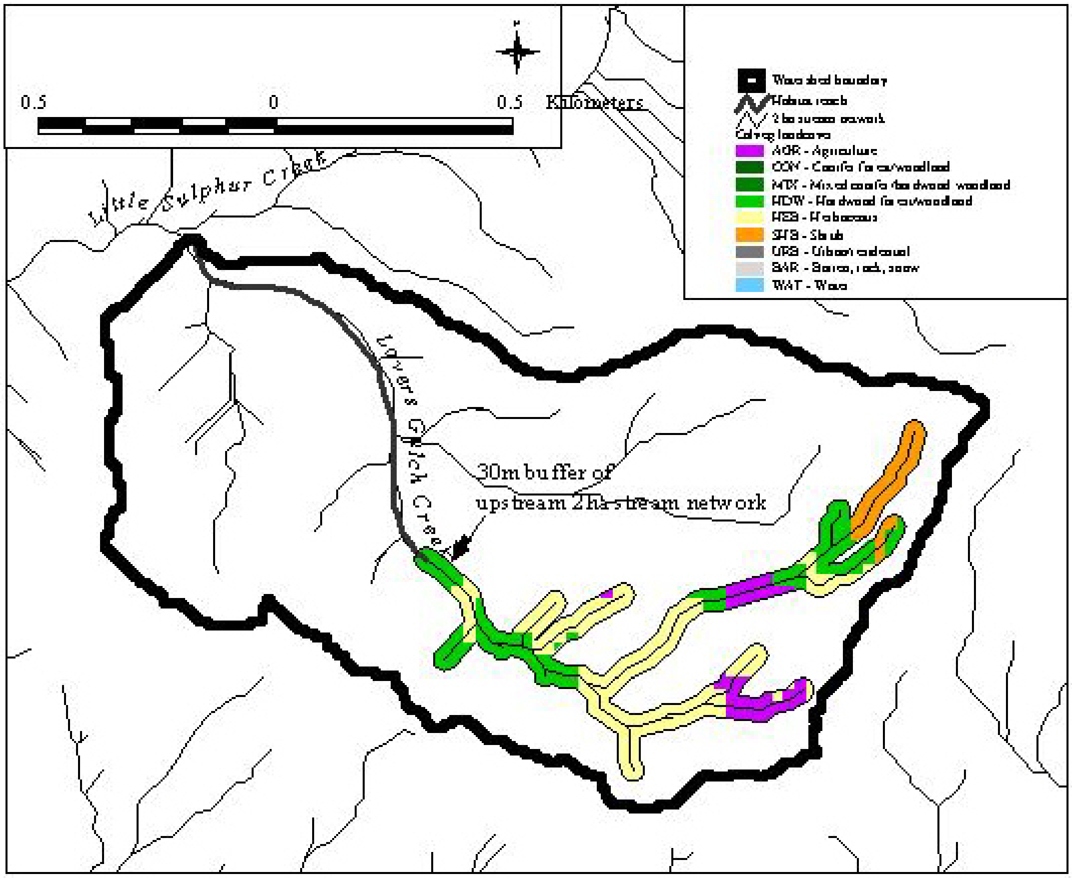


Figure 4. A comparison of stream network density on the same two adjacent quad sheets using: A) USGS maps updated on different years (illustrating that more recently updated maps display more dense stream networks); B) a stream network derived from 10 m DEMs with a 10 ha threshold for channel initiation; and C) stream network derived from 10-m DEMs with a 2-ha threshold for channel initiation.

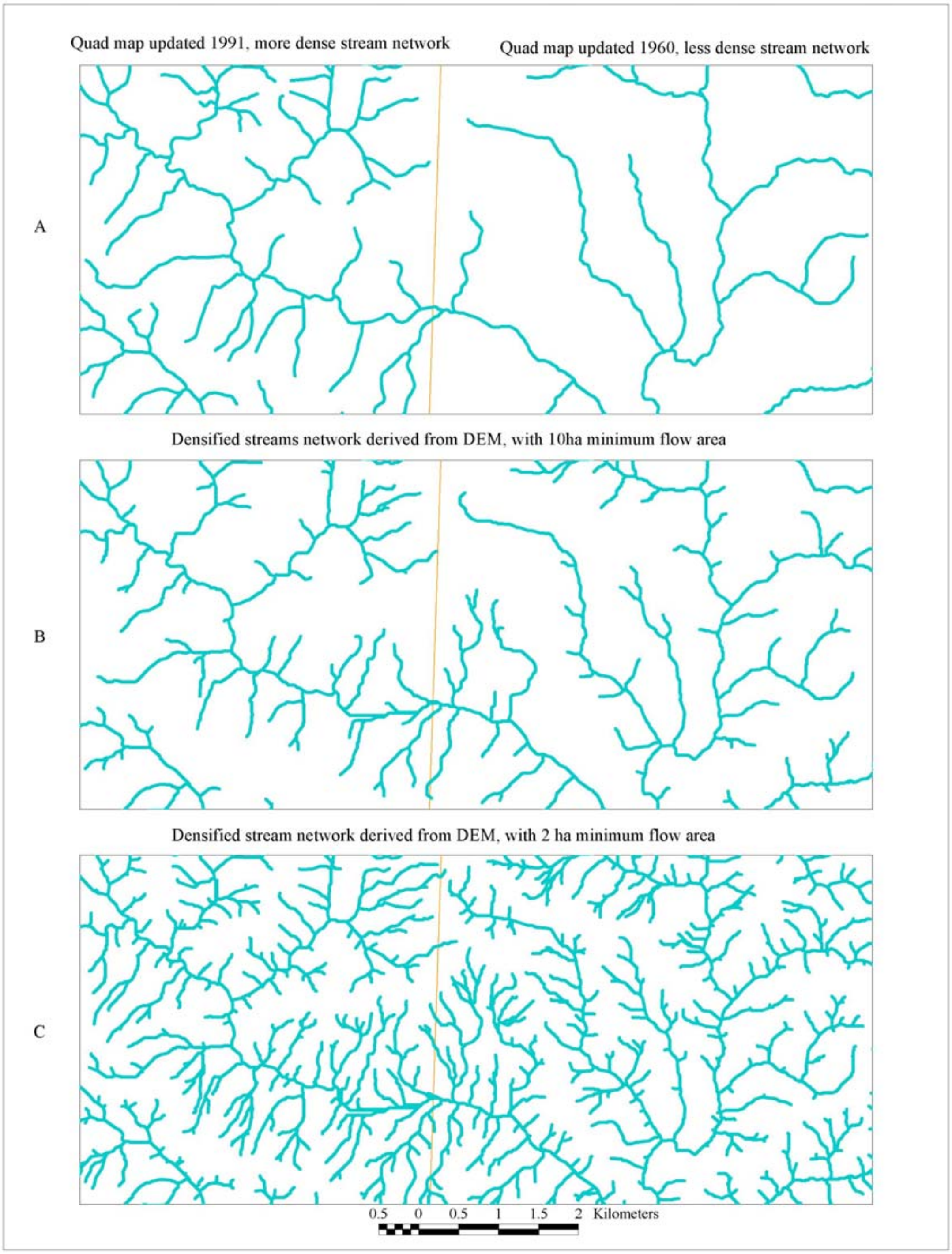


Figure 5. A comparison of the embeddedness index (‘index’) to other methods of analyzing the embeddedness data: A) proportion with embeddedness = 4; B) “high embeddedness” (embeddedness ‘3’ + ‘4’); C) a weighted average composite score.

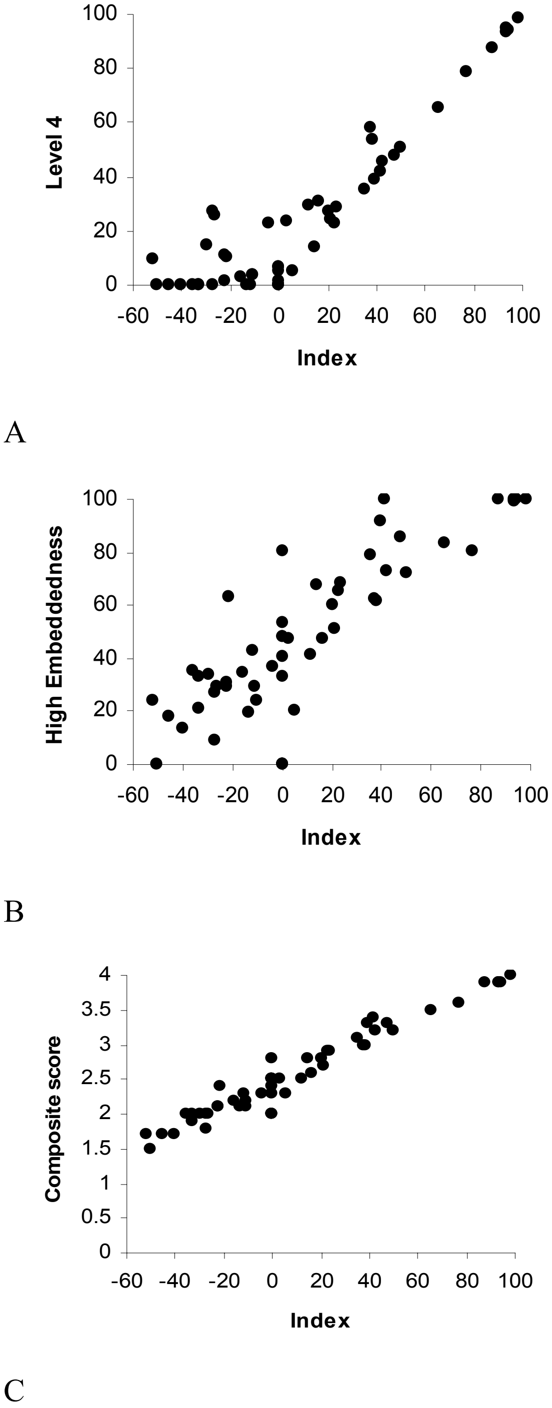
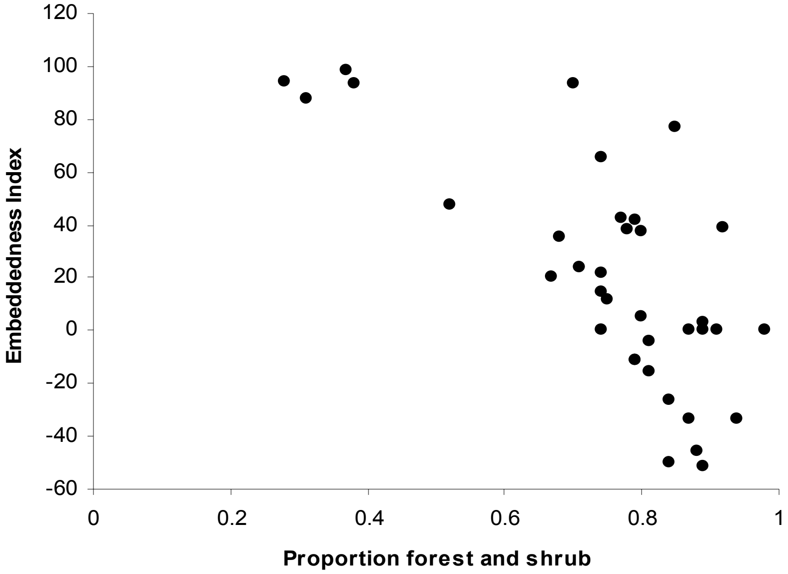
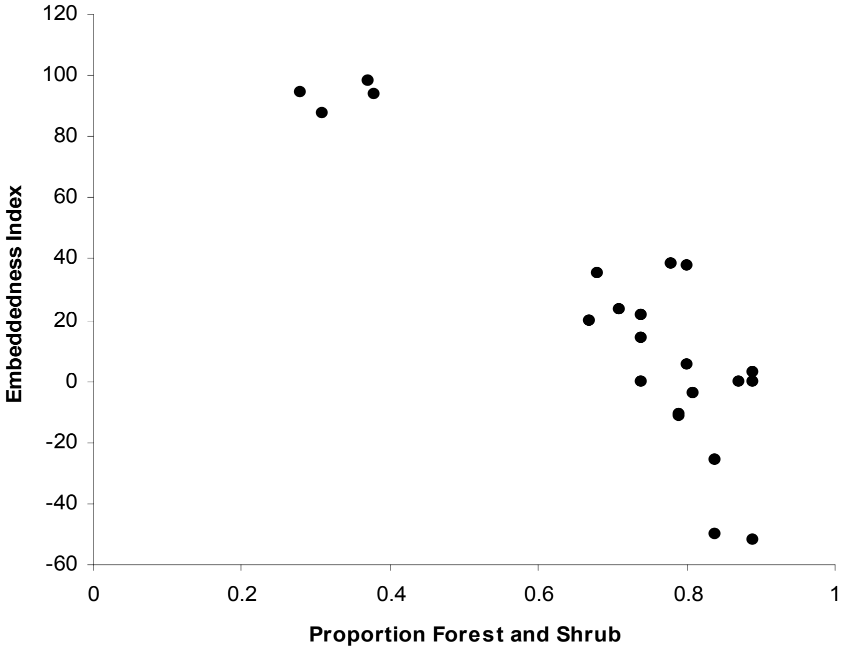


Figure 6. Relationship of embeddedness index and the proportion of a watershed with forest and shrub land cover, for A) watersheds > 1500 ha (n = 31), and; B) watersheds greater than 2500 ha (n = 22)

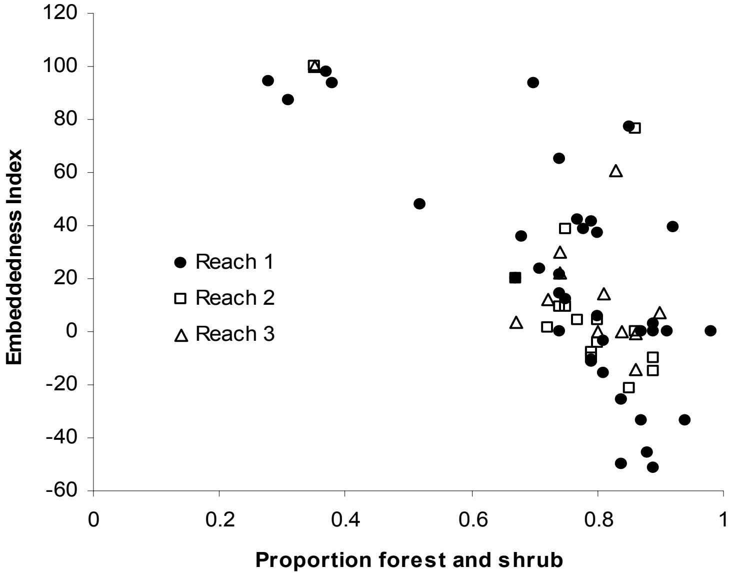


A.



B.

Figure 7. Relationship between the embeddedness index and the proportion of a watershed in forest and shrub for three data sets (watersheds > 1500 ha). Reaches 2 and 3 are spatially nested within the watersheds of Reach 1.



CHAPTER 7

CONCLUSIONS AND FUTURE RESEARCH

Conclusions and Future Research

The research and analysis presented in this dissertation demonstrate that key processes for salmonids in Mediterranean-climate watersheds operate at multiple scales. At the reach scale, inputs of large woody debris are the result of vegetation and management at the site. At the watershed scale, levels of fine sediment in spawning gravels are influenced by land use and land cover. Understanding these processes can help pair restoration objectives with the appropriate scale of action. For example, revegetation, with the appropriate species, may promote increased levels of large woody debris (LWD) at the site or reach scale but have little influence on sedimentation of spawning gravels.

Similar to its role in conifer-dominated watersheds, LWD created important habitat features in Mediterranean-climate hardwood streams, although through somewhat different mechanisms. Increased loading of LWD was associated with greater frequency of pools, and those pools had significantly higher cover values than pools formed by other mechanisms. Because individual pieces of hardwood LWD are considerably smaller than conifer debris, single pieces of wood rarely influenced channel morphology. Instead, most pools caused by LWD were formed by debris jams. “Living LWD” – trees that enter the stream but remain rooted and living - played a major role forming and stabilizing these debris jams, a phenomenon that has not been reported from other systems. Preliminary results indicate that living key pieces increase the persistence of debris jams. This hypothesis can be further tested through long-term monitoring or by dendrochronological techniques. Tree-ring analysis has been used to date the age of trees growing on “nurse logs” within a debris jam and, thus, infer the age of the jam (Keller and Tally 1979, Murphy and Koski 1989). These techniques have also been used to identify changes in tree-ring patterns that reflect sudden changes in tree orientation to estimate the year of gully formation (Vandekerckhove et al. 2001) or mass wasting events (Hupp 1983). Dendrochronological evidence of the year when a living-key tree entered the channel can likely be found in either the main trunk (the onset of much smaller growth rings corresponding to the timing of tree fall), in new sprouts or in branches that have reoriented to grow vertically.

Overall, tributaries of the Russian River have few pools caused by LWD. Because LWD has been demonstrated to be so important to fish populations elsewhere (National Research Council 1996, Roni et al. 2002), and because this research has shown that streams with high LWD have superior habitat values (in terms of pool frequency and cover), this lack of LWD may be a primary limiting factor for salmonids in the Russian River basin. However, to demonstrate that low levels of LWD are contributing to diminished salmonid populations, future research should expand on the relationship between LWD and *habitat* characteristics to the relationship between LWD and fish-population parameters (e.g. productivity, survival). The California Department of Fish and Game (CDFG) has been conducting electro-fishing surveys in Russian River tributaries each fall since 1997, often within reaches that were habitat-typed the previous summer. Hopefully, these data can clarify the relationship between LWD and fish-population dynamics. Electro-fishing data should be spatially linked to the habitat-typing data to investigate relationships between habitat values at the reach scale and fish presence, particularly of older age classes of steelhead.

Fish and Game could improve this research by collecting more detailed habitat data within the specific reaches where electro-fishing occurs and ensuring that all future electro-fishing surveys are conducted with consistent methods and an effective spatial (e.g. GPS) protocol. Rather than one-time surveys in a single reach, annual censuses within a select group of reaches will greatly improve the understanding of the relationship between LWD and fish-population parameters such as over-winter and over-summer survival.

Such research will help resolve one of the hypotheses posed here: LWD, and the habitat it creates, may be particularly important for stream-rearing salmonids within Mediterranean-climate streams. The extended summer drought reduces streams to a series of pools that are either completely isolated or connected by riffles and runs with minimal flow. Due to the contracted stream area, fish are more vulnerable to predators (Shirvell 1990, Giannico 2000). Juvenile steelhead need to add weight during the summer, but their feeding efficiency may be reduced by low flows resulting in lack of invertebrate drift and decreased access to riffles (Stillwater Sciences and Dietrich 2002). Therefore, salmonids in these systems will require reaches with sufficient pool habitat and pools that have cover from predators and provide sufficient food resources. Because LWD increases the frequency of pools within a reach while also producing pools with the highest cover value, it may be particularly important to salmonids in these systems. Further, pools formed by LWD may provide improved food resources when compared to other pool types because aquatic and terrestrial invertebrates utilize LWD for food and rearing substrate (Dudley and Anderson 1982, Hilderbrand et al. 1997, Braccia and Batzer 2001). In addition, LWD within pools trap fine organic matter to further promote productivity of the food web (Bilby and Likens 1980, Bilby 1981). Living LWD may be particularly important by providing a higher rate of input of terrestrial invertebrates, as well as direct input of leaves and twigs during baseflow conditions. These specific advantages of LWD to fish populations in Mediterranean climates merit further study.

Because most research on LWD has taken place within conifer-dominated forests with little research on LWD in other systems, management regulations focused on LWD have been developed primarily or exclusively for conifer timberlands. Thus, although anadromous fish in California utilize both conifer and hardwood watersheds, they receive less protection through environmental regulation within hardwood watersheds. For example, in California, landowners who seek to remove conifers must file a Timber Harvest Permit and comply with the Forest Practice Rules. These rules have provisions protecting riparian corridors, in part to retain LWD recruitment. Conversely, conversion of oak woodlands to vineyards does not trigger similar oversight (Giusti and Merenlender 2002). Recently, some counties have proposed stream setbacks that would increase protection for riparian corridors in the vineyard landscape (e.g. Napa and Sonoma counties). However, stream setbacks alone will not increase levels of instream LWD if management practices continue to promote removal of large trees that fall into streams.

Landowners may remove wood due for several reasons, including concerns about bank erosion and damage to bridges and culverts, for firewood, due to a tradition of maintaining “clean streams,” and due to misplaced concerns that debris jams impede fish migration. Fish and Game agencies across the West engaged in debris-removal programs ostensibly to improve fish habitat up until the 1970s, and the legacy of this practice may contribute to landowner confusion about LWD blocking fish passage. Effective extension and outreach programs can inform landowners that LWD rarely prevents fish migration (Bisson et al. 1987) and that, in fact, wood in streams creates critical habitat for salmonids. Although owners of timber land have received information about LWD through both regulations (such as the California Timber Harvest Rules) and extension programs, landowners within hardwood (i.e. non-timber) forests have generally not received information about the importance of LWD. Thus, future extension programs on the link between LWD and healthy fish populations should target private landowners in hardwood-dominated watersheds.

Results presented in Chapter 6 indicate that levels of fine sediment in spawning gravels increase as the amount of forest within a watershed decreases. While this highlights a potential link between watershed-scale land use and salmonid habitat, future research should explore how this affects fish populations. In other words, to what extent are salmonid populations in the Russian River basin limited by fine sediment in spawning gravels? The CDFG electro-fishing data may also be useful for addressing this question. Data on steelhead and coho population structures will allow researchers to parameterize population viability models. These models may indicate whether spawning success or other factors, such as juvenile survival, limit current populations (Ratner et al. 1997). As noted in Chapter 6, sediment that impedes spawning is unlikely to be a limiting factor for fish populations, particularly in streams with strong seasonal pressure on available habitat. Because of the contraction of available habitat during the summer drought, even a small amount of successful spawning may be sufficient to “seed” the limited available rearing habitat. However, if the amount of rearing habitat is increased, the impacts of sediment on spawning success may have a greater influence on population dynamics.

Finally, a factor not addressed in this dissertation – the presence of water throughout the year – may be the greatest limiting factor for salmonids in Mediterranean-climate streams. Currently, techniques that inventory stream habitat and restoration models within Mediterranean-climate streams have been imported from more mesic regions (e.g. the Pacific Northwest) where a greater proportion of the stream network is composed of perennial streams. Habitat-typing, which emphasizes quantification of rearing habitat, may overestimate potential habitat if intermittent streams are surveyed during the early summer while still flowing (Table 1). Streams within Mediterranean-climate basins, already subject to greater inter-annual and intra-annual variability in flow than streams in more mesic systems, are also more vulnerable to dewatering from human activities. Thus, restoration and protection strategies that do not directly consider both the natural variability in surface water (temporal and spatial) and its vulnerability to human influences may ignore the primary factor shaping habitat availability.

Parsons Creek, a second-order tributary to the Russian River in Mendocino County, illustrates the importance of water as a limiting habitat feature. At the Hopland Research and Extension Center (HREC), steelhead fry are generally observed in the spring distributed throughout the riffles of a 2.5 km length of stream. However, each year most of the creek on HREC becomes dry, and steelhead become restricted to isolated pools and short stretches of flowing water. Nearly all of these pools become too warm or eventually dry up. Only two short reaches of this portion of the creek consistently retain perennial surface water; each is directly below a spring (Figure 1). Thus, although steelhead fry initially occupy the entire creek on HREC, over-summer habitat exists only where springs provide localized surface water. Older age-class steelhead have been observed in pools within both of these reaches.

Parsons Creek illustrates the vulnerability of salmonids in Mediterranean-climate streams. The short reaches of perennial surface water and suitable habitat could be easily disrupted by groundwater pumping or by direct diversion of spring flow. The impacts of small-scale diversions and groundwater pumping on surface water in tributaries have not been quantified for the Russian River or, presumably, for most Mediterranean-climate systems. These interventions into watershed hydrology may have individually small but collectively large impacts on surface water in streams (Moyle and Kondolf 2000). Human alterations to the hydrology of Mediterranean-climate watersheds should be elevated to a primary focus for restoration and protection strategies for salmonid habitat. Restoration strategies to increase LWD and decrease fine sediment will do little to restore fish populations if perennial surface water is further diminished by human activities.

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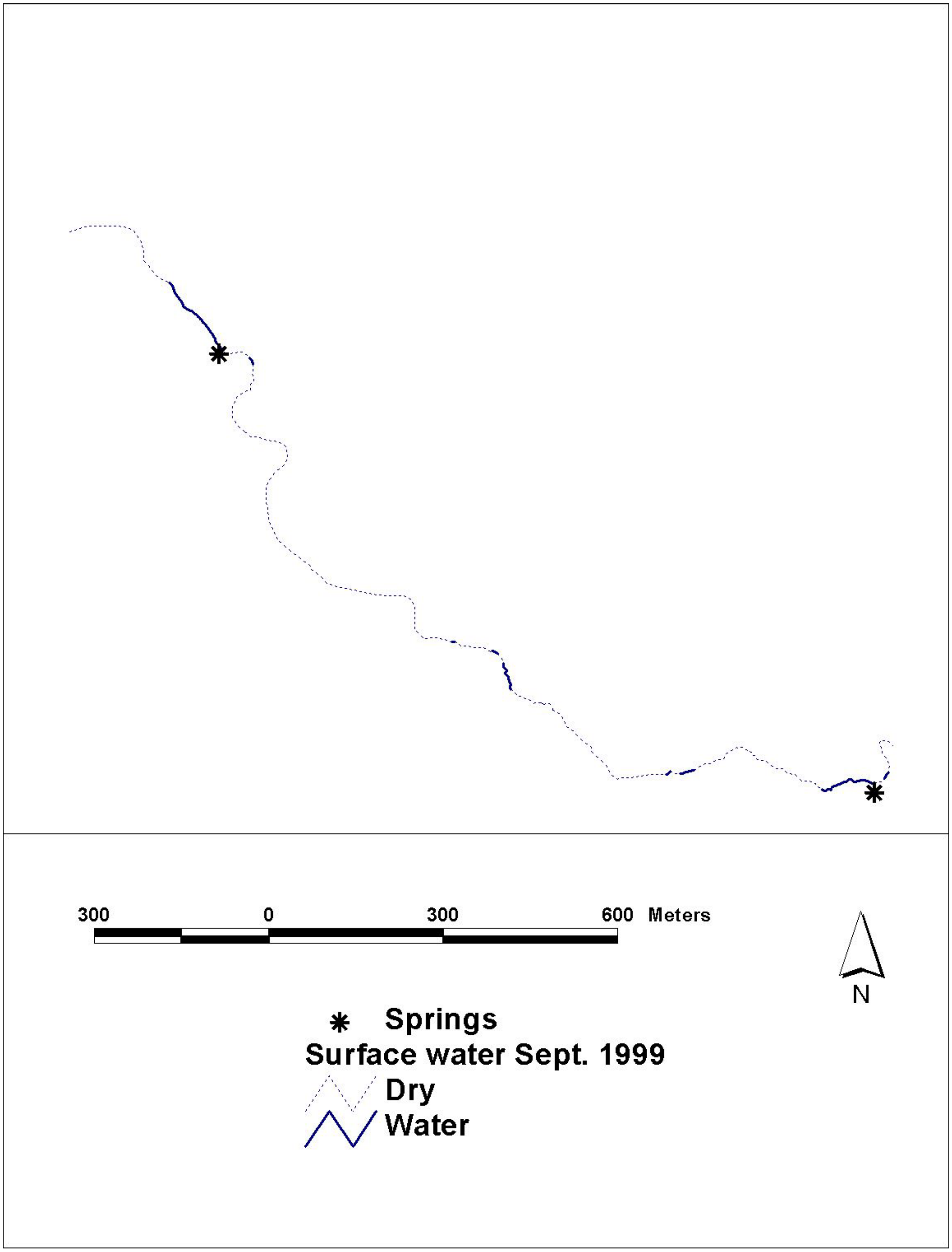
Table 1. Characteristics of pools within a 700-m reach of Redwood Creek (Alameda County) during 2001. In July, juvenile rainbow trout were observed in 13 of the 16 pools. In September juvenile trout were observed in 3 of the 10 pools.

	# pools	Avg. max depth (cm)	Avg. volume (m^3)	Total pool volume in reach (m^3)
Baseflow (spring)	49	54	3.3	162
July	16	55	3.1	50
September	10	18	0.5	5

Figure 1. This section of Parsons Creek, a second-order tributary to the Russian River in Mendocino County, dries each summer and surface water is restricted primarily to two short reaches below springs. Although steelhead fry are generally observed each spring within riffles along the entire 2.5 km length of stream depicted here, perennial habitat is restricted to these two short reaches.



APPENDICES

APPENDIX I

POOL TYPES FROM CALIFORNIA DEPARTMENT OF FISH AND GAME HABITAT-TYPING SURVEYS

Code	Pool Type
4.1	Trench
4.2	Mid-channel
4.3	Confluence
4.4	Step
5.1	Corner
5.2	Lateral-scour-log
5.3	Lateral-scour-rootwad
5.4	Lateral-scour-bedrock
5.5	Lateral-scour –boulder
5.6	Plunge
6.1	Secondary Channel
6.2	Backwater-boulder
6.3	Backwater-rootwad
6.4	Backwater-log
6.5	Dammed

From: California Department of Fish and Game (CDFG). 1998. California Salmonid Stream Habitat Restoration Manual, Third Edition. CDFG, Sacramento, CA.http://www.dfg.ca.gov/nafwb/pubs/manual3.pdf

APPENDIX II

DATA CATEGORIES USED IN LWD FIELD SURVEYS

Distributed to participants at a University of California Center for Forestry Workshop on Forestry and Riparian Systems at the UC Forestry Camp, Quincy, California, September, 2001.

Example Field Measurements for Large Woody Debris

Riparian Workshop
September 20, 2001
Jeff Opperman
(opperman@nature.berkeley.edu)

Basic criteria: at least 10 cm dbh (4 inches) and 1 m in length (40 inches).

Variables to record about LWD

1. **Species**

- use codes for genus and species: e.g. PSME, ALRH (Douglas fir, *Pseudotsuga menziesii*; White alder, *Alnus rhombifolia*)
- more general: SALI, CONI, HARD (willow species, *salix*; conifer, hardwood)

2. **Type of piece:** L (log); R (rootwad, no log); B (log with rootwad)

3. **Dimensions**

- length
  - partition length between: within bankfull channel, and beyond bankfull channel
- diameter (use dbh tape or calipers)
- Volume = [length x Π x $(\text{diam}/2)^2$]

4. **Decay class**

Decay class	**Description**
0	Still living
1	Recently entered channel; bark intact; also small branches, twigs, dried leaves, needles may be present
2	Bark mostly intact, wood still firm; small branches, twigs, leaves etc. absent; wood still original color.
3	Most bark absent, wood still firm; wood may be darkening
4	Most bark absent, some indications of decay present; wood dark
5	Significantly decayed; wood dark

5. **Position and stability**

Code	**Description**
BF	Within bankfull channel
BF1	Within bankfull; one end buried
BFB	Within bankfull; both ends buried
B-BF	Extending from bank to bankfull
WC	Within wetted channel (or part of piece is within WC)
WC1	Within wetted channel; one end buried
WCB	Within wetted channel; both ends buried
B-WC	Extending from bank to wetted channel
SUSP	Suspended above bankfull
SPAN	Spanning channel, interacting with bankfull flow
SNAG	Snagged on standing trees (generally at or near bankfull)
JAM	Within a jam

6. **Stability**
   R: root system present
   P: pinned (typically in a jam or stable accumulation)
   B: buried (in channel or terrace)
   0: no evidence of stability

7. **Function** (some examples)

Code	**Function**
KEY	Key piece in a jam
POOL	Forming or contributing to a pool
COV	Providing cover
SED	Storing sediment
REGEN	Protecting site for riparian regeneration

8. **Input mechanism**

Code	**Mechanism**
BANK	Bank erosion, undercutting
MASS	Mass wasting event
WIND	Windthrow
TREE	Whole tree enters, mechanism uncertain (e.g. could be mortality of tree from disease)
BRANCH	Breaking of major limb or branch from standing tree
RAM	Rammed by floating logs or jams
TACK	"Tackled" by other falling tree
MGT	From some type of management

9. **Input distance and source** (e.g. bank, floodplain, terrace, hill)

10. **Orientation relative to streamflow**

Variables to record about jams

1. **Position**
   SPAN: spanning wetted channel
   BANK: on bank
   PART: on bank, partially spanning wetted channel
   FLOOD: on floodplain

2. **Jam size**

- ❖ Direct enumeration (record every or most piece, link these pieces to a specific jam code)
- ❖ Dimensions: L x W x H
- ❖ Classes for number of pieces of LWD: e.g < 5, 5-10, 10-20, > 20

3. **Causes a step in profile?** (Y or N)

4. **Stabilized by standing trees?** (Y or N; can also record dbh and species of standing trees)

References

Beechie, T. J. and T. H. Sibley. 1997. Relationships between channel characteristics, woody debris, and fish habitat in northwestern Washington streams. Trans. Am. Fish. Soc. 126: 217-229.

Berg, N., A. Carlson, et al. 1998. Function and dynamics of woody debris in stream reaches in the central Sierra Nevada, California. Can. J. Fish. Aquat. Sci. 55: 1807-1820.

Harmon, M. E., J. F. Franklin, et al. 1986. Ecology of coarse woody debris in temperate ecosystems. Advances in Ecological Research 15: 133-302.

Lassettre, N. S. and R. R. Harris. 2000. The geomorphic and ecological influence of large woody debris in streams and rivers. Berkeley, California, UC Berkeley Center for Forestry.

O'Connor Environmental Inc. 2000. Garcia River Large Woody Debris Instream Monitoring. Healdsburg, CA. Prepared for: Mendocino County Resource Conservation District.

also see annotated bibliography of LWD references (N. Lassettre) at:

http://www.CNR.Berkeley.EDU/forestry/woodbiblio.html

Pool data entry definitions

- pool type (**bold** codes are LWD influenced)

BACK	Backwater pool
CON	Confluence pool
DAM	Dammed pool
PPL	Plunge pool, LWD
PPR	Plunge pool over roots that cross channel
PPJ	Plunge pool over a debris jam
PPB	Plunge pool over boulders or bedrock
LSR	Lateral scour, rootwad
LSL	Lateral scour, LWD
LSJ	Lateral scour, debris jam
LSB	Lateral scour, boulder
LSBR	Lateral scour, bedrock
MEA	Meander pool
MCP	Mid-channel pool
SIDE	Side-channel pool

LWD influence

PRI	Pool is clearly caused by presence of LWD.
ENH	LWD significantly enhances pool, e.g. 50-50 contribution along with other factor.
CON	LWD contributes to making the pool deeper or longer, but other feature, e.g. a boulder, is clearly primary cause of pool.
NONE	Pool caused by factor other than LWD.

APPENDIX III

COPIES OF FIELD DATA SHEETS

Pool Data Sheet

Survey and station___________
Type of pool_________________
Comments:

Length (up to down): _______________
Width: ___________
Max Depth:______________ Riffle Crest_______________
Depth:

Cover:

Source	**%**	**Notes**

Survey and station___________
Type of pool_________________
Comments:

Length (up to down): _______________
Width: ___________
Max Depth:______________ Riffle Crest_______________
Depth:

Cover:

Source	**%**	**Notes**

Debris Jam Data Sheet

Reach-Survey___________ Station__________
Position___________ Step in profile?____________ Extends on FP_________
Live key member?______________ Branched Key?_________ Causes pool?________
Stabilized by standing trees?____________
Spp. DBH Spp. DBH Spp. DBH
W________H__________L_________
Comments or sketch:

Reach-Survey___________ Station__________
Position___________ Step in profile?____________ Extends on FP_________
Live key member?______________ Branched Key?_________ Causes pool?________
Stabilized by standing trees?____________
Spp. DBH Spp. DBH Spp. DBH
W________H__________L_________
Comments or sketch:

Riparian data sheet
Date__________ Time__________
Location_______________ Survey________________
Comments:

Station	Species	Dbh	Height	Snag?	Landform	Notes

LWD Field Data Sheet

Date__________ **Collected by**__________ **Location**__________________________________

Surv	Station	Spp.	L. in BF (ft.)	L. out (ft.)	Diam (cm)	Chan Pos.	Dec -ay	Key ?	BR ?	Source	Dist. (ft.)	Input Mech	Function