GREEN DIAMOND RESOURCE COMPANY

Aquatic Habitat Conservation Plan and Candidate Conservation Agreement with Assurances

FINAL

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Aquatic Habitat Conservation Plan and Candidate Conservation Agreement with Assurances

Prepared for:

National Marine Fisheries Service
U.S. Fish and Wildlife Service

Prepared by:

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Contents

Appendix A. Profile of the Covered Species

CONTENTS

Tables

A.1 DESCRIPTION OF THE COVERED SPECIES

A.1.1 Chinook Salmon (*Oncorhynchus tshawytscha*)

A.1.1.1 Listing Status

On March 14, April 4, and May 23, 1994, NMFS received petitions to list several populations of salmon comprising four biological species of Pacific salmon, including chinook salmon, and subsequently initiated comprehensive coastwide status reviews to determine if listings were warranted (September 12, 1994, 59 FR 46808). On February 1, 1995, NMFS was again petitioned to list chinook salmon throughout its range in California, Oregon, Washington and Idaho and again initiated a status review to determine if the petitioned action was warranted (June 8, 1995, 60 FR 30263). On March 9, 1998 (63 FR 11482), NMFS proposed to list the Southern Oregon and California Coastal chinook salmon ESU as threatened. This ESU includes all naturally spawned coastal spring and fall run chinook salmon spawning from Cape Blanco (inclusive of the Elk River) to the southern extent of the current range for chinook salmon at Point Bonita (the northern land mass marking the entrance to San Francisco Bay).

A.1.1.1.1 California Coastal Chinook Salmon ESU

On September 16, 1999 (64 FR 50394), NMFS determined that new information supports a threatened listing for a revised California Coastal chinook salmon ESU, that was part of the larger Southern Oregon and California Coastal chinook salmon ESU, This ESU consists of California coastal chinook salmon populations from Redwood Creek in Humboldt County south through the Russian River in Sonoma County.

Critical habitat for this ESU is designated to include all river reaches and estuarine areas accessible to listed chinook salmon from Redwood Creek (Humboldt County, California) to the Russian River (Sonoma County, California), inclusive (February 16, 2000, 65 FR 7764). Rivers, estuaries, and bays known to support California Coastal chinook salmon include Humboldt Bay, Redwood Creek, and the Mad, Eel, Mattole, and Russian Rivers. Also included are adjacent riparian zones. Excluded are tribal lands and areas above specific dams or above longstanding, naturally impassable barriers (i.e., natural waterfalls in existence for at least several hundred years). Major river basins containing spawning and rearing habitat for this ESU comprise approximately 8,061 square miles in California. The following counties lie partially or wholly within these basins (or contain migration habitat for the species): Glenn, Humboldt, Lake, Marin, Mendocino, Sonoma, and Trinity.

A.1.1.1.2 Southern Oregon and Northern California Coastal Chinook Salmon ESU

The Southern Oregon and Northern California Coastal chinook salmon ESU was determined not to warrant listing (September 16, 1999, 64 FR 50394). It includes all naturally spawned populations of chinook salmon from rivers and streams between Cape Blanco, Oregon (excluding the Elk River), and the lower Klamath River, California, excluding populations in the Klamath River Basin upstream from the confluence of the Klamath and Trinity Rivers. Major river basins containing spawning and rearing habitat for this ESU comprise approximately 6,528 square miles in California and Oregon. The following counties lie partially or wholly within these basins: California - Del Norte, Humboldt, and Siskiyou; Oregon - Coos, Curry, Douglas, Jackson, Josephine, and Klamath.

A.1.1.1.3 Upper Klamath and Trinity River Chinook Salmon ESU

The Upper Klamath and Trinity River Chinook salmon ESU was determined not to warrant listing (March 8, 1998, 63 FR 11482). It includes all naturally spawned populations of chinook salmon in the Klamath and Trinity River basins upstream of the confluence of the Klamath and Trinity Rivers. Major river basins containing spawning and rearing habitat for this ESU comprise approximately 6,429 square miles in California. The following counties lie partially or wholly within these basins: Del Norte, Humboldt, Siskiyou and Trinity.

A.1.1.2 Status of ESU Populations

A.1.1.2.1 Southern Oregon and Northern California Coastal Chinook Salmon ESU

A summary of the status of populations of this ESU are shown in Table A-1. Previous assessments of stocks within this ESU have identified several stocks as being at risk or of concern. Nehlsen et al. (1991) identified seven stocks as at high extinction risk and seven stocks as at moderate extinction risk. Higgins et al. (1992) provided a more detailed analysis of some of these stocks, and identified nine chinook salmon stocks as at risk or of concern. Four of these stocks agreed with the Nehlsen et al. (1991) designations, while five fall-run chinook salmon stocks were either reassessed from a moderate risk of extinction to stocks of concern (Redwood Creek, Mad River, and Eel River) or were additions to the Nehlsen et al. (1991) list as stocks of special concern (Little and Bear Rivers). In addition, two fall-run stocks (Smith and Russian Rivers) that Nehlsen et al. (1991) listed as at moderate extinction risk were deleted from the list of stocks at risk by Higgins et al. (1992), although the USFWS (1997a as cited by NMFS, 1998) reported that the deletion for the Russian River was due to a finding that the stock was extinct. Nickelson et al. (1992) considered 11 chinook salmon stocks within the ESU, of which 4 (Applegate River fall run, Middle and Upper Rogue River fall runs, and Upper Rogue River spring run) were identified as healthy, 6 as depressed, and 1 (Chetco River fall run) as of special concern due to hatchery strays. Huntington et al. (1996 as cited by NMFS 1998) identified three healthy Level II fall-run stocks in their survey (Applegate and Middle and Upper Rogue Rivers).

No current information was available for many river systems in the southern portion of this ESU, which historically maintained numerous large populations. These populations form a genetically distinct subgroup within the ESU. NMFS concluded these California coastal populations do not form a separate ESU. However, they represent a considerable portion of genetic and ecological diversity within this ESU.

Current hatchery contribution to overall abundance is relatively low except for the Rogue River spring run, which also contains almost all of the documented spring-run abundance in this ESU. Fall-run chinook salmon in the Rogue River represent the only relatively healthy population NMFS could identify in this ESU. And found it questionable whether there are sustainable populations outside the Rogue River Basin. All river basins have degraded habitats resulting from agricultural and forestry practices, water diversions, urbanization, mining, and severe recent flooding.

Table A-1. Status of Southern Oregon and Northern California Coastal Chinook Salmon ESU (NMFS 1998).

				Status summaries[3]							Recent abundance		Trends		
River Basin	Sub-basin	Run[1]	Production[2]	A	B	C	D	E	P?[4]	Data Years	Data Type[5]	5-Year Geo. Mean[6]	Long-term[7]	Short-term[8]	Data References
Hunter Creek		Fa		A		D			P						
	Upper	Fa	Natural							1986-96	PI		36.3	36.3	BE and LGL 1995, ODFW 1997e, PSMFC 1997b
Winchuck R	Bear Cr	Fa	Natural	B		D			P	1964-96	AC/PI	592	-2.3	12.0	Nicholas and Hankin 1988, ODFW 1993, BE and LGL 1995, PFMC 1997, PSMFC 1997b
Smith R		Sp		A	A										
		Fa		B					P						
	South Fork	Sp							P	1991-97	SC		30.7	+30.7 (1987-97)	USFS 1997a
	Middle Fork	Sp							P	1991-97	SC		-4.4	-4.4 (1987-97)	USFS 1997a
	North Fork	Sp							P	1992-96	SC		26.2		USFS 1997a
	Mill Cr	Fa	Mixed							1980-96	SC		-1.1	1.9	BE and LGL 1995, PSMFC 1997b, Waldvogel 1997
Klamath R	Lower tributaries	Fa		B	B				P						
	Blue Cr	Fa								1988-96	SN		14.9	14.9	YTFP 1997b
Redwood Cr		Fa		B	C				P						
	Little R	Fa			C				P						
Mad R		Fa		B	C				P						
	North Fork	Fa	Mixed							1985-93	SC		-29.0		BE and LGL 1995, PSMFC 1997b
	Canon Cr	Fa	Natural					<TD DTH ="3 %"		1964-97	PI		-4.9	+0.1 (1987-97)	PFMC 1997
Humboldt Bay	Tributaries	Fa		A	A				P						
Eel R		Fa			C				P	1951-97	DC	16	3.6	-29.7 (1987-97)	PSMFC 1997b, SEC 1997
	Lower	Fa		B											
	Sprowl Cr	Fa	Natural							1967-97	PI		-4.7	-12.4 (1987-97)	PFMC 1997
	Tomki Cr	Fa	Natural							1964-97	TE	25	-15.6	-37.5 (1987-97)	BE and LGL 1995, PFMC 1997, PSMFC 1997b
	South Fork	Fa	Natural							1938-75	WC	4,022	-0.2		BE and LGL 1995

Table A-1. Status of Southern Oregon and Northern California Coastal Chinook Salmon ESU (NMFS 1998) (Continued)

NOTES

1 Run timing designations: Fa -- fall; Sp -- spring; Su -- summer; Wi -- winter (as reported by data reference).

2 Production: (as reported by data reference).

3 Status summaries from the following sources:

A--Nehlsen et al. (1991):

E, endangered (US); X, extinct; A+, possibly extinct; A, high extinction risk; B, moderate extinction risk; C, special concern.

B--Higgins et al. (1992):

A, high risk of extinction; B, moderate risk of extinction; C, stock of concern.

C--Nickelson et al. (1992):

H, healthy; D, depressed; S, special concern; U, unknown.

1, May not be a viable population; 2, Hatchery strays; 3, Small, variable run.

D--WDF et al. (1993): Three characters represent stock origin, production type, and status, in that order.

Origin: N, native; M, mixed; X, non-native; U, unknown; -, unresolved by state and tribes.

Production: W, wild; C, composite; A, cultured; U, unknown; -, unresolved.

Status: H, healthy; D, depressed; C, critical; U, unknown.

E--Huntington et al. (1996):

H-I, healthy Level I (abundance at least two-thirds as great as would be found in the absense of human impacts).

H-II, healthy Level II (abundance between one-third and two thirds as great as expected without human impacts).

4 Petition status [P?]: Indicates (by 'P') stocks included in the ONRC and Nawa petition dated 31 January 1995. Parentheses indicate stock is included as part of a larger unit in the petition.

5 Data Type Codes:

AC, angler catch expanded (1988-92); CS, carcass; DC, dam count; FM, fish per mile; HE, total estimated hatchery escapement; IT, index total; PC, peak or index live fish, surveys combined; PI, peak or index live fish; PR, peak redd count; RC, redd count; RH, resting hole counts; RM, redds per mile; RMC, redds per mile (surveys combined); SC, spawner counts; SN, snorkle counts; TC, trap count; TE, total estimated escapement (includes hatchery escapement only for mixed production type); TL, total live fish count; WC, wier count.

6 Most recent 5 years of data used to calculate spawning escapement geometric mean. (Expanded angler catch = 1988-92).

7 Trend (Long-term): Calculated for all data collected after 1950.

8 Short-term Trend: Calculated for most recent 7-10 years during the period 1987-96, except as noted.

NMFS was very concerned about the risks to spring-run chinook in this ESU; their stocks are in low abundance and they have continued to decline dramatically in recent years. In addition, the lack of population monitoring, particularly in the California portion of the range, led to a high degree of uncertainty regarding the status of these populations.

NMFS (1998) concluded that chinook salmon in this ESU are likely to become at risk of extinction in the foreseeable future. Overall abundance of spawners is highly variable among populations, with populations in California and spring-run chinook salmon throughout the ESU being of particular concern. There is a general pattern of downward trends in abundance in most populations for which data are available, with declines being especially pronounced in spring-run populations. NMFS found that extremely depressed status of almost all coastal populations south of the Klamath River is an important source of risk to the ESU.

A.1.1.3 Distribution

Native spawning populations of chinook salmon are distributed along the Asian coast from Hokkaido, Japan to the Anadyr River and along the North American coast from central California to Kotzebue, Alaska (Moyle 1976; Allen and Hassler 1986; Healey 1991). Chinook salmon spawning may occur from near tidewater in coastal watersheds to over 3,200 km upstream in headwaters of the Yukon River (Major et al. 1978). Introductions of juvenile chinook salmon have also established naturally reproducing populations in New Zealand, Chile and the Great Lakes.

A.1.1.4 Life History

The variable life history patterns of chinook salmon have been thoroughly reviewed by Allen and Hassler (1986) and Healey (1991). Healey (1991) presented a conceptual model that summarized two main components of variation within chinook salmon life histories. The first component is racial, which accounts for the two main behavioral types. “Stream-type” chinook typically spend one or more years as juveniles in fresh water, undertake extensive salt water migrations and return to natal watersheds in the spring or summer several months prior to spawning (Healey 1991). Stream-type chinook are typical of Asian populations and of northern populations and headwater tributaries of southern populations in North America. “Ocean-type” chinook generally migrate to the ocean within three months after emergence, stay within coastal waters during their ocean phase and return to natal watersheds in the fall several days or weeks prior to spawning (Healey 1991). Ocean-type chinook are typical of populations along the North American coast south of 56^{o} N (Healey 1991).

The second component of the life history model is tactical and accounts for variation within each race (Healey 1991). Chinook salmon populations have evolved a range of juvenile and adult behavior patterns that spreads risk across years and across habitats. These patterns include variations in timing of juvenile migrations, variations in length of estuarine residency, variations in age of maturity and variations in adult run timing (Allen and Hassler 1986; Healey 1991).

Chinook salmon in California return to spawn at two to seven years of age, with three `and four year olds comprising the bulk of spawning populations. Two year old males are called jacks or grilse and may comprise 10% to 25% of a spawning run (Allen and

Hassler 1986). Spring runs of chinook (stream-type) generally enter watersheds in May and June, but will not spawn until September and October. The chinook population in the Klamath River is predominately a late August/September run, with spawning occurring from October through December (Snyder 1931; Allen and Hassler 1986). The timing of fall runs (ocean-type) in coastal watersheds is variable and highly influenced by rainfall and stream discharge. Sand bars at the mouths of coastal watersheds must often breach before chinook salmon can enter. Runs may occur from October through January, depending on rainfall.

The fecundity of female chinook salmon is variable, depending on the age and size of the fish and geographic location. Estimates range from 2,000 to 14,000 eggs (Moyle 1976). Klamath River chinook average 3,600 eggs, while Sacramento River fish average 7,300 eggs (Allen and Hassler 1986). After completing her redd female chinook may defend the redd site from four to 25 days, depending on her condition (Neilson and Geen 1981, Neilson and Banford 1983). All chinook salmon eventually die after spawning.The incubation of chinook salmon eggs is inversely related to water temperature. Eggs in 16° C water hatch in about 32 days (Healey 1991). Chinook alevins then spend two to four weeks in the gravel prior to emergence. Survival to emergence is variable and influenced by numerous environmental factors.

A.1.1.5 Habitat Requirements

A.1.1.5.1 Spawning Habitat

Redd sites are selected by female chinook salmon and are usually in pool tails with adequate flow, depth and substrate (Briggs 1953; Allen and Hassler 1986). Velocities of 0.15 to 1.89 m/sec have been recorded at chinook redds (Briggs 1953; Smith 1973; Chapman et al. 1986). Riffle depths at redd sites may range from five to 700 cm (Chapman et al. 1986, Healey 1991). Typically, spring and fall run chinook spawn in 30 to 120 cm of water (Chapman 1949). Chinook salmon construct redds in gravels ranging from 1.3 to 10.2 cm in diameter (Allen and Hassler 1986). Eggs are usually buried 20 to 60 cm below the surface of a completed redd (Briggs 1953). The requirement of sufficient subgravel water flow seems to be of more importance to chinook salmon spawning success relative to other salmonid species (Healey 1991). Chinook produce the largest eggs which have the smallest surface area -to-volume ratio of all salmonid species. Healey (1991) speculates that chinook eggs would be more sensitive to reduced oxygen levels and require a more certain rate subgravel water flow.

A.1.1.5.2 Rearing Habitat

A large downstream migration of chinook fry right after emergence is common in most populations, and may be a dispersal mechanism to distribute fry among all suitable rearing habitats (Bjornn 1971; Reimers 1971). Once started downstream, chinook fry may continue to the estuary or take up residence in the watershed for a period ranging from several weeks to a year or more (Healey 1991). Residing fry will initially seek cover along channel margins or in low velocity areas associated with the channel bottom. As they grow larger they tend to establish territories in faster, deeper habitats (Everest and Chapman 1972). Overwintering (stream-type) juveniles seek shelter under large boulders and woody debris, a habitat shift probably caused by lower water temperatures and increased flows (Chapman and Bjornn 1969).

Estuaries play a vital role in the life cycle of chinook salmon. Fry of ocean-type chinook often migrate downstream immediately after emergence and rear to smolt size in estuaries (Healey 1991). Chinook migrating as young-of-the-year or yearling smolts also rely on estuarine habitat for additional growth and acclimation to saline water prior to oceanic migrations. There is a tendency for ocean-type chinook juveniles to make extensive use of estuarine habitat, whereas stream-type chinook juveniles briefly utilize their watershed's estuary (Healey 1991).

A.1.2 Coho Salmon (*Oncorhynchus kisutch*)

A.1.2.1 Listing Status

On October 20, 1993, the National Marine Fisheries Service (NMFS) received a petition to list coho salmon throughout its range in Washington, Oregon, Idaho, and California, and subsequently initiated a status review to determine if the petitioned action was warranted (January 26, 1994, 59 FR 3662). On July 25, 1995 (60 FR 38011), NMFS published a proposed rule to list the Southern Oregon/Northern California Coasts (SONCC) coho salmon evolutionarily significant unit [1] (ESU) as threatened. This ESU extends from Cape Blanco in Curry County, Oregon, to Punta Gorda in Humboldt County, California. On May 6, 1997 (62 FR 24588), NMFS listed the SONCC coho salmon ESU as threatened.

On November 25, 1997 (62 FR 62741), NMFS published a proposed rule to designate critical habitat for SONCC coho salmon. Critical habitat for SONCC coho salmon was designated on May 5, 1999 (64 FR 24049) and encompasses accessible reaches of all rivers (including estuarine areas and tributaries) between Cape Blanco and Punta Gorda. Excluded are areas above specific dams or above longstanding, naturally impassable barriers (i.e., natural waterfalls in existence for at least several hundred years). Major river basins containing spawning and rearing habitat for this ESU comprise approximately 18,090 square miles in California and Oregon. The following counties lie partially or wholly within watersheds inhabited by this ESU: California - Del Norte, Glenn, Humboldt, Lake, Mendocino, Siskiyou, and Trinity; Oregon - Coos, Curry, Douglas, Jackson, Josephine, and Klamath. More detailed critical habitat information (i.e., specific watersheds, migration barriers, habitat features, and special management considerations) for this ESU can be found in 64 FR 24049.

In February 2004, the California Fish and Game Commission determined that coho from Punta Gorda to the Oregon border should be listed as threatened. The regulation listing the coho became effective on March 30, 2005.

A.1.2.2 Status of ESU Populations

Risk to Populations of Southern Oregon/Northern California Coasts Coho Salmon ESU (from: NOAA-NWFSC Tech Memo-24: Status Review of Coho Salmon; (NMFS, 1994a)

[1] An Evolutionarily Significant Unit is a distinct population segment that is substantially reproductively isolated from other conspecific population units and represents an important component in the evolutionary legacy of the species (Waples 1991).

All coho salmon stocks between Punta Gorda and Cape Blanco are depressed relative to past abundance, but there are limited data to assess population numbers or trends. The main stocks in this region (Rogue River, Klamath River, and Trinity River) are heavily influenced by hatcheries and, apparently, have little natural production in mainstem rivers. The apparent declines in production in these rivers, in conjunction with heavy hatchery production, suggest that the natural populations are not self-sustaining. The status of coho salmon stocks in most small coastal tributaries is not well known, but these populations are small. There was unanimous agreement among the Biological Review Team (BRT) that coho salmon in this ESU are not in danger of extinction but are likely to become endangered in the foreseeable future if present trends continue (Table A-2).

Table A-2. Summary of risk considerations for Southern Oregon/Northern California Coast Coho Salmon ESU (NMFS 1994a).

Risk category	Considerations
Absolute numbers (Recent average)	Run size ca. 10,000 natural, 20,000 hatchery. Current production largely in the Rogue and Klamath basins.
Numbers relative to historical abundance and carrying capacity	Substantially below historical levels. In California portion of ESU, ca. 36% of coho streams no longer have spawning runs. Widespread habitat degradation.
Trends in abundance and production	Long-term trends clearly downward. Main data are for Rogue River basin, where runs declined to very low levels in 1960s and 1970s, then increased with start of hatchery production.
Variability factors	Low abundance or degraded habitat may increase variability.
Threats to genetic integrity	Most existing populations have hatchery plantings, with many out-of-state stock transfers in California portion of the ESU.
Recent events	Recent droughts and change in ocean production have probably reduced run sizes.
Other Factors	None identified.
Conclusion	Not presently in danger of extinction, but likely to become so.

A.1.2.3 Distribution

Globally, coho salmon spawn in coastal watersheds in both Asia and North America. In Asia they are distributed from Hokkaido, Japan to the Anadyr River in Russian Siberia (Moyle 1976; Hassler 1987). In North America coho salmon are distributed from Point Hope, Alaska south to the northern edge of Monterey Bay (Moyle 1976). Along the North American coast coho salmon are most abundant between southern Oregon and southeast Alaska. In California, coho salmon are the second most abundant of the five species of Pacific salmon. They are found in numerous coastal drainages from the Oregon/California border to Waddel and San Lorenzo Creeks of Santa Cruz county (Sandercock 1991). Coho salmon are uncommon and, in the Sacramento River despite several attempts (1956, 1957 and 1958) to establish populations through plantings of juveniles (Hallock and Fry 1967).

The Southern Oregon/northern California coasts coho ESU includes coho salmon from Cape Blanco in southern Oregon to Punta Gorda in northern California. Geologically, this region includes the Klamath Mountains Geologic Province, which has soils that are not as erosive as those of the Franciscan Formation to the south (NMFS, 1994a). Dominant vegetation along the coast is redwood forest, while some interior basins are much drier than surrounding areas and are characterized by many endemic plant species. Elevated stream temperatures are a factor in some areas, but not to the extent that they are in areas south of Punta Gorda.

Rivers in this ESU are relatively long compared to those to the south. With the exception of major river basins such as the Rogue and Klamath, most streams in this region have short duration of peak flows and relatively low flows given both peak flow levels and basin sizes, compared to rivers farther north (NMFS 1994a). Freshwater fishes include elements of the Sacramento River fauna as well as from the Klamath-Rogue ichthyofaunal region. Strong and consistent coastal upwelling begins around Cape Blanco and continues south into central California, resulting in a relatively productive nearshore marine environment. In contrast to coho salmon from north of Cape Blanco, which are most frequently captured off the Oregon coast, coho salmon from this region are captured primarily in California waters.

Genetic data indicate that most samples from this region differ substantially from coho salmon from south of Punta Gorda. In general, populations from southern Oregon also differ from coastal Oregon populations north of Cape Blanco. However, some samples from the Rogue River show an unexplained genetic affinity to samples from outside the region, including some from the Columbia River. In addition, a sample from the Elk River (just south of Cape Blanco) clustered with samples from the Umpqua River (NMFS 1994a).

The southern boundary of this ESU is farther south than the boundary designated for the Klamath Mountains Province steelhead ESU, which includes the Klamath River but not drainages to the south (Busby et al. 1994 as cited by NMFS 1994a). Both the steelhead and coho salmon ESUs share the northern boundary of Cape Blanco. Although the Klamath River (inclusive) serves as the southern boundary for the Klamath Mountains Geological Province and for freshwater fish faunas, major changes in ocean currents and environmental characteristics, as well as the southern limit of the steelhead half-pounder life history strategy, occur at Cape Mendocino/Punta Gorda.

Consequently, the southern limit of the steelhead ESU was based primarily on strong genetic discontinuity between Klamath River steelhead and steelhead populations to the south (Busby et al. 1994 as cited by NMFS 1994a). In contrast, Punta Gorda serves as the southern boundary of the southern Oregon/northern California coho salmon ESU because of the strong environmental transition at Punta Gorda, and because genetic data indicate Punta Gorda, rather than the Klamath River, as an approximate transition area for coho salmon.

For California coho salmon, Pacific Rivers Council et al. (1993 as cited by NMFS, 1994a) reported that Brown and Moyle (1991) estimated that naturally spawned adult coho salmon (regardless of origin) returning to California streams were less than 1% of their abundance at mid-century, and indigenous, wild coho salmon populations in California did not exceed 100 to 1,300 individuals. They further state that Brown and Moyle (1991) found that 46% of California streams, which historically supported coho salmon populations, and for which recent data were available, no longer supported runs (NMFS 1994a).

A.1.2.4 Life History

The life history of coho salmon in California has been well documented by several authors (Shapovalov and Taft 1954; Moyle 1976; Hassler 1987; Sandercock 1991). The life cycle of coho salmon is from two to five years, with three years being most common. Juveniles usually spend at least one year in freshwater before out-migrating to the ocean (juveniles in Alaskan watersheds commonly reside for two years). Coho salmon from California watersheds then spend one to two years at sea before returning to spawn in their natal watersheds (Alaskan coho may stay at sea for three years). The primary exception to this pattern are jacks, sexually mature males that return to freshwater to spawn after only 5-7 months in the ocean (NMFS 1994a). Jacks are a highly variable component of a spawning run. For example, Murphy (1952) summarized counts of coho salmon passing over Benbow Dam on the South Fork of the Eel River from 1939-51 and jacks comprised from 6.9% to 33.8% of the total coho escapement. There is a latitudinal cline in the proportion of jacks in a coho salmon population, with populations in California having more jacks and those in British Columbia having almost none (Drucker 1972 as cited by NMFS 1994a). Although the production of jacks is a heritable trait in coho salmon (Iwamoto et al. 1984), it is also strongly influenced by environmental factors (Shapovalov and Taft 1954, Silverstein and Hershberger 1992 as cited by NMFS 1994a). The proportion of jacks in a given coho salmon population appears to be highly variable and may range from less than 6% to over 43% over 9-35 years of monitoring (Shapovalov and Taft 1954, Fraser et al. 1983, Cramer and Cramer 1994 as cited by NMFS 1994a).

Spawning occurs from early September through March, with peak periods between November and January. In the Klamath River, returning coho enter between September and December, with most spawning occurring during October and November. However, many spawning runs in California occur only after heavy rains have elevated stream flows to breach sand bars at the mouths of some coastal watersheds. If conditions (flow, temperature) in a coastal watershed are unsuitable, coho will postpone migration for weeks or months until conditions change (Sandercock 1991). Coho in large watersheds such as the Klamath River may migrate 65 to 130 miles to spawning sites in tributaries. Coho in smaller coastal watersheds rarely migrate more than 60 miles before spawning in upper sections of main channels or in smaller tributaries. There is also a tendency of earlier run fish to migrate further upstream than late run fish (Briggs 1953). After completing her redd, female coho salmon may remain near the redd for three to 23 days and defend the redd site until too weak to do so (Briggs 1953). All coho salmon die after spawning.

Fecundity of female coho salmon is variable depending on size of female, geographic location and age of spawner. Hassler (1987) cited values of 1,440 to 5,700 eggs for spawners of 44 to 72 cm from Washington. Shapovalov and Taft (1957) reported an average fecundity of 2,700 eggs from Waddell and San Lozenzo Creeks. Ocean distribution of coho salmon, inferred from marine recoveries of coded-wire-tagged fish, show distinctive differences between regions. Coded-wire tags (CWTs) are primarily recovered in salt or fresh water as the salmon return to their natal streams after overwintering in the ocean (NMFS 1994a). Ocean distribution patterns based on CWT marine recovery patterns have been determined from CWT recovery data for 66 North American hatcheries from the Pacific States Marine Fisheries Commission's (PSMFC 1994 as cited by NMFS 1994a). Ocean distribution patterns for California coho salmon are shown in Table A-3.

Table A-3. Marine recoveries of coded wire tags, expanded for sampling, from selected production facilities in Alaska (AK), British Columbia (BC), Washington (WA), Oregon (OR), and California (CA) by release location, including years released, expanded number of tags recovered by state or province, total number of tags recovered, and percent recoveries by state or province (Data from PSMFC 1994 as cited by NMFS 1994a).

Hatchery	Brood years	Expanded number of marine recoveries (% of total)					
		AK	BC	WA	OR	CA	Total
Iron Gate	1974, 77-84, 88-89	0.0 (0.0)	6.4 (0.1)	14.5 (0.2)	1,715.6 (19.4)	7,098.5 (80.3)	8,835.0
Trinity River	1976-85, 89	0.0 (0.0)	4.0 (0.0)	27.5 (0.1)	4,610.5 (22.5)	15,820.5 (77.3)	20,462.5
Mad River	1975, 78-79, 84-86	0.0 (0.0)	1.1 (0.0)	16.3 (0.7)	495.2 (20.2)	1,933.1 (79.0)	2,445.7
Warm Springs	1984-87	0.0 (0.0)	0.0 (0.0)	2.7 (0.3)	59.9 (7.2)	764.0 (92.4)	826.6

The patterns of recoveries showed marked differences between areas, with extremely limited transition zones between areas (NMFS, 1994a). Eight general CWT recovery patterns were identified, one of which includes Northern California and Oregon south of Cape Blanco. Coho salmon released from the southernmost facilities (those south of Cape Blanco) had the most southerly recovery patterns: these fish were recovered primarily in California (65-92%), with some recoveries in Oregon (7-34%) and almost none (<1%) in Washington or British Columbia. The recovery pattern of coho salmon released from the southernmost hatchery, Warm Springs (Russian River), had a much higher proportion of California recoveries (92%) than the other California and southern Oregon facilities. Whether this represents a unique recovery pattern, or results from the southerly location of the hatchery, is not known (NMFS 1994a).

A.1.2.5 Habitat Requirements

A.1.2.5.1 Spawning Habitat

Redd sites are selected by females and are located in pool tails or slightly upstream of the hydraulic control, where the water changes from a laminar to more turbulent flow. Water depths at redd locations range from 0.18 to 0.46 meters (Smith 1973; Hassler 1987). Redds are located in relatively fast water (0.3 to 0.5 m/sec) which ensures adequate aeration and circulation to facilitate embryo development and fry emergence (Smith 1973; Hassler 1987). Coho salmon utilize small to medium sized substrate ranging from 1.3 to 15.0 cm in diameter (Reiser and Bjornn 1979; Sandercock 1991). Developing coho salmon appear able to tolerate higher concentrations of fines (up to 10%) than other salmonid species, although redds situated in gravels with lower amounts of fines (5% or less) have higher rates of juvenile emergence (Emmett et al. 1991). Excessive amounts of fines deposited on redds reduces oxygen flow to developing eggs and young and impedes successful emergence of juveniles. Briggs

(1953) reported that coho salmon in California spawn in water temperatures ranging from 5.6^{o} C to 13.3^{o} C.

Incubation of eggs takes from 38 to 101 days and is inversely related to water temperature (Hassler 1987). Egg development is slower in colder water and faster in warmer water. After hatching, coho alevins remain in the gravel until their yolk sacs have been absorbed, usually a period of two to ten weeks (Moyle 1976; Hassler 1987). Survival of eggs and alevins to emergence is highly variable and dependent on numerous environmental factors. Under extreme conditions none may survive; under average conditions 15%-27% may survive (Neave 1949); and under ideal conditions 65%-85% may survive (Shapovalov and Taft 1954).

A.1.2.5.2 Rearing Habitat

Newly emerged coho fry seek out shallow water along stream margins, backwaters and side channels (Sandercock 1991). Initially coho fry form schools, but as they grow larger the schools break up and juveniles (parr) tend to establish individual territories (Hassler 1987). Larger, more dominant parr tend to occupy the heads of pools; while smaller parr are found farther downstream (Chapman and Bjornn 1969). As the parr grow, their territories expand until by summer they are located in deep pools. Ideal rearing habitat consists of a mixture of pools and riffles with abundant instream and overhead cover (especially large woody debris), water temperatures between 10^{o} and 15^{o} C, dissolved oxygen near saturation and low amounts of fines (Hassler 1987). Scrivener and Andersen (1984) reported that streams with larger amounts of complex habitat (cobbles, boulders, logs and overhanging riparian vegetation) supported larger numbers of juvenile coho salmon.

By the onset of autumn coho parr decrease feeding activity and migrate into deeper pools with LWD and undercut banks, seeking protection from elevated flows. In some watersheds coho parr will move into tributaries that maintain more stable flows throughout the winter (Tripp and McCart 1983). Towards the end of March coho parr start to migrate downstream and into the ocean. In California, out-migration from small coastal watersheds peaks from mid-April to mid-May (Shapovalov and Taft 1954). Factors affecting time of out-migration include: size of juveniles, flow conditions, water temperature, dissolved oxygen, day length and food availability (Shapovalov and Taft 1954). At the onset of out-migration, juveniles defend territories less aggressively and form aggregations. Out-migrants move in groups of 10 to 50 fish and are of similar size (Shapovalov and Taft 1954). Parr marks are still obvious on early migrants, but later migrants are more silvery, having transformed into smolts. Size of coho smolts seems to be consistent throughout the species geographic range. Several authors have reported an average fork length of 10 to 12 cm for coho smolts (Sumner 1953; Shapovalov and Taft 1954; Salo and Bayliff 1958).

A.1.3 Steelhead and Resident Rainbow Trout (*Oncorhynchus mykiss irideus*)

A.1.3.1 Listing Status

A.1.3.1.1 *Steelhead*

Steelhead from the Illinois River, a Rogue River tributary, were initially petitioned for listing on 5/5/92. On 7/31/92, NMFS published in the *Federal Register* that the listing may be warranted. On May 29, 1993 (58 FR 29390), NMFS concluded that Illinois River winter steelhead did not constitute a "species," and therefore, did not qualify for listing under the ESA. However, NMFS requested biological information for all coastal steelhead populations. On February 16, 1994, NMFS received a petition to list steelhead throughout its range in Washington, Oregon, Idaho, and California, and subsequently initiated a status review to determine if the petitioned action was warranted (May 27, 1994, 59 FR 27527).

Klamath Mountains Province Steelhead

On March 16, 1995 (60 FR 14253), NMFS published a proposed rule to list steelhead in the Klamath Mountains Province (KMP) ESU as threatened. The KMP steelhead ESU was proposed for listing again on August 9, 1996 (61 FR 41541). The KMP steelhead ESU extends from Cape Blanco, Oregon, to the Klamath River Basin, California, inclusive. On March 19, 1998 (63 FR 13347), NMFS determined that listing was not warranted for this ESU. The ESU was reclassified as a candidate for listing due to concerns over specific risk factors, but it was again determined that listing was not warranted for this ESU (66 FR 17845).

Northern California Steelhead

On August 9, 1996 (61 FR 41541), NMFS published a proposed rule to list the Northern California steelhead ESU as threatened. The ESU includes steelhead in California coastal river basins from Redwood Creek south to the Gualala River, inclusive. As with KMP steelhead, on March 19, 1998 (63 FR 13347), NMFS determined that listing was not warranted for the Northern California steelhead ESU. However, the ESU was reclassified as a candidate for listing due to concerns over specific risk factors. Because the State of California has failed to implement conservation measures that NMFS considered critically important in its decision not to list the Northern California steelhead ESU, NMFS completed an updated status review and has reconsidered the status of this ESU under the ESA. On February 11, 2000 (65 FR 6960), NMFS proposed to list Northern California steelhead as threatened. The Northern California steelhead ESU was listed as threatened on June 7, 2000 (65 FR 36075). On January 2, 2006 (70 FR 52488), NMFS made the final designation of critical habitat for the Northern California steelhead. Effective February 6, 2006 (71 FR 834), NMFS utilized the distinct population segment (DPS) policy to delineate steelhead populations rather than the evolutionarily significant unit (ESU) policy. The change is also consistent with the U.S. Fish & Wildlife Service's approach to making listing determinations.

A.1.3.1.2 Resident Rainbow Trout

USFWS recently asserted jurisdiction over the resident form of the rainbow trout, which is genetically indistinguishable from steelhead.

A.1.3.2 Status of Steelhead Populations

A.1.3.2.1 Klamath Mountains Province Steelhead

(From: NOAA-NMFS Tech Memo-19. Status Review for Klamath Mountains Province Steelhead [NMFS 1994b]).

Historical information for northern California populations of steelhead are scarce, although Snyder (1925 as cited by NMFS1994b) noted that trout (including steelhead) were declining in the Klamath River Basin at that time.

Qualitative evaluations considered recent published assessments by agencies or conservation groups of the status of steelhead stocks from Cape Blanco to the Klamath River Basin (Nehlsen et al. 1991; Nickelson et al. 1992; USFS 1993a,b; McEwan and Jackson in prep. (as cited by NMFS 1994b). Results of these assessments are summarized in Table A-4.

Table A-4. Summary of recent qualitative assessments of steelhead abundance for all river basins reviewed. Blanks indicate that a particular run was not evaluated (NMFS 1994b).

River basin	Run-type	Nehlsen risk level[a]	ODFW/CDFG assessment[b]	USFS assessment[c]
Oregon				
Elk River	Winter			Healthy
Euchre Creek	Winter			
Rogue River	Winter		Healthy	Healthy
	Summer	Moderate	Depressed	Depressed
Applegate River	Winter			
	Summer			
Illinois River	Winter	Moderate	Depressed	Depressed
Hunter Creek	Winter			
Pistol River	Winter		Depressed	
Chetco River	Winter		Depressed	Depressed
Winchuck River	Winter		Healthy	Healthy
California				
Smith River	Winter		Healthy	Low abundance
	Summer	High		Depressed
Klamath River	Winter			Low abundance, insufficient information
	Summer	Moderate		Depressed, moderate to high risk
Trinity River	Winter			Stable, depressed
	Summer			Stable, high risk

[a] - Risk of local extinction, as defined in Nehlsen et al. (1991).
[b] - Assessments in state agency documents: Oregon, Nickelson et al. (1992); California, McEwan and Jackson (in prep.).
[c] - General assessments of condition of portions of runs on U.S. Forest Service lands (USFS 1993a,b).

NMFS (NMFS 1994b) attempted to distinguish naturally produced fish from hatchery produced fish in compiling these summary statistics. All statistics were based on data for adults that spawn in natural habitat ("naturally spawning fish"). The total of all naturally spawning fish ("total run size") is divided into two components "Hatchery produced" fish are reared as juveniles in a hatchery but return as adults to spawn naturally; "naturally produced" fish are progeny of naturally spawning fish (NMFS 1994b).

The quantitative and qualitative risk evaluation analyses (NMFS 1994b) revealed the following:

- Although historical trends in overall abundance within the ESU are not clearly understood, there has been a substantial replacement of natural fish with hatchery produced fish.

- Since about 1970, trends in abundance have been downward in most steelhead populations within the ESU, and a number of populations are considered by various agencies and groups to be at moderate to high risk of extinction.

- Declines in summer steelhead populations are of particular concern.

- Most populations of steelhead within the area experience a substantial infusion of naturally spawning hatchery fish each year. After accounting for the contribution of these hatchery fish, we are unable to identify any steelhead populations that are naturally self-sustaining.

- Total abundance of adult steelhead remains fairly large (above 10,000 individuals) in several river basins within the region, but several basins have natural runs below 1,000 adults per year.

The Klamath Mountains Province steelhead ESU was recently reevaluated by NMFS Biological Review Team (66 FR 17845). They reviewed updated abundance and trend information available for this ESU and concluded that the ESU was not in danger of extinction nor likely to become so in the foreseeable future (66 FR 17845).

A.1.3.2.2 *Northern California Steelhead*

(From: NOAA-NMFS NMFS-NWFSC-27 Status Review for West Coast Steelhead from Washington, Idaho, Oregon, and California [NMFS, 1996]).

NMFS review team concluded that the Northern California steelhead DPS (classified as an ESU at the time) is not presently in danger of extinction, but that it is likely to become endangered in the foreseeable future. Nehlson et al.'s (1991) finding's of risk for extinction for Northern California Steelhead are summarized in Table A-5. below.

Population abundances are very low relative to historical estimates (1930s dam counts), and recent trends are downward in stocks for which data were available, except for two small summer steelhead stocks. Summer steelhead abundance is very low. There is particular concern regarding sedimentation and channel restructuring due to floods, apparently resulting in part from poor land management practices. The abundance of introduced Sacramento squawfish as a predator in the Eel River is also of concern.

Table A-5. Northern California Steelhead stocks identified by Nehlsen et al. (1991) as at some risk of extinction.

Extinct	Possibly extinct	High risk	Moderate risk	Special concern
None	None	Redwood Creek Mad River	Eel River	None

For certain rivers (particularly the Mad River), NMFS is concerned about the influence of hatchery stocks, both in terms of genetic introgression and of potential ecological interactions between introduced stocks and native stocks. They found that there are two major areas of uncertainty. Information on steelhead run sizes throughout the DPS is lacking. Their conclusions were based largely on evidence of habitat degradation and the few dam counts and survey index estimates of stock trends in the region. Also, the genetic heritage of the natural winter steelhead population in the Mad River is uncertain. Table A-6. summarizes the spawning escapement estimates for rivers within the Northern California Coastal Steelhead DPS as of the 1960's. Table A-7 provides additional abundance estimates.

Risk factors identified for this DPS include freshwater habitat deterioration due to sedimentation and flooding related to land management practices and introduced Sacramento squawfish as a predator in the Eel River. For certain rivers (particularly the Mad River), NMFS is concerned about the influence of hatchery stocks, both in terms of genetic introgression and potential ecological interactions between introduced stocks and native stocks.

Table A-6. Estimated steelhead spawning populations for Northern California Steelhead ESU rivers in the mid-1960s (CDFG 1965 as cited in NMFS 1996), with comparable recent maximum estimates.

Stream	Population Estimate
Redwood Creek	10,000
Mad River	6,000
Eel River System (Total)	82,000
Mattole River	12,000
Ten Mile River	9,000
Noyo River	8,000
Big River	12,000
Navarro River	16,000
Garcia River	4,000
Gualala River	16,000
Other streams (Humboldt, Mendocino Counties)	23,000
Total	**198,000**

Table A-7. Summary of historical abundance estimates for the Northern California evolutionarily significant unit (as cited in NMFS 1996).

River Basin	Abundance*	Years	Reference
Eel River			
Cape Horn Dam	4,400	1930s	McEwan and Jackson 1996
	1,000	1980s	McEwan and Jackson 1996
Benbow Dam	18,784	1940s	Shapovalov and Taft 1954
	3,355	1970s	McEwan and Jackson 1996
Mad River			
Sweasy Dam	3,800	1940s	Murphy and Shapovalov 1951
	2,000	1960s	McEwan and Jackson 1996
Casper Creek	114	1964	Graves and Burns 1970
	102	1968	Graves and Burns 1970
* Excludes estimates from CDFG 1965.			

A.1.3.3 Distribution

Steelhead are widely distributed from the Kuskokwin River of western Alaska to Baja California (Moyle 1976; Behnke 1992). The anadromous rainbow trout is called the steelhead, which accounts for most of the variable life history patterns. Steelhead populations occur throughout the range of steelhead except in the northern and southern extremities (Behnke 1992). The present southern limit of steelhead distribution is Malibu Creek, California. The southern range of summer run steelhead is the Middle Fork of the Eel River (Barnhart 1986).

A.1.3.4 Life History

The life histories of rainbow trout have been reviewed by numerous authors (Smith 1973; Jones 1976; Moyle 1976; Barnhart 1986; Behnke 1992). The anadromous and resident forms are genetically indistinguishable, and the life history of resident rainbow trout are similar to those of steelhead while in the freshwater phase.

Steelhead populations may be grossly categorized as summer run or fall/winter run fish, depending when spawning adults enter fresh water. This is an oversimplification and adult steelhead probably enter fresh water every month of the year somewhere in their widespread distribution (Behnke 1992). Summer run steelhead are not abundant throughout the Pacific southwest and the runs in many watersheds consist of less than 100 adults (Roelofs 1983).

Summer run fish usually enter fresh water from May through August and move upstream to hold in deep pools until the following winter or spring to spawn. These stream-maturing type steelhead enter fresh water in a sexually immature condition and require several months in freshwater to mature prior to spawning Fall/winter run fish generally enter fresh water from September through November, whereas many coastal watersheds have late runs of winter steelhead that enter fresh water from January through April. These ocean-maturing type steelhead enter fresh water with well-developed gonads and spawn shortly after river entry The partitioning of an anadromous species into distinct races is an excellent reproductive strategy since this enlarges the use of its environment and increases productivity (Behnke 1992).

Adult steelhead are iteroparous and can spawn more than once before dying. Repeat spawners are a significant contribution to many populations. Most populations consist of 10% to 20% repeat spawners (Behnke 1992). Forsgren (1979) reported that second time spawners comprise 70% to 85% of repeat spawners and third time spawners comprise 10% to 25% of repeat spawners. Spawning survival is highly variable and influenced by genetic factors, habitat quality, fishing pressure and management plans.

The fecundity of rainbow trout (either resident or anadromous) is highly variable, from 200 to 12,000 eggs depending on the size of the female (Moyle 1976). Moyle (1976) reported that resident fish usually produce less than 1,000 eggs and that steelhead average about 2,000 eggs per kilogram of body weight.

Incubation of steelhead eggs, as with all salmonids, is inversely related to water temperature. Eggs in 15° C water hatch in approximately 19 days, whereas eggs in 5°C hatch in about 80 days (Barnhart 1986). Steelhead alevins remain in the gravel for two to four weeks and are sustained by their yolk sacs. Survival of eggs and alevins to emergence is highly variable and dependent on numerous environmental factors.

Steelhead reside in fresh water from one to four years before smolting and out-migrating to the ocean. Juveniles in the Pacific southwest typically spend one to two years before smolting (Barnhart 1986). Steelhead then spends one to four years at sea before returning to spawn. The length of both instream and oceanic residency increases from south to north along the species' distribution (Barnhart 1986).

A.1.3.5 Habitat Requirements

The anadromous and resident forms of rainbow trout are genetically indistinguishable, and habitat requirements of resident rainbow trout are similar to those of steelhead while in the freshwater phase (with the possible exception of estuary and some mainstem habitats).

A.1.3.5.1 Spawning Habitat

Spawning usually occurs in pool tails with cool, clear, well-oxygenated water with suitable current velocity, depth and gravel size (Reiser and Bjornn 1979). Depending on the watershed and size of the fish (resident or anadromous), steelhead spawn at depths of 0.10-1.5 meters, in current velocities of 0.23-1.55 m/sec and in gravel of 0.64-12.7 cm in diameter (Smith 1973; Barnhart 1986). Generally summer run steelhead spawn in the upper sections of watersheds, utilizing habitat inaccessible to fall/winter run fish. Steelhead often utilize intermittent streams for spawning purposes (Kralik and Sowerwine 1977; Carrol 1984).

A.1.3.5.2 Rearing Habitat

After emergence, steelhead fry tend to school and seek out shallow water along stream margins. As the fry grow they start to establish and defend individual territories. Most young-of-the-year steelhead fry inhabit riffles or runs (Barnhart 1986). Mortality of juvenile steelhead is highest the first few months after emergence as fry move about and attempt to establish territories (Shapovalov and Taft 1954; Chapman 1966). Larger steelhead fry (age 1+ year and older) generally maintain territories in pool and run habitats. A productive steelhead stream should have summer temperatures of 10° C to 15° C and an upper limit of around 20° C (Barnhart 1986).

A.1.4 Coastal Cutthroat Trout (*Oncorhynchus clarki clarki*)

A.1.4.1 Listing Status

Coastal cutthroat trout were listed as endangered in the Umpqua ESU in 1996. On April 5, 1999, NMFS determined that listing was not warranted for the Oregon Coast ESU. However, the ESU was designated as a candidate for listing due to concerns over specific risk factors. This ESU included populations of coastal cutthroat trout in Oregon coastal streams south of the Columbia River and north of Cape Blanco (including the Umpqua River Basin. On April 5, 1999, NMFS also determined that listing was not warranted for the Southern Oregon/California Coast Cutthroat trout ESU. The ESU included populations of coastal cutthroat trout from south of Cape Blanco to the southern extent of the subspecies' range (approximately the Mattole River in California). This species is now formally under the jurisdiction of the U.S. Fish and Wildlife Service and at the current time a review of the status of this species in being conducted.

A.1.4.2 Distribution

Coastal cutthroat trout are found in coastal drainages from the Eel River in northern California (Dewitt 1954) to Prince William Sound in Alaska (Trotter 1989). The inland limits of coastal cutthroat trout distribution are most likely the Fraser River in British Columbia and Celilo Falls on the Columbia River (Crawford 1979; Trotter 1989).

A.1.4.3 Life History

The life history of coastal cutthroat trout has been reviewed by numerous authors (Dewitt 1954; Sumner 1962; Armstrong 1971; Johnson 1981; Pauley et al. 1989; Trotter 1989; Behnke 1992). Trotter (1989) described three typical life history forms of coastal cutthroat trout: an anadromous form, a potamodromous form that includes lake and stream-dwelling populations and a non-migratory form which lives in small streams and headwater tributaries. Anadromy tends to be poorly developed. Anadromous populations occur sympatrically and allopatrically with resident populations throughout their distribution (Michael 1989; Pauley et al. 1989; Trotter 1989).

Depending on time of entry, coastal cutthroat trout spawn from December to May. In California, Oregon, Washington and southern British Columbia the peak month is February, whereas in Alaska spawning peaks in April and May. The age of first time spawning females ranges from two to five years old.

Coastal cutthroat trout may spawn more than once. Sumner (1962) reported that in an Oregon coastal stream 39% of coastal cutthroat survived their intial spawning migration, 17% survived a second spawn and 12% survived a third spawn. These data were collected on a watershed lacking an intensive coastal cutthroat fishery.

The fecundity of female coastal cutthroat varies with age and size. Scott and Crossman (1973) reported a range of values from 226 eggs from a 20 cm fish to 4,420 eggs from a 43 cm fish. Forty coastal cutthroat trout collected from McDonald Creek in northern California had an average fecundity of 1,400 eggs (Taylor 1996).

Eggs of coastal cutthroat trout hatch after six to seven weeks of incubation, depending on water temperature. The alevins remain in the gravel approximately two weeks before emergence. The emergence of coastal cutthroat trout fry occurs from March through June, depending on the locale and time of spawning (Trotter 1989).

Anadromous coastal cutthroat trout smolt and migrate to the ocean between the ages of one to six years old (Trotter 1989). There seems to be a relationship between the age and size of smolting and the type of marine environment the smolts enter. For example, in McDonald Creek where smolts enter an enclosed brackish lagoon, a majority of cutthroat smolts out-migrated as one year olds (Taylor 1996). Smolts from coastal watersheds which flow directly into rough surf that forces them offshore tend to out-migrate as three to six year olds. Johnson (1981) speculated that physical and biological characteristics of the marine environment have exterted selective pressures to account for the differences in smolt age and size.

Potamodromous coastal cutthroat trout display migratory patterns similar to anadromous cutthroat, except the resident fish do not migrate to the ocean. Instead their migrations consist of foraging during the spring and summer in main channels of watersheds or in lakes and then migrating into tributaries for spawning purposes (Trotter 1989). Spawning tributaries may be either upstream or downstream from feeding areas. Potamodromous coastal cutthroat trout utilize similar spawning habitat as anadromous forms, and may even contribute to or maintain anadromous populations (Royal 1972; Jones 1979).

Non-migratory coastal cutthroat trout that live in isolated headwater tributaries, remain small in size (150-200 mm), and seldom live past the age of three or four years (Trotter 1989). Females tend to mature by the age of two or three years. The entire life cycle of non-migratory cutthroat trout may be completed in less than 200 meters of stream channel (Wyatt 1959).

A.1.4.4 Habitat Requirements

Coastal cutthroat trout spawn in cool, well oxygenated water with suitable velocity, depth and substrate composition. Coastal cutthroat tend to spawn in first and second order tributaries and isolated headwaters where interactions with other salmonids (primarily steelhead and coho salmon) are reduced (Johnson 1981). Redd sites are generally located in pool tails with protective cover nearby. Spawning has been observed in velocities ranging from 0.11 to 1.02 m/sec, in riffle depths of 0.10 to 1.00 meters and in substrate 0.6 to 10.2 cm in diameter (Smith 1973; Pauley et al. 1989; Taylor 1996).

Total length of newly emerged fry is about 25 mm. They move into low velocity areas along the stream margin, backwater pools and side channels (Moore and Gregory 1988). Fry will remain in these habitats for the entire summer if there is little or no competition from other salmonid species. However, larger coho salmon fry exert social dominance over cutthroat fry and force cutthroat fry into riffles, where they stay until autumn when lower water temperatures reduce aggression in coho and/or elevated flows displace them from the riffles (Glova and Mason 1976).

A.1.5 Tailed Frog (*Ascaphus truei*)

A.1.5.1 Listing Status

This species previously was considered a Category 2 candidate for listing; USFWS subsequently has dropped the "C2" category in its list of species that are listed, proposed for listing, or candidates for listing.

A.1.5.2 Distribution

The tailed frog is the only member of the genus *Ascaphus.* It is endemic to the Pacific Northwest and is widely distributed from northwestern California to British Columbia and western Montana (Nussbaum et al. 1983). Tailed frogs are found at elevations from sea level to near timber line throughout the coastal mountains from British Columbia south to Mendocino County and in the inland mountains of southeast Washington, Idaho, and Montana (Metter 1968). In California, they occur from sea level to 6500 feet, mostly at sites receiving over 40" of precipitation annually in Siskiyou, Del Norte, Trinity, Shasta, Tehama, Humboldt, Mendocino, and possibly Sonoma counties (Bury 1968). Throughout much of its range the species is distributed as disjunct populations (Metter 1968). Bury and Corn (1988a) believed that isolated, discrete populations most likely occurred in drier forests and heavily managed lands.

A.1.5.3 Life History

Tailed frogs have been shown to breed in both the spring and early fall in different portions of their range. Breeding occurs in the water with the males utilizing the "tail" as a copulatory organ to accomplish internal fertilization. Eggs are deposited in the summer and hatch after four to six weeks (Brown 1990). In coastal regions, the tadpoles typically do not emerge from the nest site until later in the fall (Wallace and Diller, 1998); in interior regions, they over-winter at the nest site and emerge the next spring (Metter 1964). The tadpoles metamorphose into adults in varying time periods depending on the characteristics of the regional population. The larval period may last for 1-4 years; it is shorter in more coastal and lower elevation populations and longer in more inland and higher elevation populations (Daugherty and Sheldon 1982, Nussbaum et al. 1983, Metter 1964 and 1967, Brown 1990, and Wallace and Diller, 1998t).

Adult tailed frogs feed on a wide variety of terrestrial and aquatic invertebrates (Metter 1964). They feed both in the water and on land, and may actively forage in adjacent forests on wet or rainy nights (Nussbaum et al. 1983). The tadpoles feed primarily on diatoms which they scrape off rocks with an enlarged suction-like mouth. Their suction-like mouth also enables them to attach themselves to rocks and other objects in swift flowing water to prevent being washed downstream.

A.1.5.4 Habitat Requirements

Tailed frog habitat has been characterized as perennial, cold, fast flowing mountain streams with dense vegetation cover, or streams in steep-walled valleys in nonforested areas (Bury 1968, Nussbaum et al. 1983). The frogs may inhabit spray drenched cliff walls near waterfalls (Zeiner et al. 1988), but avoid marshes, lakes, and slow sandy streams (Daugherty and Sheldon 1982).

To support larval tailed frogs, streams must have suitable gravel and cobble for attachment sites and diatoms for food (Bury and Corn 1988a). Streams supporting tailed frogs have been found primarily in mature (Bury and Corn 1988a, Welsh 1990) and old growth coniferous forests (Bury 1983, Welsh 1990). Bury and Corn (1988a) reported that the frogs seem to be absent from clearcut areas and managed young forests (Welsh 1990), although they have been observed to occur commonly in young managed forests in coastal California Diller and Wallace, 1999). Welsh (1990) also observed them in young naturally regenerated forests and suggested that structure rather than age per se of old growth was important to the animals. In California, tailed frogs have been found in Sitka spruce, redwood, Douglas-fir, and ponderosa pine forests (Bury 1968).

A.1.6 Southern Torrent Salamander (*Rhyacotriton variegatus*)

A.1.6.1 Listing Status

This species previously was considered a Category 2 candidate for listing; USFWS subsequently dropped the “C2” category in its list of species that are listed, proposed for listing, or candidates for listing. On June 6, 2000 the USFWS announced that, after review, the southern torrent salamander did not warrant listing as endangered or threatened at this time. USFWS recommended that the species remain on the Federal Species of Concern list.

A.1.6.2 Distribution

The southern torrent salamander is one of four species in the genus *Rhyacotriton* and is the most southerly ranging. Recent genetic studies (Good and Wake 1992) split the former Olympic salamander (*R. olympicus*) into four separate species. It is the only species of the genus that occurs in California. Southern torrent salamanders occur west of the Cascades from northwestern Oregon south to Mendocino County in California (Good and Wake 1992). Bury and Corn (1988a) believed that the salamanders are distributed as isolated, discrete populations, especially in heavily managed or drier forests. In California, the species is found in the coastal forests of northwestern California south to Mendocino County (Anderson 1968).

A.1.6.3 Life History

The southern torrent salamander has an aquatic dependent larval stage that may last for two to four years (Nussbaum and Tait 1977) followed by metamorphosis into an adult form. The larvae occupy the interstices among gravels and cobble in the stream. Transformed adults occur in the same microhabitats as the larvae, but are also found under objects along stream edges and in splash zones. Both larvae and adults feed on a variety of small aquatic and semiaquatic invertebrates that are located in the stream or along the margins of the stream (Bury and Martin 1967, Bury 1970). These salamanders are generally believed to have low dispersal capabilities, with annual in-stream movements reported to be usually only several meters (Nussbaum and Tait 1977, Welsh and Lind 1992). However, there is evidence based on pitfall traps that adults can disperse significant distances of up to about 100 meters from streams during wet periods of the year (Grialou et al. 1995).

Breeding is thought to occur for an extended period of time, with the peak of egg-laying probably in spring or early summer (Nussbaum and Tait 1977). Little is known about the selection of sites for egg-laying, but the incubation period is believed to be long, which would result in the eggs over-wintering in the stream.

A.1.6.4 Habitat Requirements

In general, these salamanders occupy humid coastal (Anderson 1968) coniferous forests at maximum elevations that were thought to be 3900 feet (Welsh 1990); but recent field surveys (Diller unpubl. Report) indicate that they can be found up to approximately 5000 feet. They are most commonly associated with the uppermost portions of cold, well shaded permanent streams with a loose gravel substrate (Anderson 1968, Nussbaum et al. 1983), springs, headwater seeps (Welsh 1990), waterfalls (Bury and Corn 1988a), and moss covered rock rubble with flowing water (Anderson 1968). Torrent salamanders also can be found in streams with little surface flow, and they may persist in streams with segments of subsurface flow during the dry summer season. The adult salamanders may inhabit moist stream banks and splash zones, but are rarely found more than 1 m from water (Nussbaum and Tait 1977). They have been observed wintering in talus slopes (Herrington 1988). Bury (1983) did not find torrent salamanders in 6-14 year old logged streams and Bury and Corn (1988a) found the salamanders to be more numerous in streams in uncut 60-500 year old stands than in 14-40 year old regenerated area stands (Bury and Corn 1988a). However, in coastal young growth forests, Diller and Wallace (1996) reported finding no relationship between torrent salamander occurrence and stand age and found salamanders in a high proportion of streams, including recently logged areas.

The other salamander that most closely occupies the same stream microhabitat as the torrent salamander is the larval stage of the Pacific giant salamander (*Dicamptodon tenebrosus*). The Pacific giant larvae grow larger in size and not only compete with torrent salamanders, but probably also prey on them. It is unknown whether Pacific giant salamanders exclude or limit torrent salamanders from certain streams or segments of streams, but have been reported to eat torrent salamander eggs (Nussbaum et al. 1983).

A.2 SENSITIVITY OF THE COVERED SPECIES TO IMPACTS

A.2.1 Anadromous Salmonids

The causes of decline of anadromous salmonids in California are numerous and often interactive but can be grouped into four general categories:

- Degradation or loss of freshwater habitat.
- Interactions with hatchery salmonids.
- Overexploitation of stocks by commercial fishing.
- Climatic factors such as ocean conditions and precipitation timing and amounts.

A.2.1.1 Habitat Degradation and/or Loss

According to Nehlsen et al. (1991) and Reeves and Sedell (1992), degradation and/or loss of freshwater habitat is the single largest cause in the decline of anadromous salmonids along the Pacific northwest Watershed disturbances associated with urbanization, timber harvesting, mining, agriculture, livestock grazing, dams, and water diversions have all contributed to the loss of freshwater habitat.

These human activities have typically reduced the complexity of habitat often associated with productive salmonid streams, especially reductions of LWD and increased sedimentation in pools and spawning riffles (Sandercock 1991). Sedimentation (resulting in shallowing of pools) and removal of riparian vegetation has also lead to excessive increases in summer water temperatures in some salmonid watersheds.

Loss of spawning and rearing habitat has also occurred through human activities which denied migrating adults access to traditional spawning areas. Dams on the Klamath, Trinity, Mad, Eel, Sacramento and San Joaquin Rivers have all severely impacted runs of salmon and steelhead in California. These dams have either prohibited fish access to traditional spawning and rearing areas and/or degraded downstream habitat conditions. Improperly installed culverts have reduced or prohibited access of migrating spawners to tributaries within numerous coastal watersheds.

A.2.1.2 Interactions with Hatchery Salmonids

Interactions with hatchery salmonids have possibly impacted wild stocks of salmonids through:

- potential loss of genetic integrity;
- competition between juveniles;
- transmission of diseases.

Although widely cited as occurring, the loss of genetic integrity is difficult to determine because the amount of interbreeding between native and non-native stocks is poorly understood (Hindar et al. 1991). Stocks of coho in California do not appear to be strongly differentiated genetically (Bartley et al. 1992). This lack of differentiation may be caused by transplants of stocks within California plus the introduction of coho from various Oregon and Washington watersheds decades prior to the ability to determine an individual’s genetic composition (Bartley et al. 1992).

Several studies have reported reduced densities of wild juvenile coho after the release of hatchery juveniles (Nickelson et al. 1986; Miller et al. 1990). Miller et al. (1990) also reported similar reductions in the subsequent adult returns. In subsequent years, Nickelson et al. (1986) detected a shift towards earlier returning adult spawners, which is indicative of hatchery fish (Brown et al. 1994). These reductions in native juvenile densities may occur because juvenile coho are territorial and the larger hatchery fish displace the natives from preferred habitat (Nickelson et al. 1986). When displaced from established territories, juvenile coho are more suceptible to predation and may also experience reduced growth rates which may further affect survival to maturity (Puckett and Dill 1985; Steward and Bjornn 1990).

The transmission of diseases from hatchery salmonids to native stocks is potentially a serious problem, yet little information exists to confirm the extent of this concern because of limited field investigations (Steward and Bjornn 1990; Kruger and May 1991). An example of hatchery salmonids passing diseases to wild fish recently occurred in the Madison River in Montana where planted rainbow trout infected the wild population with whirling disease. In three years the Madison River's rainbow trout population declined by more than 90% (Holt 1995). The following virulent diseases affect hatchery salmonids and have the potential to infect wild stocks: viral hemorrhagic septicemia, bacterial kidney disease, infectious hematopoietic necrosis, herpes virus and infectious pancreatic necrosis (Håstein and Lindstad 1991).

A.2.1.3 Over-exploitation

Excessive harvest by commercial fishing is commonly cited as a significant factor in the decline of chinook and coho salmon, but the effects are hard to quantify since catch records rarely distinguish between wild and hatchery stocks (Steward and Bjornn 1990). In mixed-stock commerical fisheries, wild stocks may be overfished because they are unable to sustain the same harvest rates as hatchery fish.

Female coho salmon in California mainly have a three year life cycle, thus they lack the ability to withstand overharvest compared to other salmonids in which a single year class matures at a variety of ages. For example, the coho runs in Scott and Wadell Creeks (the southern most coho populations) have exclusively three year life cycles and only experience a strong return once every three years because two of the year classes are severly depressed (Brown et al. 1994).

Although steelhead are not fished commercially in the United States, exploitation by foreign fleets has been blamed in the decline of steelhead stocks. Asian fleets gillnetting squid in the Gulf of Alaska have been long suspected as a major harvester of steelhead from North American watersheds.

In-river gillnetting by native American tribes has also been suggested in the decline of some salmonid stocks. While these fisheries are currently regulated to allow sufficient escapement of adults, regulations concerning the timing and gear restrictions of these fisheries may impact certain segments of salmon runs. For example, timing of the fishery may over-harvest an early or late segment of a run. On the Klamath River, regulations require large gillnet mesh sizes to prevent the harvest of steelhead. However, large mesh sizes target larger chinook salmon and may have contributed to the decline of older age classes of spawning adults.

A.2.1.4 Climatic Factors

Although extremely difficult to quantify, recent natural climatic events have most likely contributed to the decline of numerous stocks of anadromous salmonids along the Pacific northwest coast. A warming trend in the ocean along the Pacific northwest coast during 1976-1983 coincided with: 1) an abrupt drop in coho adult numbers in the Oregon Production Zone; 2) elevated sea-surface temperatures; and 3) reduced biological productivity in the California Current (Nickelson 1986; Lawson 1993). The 1982-1983 El Niño event, the largest ocean warming event of the century, severly impacted primary and secondary productivity thus impacting the entire northeast Pacific food-web (Pearcy 1992).

California is the southernmost range of coho salmon and these populations are well adapted to the extreme hydrologic, physical and climatic conditions (for salmonids) of coastal watersheds. However, the recent drought conditions of 1976-177 and 1986-1992 have made survival of the species in the southern part of its range even more demanding. Instream salmonid habitat conditions during the droughts were impaired by the sucessive years of low rainfall.

Conversely, past flood events have also impaired coho salmon habitat along the Pacific northwest coast. The recent floods of 1955 and 1964, in combination with intensive pre-Forest Practice Rules timber harvesting, severely degraded the quantity and quality of salmonid habitat in northern California watersheds. Salmonids in California have certainly experienced catastrophic natural events in the distant past, but these past salmonid populations were not simultaneously confronted with widespread, continuous human-related impacts to instream habitat.

A.2.2 Tailed Frog

Tailed frogs were considered rare for many years, but are now known to occur in high densities in suitable habitats (Nussbaum et al. 1983). Welsh (1990) expected numbers of frogs to decline due to timber harvest, to which they seem sensitive (Bury and Corn 1988b). He also speculated that the narrow niche, isolated population distribution, and long generation time of tailed frogs in conjunction with the lack of protection of headwater habitats make the species susceptible to local extirpations. Bury and Corn (1988a) predicted that populations subjected to clearcutting in the Coast Range of Oregon and northern California would probably go extinct following clearcutting, whereas those in the Cascades of Oregon and Washington had a higher probability of surviving. However, Bury (1968) noted that deforestation had a less detrimental effect on tailed frog populations where an influence of maritime climate was present. Studies in the coastal areas of northern California (Diller and Wallace, 1999) support the hypothesis that the impacts of timber harvest are less in coastal areas. Similar too what was noted above for the torrent salamander, tailed frogs were found in a high proportion of streams in previously logged areas. Geology was also the most important landscape-scale variable associated with occurrence of tailed frogs.

Bury and Corn (1988a) and Welsh (1990) believed that long-term, range-wide reductions or extinctions of tailed frogs were likely due to local extirpations, increased population fragmentation, habitat loss, restricted gene flow, and limited recolonization of streams when habitats are re-established (Bury and Corn 1988a).

Removal of timber by logging or fire is believed to result in the disappearance of tailed frogs due to increased stream temperatures (Noble and Putnam 1931, Nussbaum et al. 1983, Bury and Corn 1988a) and sedimentation (Nussbaum et al. 1983, Bury and Corn 1988a). The effects may affect the frogs directly, or indirectly by shifting the larval food base from diatoms to green algae (Bury and Corn 1988a). However, Bury (1968) stated "Presence of the frog in logged areas of coastal Humboldt County suggests that deforestation is less of a threat to the disappearance of *Ascaphus* in coastal than inland streams".

Although the survival of tailed frogs may depend on protection of cool flowing streams and adjacent forest habitats (Bury and Corn 1988b), timber harvest is not incompatible with such protection (Welsh 1990). Bury and Corn (1988a) and Welsh (1990) suggested eliminating harvest adjacent to aquatic habitats and establishing buffer strips to protect current frog populations and act as sources for future repopulation of logged areas. Bury and Corn (1988a) recommended establishing protection zones by retaining deciduous and small (cull) trees around streams while felling merchantable timber away from the streams. They noted that small clumps of trees around streams rather than cover along whole stream courses may be adequate to protect local populations (Bury and Corn 1988a). Retention of coarse woody debris for nutrient sources and sediment traps, further studies and surveys of tailed frogs, and protection of headwater habitats have also been recommended (Bury and Corn 1988a).

A.2.3 Southern Torrent Salamander

Welsh (1990) believed that logging and fragmentation of old growth coniferous forests would cause numbers of torrent salamanders to decline, with local extirpation of populations due to the species microhabitat requirements and lack of protection of headwater habitats. Bury and Corn (1988a) suggested that recolonization of logged areas would be rare and slow due to isolated population distribution, long generation time, narrow temperature requirements, and susceptibility to water loss limiting overland dispersal of the species (Welsh 1990). Recolonization may be more likely to occur in high gradient streams (Bury and Corn 1988a), but Welsh (1990) thought that local extirpations, increased population fragmentation and habitat loss, and restricted gene flow made populations vulnerable to long-term range-wide extinctions. The impacts of timber harvest on torrent salamanders appear to be less severe in coastal areas. Diller and Wallace (1996) found a high proportion of salamanders in streams that previously had been logged, including recently clearcut areas. In these coastal areas, geology was the only landscape-scale variable that strongly correlated with the occurrence of salamanders. In areas of a consolidated geologic type (e.g., Franciscan), torrent salamanders were found in high gradient reaches of almost all streams that were searched. It was hypothesized that the cool moist conditions of the coastal areas ameliorate the impacts of canopy removal for this species.

Short-term detrimental effects of logging on salamander habitat include increased sedimentation which fills crevices, and increased water temperatures (Bury and Corn 1988a). Bury and Corn (1988b) noted that these salamanders were sensitive to timber harvest and suggested that their survival was dependent on the protection of cool flowing streams and adjacent forested habitats which provide shade and maintain stream quality. Timber harvest plans should be designed and implemented to provide such protection (Welsh 1990). Bury and Corn (1988a) recommended protecting streams by felling merchantable timber away from streams and leaving deciduous and small (cull) trees to provide shade cover. To reduce the expense of leaving merchantable timber along whole stream courses, small clumps of trees may be retained to protect current populations and provide sources for future repopulation of logged areas (Bury and Corn 1988a). Retaining coarse woody debris, conducting preharvest surveys, and obtaining more data on the species' habitat preferences and environmental tolerance have also been recommended (Bury and Corn 1988a).

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Appendix B Evaluation of the Impact of Timber Harvest on Future Potential Recruitment of Large Woody Debris in Class I Watercourses

Evaluation of the Impact of Timber Harvest on Future Potential Recruitment of Large Woody Debris in Class I Watercourses

Green Diamond Resource Company

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Riparian management zones (RMZs) provide several important biological and watershed functions. In addition to functions such as maintaining the riparian microclimate and providing nutrient inputs, one of the most important functions of the RMZs is to provide for the recruitment of large woody debris (LWD) to the watercourse. LWD is recognized as a vital component of salmonid habitat. The physical processes associated with LWD include sediment sorting and storage, retention of organic debris, and modification of water quality (Bisson et al. 1987). The biological functions associated with LWD structures for the salmonid species include important rearing habitats, protective cover from predators and elevated stream flow, retention of gravels for salmonid redds, and regulation of organic material for the in-stream community of aquatic invertebrates (Murphy et al. 1986; Bisson et al. 1987). Decreased supply of LWD can result in increased vulnerability to predators, reduction in winter survival, reduction in carrying capacity, reduced spawning habitat availability, reduction in food productivity and loss of species diversity (Hicks et. al. 1991 as cited by Spence et. al. 1996). Long-term reductions in LWD can result in less stream complexity and reduce the amount of high quality rearing habitat for salmonids and other fish species.

The minimum width of RMZs on Class I (fish bearing) watercourses is 150 feet with 85% overstory canopy retention in the inner zone (50-70 feet depending on slope class) and 70% overstory retention in the remaining outer zone. However, probably the most important measure relative to the potential recruitment of LWD is that no trees will be harvested that are judged likely to recruit to the watercourse. There are a variety of criteria that will be used to make this judgment including, but not restricted to, distance from the stream, direction of the lean, a clear fall path to the channel, and potential for stream undercutting. However, some of these criteria are inherently subjective and concerns have been raised that the “likely to recruit” language in Green Diamond’s draft Aquatic Habitat Conservation Plan/Candidate Conservation Agreement with Assurances (AHCP/CCAA) is not sufficient to insure that there will be no loss of important future LWD. Numerous attempts were made to improve the likely to recruit language, but none were entirely successful. As a result, the Services (NMFS and the Fish and Wildlife Service) and Green Diamond agreed to gather empirical data from Watercourse and Lake Protection Zones (WLPZs) in Class I watercourses to assess the extent to which current guidelines were successful in maintaining future potential LWD. The objective of this study was to gather data from WLPZs that have been marked, but not yet harvested, and from those that already have been harvested, following Green Diamond’s internal guidelines relative to retaining trees that are likely to recruit.

To permit quantification of future potential LWD, we made several assumptions concerning recruitment and quantified trees in terms of “Full Tree Equivalents” (FTE). One FTE is defined as a tree with a probability of 1.0 that it would some day fall into the stream and eventually become a “fully functional” piece of LWD. Fully functional LWD interacts with the hydrology of the stream in such a way that it provides for all the benefits described above. To calculate FTE’s, we developed a tree recruitment potential model based on tree height and the distance of each tree from the channel. The model assumes the stream is a straight line and each tree has an equal probability of falling in any direction. The FTE was calculated as the proportion of an area of a circle that extends beyond the closest watercourse transition line (WTL). The circle was circumscribed by the falling radius of the tree. For example, a 150-foot tall tree located 100 feet from the WTL has the potential to fall into the channel with a maximum of 50 feet of the tree being recruited. The FTE value of this tree would be 0.110 meaning that 11.0% of the area of the circle represented by the falling radius of that tree could extend beyond the WTL and into the channel. This calculation gives a greater weighting factor to trees that would provide greater functionality to the stream in terms of having a greater proportion of the tree potentially interacting with the fluvial processes of the stream. A tree that is farther from the WTL than it’s height received an FTE value of 0.0. A tree located within the WTL (growing within the active channel) received a FTE value of 1.0. These trees were considered recruited and 100% functional regardless of the falling direction. We also assumed that 10” DBH was the minimum size tree that would be functional in most Class I watercourses. Quantifying of the impact of timber harvest on the potential recruitment of LWD was based on the summation of FTE’s before and after harvest of trees greater than or equal to 10” DBH.

The initial analysis was based on the current height of trees in the WLPZ, recognizing that most trees will continue to grow and will not recruit (blow down, be recruited by fluvial or geological processes or die and fall into the watercourse) for many years into the future. Green Diamond recalculated potential impacts from tree harvest within the RMZ after adding 50 years of average growth to the trees in the WLPZ. This provided a view on recruitment potential of trees within the WLPZ retained on site for the life of the permit. The difference in impacts from harvesting on FTE’s at current rotation age versus impacts at rotation plus 50 years could then be evaluated.

Field Methods

Five Class I WLPZs were inventoried for LWD recruitment potential. Two of these WLPZs were located in Maple Creek (T8&9N, R1E HBM) and three in Ryan Creek (T4N, R1E HBM). The two WLPZs in Maple Creek were each from separate THPs that were harvested and logged during the summer of 2003 (Attachment A, Figures A1 and A2). The three WLPZs in Ryan Creek were located within a single Timber Harvest Plan (THP) unit that had been laid out and marked, but had not been harvested (Figure A3). All the WLPZs were administered under the Threatened and Impaired Watershed package of the California Forest Practice Rules and therefore are nominally 150 feet wide.

The inventory crews worked in groups of four. One person with a hip chain walked the stream channel along the edge of the riparian zone. This person took notes, kept track of channel distance for each conifer, measured the channel gradient (every 300') and kept the rest of the crew in a perpendicular line with the stream as they measured the conifers and snags in the WLZP. The upslope crewmembers measured the DBH, the distance of the tree from channel (Y coordinate), distance up the channel (X coordinate), hillslope gradient, and noted the species of any conifer tree that was 10" DBH or larger. DBH was measured with a Biltmore stick to the nearest inch at 4.5' on the uphill side of any standing tree. Each standing conifer was evaluated for an obvious lean of greater than or equal to 5 degrees from vertical. If a tree had an obvious lean, the angle of lean and the direction of lean were measured in relation to the stream channel. A tree that was leaning perpendicular towards the channel was given a direction of lean of 90 degrees. Therefore 0 to 179 degrees was assigned to trees with a downslope direction of lean and 180 to 359 degrees to trees with an upslope direction of lean. The diameter, height, species and decay class of all snags greater than 10" DBH were noted. In the unharvested WLPZs, each tree that was marked for harvest was noted as a "stump". A marked tree typically has a blue painted stripe and a basal mark. It was assumed that all trees that were marked will be harvested when the THP unit is operated. In the harvested WLPZs, the species, diameter and location (X and Y coordinates in relation to the channel) of stumps of the recently harvested trees were noted.

Within each sampling location, a representative sample of conifer trees of each species (grouped by redwood and other conifer) were measured for tree height in addition to DBH. Trees selected for height measurement were representative dominant and co-dominant trees of the WLPZ. The actual selection depended on the ability to see both top and bottom of the tree at a reasonable distance from the tree (e.g. within the % range of the clinometer). These sampled conifers were used to estimate the heights of the trees in the WLPZs.

Analysis

In order to calculate the FTE for each tree, the height of each tree was needed. The exponential form of the height-diameter model from Krumland and Wensel (1978) was used to estimate tree heights in the various WLPZs. The trees that were selected for height measurement were used in the model to develop individual height-diameter relationships for each WLPZ, except the data from Ryan Creek were pooled since the three WLPZs were in close proximity to each other. The FTE of each tree was then calculated and summed for the pre-harvest condition. The FTE of harvested trees (stumps) in the Maple Creek WLPZs were estimated from the diameters of the stumps. The post harvest condition was determined by setting the FTE value for each marked (Ryan Creek WLPZs) or harvested tree (Maple Creek WLPZs) to a value of zero. The difference between the summed pre-harvest FTE values and summed post-harvest FTE values was expressed as a percent post-harvest reduction in cumulative FTE for each WLPZ.

In order to evaluate the potential impact of harvest over the term of the permit, we assumed that all the WLPZs were in the 50 year age class and then grew the trees an additional 50 years. Based on the average site index for Green Diamond's property, we would expect redwood and Douglas fir in the 50-year age class to grow approximately 50 feet taller in 50 years. Conifer trees that were less than 10" DBH at the 50-year age class were not added to the analysis of the 100-year age class.

Additional information was measured and summarized for each of the WLPZs which could be used to adjust the FTE value of individual trees, numerically. This information can be used to refine the probability of individual trees being recruited to the stream channel based on the side slope gradient and the amount and direction of lean of individual trees. Each standing conifer was evaluated for an obvious lean and if present the angle of lean and the direction of lean were measured in relation to the stream channel. The channel and side slopes were also measured, in percents, and summarized for each WLPZ. The channel slope was measured approximately once every 300 feet of channel or at any obvious changes. A weighted average was then calculated for the entire channel. The bank slope measurements were treated similarly and presented as a range of slope values for the WLPZ. The analysis presented here assumed all the trees were vertical and had an equal probability of falling in any direction. No FTE values were modified to account for the amount or direction of lean or the slope gradient. The information was collected and presented for discussion purposes.

<u>Results and Discussion</u>

The cumulative FTE reduction is the total affect that timber harvest had (or will have once harvested), on the recruitment potential of conifers to the watercourse. Figures 1-5 are graphical representations of each measured live conifer, stump, and snag in relation to the WTL. A red circle with a radius equal to the corresponding tree height is drawn around each tree that was harvested (or will be harvested). Each stump's FTE is represented by the proportion of the circle that extends beyond the WTL. When a circle does not extend beyond the WTL, the pre-harvest FTE values equal zero. The reduction in FTE values for all WLPZs post-harvest ranged from 0.0 to 0.62% (Table 1). Fifty years from now, all the conifer trees within these WLPZs were assumed to grow on average 50 feet taller. If the same trees were marked within these WLPZs, but were harvested 50 years from now, the reduction in FTE values post-harvest would range from 0.29 to 1.58%. A summary of the pre- and post-harvest stand component within each WLPZ is presented in Attachment B.

In the three Ryan Creek WLPZs, we assumed that each tree that was marked for harvest will be cut when the THP unit is operated. We observed cases in the two Maple Creek WLPZs (which were harvested) where several trees where originally marked for harvest, but not actually cut. In a few instances an adjacent unmarked tree was traded for the marked tree. It is likely the timber fallers determined that cutting the marked tree would be unsafe or infeasible to fall. The marked trees may have been limb-locked or located behind another tree, an old growth stump or a snag. In some cases the faller would make a trade and sometimes decide not to cut anything from that particular area.

In the North Fork Maple WLPZ, 5 of 251 conifer trees were harvested (98.0% conifer retention). This equates to approximately 1 tree harvested for every 260 feet of WLPZ length. Of the 5 trees harvested, none had a FTE value greater than zero. The harvest of the 5 trees did not change the recruitment potential of the WLPZ (Table 1). If the harvest was delayed 50 years, 4 of the 5 trees harvested would have a FTE value greater than zero. This would result in a 0.29% reduction in the recruitment potential of conifers in the WLPZ (Table 1).

In the CR1500 WLPZ, 88 of 1115 conifer trees were harvested (92.1% conifer retention). This equates to approximately 1 tree harvested for every 25 feet of WLPZ length. Of the 88 trees harvested, 14 had a pre-harvest FTE value greater than zero. After harvest, the removal of the 14 trees resulted in a 0.62% reduction in the recruitment potential of conifers in the WLPZ (Table 1). If the harvest was delayed 50 years, 44of the 88 trees harvested would have a FTE value greater than zero. This would result in a 1.58% reduction in the recruitment potential of conifers in the WLPZ (Table 1).

In Ryan Creek Tributary #1, 8 of 296 conifer trees were marked for harvest (97.3% conifer retention). This equates to approximately 1 tree harvested for every 135 feet of WLPZ length. Of the 8 trees harvested, 7 had a pre-harvest FTE value greater than zero. After harvest, the removal of the 7 trees resulted in a 0.48% reduction in the recruitment potential of conifers in the WLPZ (Table 1). If the harvest was delayed 50 years, all of the trees harvested would have a FTE value greater than zero. This would result in a 1.20% reduction in the recruitment potential of conifers in the WLPZ (Table 1).

In Ryan Creek Tributary #2, 10 of 420 conifer trees were marked for harvest (97.6% conifer retention). This equates to approximately 1 tree harvested for 120 feet of WLPZ length. Of the 10 trees, 7 had a pre-harvest FTE value greater than zero. After harvest, the removal of the 7 trees resulted in a 0.23% reduction in the recruitment potential of conifers in the WLPZ (Table 1). If the harvest was delayed 50 years, all of the trees harvested would have a FTE value greater than zero. This would result in a 0.80% reduction in the recruitment potential of conifers in the WLPZ (Table 1).

An inexperienced crewmember, who was unfamiliar with the use of a Biltmore stick, created a minor bias in the calculation of total FTE for Ryan Creek tributary #2. The incorrect use of the Biltmore stick resulted in a positive bias of DBH on larger diameter trees and therefore an overestimation of tree height. This crew member only worked one of the three days it took to survey this WLPZ, and due to where he worked (within the first 50 feet from the channel and from a channel distance of 551 feet to 938 feet), the potential error can be evaluated as to its affect on the survey. There were no trees harvested from this area of the WLPZ. As a result the post-harvest FTE values were not reduced from activity in this part of the WLPZ. The pre- and post- harvest FTE calculations will be off by an identical amount resulting in a slightly higher cumulative FTE. Therefore any reduction in FTE due to harvest would have a slightly lower influence in the reduction in the overall recruitment potential of conifers.

In Ryan Creek Tributary #3, 10 of 521 conifers were marked for harvest (98.1% conifer retention). This equates to approximately 1 tree harvested for every 60 feet of WLPZ length. Of the 10 trees harvested, 7 had a pre-harvest FTE value greater than zero. After harvest, the removal of the 7 trees resulted in a 0.19% reduction in the recruitment potential of conifers in the WLPZ (Table 1). If the harvest was delayed 50 years, all of the trees harvested would have a FTE value greater than zero. This would result in a 0.63% reduction in the recruitment potential of conifers in the WLPZ (Table 1).

The pre- versus post-harvest difference in FTE indicated that timber harvest was having a very minor impact (maximum of <1%) on the cumulative total of future potential LWD recruitment. However, even more important is that the reduction comes from future LWD that has the lowest probability of becoming functional LWD. This is further supported by the analysis where the impact was evaluated over the life of the Plan. Fifty years from now, the pre- versus post-harvest difference in FTE would result in a maximum of <2% reduction of future potential LWD recruitment. Given this outcome, Green Diamond believes that its current internal guideline of not harvesting trees in Class I WLPZs that are likely to recruit is successful at maintaining a high level of future potential LWD recruitment.

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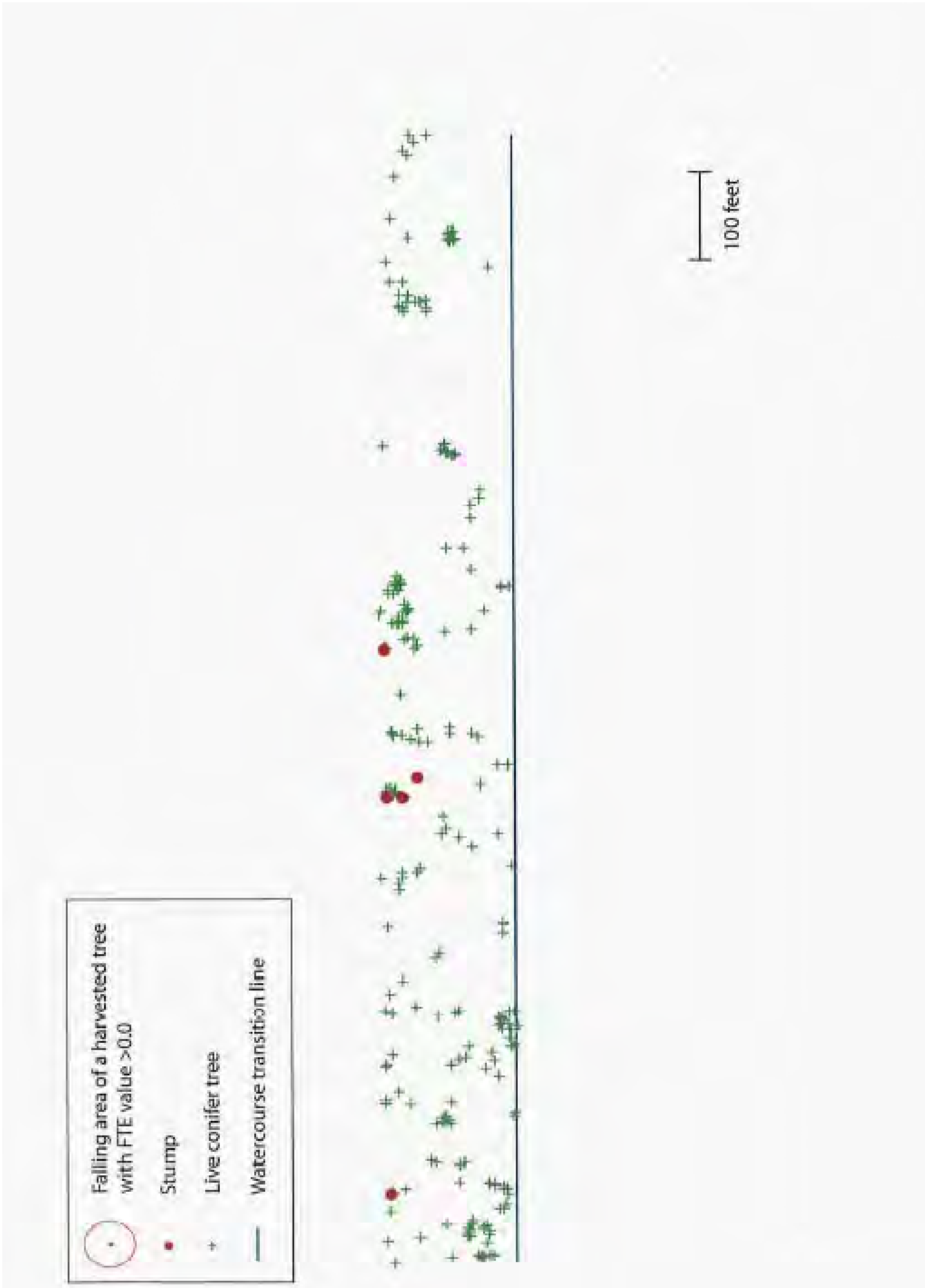

Figure 1. NF Maple Creek Class I WLPZ.

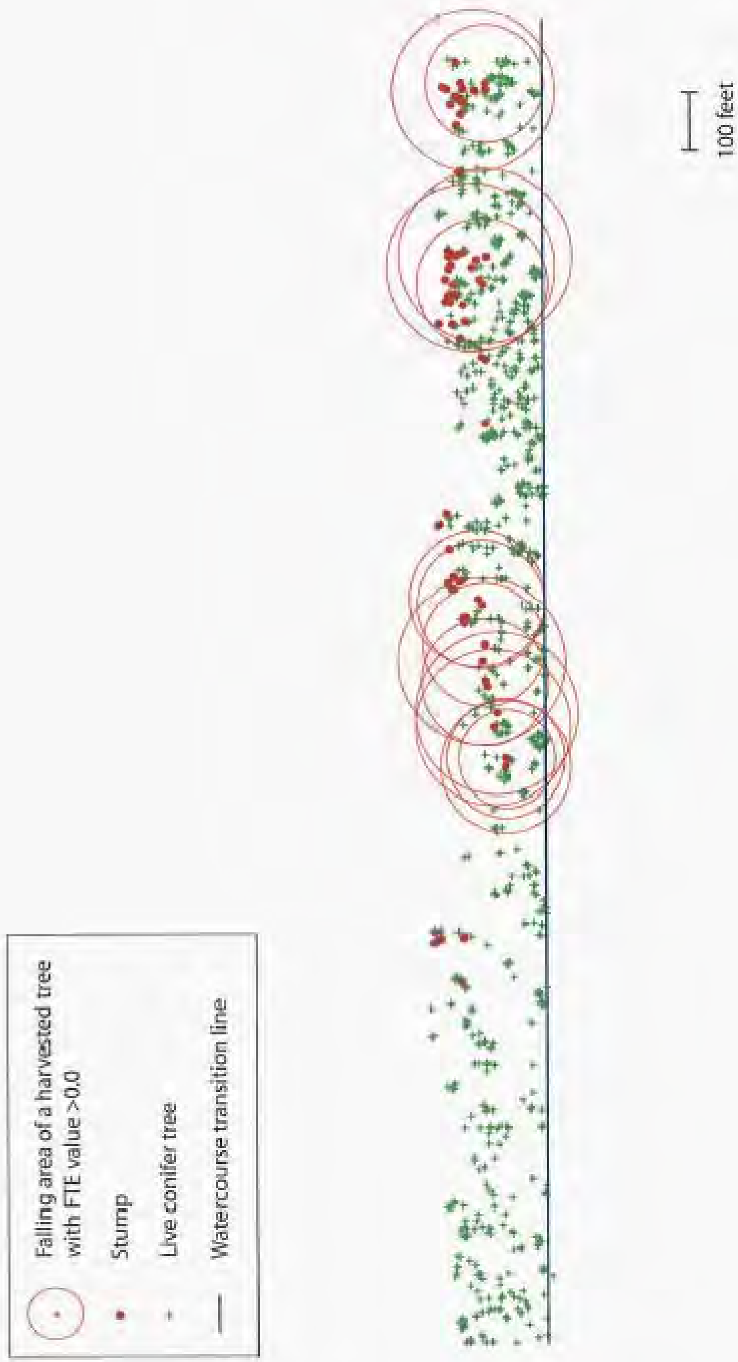

Figure 2. CR1500 Class I WLPZ.

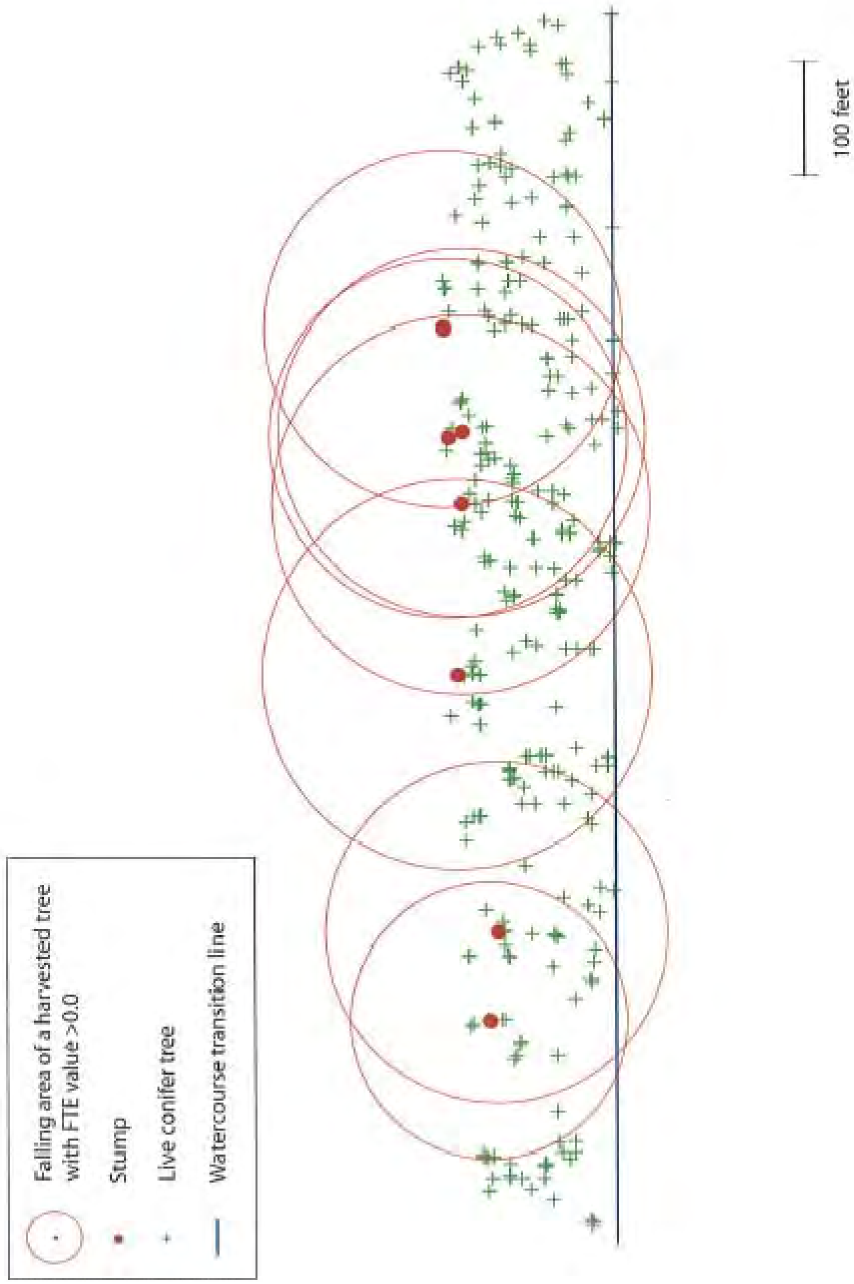

Figure 3. Ryan Creek #1 Class I WLPZ.

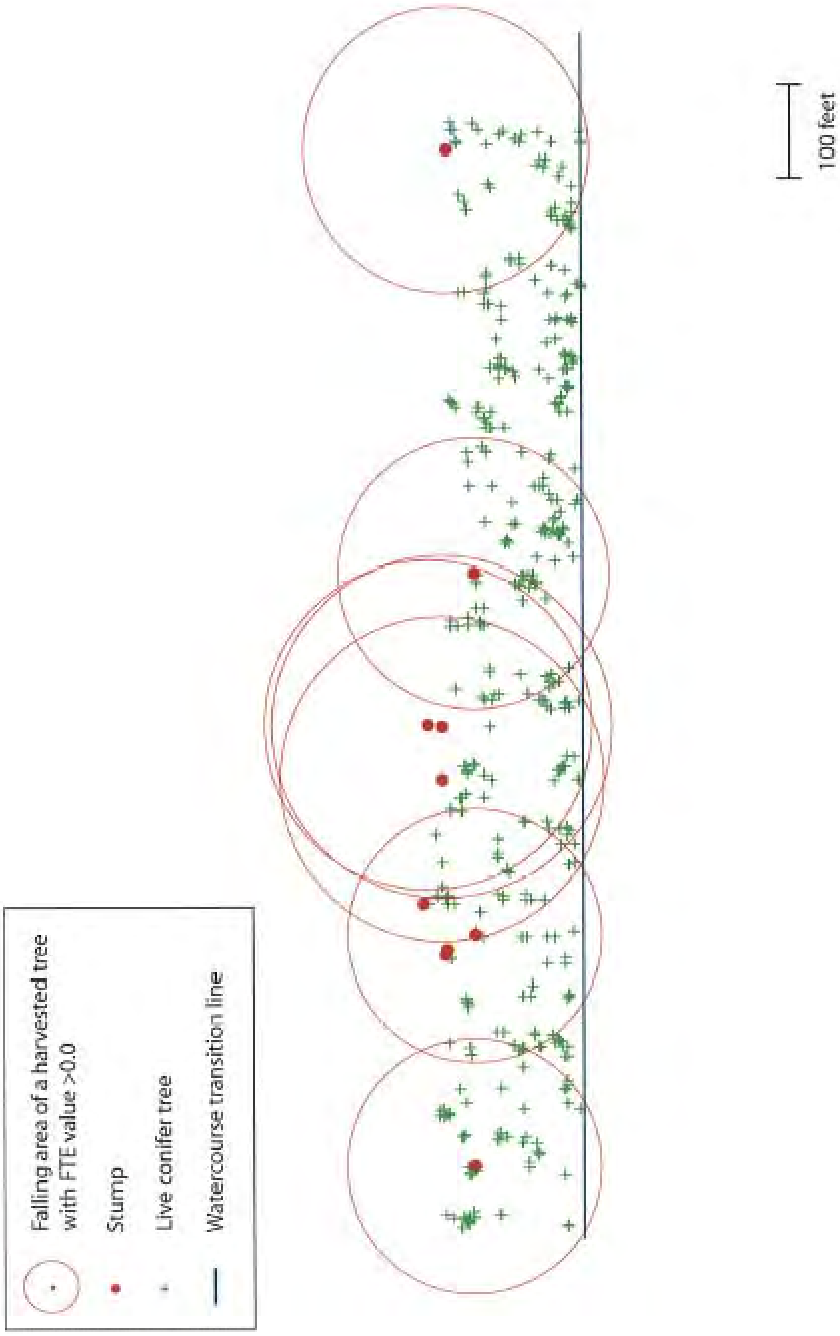

Figure 4. Ryan Creek #2 Class I WLPZ.

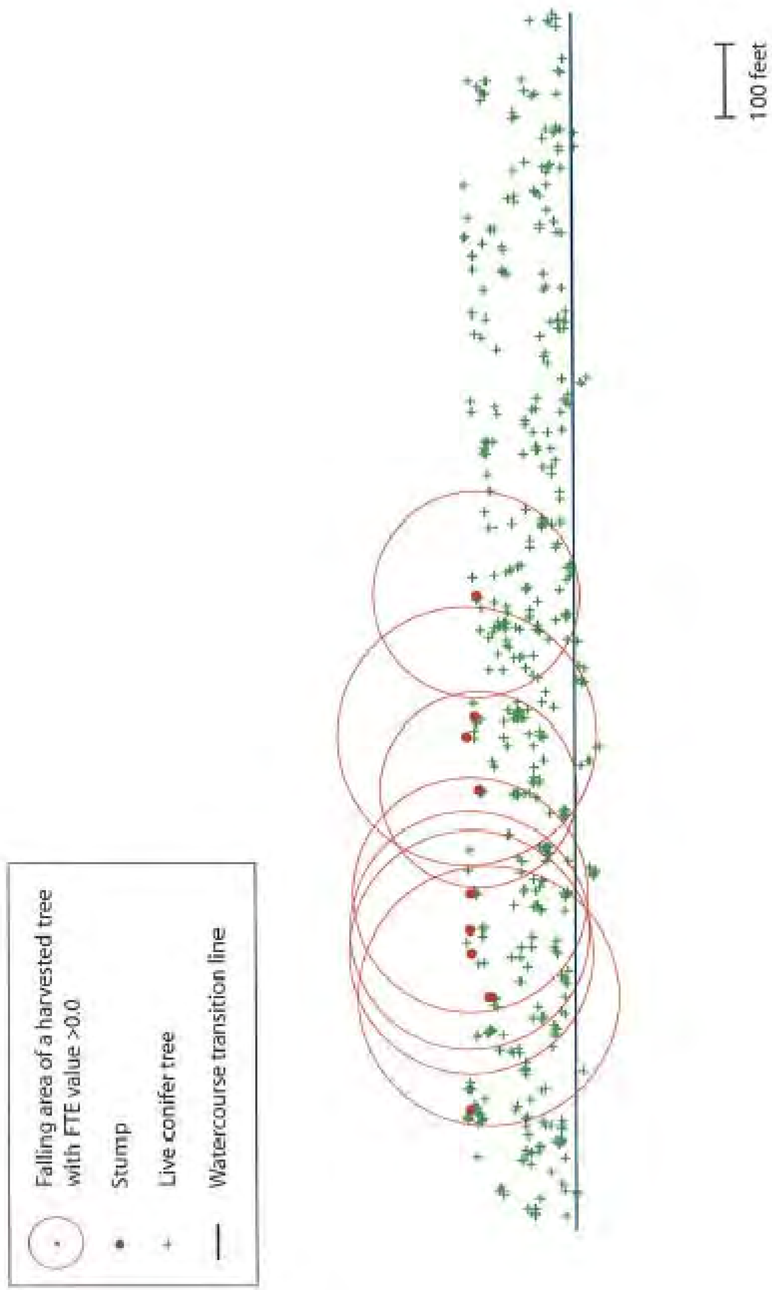

Figure 5. Ryan Creek #3 Class I WLPZ.

Table 1. Full Tree Equivalents (FTE) and associated parameters.

	Ryan #1	**Ryan #2**	**Ryan #3**	**CR1500**	**NF Maple**
Zone survey length (feet)	1086	1203	1689	2183	1299
Total # of live and recently harvested conifers in zone	296	420	521	1115	251
Total # of live trees marked or recently harvested	8	10	10	88	5
Percent conifer retention	97.3	97.6	98.1	92.1	98.0
Current Full tree equivalents (FTE)					
Pre-harvest	56.65	88.36	124.19	134.51	28.30
Post-harvest	56.37	88.16	123.95	133.68	28.30
Percent reduction	0.48	0.23	0.19	0.62	0.00
# of harvested trees with a FTE value >0.0 (current)	7	7	7	14	0
Predicted Full tree equivalents (+ 50 years)					
Pre-harvest	75.86	111.61	153.57	204.38	41.99
Post-harvest	74.95	110.72	152.59	201.15	41.87
Percent reduction	1.20	0.80	0.63	1.58	0.29
# of harvested trees with a FTE value >0.0 (+ 50 years)	8	10	10	44	4

Attachment A

Class I WLPZ survey areas

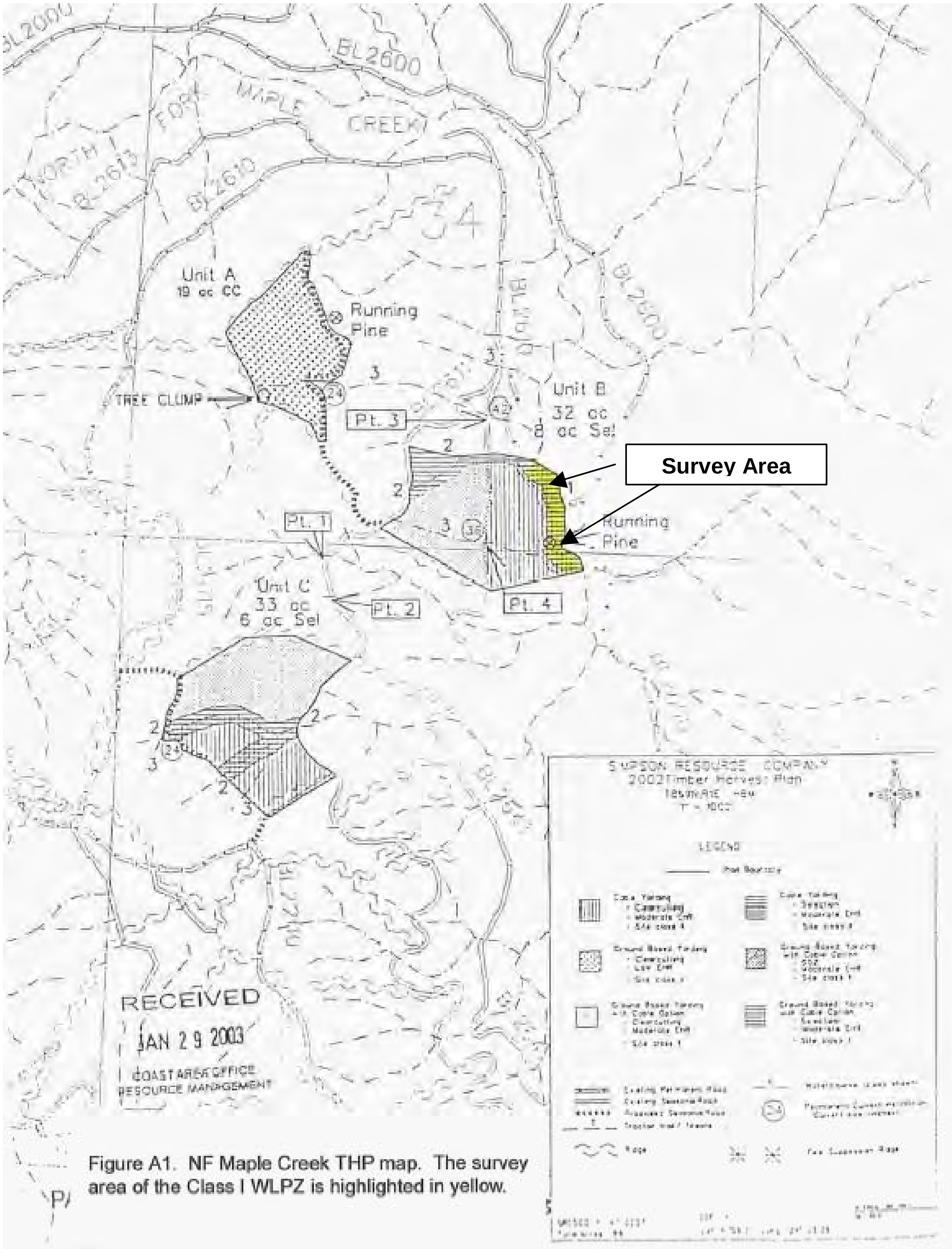

Figure A1. NF Maple Creek THP map. The survey area of the Class I WLPZ is highlighted in yellow.

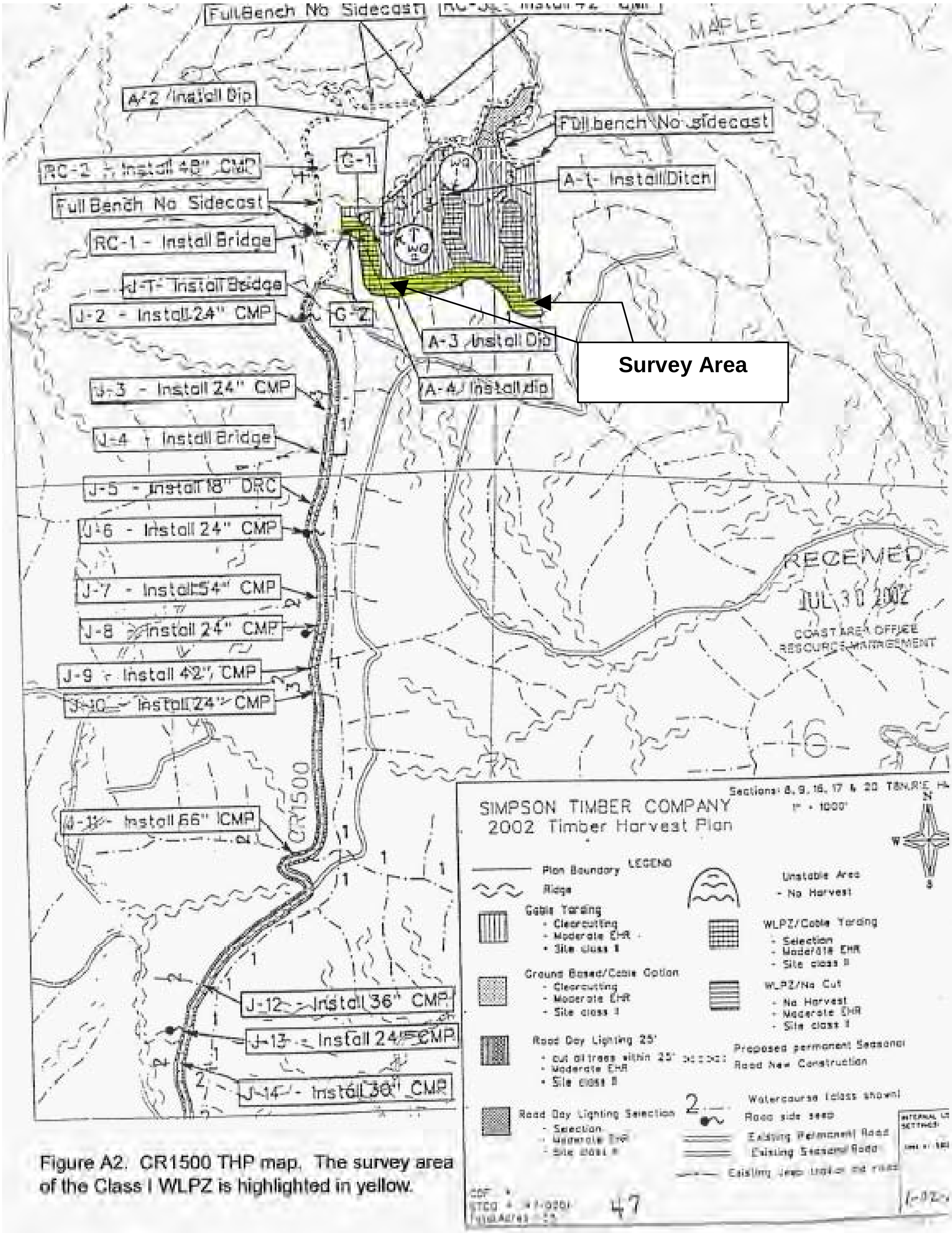

Figure A2. CR1500 THP map. The survey area of the Class I WLPZ is highlighted in yellow.

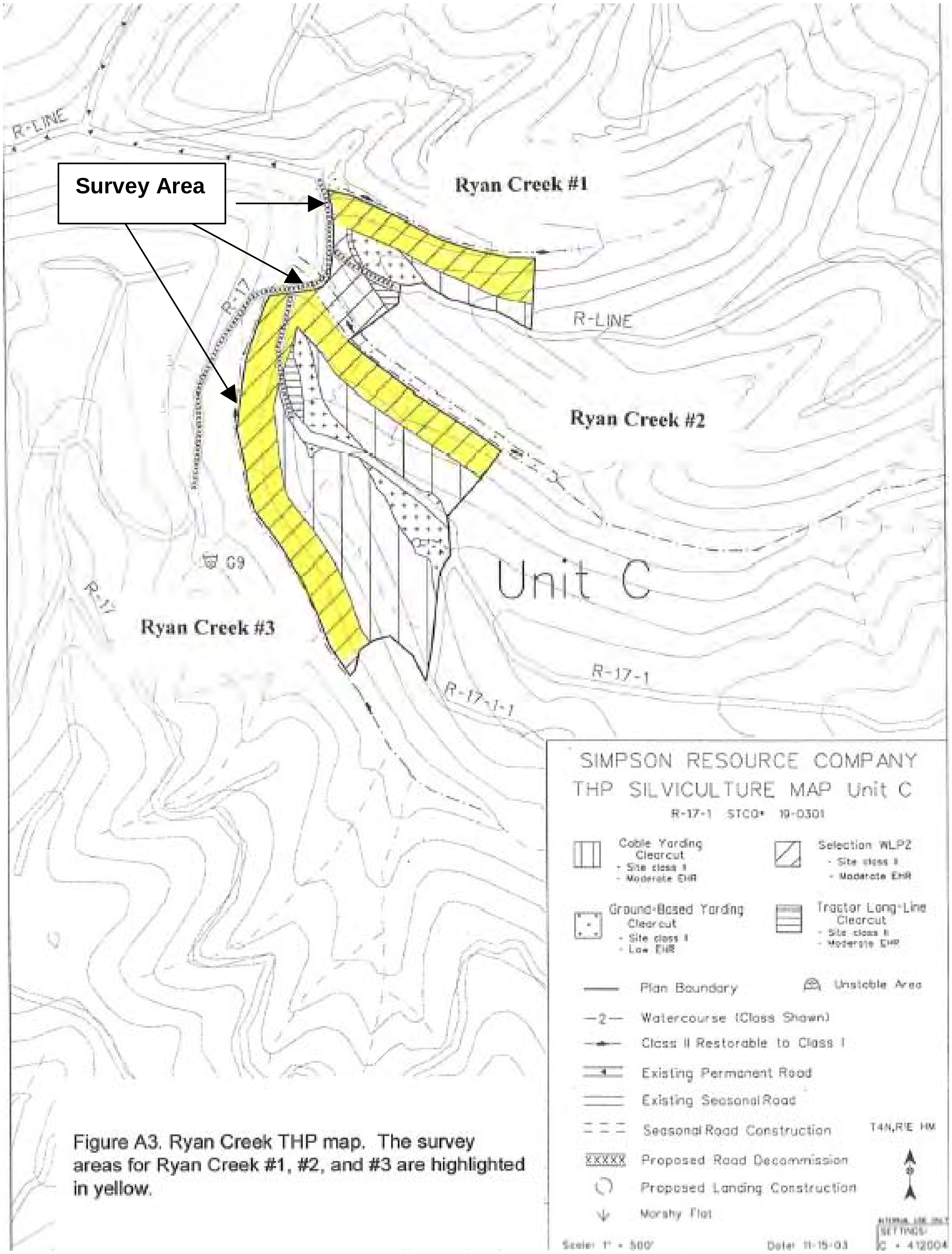

Figure A3. Ryan Creek THP map. The survey areas for Ryan Creek #1, #2, and #3 are highlighted in yellow.

Attachment B

Summary data

Table B1. Full Tree Equivalents (FTE) and associated summary information for each WLPZ.

	Ryan #1	**Ryan #2**	**Ryan #3**	**CR1500**	**NF Maple**
Zone survey length (feet)	1086	1203	1689	2183	1299
Total # of live and recently harvested conifers in zone	296	420	521	1115	251
# of redwood	168	342	426	982	184
# of Douglas fir	126	76	95	129	23
# of other conifer	2	2	0	4	44
Total # of live trees marked or recently harvested	8	10	10	88	5
# of redwood	2	7	9	81	5
# of Douglas fir	6	3	1	7	0
# of other conifer	0	0	0	0	0
Percent conifer retention	97.30	97.62	98.08	92.11	98.01
Current Full tree equivalents (FTE)					
Pre-harvest	56.65	88.36	124.19	134.51	28.30
Post-harvest	56.37	88.16	123.95	133.68	28.30
Percent reduction	0.48	0.23	0.19	0.62	0.00
# of harvested trees with a FTE value >0.0 (current)	7	7	7	14	0
Predicted Full tree equivalents (+ 50 years)					
Pre-harvest	75.86	111.61	153.57	204.38	41.99
Post-harvest	74.95	110.72	152.59	201.15	41.87
Percent reduction	1.20	0.80	0.63	1.58	0.29
# of harvested trees with a FTE value >0.0 (+ 50 years)	8	10	10	44	4
# of trees with obvious lean ($\geq 5^0$)	53	100	171	166	55
Range of lean from vertical (degrees)	5 - 60	5 - 50	5 - 55	5 - 60	5 - 46
# of trees with downslope lean ($0\text{-}179^0$)	39	69	122	112	26
# of trees with upslope lean ($180\text{-}359^0$)	14	31	49	54	29
Channel gradient (%)	2.2	2	2	2	3
Slope gradient range (%)	35 - 76	0 - 82	0 - 62	5 - 100	3 - 18

Table B2. Diameter and height summary of conifers and snags for each WLPZ.

	Ryan #1	Ryan #2	Ryan #3	CR1500	NF Maple
Average diameter of WLPZ conifers (inches)	24.1	29.1	26.7	20.8	21.7
Redwood	23.1	27.7	25.8	21.0	23.0
Douglas fir	25.5	35.5	31.1	19.3	18.9
Other conifer	12.0	36.0	none	20.0	17.6
Average diameter harvested conifers (inches)	22.8	31.1	33.2	23.2	31.8
Redwood	28.0	29.3	33.0	22.9	31.8
Douglas fir	21.0	35.3	35.0	26.6	none
Other conifer	none	none	none	none	none
Live conifer diameter range (inches)	10 - 56	10 - 100	10 - 78	10 - 80	10 - 60
Redwood	10 - 56	10 - 100	10 - 78	10 - 80	10 - 52
Douglas fir	10 - 48	10 - 100	10 - 64	10 - 50	10 - 36
Other conifer	12	23 - 49	none	12 - 30	10 - 60
Harvested conifer diameter range (inches)	11 - 34	22 - 41	18 - 52	8 - 47	25 - 39
Redwood	22 - 34	22 - 36	18 - 52	8 - 47	25 - 39
Douglas fir	11 - 28	32 - 41	35	24 - 31	none
Other conifer	none	none	none	none	none
Average height of WLPZ conifer (feet)	139	138.5	135.0	92.7	87
Redwood	119.9	130.7	126.7	88	84.2
Douglas fir	161.7	174.1	172.0	127.3	95.7
Other conifer	126.6	177.3	none	129.1	94.2
Live Conifer height range (feet)	75.5 - 189.9	75.5 - 226.7	75.5 - 209.4	58.1 - 164.3	53.4 - 131.2
Redwood	75.5 - 186.1	75.5 - 226.7	75.5 - 209.4	58.1 - 164.3	53.4 - 126.2
Douglas fir	100.7 - 189.9	126.7 - 209.2	126.7 - 198.2	104.1 - 162.8	80.3 - 131
Other conifer	126.6 - 135.5	164 - 190.5	none	111.7 - 146.4	80.3 - 131.2
# of snags	21	33	36	29	6
Redwood	7	16	26	22	6
Douglas fir	14	9	10	7	0
Other conifer	0	0	0	0	0

Appendix C. Studies, Surveys, and Assessments of Covered Species and their Habitats Conducted in the Current Plan Area

CONTENTS

INTRODUCTION

The following are summaries of methods, results and conclusions of numerous investigations Green Diamond has undertaken on Plan Area properties since at least 1994. These are organized into physical habitat assessments, fish population studies, amphibian surveys, and an analysis and projection of future habitat conditions. Many of these projects have evolved from narrowly focused studies initially employed to answer a single question or monitor relatively few parameters into a comprehensive program across a wide geographic and temporal landscape. The results of these investigations, along with continuing scientific progress in assessing habitat and populations of species inhabiting Green Diamond's properties have driven the evolution of the methodologies described herein. As they have evolved, many of the monitoring investigations described in this appendix have become the basis for many of the protocols presented and described in the Appendix D of this Plan.

Appendix C1. Channel and Habitat Typing Assessment

CONTENTS

Figures

Tables

C1.1 METHODS

Initial channel and habitat typing assessments were conducted by Green Diamond Fisheries personnel in 1994 and 1995 following the CDFG methods described by Flosi and Reynolds (1994). Prior to the onset of assessments, Green Diamond's fisheries field technicians participated in a four-day training seminar sponsored by CDFG in order to become familiar with the methodology. In the 1995 season, Green Diamond field personnel followed the 10% sampling scheme modification proposed by CDFG to reduce the time required for this assessment (Hopelain 1995). All field data was entered into the Habitat Program (Flosi and Reynolds 1994) and resulting data tabulated, summarized, and discussed below.

During those two years Green Diamond fisheries personnel assessed sixteen streams on Green Diamond's ownership in the HPAs, identifying 75 reaches by channel type for a total of over 94 miles of stream channel examined (Table C1-1). The sixteen streams assessed were selected based on their biological significance as producers of salmonids, and the size of Green Diamond's ownership in the watershed's anadromous reaches.

Additionally, channel and habitat typing assessments of streams on Green Diamond's ownership in the HPAs also were conducted by the Yurok Tribal Fisheries Program (YTFP) (31 streams during VN1996-1998), the California Conservation Corp (CCC) (3 streams in 1995), the Louisiana Pacific Corp. (4 streams in 1994), and the California Department of Fish and Game (CDFG) (4 streams in 1991 and 1998). Assessments by those entities were conducted on 42 streams covering more than 149 reaches for a total of over 135 miles of channel (Table C1-1).

For the purposes of summarizing and comparing stream channel and habitat parameters several of the channel and habitat typing variables (canopy closure, % conifer canopy, % LWD as structural shelter, and % of stream length in pool) were plotted against stream watershed area. These variables were mean values for the entire length of stream that was surveyed. For comparison purposes to other surveyed streams within each HPA the watershed area was determined at the midpoint of the surveyed reach of stream. The dry sections of channel in the lower portion of the watershed were not included in the overall stream length. The mid point of the wetted channel length normalizes the stream size based on the relative position in the watershed where the survey occurred and the mean values of interest. The least squares regression displayed on these figures was added for comparison purposes only and not intended for statistical analysis. These data were not transformed to find the best fit but just to get a general sense of how conditions in certain HPAs compare with those other HPAs. The R^2 and p-values are also shown on the figures.

To allow the comparison of pool tail-out embeddedness between assessed streams, a stream gradient was determined from the channel types. Each channel type has a delineation criteria based on a range of channel gradients. To derive an average stream gradient, the mean gradient of each channel type criteria was weighted according to the length of each channel type.

Table C1-1. Summary of the channel and habitat typing assessments conducted during 1991-1998 on Green Diamond's ownership in the HPAs.

HPA	Surveyed By:										Totals	
	Green Diamond		Yurok Tribal Fisheries Program		Louisiana-Pacific		CCC(1)		CDFG(2)			
	No. streams	Miles	No. streams	Miles	No. streams	Miles	No. streams	Miles	No. streams	Miles	No. streams	Miles
Smith River	4	22.99	x	x	x	x	X	x	x	x	4	22.99
Coastal Klamath	6	35.35	16	52.46	x	x	X	x	x	x	22	87.81
Blue Creek	x	X	4	21.63	x	x	X	x	x	x	4	21.63
Interior Klamath	x	X	11	30.23	x	x	X	x	x	x	11	30.23
Redwood Creek	x	X	x	x	x	x	X	x	x	x	0	0
Coastal Lagoons	x	X	x	x	x	x	X	x	x	x	0	0
Little River	x	X	x	x	4	18.02	X	x	x	x	4	18.02
Mad River	3	11.29	x	x	x	x	X	x	x	x	3	11.29
NF Mad River	2	18.03	x	x	x	x	X	x	x	x	2	18.03
Humboldt Bay	1	7.04	x	X	x	x	3	7.04	x	x	4	14.08
Eel River	x	X	x	X	x	x	X	x	4	5.84	4	5.84
TOTALS	16	94.70	31	104.32	4	18.02	3	7.04	4	5.84	58	229.92

(1)California Conservation Corps
(2)California Department of Fish and Game

C1.2 RESULTS

Results of the channel and habitat typing assessments for the 58 streams are summarized in Tables C1-2 through C1-8. These results are discussed in more detail in the following discussion and conclusions section below.

C1.3 DISCUSSION

The following discussion is based on the results of the channel and habitat typing assessments presented in Tables C1-2 through C1-8.

Table C1-2. Stream assessment summaries for four Plan Area streams in the Smith River HPA.

Parameters	Streams			
	SF Winchuck River	**Dominie**	**Wilson**	**Rowdy**
Year Assessed	1995	1995	1994	1995
Assessed by	Green Diamond	Green Diamond	Green Diamond	Green Diamond
Total Length of Channel Assessed (feet)	31,961	17,118	35,640	36,668
Mean % Canopy Density	92	94	79	63
% deciduous	98	93	94	97
% conifer	2	7	6	3
% LWD as Structural Shelter in All Pools	6.4	18.2	21.8	5.6
Habitat Types as % of Total Length				
Riffles	41	51	25	24
Flat-water	32	29	41	42
Pools	27	20	28	33
Dry Channel	0	0	7	1
Pool Tailout Embeddedness as % Occurrence				
0-25%	27.3	0.5	37.0	32.5
26-50%	37.2	31.3	35.5	41.0
51-75%	19.1	21.5	28.0	17.5
76-100%	16.4	46.8	0.0	6.3
Maximum Pool Depths as % Occurrence				
<1' deep	0.6	0.9	0.0	20.4
1'-2' deep	4.3	53.7	5.9	2.0
2'-3' deep	40.2	41.7	39.1	7.1
3'-4' deep	39.6	3.7	27.2	33.7
>4' deep	15.2	0.0	27.8	36.7
Index of Embeddedness	3.5	3.1	3.3	2.6
Mid-point Gradient (%)	2.1	4.2	1.1	2.4
Mid-point Watershed Area (acres)	4,336	1,356	5,092	10,990

Table C1-3. Stream assessment summaries for 22 Plan Area streams the Coastal Klamath HPA.

Parameters	Streams							
	Hunter	**EF Hunter**	**High Prairie**	**Mynot**	**HPW**	**NF HPW**	**Terwer**	**EF Terwer**
Year Assessed	1994	1996	1996	1996	1996	1996	1994	1996
Assessed by:	Green Diamond	YTFP	YTFP	YTFP	YTFP	YTFP	Green Diamond	YTFP
Total Length of Channel Assessed (ft)	54,399	11,846	18,336	10,880	23,404	4,413	62,416	16,131
Mean % Canopy Density	80	88	80	76	90	95	36	71
% deciduous	93	93	77	85	91	73	75	95
% conifer	7	7	23	15	9	27	25	5
% LWD as Structural Shelter in all Pools	35	55.1	36.4	15.8	46.1	33.1	16.5	6.8
Habitat Types as % of Total Length								
Riffles	8.0	1	8	0	15	22	19.0	7
Flat-water	32.0	41	35	6	28	9	43.0	59
Pools	17.0	15	37	6	19	52	31.0	34
Dry Channel	43.0	44	19	86	38	14	7.0	0
Culvert	0	0	1	0	0	0	0	0
Pool Tailout Embeddedness as % Occurrence								
0-25%	24.7	0	2.3	0	1	0	31.3	9.0
26-50%	57.0	19	46.0	11	19.4	35	45.0	76.0
51-75%	18.2	47	49.4	79	69	63	21.3	15.0
76-100%	0	33	2.8	11	10.6	2	0	0
Maximum Pool Depths as % Occurrence								
<1' deep	0.0	1.8	9.7	21.1	5.0	10.4	0.5	1.6
1'-2' deep	8.0	56.1	55.7	57.9	70.5	60.4	1.5	48.4
2'-3' deep	38.3	31.6	27.8	15.8	22.7	29.2	19.8	36.3
3'-4' deep	32.5	8.8	6.1	0	1.8	2.1	28.9	9.3
>4' deep	21.4	1.8	1.0	5.3	0	0	49.2	4.4
Index of Embeddedness	2.7	2.9	3.0	3.0	1.6	2.6	2.5	1.6
Mid-point Gradient (%)	1.6	NA	3.6	NA	1.7	3.0	1.5	NA
Mid-point Watershed Area (acres)	4,898	1,031	2,134	526	1,012	522	8,602	3,523

Codes

HPW	Hoppaw Creek	NF HPW	North Fork Hoppaw
EF	East Fork	NA	Not applicable, or not available

Table C1-3 Continued. Stream assessment summaries for 22 Plan Area streams in the Coastal Klamath HPA.

Parameters	Streams							
	McG	**WF McG**	**Tarup**	**Omagar**	**APCM**	**APCS**	**APCN**	**A-P Trib**
Year Assessed:	1996	1996	1996	1996	1995	1995	1995	1997
Assessed by	YTFP	YTFP	YTFP	YTFP	Smpsn	Smpsn	Smpsn	YTFP
Total Length of Channel Assessed (feet)	29,085	13,033	26,343	13,276	17,299	8,284	26,669	3,132
Mean % Canopy Density	89	94	97	95	91	95	93	84
% deciduous	92	89	93	90	97	94	89	90
% conifer	8	11	7	10	3	6	11	10
% LWD as Structural Shelter in all Pools	37.8	41.2	25.4	43.4	15.1	35.8	9.6	27.1
Habitat Types as % of Total Length								
Riffles	4	6	10	10	28.0	46.0	37.0	6
Flat-water	25	20	19	39	31.0	29.0	29.0	54
Pools	69	73	71	26	17.0	24.0	25.0	39
Dry Channel	1	1	0	0	24.0	1.0	9.0	1
Culvert	0	0	0	23	0	0	0	0
Pool Tailout Embeddedness as % Occurrence								
0-25%	0.4	0	1.6	7.0	9.0	15.0	9.8	44.1
26-50%	15.5	2.7	26.5	51.0	33.3	23.0	19.3	55.9
51-75%	66.7	62	71.1	38.3	27.9	21.0	27.0	0
76-100%	17.7	35.5	0.9	3.7	24.9	41.0	43.7	0
Maximum Pool Depths as % Occurrence								
<1' deep	6.5	13.9		15.1	2.2	1.5	0.6	19.2
1'-2' deep	42.8	47.5	30.3	56.0	30.1	67.6	29.3	56.2
2'-3' deep	32.1	27	43.9	16.4	45.2	29.4	48.1	20.5
3'-4' deep	10.7	25	16.8	5.0	17.2	1.5	17.1	4.1
>4' deep	7.8	1.6	9.0	0.0	5.4	0.0	5.0	0.0
Index of Embeddedness	1.9	2.1	2.7	2.9	2.7	3.1	3.0	2.2
Mid-point Gradient (%)	1.8	2.7	5.6	3.9	1.7	4.5	2.1	5.6
Mid-point Watershed (acres)	1,672	1,296	1,971	773	2,573	1,290	2,437	1,076

Codes

McG	McGarvey Creek WF	McG	West Fork McGarvey Creek
APCN	North Fork Ah Pah Creek	APCM	Main stem Ah Pah Creek
A-P Trib	Tributary to Main stem Ah Pah	APCS	South Fork Ah Pah Creek

Table C1-3 Continued. Stream assessment summaries for 22 Plan Area streams in the Coastal Klamath HPA.

Parameters	Streams					
	Bear	**Bear (Trib 1)**	**Bear (Trib 2)**	**Surpur**	**Little Surpur**	**Tectah**
Year Assessed	1995	1996	1996	1996	1996	1996
Assessed by	Smpsn	YTFP	YTFP	YTFP	YTFP	YTFP
Total Length of Channel Assessed (feet)	17,581	7,102	4,242	18,046	11,072	66,632
Mean % Canopy Density	88	77	78	89	93	86
% deciduous	93	93	91	94	91	89
% conifer	7	7	9	6	9	11
% LWD as Structural Shelter in all Pools	19.8	9.8	22.7	13.2	18.2	14.6
Habitat Types as % of Total Length						
Riffles	58	14	3	4	0	6
Flat-water	24	53	64	23	33	44
Pools	16	33	31	73	61	48
Dry Channel	2	0	2	0	6	2
Culvert	0	0	0	0	0	0
Pool Tailout Embeddedness as % Occurrence						
0-25%	4.5	1.9	0.0	1.0	0.0	0.0
26-50%	22.3	79.4	73.0	36.0	31.3	68.0
51-75%	54.3	18.4	27.0	61.0	66.7	32.0
76-100%	19.0	0.0	0.0	3.0	2.1	0.0
Maximum Pool Depths as % Occurrence						
<1' deep	60.0	8.2	24.2	0.6	1.6	5.7
1'-2' deep	6.0	71.4	56.1	42.3	42.6	35.9
2'-3' deep	19.0	15.3	15.2	37.2	36	30.6
3'-4' deep	6.0	4.1	4.5	17.3	18.2	14.3
>4' deep	9.0	2.0	0.0	2.6	1.6	13.5
Index of Embeddedness	2.7	2.3	2.9	2.4	2.5	2.3
Mid-point Gradient (%)	3.4	4.2	NA	NA	4.0	NA
Mid-point Watershed (acres)	5,112	1.186	1.442	2.712	1.363	7,434

Table C1-4. Stream assessment summaries for four Plan Area streams in the Blue Creek HPA.

Parameters	Streams			
	Blue	**WF Blue**	**Potato Patch**	**Slide**
Year Assessed	1998	1995	1997	1997
Assessed by	YTFP	YTFP	YTFP	YTFP
Total Length of Channel Assessed	77,144	22,842	2,162	12,050
Mean % Canopy Density	42	87	95	38
% deciduous	66	94	90	23
% conifer	34	6	10	77
% LWD as Structural Shelter in all Pools	4.0	6.0	1.5	3.3
Habitat Types as % of Total Length				
Riffles	16	49	13	16
Flat-water	61	23	56	65
Pools	23	27	30	19
Dry Channel	0	1	0	0
Pool Tailout Embeddedness as % Occurrence				
0-25%	6.1	10.2	0.0	0.9
26-50%	75.1	31.3	28.7	65.3
51-75%	17.5	53.1	68.7	31.0
76-100%	1.3	4.7	2.7	2.8
Maximum Pool Depths as % Occurrence				
<1' deep	0.6	78.4	0	0
1'-2' deep	6.3	1.1	45.5	12.9
2'-3' deep	5.0	8.7	39.4	44.7
3'-4' deep	21.4	8.3	12.1	32.9
>4' deep	66.4	3.5	3.0	9.4
Index of Embeddedness	2.9	2,2	2,1	2,7
Mid-point Gradient (%)	2.0	6.1	5.7	6.6
Mid-point Watershed Area (acres)	38,563	4,372	2,820	3,414

Table C1-5. Stream assessment summaries for 11 Plan Area streams in the Interior Klamath HPA.

Parameters	Streams				
	Johnson	Pecwan	EF Pecan	Mettah	SF Mettah
Year Assessed	1996	1997	1997	1997	1997
Assessed by	YTFP	YTFP	YTFP	YTFP	YTFP
Total Length of Channel Assessed	11,906	4,239	1,836	36,801	8,482
Mean % Canopy Density	94	74	86	86	89
% deciduous	97	69	76	83	78
% conifer	3	31	24	17	22
% LWD as Structural Shelter in all Pools	9.3	1.7	4.3	10.3	19.9
Habitat Types as % of Total Length					
Riffles	3	14	16	10	12
Flat-water	24	62	30	51	64
Pools	60	24	54	40	24
Dry Channel	13	0	0	0	0
Pool Tailout Embeddedness As % Occurrence					
0-25%	0	0	0	0.0	0
26-50%	6.0	7.1	0	23	5.0
51-75%	93.0	92.9	100	76.6	92.0
76-100%	1.0	0	0	0.8	3.0
Maximum Pool Depths as % Occurrence					
<1' deep	4.2	0	0	4.7	0
1'-2' deep	46.9	19.0	10.0	56.5	54.1
2'-3' deep	33.3	33.3	35.0	27.7	38.8
3'-4' deep	11.5	33.3	30.0	8.4	7.1
>4' deep	4.2	14.3	25.0	2.9	0
Index of Embeddedness	3.0	3.0	3.0	2.8	3.0
Mid-point Gradient (%)	NA	3.5	4.1	2,8	3.0
Mid-point Watershed Area (acres)	1,307	17,574	8,401	2,959	1,558

Table C1-5 Continued. Stream assessment summaries for 11 Plan Area streams in the Interior Klamath HPA.

Parameters	Streams					
	Roach	Roach (Trib)	Morek	Cappel	Tully	Robbers Ck
Year Assessed	1997	1997	1997	1997	1997	1997
Assessed by	YTFP	YTFP	YTFP	YTFP	YTFP	YTFP
Total Length of Channel Assessed	38,876	6,235	2,060	3,529	41,995	3,643
Mean % Canopy Density	78	80	85	79	79	84
% deciduous	70	73	66	59	92	92
% conifer	30	27	34	41	8	8
% LWD as Structural Shelter in all Pools	3.5	16.6	6.4	5.7	12.7	10.5
Habitat Types as % of Total Length						
Riffles	4	2	22	27	5	8
Flat-water	48	41	45	31	70	52
Pools	45	53	21	42	24	31
Dry Channel	3	3	13	0	2	1
Pool Tailout Embeddedness As % Occurrence						
0-25%	0	0	0	0	27.6	4.8
26-50%	0	0	16.6	2.0	54.6	32.1
51-75%	100	100	83.4	98.0	0	63.2
76-100%	0	0	0	0	0	0
Maximum Pool Depths as % Occurrence						
<1' deep	1.1	0	9.0	2.3	0.8	6.2
1'-2' deep	30.6	52.4	40.1	14.0	28	43.7
2'-3' deep	30.6	30.2	45.4	65.1	41.4	37.4
3'-4' deep	21.0	12.7	4.5	14.0	19.2	10.4
>4' deep	16.7	4.8	0	4.7	10.7	2.1
Index of Embeddedness	2.4	3.0	2.8	3.0	1.9	3.0
Mid-point Gradient (%)	2.2	2.6	4.7	7.0	4.1	5.0
Mid-point Watershed Area (acres)	10,808	3,548	2,562	5,312	7,264	2,106

Table C1-6. Stream assessment summaries for four Plan Area streams in the Little River HPA.

Parameter	Streams			
	USFLR	LSFLR	RR	LR
Year Assessed	1994	1994	1994	1994
Assessed by	L-P	L-P	L-P	L-P
Total Length of Channel Assessed (feet)	10539	14998	7,262	62,373
Mean % Canopy Density	99	98	98	91
% deciduous	76	67	69	84
% conifer	24	33	31	16
% LWD as Structural Shelter in All Pools	25.9	38.5	26.6	17.3
Habitat Types as % of Total Length				
Riffles	32	30	37	19
Flat-water	20	11	7	25
Pools	45	56	46	53
Dry Channel	3	3	10	3
Pool Tailout Embeddedness as % Occurrence				
0-25%	21.7	14.2	10.5	8.1
26-50%	44.0	46.3	49.2	41.1
51-75%	17.2	31.4	31.9	38.7
76-100%	16.6	8.3	8.1	12.1
Maximum Pool Depths as % Occurrence				
<1' deep	6.8	5.0	26	2.7
1'-2' deep	49.5	43.4	50.0	20.4
2'-3' deep	31.8	31.4	18.7	26.8
3'-4' deep	6.8	7.5	4.4	26
>4' deep	4.5	12.6	1.1	23.6
Index of Embeddedness	2.3	2.3	1.9	3.2
Mid-point Gradient (%)	3.1	1.6	2.9	3.0
Mid-point Watershed Area (acres)	3,095	2,611	1,205	9,475

Codes

USFLR	Upper South Fork Little River
LSFLR	Lower South Fork Little River
RR	Railroad Creek
LR	Mainstem Little River
NA	Not applicable or not available

Table C1-7. Stream assessment summaries for five Plan Area streams in the Mad River HPA and North Fork Mad River HPA.

Parameter	Mad River HPA			North Fork Mad River HPA	
	Streams			Streams	
	CC	DC	LC	NFMR	LPC
Year Assessed	1994	1994	1995	1994	1994
Assessed by	Smpsn	Smpsn	Smpsn	Smpsn	Smpsn
Total Length of Channel Assessed (feet)	24,862	4,512	30,227	80,278	14,928
Mean % Canopy Density	81	92	79	73	95
% deciduous	85	75	79	95	87
% conifer	15	25	21	5	13
% LWD as Structural Shelter in All Pools	16.7	14	26.9	12.1	10.4
Habitat Types as % of Total Length					
Riffles	26	67	9	11	47
Flat-water	27	14	41	38	23
Pools	47	16	50	42	30
Dry Channel	0	3	0	10	0
Pool Tailout Embeddedness as % Occurrence					
0-25%	16.7	30.5	3.0	18.1	6.0
26-50%	41	40.8	16.0	19.3	21.3
51-75%	32.1	18.3	22.0	28.6	20.9
76-100%	11.2	11.1	60.0	33.6	51.9
Maximum Pool Depths as % Occurrence					
<1' deep	1.0	6.1	0.4	07.4	3.5
1'-2' deep	19.6	78.8	12.7	10.7	41.6
2'-3' deep	39.0	9.1	38.3	33.6	39.8
3'-4' deep	22.7	3.03	32.8	26.6	12.6
>4' deep	17.6	3.03	15.6	28.2	2.3
Index of Embeddedness	2.4	2.1	3.4	2.8	2.5
Mid-point Gradient (%)	3.0	3.7	1.0	1.4	2.6
Mid-point Watershed Area (acres)	8,595	1,492	2,985	11,273	4,592

Codes

DC	Dry Creek	NFMR	North Fork Mad River
CC	Cañon Creek	LPC	Long Prairie Creek
LC	Lindsay Creek	NA	Not applicable or not available

Table C1-8. Stream assessment summaries eight Plan Area streams in the Humboldt Bay HPA and Eel River HPA.

Parameter	Humboldt Bay HPA				Eel River HPA			
	Streams				Streams			
	RC	**RC(a)**	**RC(b)**	**SC**	**WC**	**ST**	**HW**	**WFH**
Year Assessed	1995	1995	1995	1994	1991	1991	1998	1998
Assessed by	CCC	CCC	CCC	Smpsn	CDFG	CDFG	CDFG	CDFG
Total Length of Channel Assessed (feet)	27,682	1,139	8,342	37,153	2,481	5,063	20,975	2,342
Mean % Canopy Density	94	90	88	88	80	67	57	86
% deciduous	68	NA	NA	83	83	71	81	95
% conifer	32	NA	NA	17	17	29	19	5
% LWD as Structural Shelter in all Pools	49.1	17.1	39.8	27.5	10.0	48.2	4.0	0.0
Habitat Types as % of Total Length								
Riffles	5	3	1	27	86	33	65	74
Flat-water	29	16	37	29	10	37	29	18
Pools	65	81	61	44	4	26	6	7
Dry Channel	1	0	0	0	0	5	0	0
Pool Tailout Embeddedness as % Occurrence								
0-25%	7.5	NS*	NS*	9.8	0	63.8	0.9	0.0
26-50%	22.4			24.5	17.8	17.7	22.3	18.0
51-75%	33.5			34.5	17.8	17.3	62.3	73.0
76-100%	36.6			30.6	64.4	1.1	13.8	9.0
Maximum Pool Depths as % Occurrence								
<1' deep	6	19	2.9	0.6	0.0	0.0	0.0	0.0
1'-2' deep	44.8	54.8	43.8	12.6	83.3	43.1	42.0	81.8
2'-3' deep	30.7	19	35.1	42.5	16.7	39.4	52.0	18.2
3'-4' deep	12.2	7.1	13.9	26.5	0.0	10.6	3.8	0.0
>4' deep	6.2	0.0	4.3	17.9	0.0	7.3	2.3	0.0
Index of Embeddedness	3.0	3.0	4.0	2.8	2.9	2.3	2.4	1.9
Mid-point Gradient (%)	1.0	1.0	1.0	1.0	2.6	3.3	2.1	7.0
Mid-point Watershed Area (acres)	3,669	662	1,293	5,399	1,250	3,308	2,594	3,372

Codes

RC	Ryan Creek	WC	Wilson Creek
RC(a)	1st unnamed trib to RC	ST	Stevens Creek
RC(b)	2nd unnamed trib to RC	HW	Howe Creek
SC	Salmon Creek	WFH	West Fork Howe Creek
NS*	The CCC judged these pools as 'Not suitable for spawning', and did not record pool tailout embeddedness values.	NA	The value was either not recorded or not applicable

C1.3.1 Mean Percent Canopy Closure and Percent Canopy Cover

The mean percent canopy closure along each assessed stream as a function of watershed area is shown as Figure C1-1. The percentage of canopy closure along stream channels is important for the regulation of water temperatures and as a source of nutrients for the aquatic organisms. This assessment also provides information about the species (conifer, deciduous) composition of the riparian zone.

The mean canopy closure in the 58 assessed streams ranged from 36% in Terwer Creek ([Coastal Klamath HPA] Table C1-3), to 99% in Upper South Fork of Little River ([Little River HPA] Table C1-6) and are shown in Figure C1-1. CDFG's Salmonid Restoration Manual recommends that a mean canopy closure of approximately 80% is required/desirable to maintain suitable summer water temperatures for juvenile coho salmon (Flosi and Reynolds 1994). From the assessments conducted 69% of the streams assessed (40 of 58) had mean canopy closures greater than or equal to 80% (Figure C1-1). As shown in this figure the mean canopy closure percentage diminishes with increased stream watershed size.

The percent canopy cover by type (deciduous and conifer) for the assessed streams are shown in Tables C1-2 through C1-8. The mean percent conifer closure plotted against watershed area is shown as Figure C1-2. The percent of conifer cover ranged from a low of 2% in the South Fork Winchuck River ([Smith River HPA] Table C1-2) to 77% on Slide Creek ([Blue Creek HPA] Table C1-4) and are shown in Figure C1-2. As shown in Figure C1-2, deciduous trees dominated the riparian canopy of the assessed streams, with most of the streams (67%) containing less than 20% conifers along the riparian margin. As shown in the figure, there is a trend with a slightly larger percentage of conifer canopy in larger watersheds as compared to smaller watersheds.

C1.3.2 Percent LWD as Structural Shelter in Pool Habitats

To assess habitat complexity, the dominant structural shelter element and the contribution of other shelter components was determined on a percent basis for each habitat type. LWD is an important shelter component that facilitates numerous functions within certain channel types. LWD is a pool-forming component that adds complexity and cover to stream channels. The percentage of in-channel LWD as shelter should reflect the quantity and quality of potential salmonid habitat and possibly the effects of past management practices.

The results of assessment of LWD as structural shelter in all pools surveyed as part of the habitat assessments are summarized in Tables C1-2 through C1-8. LWD as structure in pools in the assessed streams are shown by watershed area in Figure C1-3. As shown in Figure C1-3, the percentage of LWD as shelter was greatest in stream pools. The percentage of LWD as shelter in pools ranged from a low of 0% in West Fork Howe Creek ([Eel River HPA] Table C1-8) to a high of 55% in East Fork Hunter Creek ([Coastal Klamath HPA] Table C1-3).

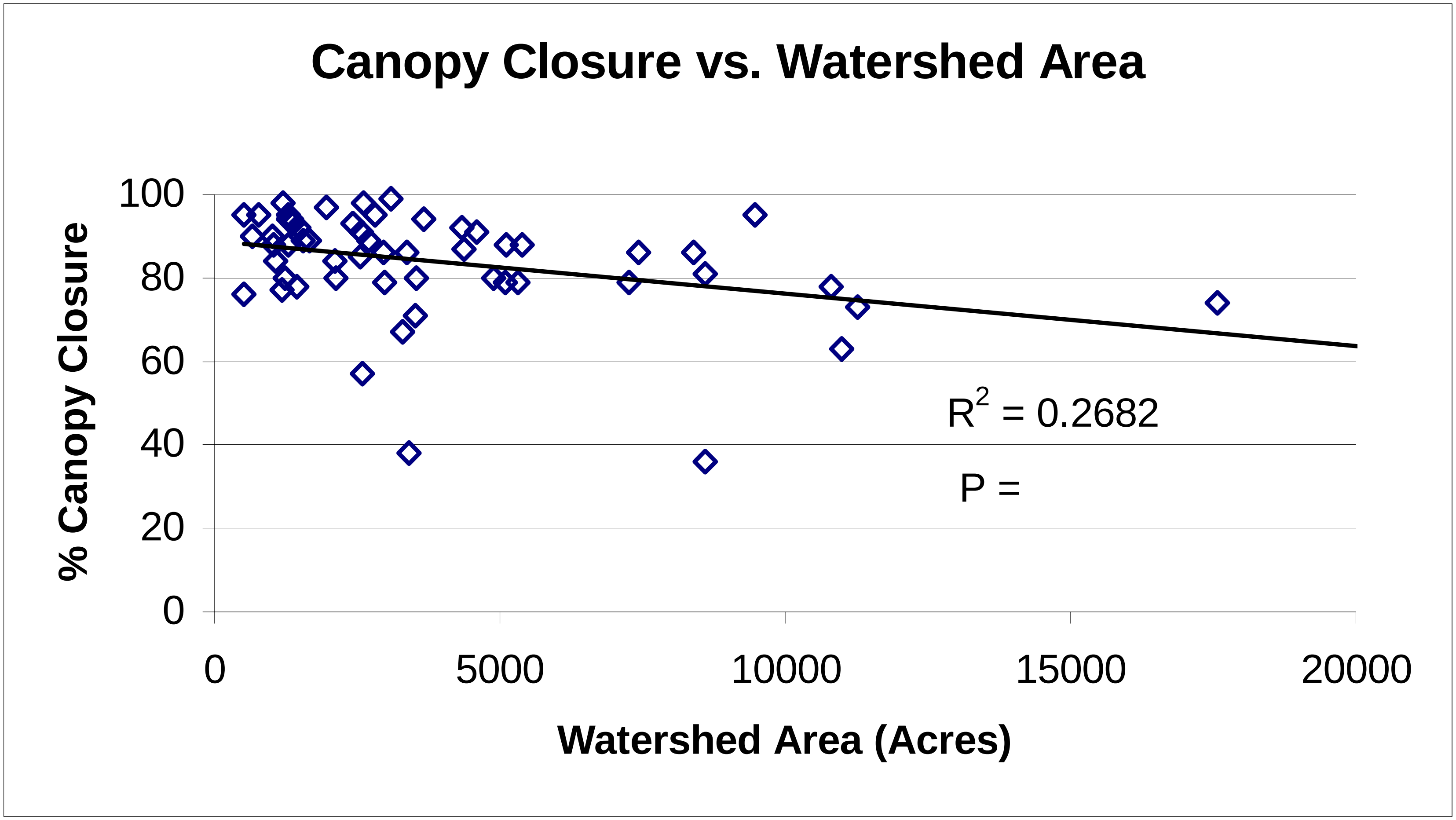

Figure C1-1. Canopy closure versus watershed area for all assessed streams in which habitat typing surveys were conducted.

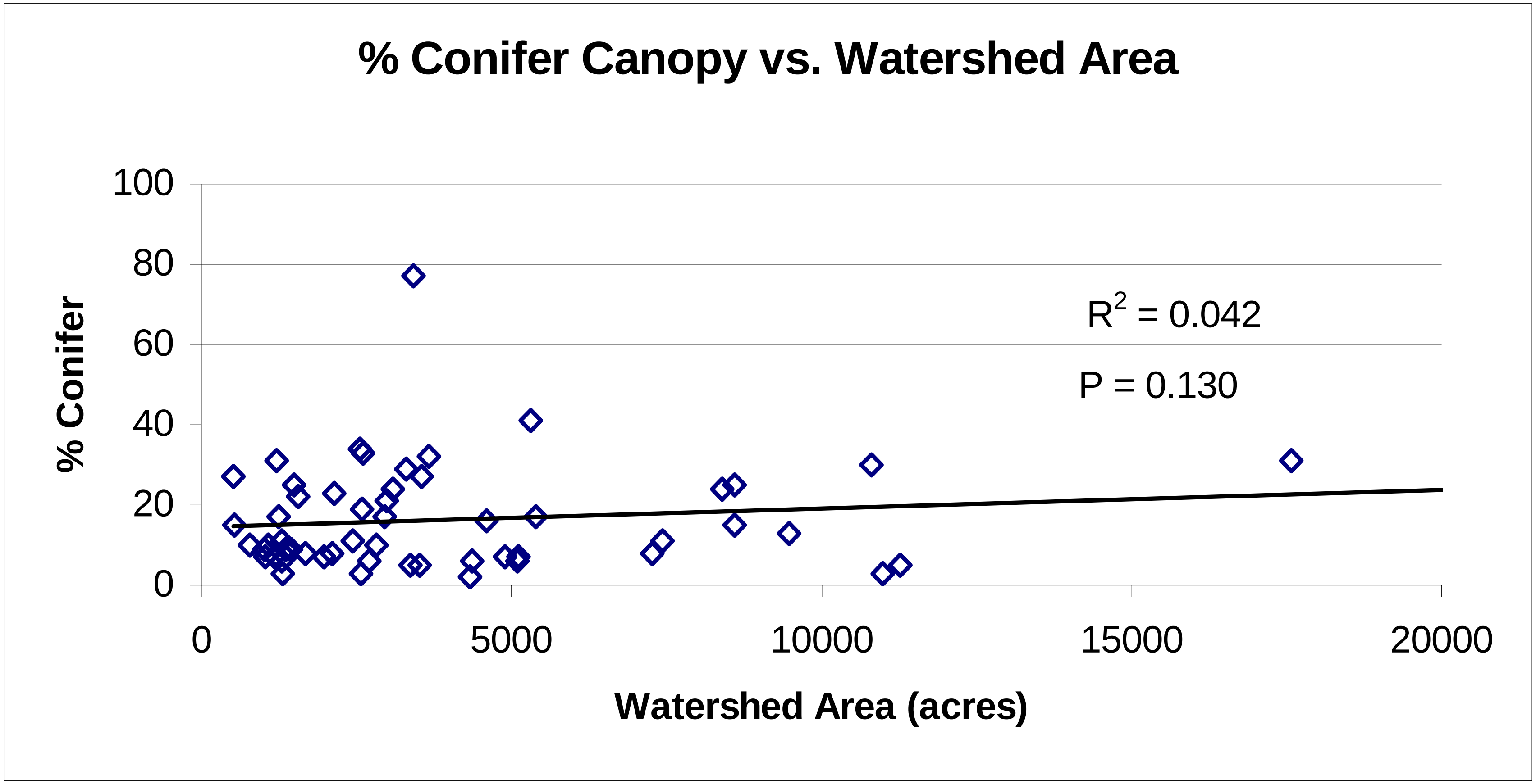

Figure C1-2. Percent conifer canopy versus watershed area for all assessed streams in which habitat typing surveys were conducted.

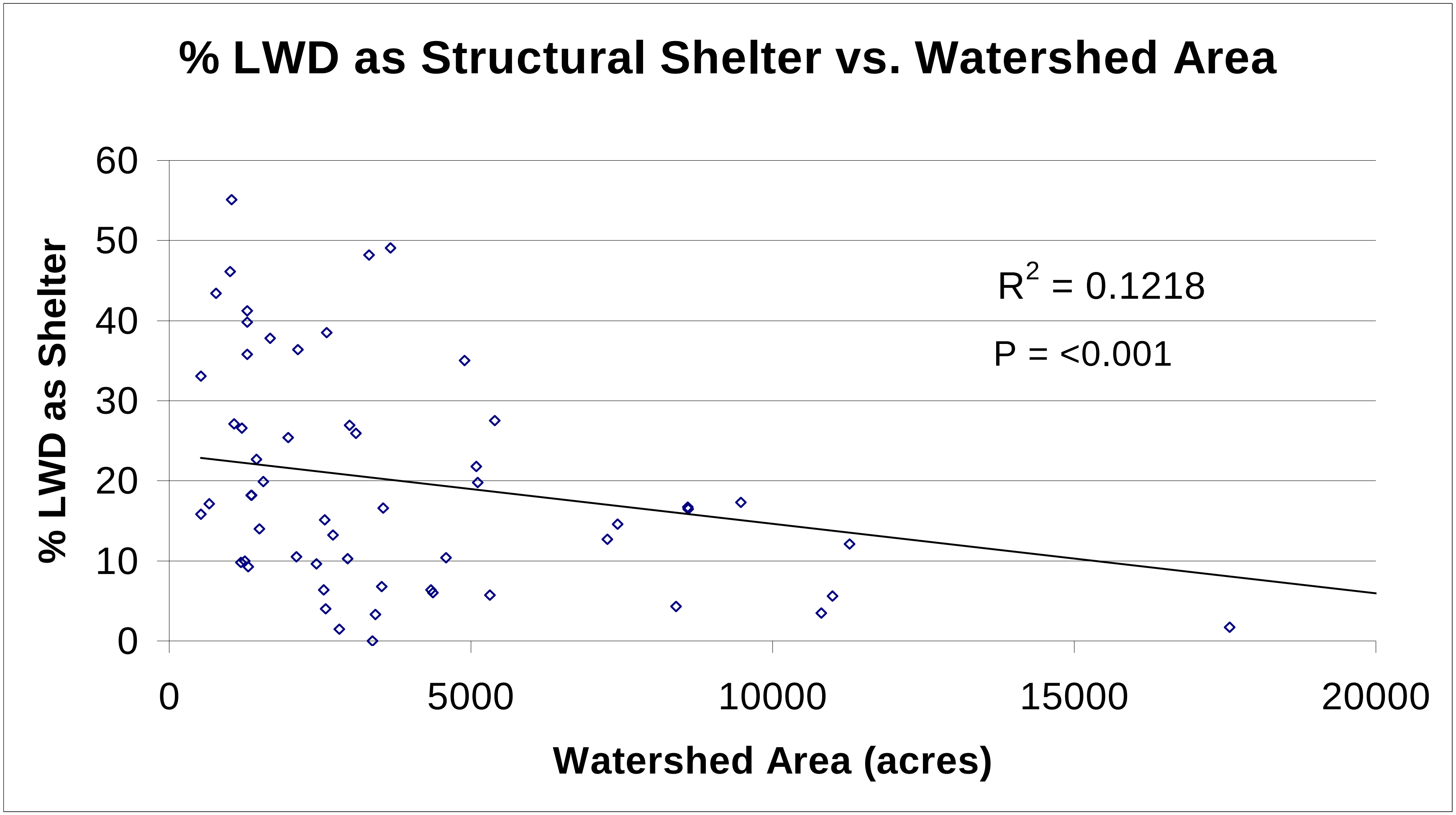

Figure C1-3. Percentage of LWD as structural shelter versus watershed area for the assessed streams.

East Fork Hunter Creek was the only stream assessed in which LWD was the dominant (>50%) structural cover. Two additional streams, Ryan Creek ([Humboldt HPA] Table C2-7) with 49%, and Stevens Creek ([Eel River HPA] Table C1-8) with 48% had nearly 50% LWD as structural cover. Of all 58 of the streams assessed, approximately 36% (21 of 58 streams) had LWD as a structural shelter component greater than 20% of all in-stream cover present (Figure C1-3). As shown in that figure there is generally a trend of lower percentages of LWD as structural shelter in pools within streams with larger watershed areas.

The relatively higher amounts of LWD as structural shelter in Hunter Creek, Ryan and Stevens Creeks are probably due to past management practices which retained some riparian cover and also did not aggressively clear the channel of LWD. These watersheds may additionally have some inherent geologic instability that still provides episodic inputs of LWD and sediments to their channels. The lower percentages of LWD in the North Fork Mad River can be attributed to extensive clearing of LWD from the channel. Historic photographs from the mid-1950's show sections of channel clogged with immense jams of logging slash and giant pieces of redwood LWD. Presently, these same sections of channel are nearly devoid of LWD as a result of aggressive stream cleaning efforts during the late 1960's and 1970's. At the time, clearing stream channels of debris jams was deemed by the best available information as a means of fisheries restoration (stream cleaning was also a response to the damage incurred to bridges and roads by debris during the 1955 and 1964 floods). Unfortunately many of these efforts went far beyond improving fish passage and removed what are now regarded as vital habitat components.

C1.3.3 Habitat Types as a Percent of Total Length

Level II (Flosi and Reynolds 1994) partitioning of habitat units separates the stream channel into riffles, flat-water, pools and dry channel. Generally, forming conclusions about the relative health of a stream with respect to salmonids from a level II partitioning of habitat units is difficult. Local geology, channel type, water level, and channel gradient will all influence the relative proportions of each habitat type. However, an extremely high proportion of a certain habitat unit may indicate a channel response to major (either natural or management influenced) watershed disturbances.

Excessive aggradation of stream reaches may lead to a high proportion of riffle habitat as well as an increase in seasonal stretches of dry channel as pools and runs get filled in with sediment. Intermittence is common in steep mountainous watersheds where a majority of the channel is confined and sediments are transported through these areas and are deposited on the wide, low gradient reaches near the mouths. Depending on the watershed this aggradation of sediment can be quite extensive. During low flow conditions the stream will go sub-surface, percolating through the sediment deposits. Many stream channel segments assessed were dry during the assessment surveys.

The summary of the habitat types as a percent of total length of each assessed stream and plotted by watershed area are shown in Tables C1-2 through C1-8. Of the 58 streams evaluated, there were 59% (34 out of 58) which had at least 1% of their total length of stream channel classified as dry channel. Three streams had greater than 40% of their total channel classified as dry: Hunter Creek (43%), East Fork Hunter Creek (44%) and Mynot Creek (86%) all within the Coastal Klamath HPA (Table C1-3).

Many watersheds within the Plan Area exhibit this naturally occurring phenomenon. However, the increased sediment loads from hillslope failures often associated with logging activities and road construction can amplify the spatial and temporal extent of intermittency (Hicks et al. 1991). The impact of intermittency on salmonid populations has not been quantified, but probably affects the out-migration of juveniles or may result in the stranding of juveniles in isolated pools where they would be susceptible to threshold temperatures and increased predation.

For the streams assessed, the percent of stream length of pools ranged from 4% in Wilson Creek ([Eel River HPA] Table C2-8) to 81% in Ryan Creek ([Humboldt Bay HPA] Table C2-7). The percent of stream length of pools by watershed area are shown in Figure C1-4. As shown in Figure C1-4 the percentage of stream length of pools were widely variable in smaller watersheds (less than 5000 acres). For the 58 streams assessed, the percent of total stream length of riffles ranged from 0% in Mynot Creek and Little Surper Creek ([Coastal Klamath HPA] Table C1-3) to 86% in Wilson Creek ([Eel River HPA] Table C1-8). The percentage of stream length of flat-water habitats ranged from 6% in Mynot Creek ([Coastal Klamath River HPA] Table C2-3) to 70% in Tully Creek in the Interior Klamath River HPA (Table C1-4). The trend is that as watershed size increases beyond 5,000 acres, the variability in pool lengths as a total of stream length decreases.

C1.3.4 Pool Tail-out Embeddedness as Percent Occurrence

Summary of pool-tail out embeddedness estimates are shown in Tables C1-2 through C1-8. The embeddedness of channel substrate in pool tail-outs is a gross indication of the amount of fines present in spawning gravels which, in turn, may reduce the survival to emergence of salmonid alevins. However, the measurement is subjective and probably not accurately repeatable. If embeddedness was considered high (>50%), a more rigorous monitoring of substrate composition may be warranted to document amount of fines within pool tail-outs. Of the 58 assessed streams, 60% (35 out of 58) had embeddedness occurrences greater than 50%. From these assessments, 3 streams: East Fork Pecwan, Roach Creek, and a tributary to Roach Creek (all in the Interior Klamath HPA) had pool tail-out embeddedness occurrences of 100%.

An index of Pool tail-out embeddedness as a function of stream gradient for the assessed streams is shown in (Figure C1-5). Using embeddedness index categories of 1 through 4 which correspond to estimates of percent embeddedness of: 0-25% =1; 26-50% = 2; 51-75% = 3; and 76-100% = 4 the streams were categorized as shown in Figure C1-5 (Flosi et al. 1998). As shown in Figure C1-5 the estimated embeddedness for all Plan Area streams assessed generally were found to fall within the range of Index values of 2 to 3 regardless of stream gradient and the average index rating only diminished slightly for streams with larger watersheds.

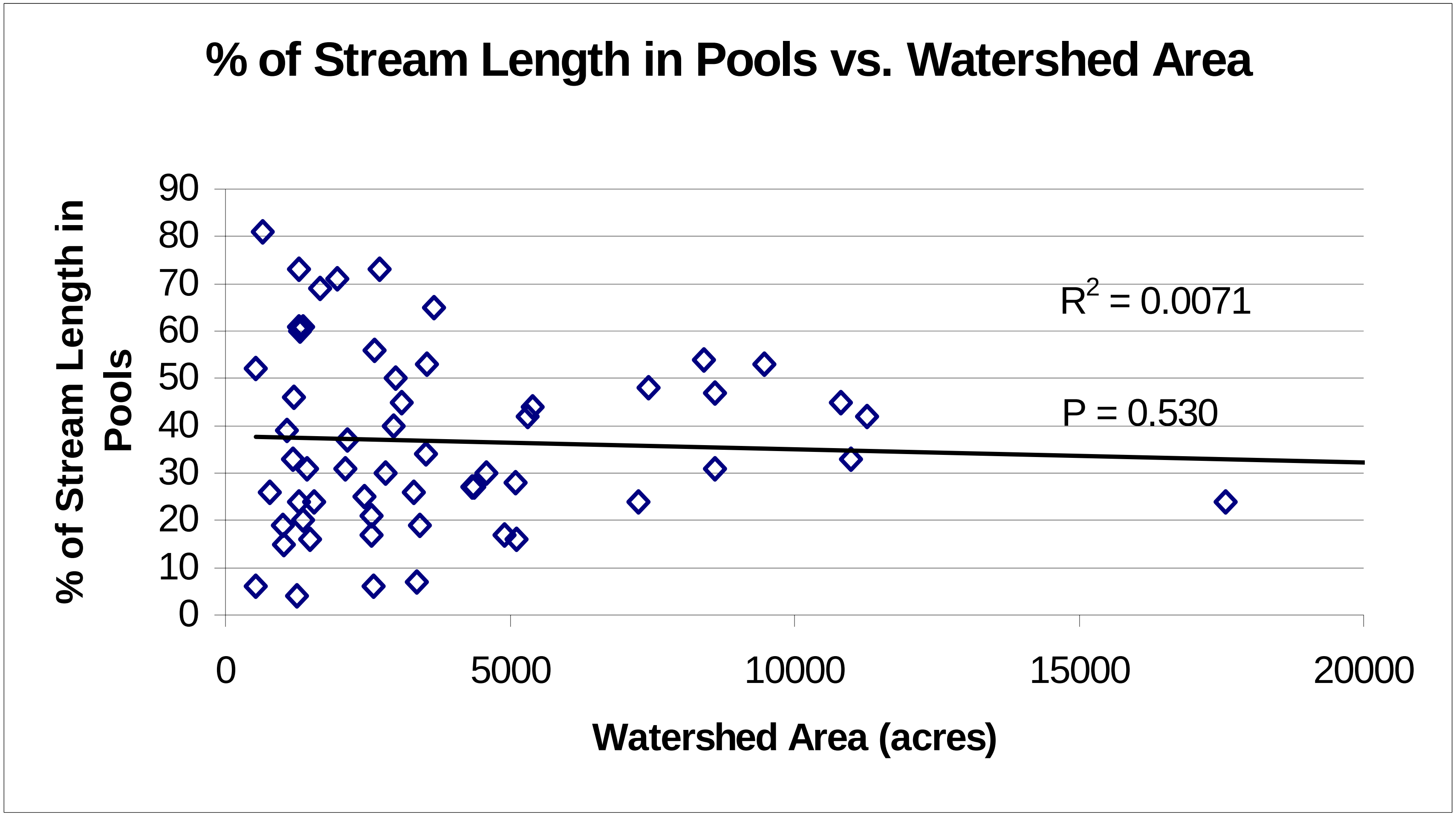

Figure C1-4. **Percent of stream length in pools plotted by watershed area for all streams assessed during the habitat assessments.**

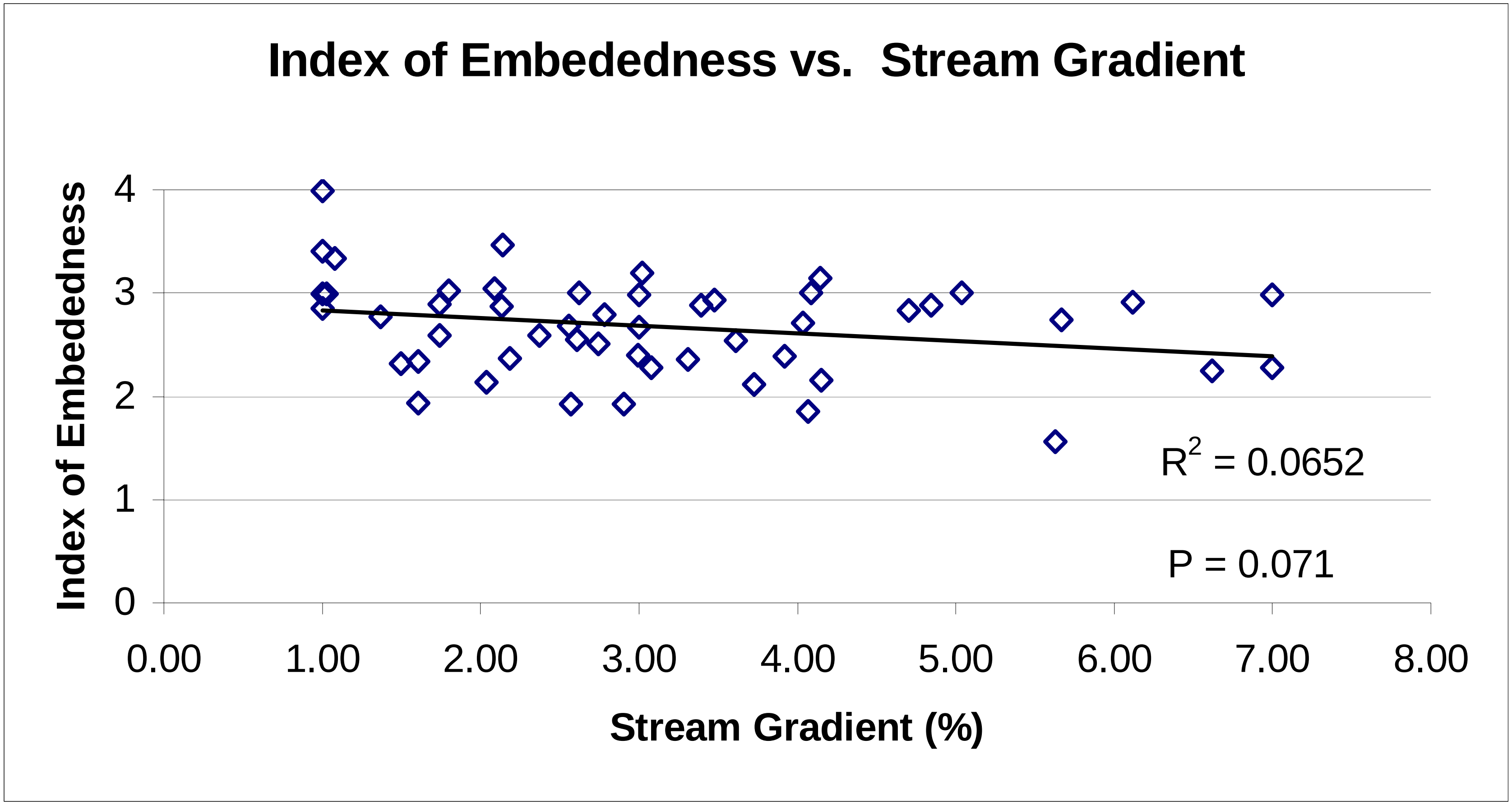

Figure C1-5. **Index of streambed embeddedness as a function of stream gradient for all assessed streams.**

C1.3.5 Maximum Residual Pool Depth as Percent Occurrence

Maximum pool depths are used by CDFG to calculate the percentage of primary pools, which are known to provide critical summer habitat for juvenile coho and steelhead under low flow conditions (Flosi et al., 1998). From CDFG's habitat typing assessments, there are indications that the better coastal coho streams may have as much as 40% of their total habitat length in primary pools (Flosi et al., 1998). A primary pool in a third order or larger stream would be expected to have a depth of three feet or greater. A primary pool in a first and second order stream is considered to be a depth of 2 feet or greater (Flosi and Reynolds 1994). Watershed area may be a confounding factor in comparing this variable, as smaller drainages with lower discharges tend to have shallower pools.

A summary of the residual pool depths for all assessed streams is shown in Tables C1-2 through C1-8. Of the 58 streams assessed, 14 (24%) had greater than 40% of their total pool habitat in primary pools (residual depths greater than 3') (Figure C1-6). These included three creeks that had in excess of 70% of their pools greater than 3' in depth: Rowdy Creek ([Smith River HPA] 70.4%), Terwer Creek ([Interior Klamath River HPA] 78.1%), and Blue Creek ([Blue Creek HPA] 87.8%) (Figure C1-6). On the average, the mean maximum residual pool depth was 2 feet for the assessed streams. In general, the streams with larger watershed areas contain deeper pools, on the average, than those with smaller watershed areas. Most of the assessed streams are in small drainages and are smaller than third order streams. Pools with residual depths greater than 2 feet or greater in many of these small streams may act as primary pools and provide temperature refugia. If these pools were considered as primary pools, functioning as summer habitat for juvenile salmonids during low flow conditions, then 71% of the assessed streams (41 out of 58) have greater than 40% of their pools classified as primary pools. Twenty-one percent of total streams assessed (12 out of 58 streams), have over 80% of their total pools greater than 2' in depth (Figure C1-6).

C1.4 CONCLUSIONS

The stream channel and habitat typing assessments indicated that habitat conditions for salmonids varied significantly among and within the 58 assessed streams. Taken together, the assessments suggested that there were:

1. A lack of complex pool habitat with low levels of LWD as shelter;

2. Dense, alder dominant riparian zones that provided excellent canopy closure, yet lacked the LWD recruitment potential of larger, more persistent, conifers;

3. Embedded gravels in many pool tails; and

4. Aggraded conditions in the lower reaches of some streams.

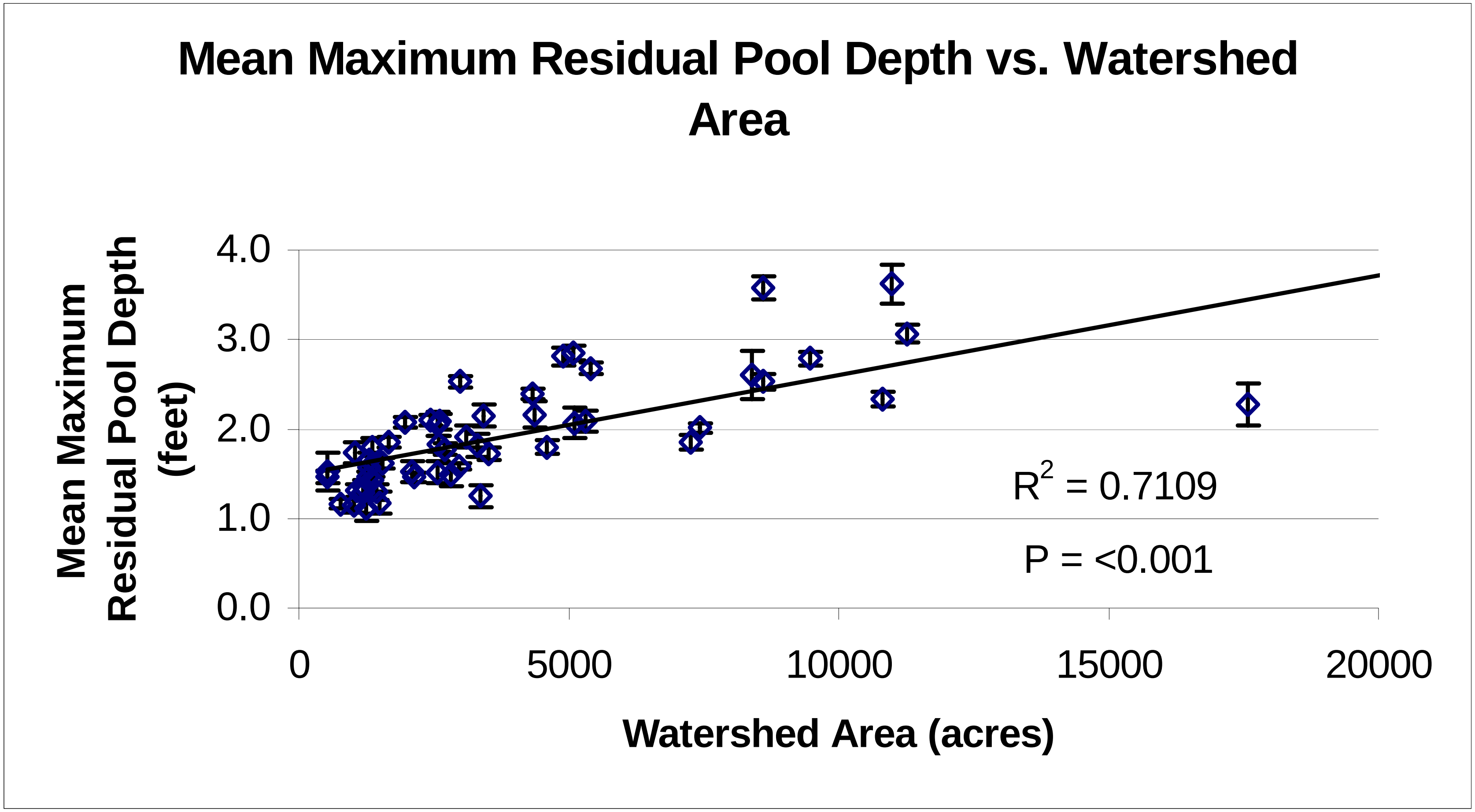

Figure C1-6. **Mean maximum pool depths plotted against watershed acres for the assessed streams. Error bars represent plus or minus one standard error.**

C1.5 REFERENCES

Flosi, G. and F.L. Reynolds. 1994. California salmonid stream habitat restoration manual. Second Edition. IFD, CDFG, Sacramento, CA.

Flosi, G., S. Downie, J. Hopelain, M. Bird, R. Coey, and B. Collins. 1998. California salmonid stream habitat restoration manual. Third Edition. IFD, CDFG, Sacramento, CA.

Hicks, B.J., J.D. Hall, P.A. Bisson and J.R. Sedell. 1991. Responses of salmonids to habitat changes. American Fisheries Society Special Publication 19:483-518.

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Appendix C2. Large Woody Debris Surveys

CONTENTS

Figures

Tables

C2.1 OBJECTIVES AND METHODS

In the following description, there is a difference between an inventory and a sample. A sample is a type of survey where the crewmember only counts and measures LWD pieces within a certain percentage (i.e. 20% sample) of the stream length. An inventory is a survey in which all pieces are counted and measured for the entire anadromous stream length.

C2.1.1 Number of Streams Sampled and/or Inventoried

An in-channel and recruitment zone large woody debris (LWD) survey was conducted on 16 streams on Green Diamond's ownership in the HPAs: eight in in 1994 and eight additional streams in 1995. Information regarding the distribution of LWD was also obtained in the channel and habitat typing assessment process, but the importance of LWD to biological and physical processes in the stream channel justified the need for a more thorough assessment of this critical habitat component. The LWD surveys covered two distinct zones:

- LWD within the bankfull discharge area of the stream channel; and
- LWD and live trees within the "recruitment zone," defined as the area encompassing the floodplain and 50 feet of the hillslope beyond the bankfull channel margin.

The objectives of the LWD survey include:

- Accurately documenting the current abundance, distribution, and characteristics of instream LWD.
- Providing a repeatable methodology for monitoring long-term changes in the abundance, distribution, and characteristics of instream LWD.
- Accurately identifying the source of instream LWD (naturally recruited or restoration structure) and the species composition of instream LWD (hardwood or conifer).

The LWD survey was conducted using the CDFG methods (Flosi and Reynolds, 1994). This methodology is a 20% sample that was designed with the objective of quickly identifying stream reaches lacking in LWD for prioritizing restoration projects. Each stream reach is delineated by Rosgen Channel Type during the CDFG Habitat Typing process. During these LWD surveys 200' out of every 1000' of each channel type would be inventoried for both inchannel LWD and recruitment zone LWD.

Little River and three of its primary tributaries were inventoried for LWD in 1994 by Louisiana Pacific (LP) Fisheries Biologists. In 1998 Green Diamond Timber acquired the LP timberlands as well as their historical fisheries data for Little River. LP's LWD survey was a 100% inventory that tallied all inchannel pieces of LWD within the Bankfull margins. In LP's survey no riparian or recruitment zone inventory was conducted and the inchannel inventory grouped the 3' – 4' category with the >4' category. This lack of information is noted in the following tables that summarize the Little River LWD data.

C2.1.2 Index of LWD Volume

An index of volume was developed for the purposes of depicting and comparing the amount of LWD in each stream to the watershed area. At the time of the survey/inventory, LWD pieces were categorized as follows based on their length: 6-20 feet, and >20 feet. In addition the LWD pieces were categorized as follows based on their maximum diameter: 1-2 feet, 2-3 feet, 3-4 feet, and >4 feet. The volume index was calculated by multiplying the mean diameter class times the "mean" length class. The mean diameter classes used for calculating the volume index were: 1.5 feet for the 1-2' class, 2.5 feet for the 2-3', 3.5 feet for the 3-4' class, and 4 feet for the >4' class. The "mean" lengths used for calculating the volume index were: 13 feet for the 6- 20' class and 20 feet for the >20' class. The index of volume was based on the instream average pieces per 100 feet. Since the actual diameters and lengths were not measured for each piece, the calculated volume in not a "true" volume but rather an index of volume. The index allows comparison between streams on Green Diamond property within the different HPAs.

C2.1.3 100% In-Channel Inventory

During Green Diamond's 1994 surveys field crews noted that a 20% sample could significantly underestimate or overestimate the actual pieces per 100 feet of channel. For example within a short channel type, where only 400 or 600 feet of channel were sampled, it is possible that one large log jam could skew the survey results to indicate that there are more pieces per 100 feet than actually exist in the reach. Conversely, if in that same short reach of channel the survey locations randomly missed most of the LWD, the results would be artificially low. To test these possibilities, an additional 100% inventory was conducted on all of the streams surveyed in 1995. The 100% inventory and the CDFG 20% sample were conducted simultaneously. This data allows a direct comparison of the CDFG methodology to a known inventory and thus is an indicator of the accuracy of a 20% sample.

C2.1.4 1999 Prairie Creek Inventory by Redwood National Park

In-channel and recruitment zone LWD data from undisturbed watersheds in coastal California are needed to compare with data from managed forests in the same area. This need led to the cooperative effort with Redwood National Park (RNP) and National Marine Fisheries Service (NMFS) to inventory inchannel LWD in Prairie Creek. In 1999 RNP and NMFS conducted a 100% inventory of 4.3 miles of Prairie Creek in Prairie Creek National Park. Prairie Creek is considered to be the best remaining example of a watershed dominated by old growth redwood forest. While this survey focused on quantifying LWD volume rather than a piece count per unit length, the data has been summarized by size categories of inchannel pieces (Kramer, pers. Comm.). This data should be considered as a known or true piece count of a relatively undisturbed watershed that may be directly compared to both the CDFG 20% samples and the 100% inventories conducted in Plan Area streams. However, when comparing Prairie Creek and many of the assessed Plan Area streams, the differences in their channel morphology must be considered. Prairie Creek is a low–gradient alluvial channel in a relatively wide valley bottom, while many of the Plan Area streams are higher gradient in more incised channels.

C2.2 RESULTS

C2.2.1 LWD Sampling Survey Results

Results of Green Diamond's 1994 and 1995 LWD surveys and the 1994 Louisiana Pacific LWD inventories are summarized in Tables C2-1 through C2-14. Tables C2-1 through C2-7 contains the estimated overall LWD piece count, displayed as average pieces per 100 feet of channel, delineated by Rosgen Channel Type, condition (dead vs. live), and live species. Figure C2-1 depicts each stream's mean count of instream LWD per 100 feet of stream channel plotted against the stream's watershed area. Figure C2-2 graphically depicts, for each stream surveyed, the mean number of LWD pieces in the riparian recruitment zone per 100 feet of stream channel. Tables C2-8 through C2-14) provides summaries of the LWD data delineated by size categories both in the channel and in the riparian recruitment zone. In Figure C2-3, the index of LWD volume for each stream surveyed is plotted against that stream's watershed area.

In the 20 streams surveyed, the average amount of inchannel LWD ranged from zero pieces per 100 linear feet of an A2 channel type in North Fork Mad River (North Fork Mad River HPA) to 16.3 pieces per 100 linear feet of an F3 channel in Salmon Creek (Humboldt Bay HPA). The average amount of live conifers in the recruitment zone (50 feet beyond the bankfull channel) that could potentially become instream LWD ranged from 0 pieces per 100 linear feet in three sections of Long Prairie Creek (Mad River HPA) to 9.5 pieces per 100 linear feet of channel in the upper reaches of Salmon Creek (Humboldt Bay HPA). The survey also divided LWD pieces into eight size classes by length (greater or less than 20') and by diameter (1'-2', 2'-3', 3'-4', and over 4') to identify dominant size classes of LWD. Of the twenty streams surveyed in 1994 and 1995, the dominant, or co-dominant size class of inchannel LWD for all streams was 1'-2' diameter and less than 20' in length. The dominant size class in the riparian zone for all sixteen streams with Recruitment Zone surveys was consistently 1'-2' diameter and greater than 20' in length. The summarized results of the LWD surveys are presented in the tables below.

As shown in Figure C2-1, the mean number of instream LWD pieces per 100 feet of stream channel decreased significantly with increased watershed area. While there is some variability the trend for streams with less than approximately 4,000 acres in the watershed, the number of instream pieces of LWD is generally greater than 3 per 100 feet of channel (Figure C2-1). For streams with watershed areas greater than approximately 4,000 acres, the mean number of instream pieces of LWD is generally less than 3 pieces per 100 feet of stream channel (Figure C2-1).

The number of pieces of LWD within the stream recruitment zone for each of the Streams surveyed is shown in Figure C2-2. As shown in Figure C2-2, the mean number of pieces of LWD per 100 feet of channel in the riparian recruitment zone ranged from approximately 3.5 in Wilson Creek (Smith River HPA) to 12.5 for the South Fork Ah Pah Creek (Coastal Klamath River HPA). Streams within in the Coastal Klamath and Blue Creek HPAs had 5 of the 7 greatest mean number of LWD pieces (7.7 to 12.6 pieces) in the recruitment zone per 100 feet of stream channel of all streams surveyed.

Table C2-1. Summary of 1994 and 1995 LWD sample (average pieces per 100 feet by channel type), Smith River HPA.

South Fork Winchuck River		Recruitment Zone					In Channel	
Reach	**Channel Type**	**Dead & Down**	**Dead & Standing**	**Perched**	**Live Conifer**	**Live Deciduous**	**LWD**	**No. of Sections**
1	C4	0.5	0.1	0.1	0.2	7.1	1.2	16
2	F4	0.2	0.3	0.0	0.5	7.8	0.3	3
3	C4	1.3	0.1	0.0	0.9	5.9	2.4	7
4	D3	0.5	0.0	0.0	0.5	3.5	0.0	1
5	A2	1.5	1.0	0.5	0.5	6.4	3.0	4
Rowdy Creek		**Recruitment Zone**					**In Channel**	
Reach	**Channel Type**	**Dead & Down**	**Dead & Standing**	**Perched**	**Live Conifer**	**Live Deciduous**	**LWD**	**No. of Sections**
1	D4	0.3	0.0	0.0	0.3	1.1	0.7	12
2	B3	0.4	0.4	0.2	0.5	3.6	1.4	16
3	B2	0.2	0.5	0.3	0.7	5.5	0.5	6
4	F3	0.8	0.3	0.0	1.5	8.5	0.2	3
Dominie Creek		**Recruitment Zone**					**In Channel**	
Reach	**Channel Type**	**Dead & Down**	**Dead & Standing**	**Perched**	**Live Conifer**	**Live Deciduous**	**LWD**	**No. of Sections**
1	F3	1.0	0.4	0.1	0.6	3.2	1.8	8
2	A3	0.8	0.2	0.0	1.7	6.2	3.3	3
3	F3	3.0	1.0	0.0	3.5	2.0	1.0	1
4	A2	0.9	0.5	1.0	2.1	2.9	6.9	4
Wilson Creek		**Recruitment Zone**					**In Channel**	
Reach	**Channel Type**	**Dead & Down**	**Dead & Standing**	**Perched**	**Live Conifer**	**Live Deciduous**	**LWD**	**No. of Sections**
1	F4	1.7	0.2	0.1	1.2	4.1	2.0	35
2	B3	2.5	2.0	0.2	1.8	2.2	2.7	3

Table C2-2. Summary of 1994 and 1995 LWD sample (average pieces per 100 feet by channel type), Coastal Klamath HPA.

Hunter Creek		Recruitment Zone					In-Channel	
Reach	Channel Type	Dead & Down	Dead & Standing	Perched	Live Conifer	Live Deciduous	LWD	No. of Sections
1	F4	0.5	0.0	0.0	0.3	8.2	0.4	8
2	D4	0.6	0.1	0.1	0.5	2.9	1.8	25
3	B4	1.2	0.2	0.0	1.5	4.7	3.4	11
4	F3	2.2	0.5	0.0	1.2	4.7	3.7	3
5	F4	3.8	0.7	0.4	1.4	2.9	5.2	9
Terwer Creek		**Recruitment Zone**					**In Channel**	
Reach	**Channel Type**	**Dead & Down**	**Dead & Standing**	**Perched**	**Live Conifer**	**Live Deciduous**	**LWD**	**No. of Sections**
1	F4	1.6	0.2	0.1	1.5	2.0	3.6	18
2	F3	2.1	1.5	0.2	2.7	5.3	3.5	13
3	F2	4.1	1.9	0.1	3.8	6.4	1.5	15
4	F4	3.3	3.9	0.2	2.6	0.8	3.3	16
North Fork Ah Pah Creek		**Recruitment Zone**					**In Channel**	
Reach	**Channel Type**	**Dead & Down**	**Dead & Standing**	**Perched**	**Live Conifer**	**Live Deciduous**	**LWD**	**No. of Sections**
1	F4	0.2	0.3	0.0	3.2	2.1	1.7	5
2	A2	5.0	1.5	0.0	2.0	7.5	6.5	1
3	B3	3.6	1.1	0.0	3.4	7.1	5.8	4
4	B2	4.8	1.8	0.0	5.8	8.5	4.5	2
5	A2	5.2	0.8	0.2	4.7	7.0	4.7	3
6	F4	2.4	1.8	0.2	4.8	6.4	5.8	13
South Fork Ah Pah Creek		**Recruitment Zone**					**In Channel**	
Reach	**Channel Type**	**Dead & Down**	**Dead & Standing**	**Perched**	**Live Conifer**	**Live Deciduous**	**LWD**	**No. of Sections**
1	B4	4.8	0.1	0.1	1.1	2.6	2.1	5
2	A3	5.8	0.2	0.4	3.0	2.8	7.9	5
Ah Pah Creek		**Recruitment Zone**					**In Channel**	
Reach	**Channel Type**	**Dead & Down**	**Dead & Standing**	**Perched**	**Live Conifer**	**Live Deciduous**	**LWD**	**No. of Sections**
1	C4	0.8	0.2	0.7	2.7	2.5	2.1	6
2	D4	3.5	1.2	0.0	2.3	2.7	3.3	3
3	F3	3.5	1.3	0.0	5.3	1.3	2.3	2
4	A2	8.0	0.0	0.0	1.5	0.5	6.0	1
5	F4	6.6	0.3	0.0	3.3	1.4	7.0	4
6	A2	7.0	0.5	0.0	2.5	5.5	7.0	1
7	F3	4.4	1.0	0.4	2.6	4.6	5.8	4

Table C2-3. Summary of 1994 and 1995 LWD sample (average pieces per 100 feet by channel type), Blue Creek HPA.

West Fork Blue Creek		Recruitment Zone					In Channel	
Reach	Channel Type	Dead & Down	Dead & Standing	Perched	Live Conifer	Live Deciduous	LWD	No. of Sections
1	B2	0.8	0.2	0.0	1.5	3.5	1.8	5
2	A2	3.7	0.7	0.1	2.6	2.8	3.2	18

Table C2-4. Summary of 1994 and 1995 LWD sample (average pieces per 100 feet by channel type), Little River HPA.

Little River		Recruitment Zone (N/A)					In Channel	
Reach	**Channel Type**	**Dead & Down**	**Dead & Standing**	**Perched**	**Live Conifer**	**Live Deciduous**	**LWD**	**Length of Survey (ft)**
1	B3	N/A	N/A	N/A	N/A	N/A	2.2	1614
2	B2	N/A	N/A	N/A	N/A	N/A	1.5	5506
3	B3	N/A	N/A	N/A	N/A	N/A	2.8	3526
4	F2	N/A	N/A	N/A	N/A	N/A	3.2	3214
5	F3	N/A	N/A	N/A	N/A	N/A	1.4	1366
6	B2	N/A	N/A	N/A	N/A	N/A	2.0	10902
7	B4	N/A	N/A	N/A	N/A	N/A	2.5	9876
8	B2	N/A	N/A	N/A	N/A	N/A	2.4	6347
9	A2	N/A	N/A	N/A	N/A	N/A	3.2	1062
10	B2	N/A	N/A	N/A	N/A	N/A	4.2	9415
11	B3	N/A	N/A	N/A	N/A	N/A	5.1	2412
12	B2	N/A	N/A	N/A	N/A	N/A	8.8	2644
13	B4	N/A	N/A	N/A	N/A	N/A	10.2	3339
14	A2	N/A	N/A	N/A	N/A	N/A	9.8	1546
Railroad Cr.		**Recruitment Zone**					**In Channel**	
Reach	**Channel Type**	**Dead & Down**	**Dead & Standing**	**Perched**	**Live Conifer**	**Live Deciduous**	**LWD**	**Length of Survey (ft)**
1	F4	N/A	N/A	N/A	N/A	N/A	4.1	748
2	B2	N/A	N/A	N/A	N/A	N/A	6.7	3901
3	B3	N/A	N/A	N/A	N/A	N/A	7.8	1998
4	B4	N/A	N/A	N/A	N/A	N/A	13.1	1244
Lower South Fork Little River		**Recruitment Zone**					**In Channel**	
Reach	**Channel Type**	**Dead & Down**	**Dead & Standing**	**Perched**	**Live Conifer**	**Live Deciduous**	**LWD**	**Length of Survey (ft)**
1	F4	N/A	N/A	N/A	N/A	N/A	5.9	7594
2	F3	N/A	N/A	N/A	N/A	N/A	8.4	2042
3	B2	N/A	N/A	N/A	N/A	N/A	9.3	961
4	C4	N/A	N/A	N/A	N/A	N/A	9.4	1679
5	F3	N/A	N/A	N/A	N/A	N/A	10.9	1628
Upper South Fork Little River		**Recruitment Zone**					**In Channel**	
Reach	**Channel Type**	**Dead & Down**	**Dead & Standing**	**Perched**	**Live Conifer**	**Live Deciduous**	**LWD**	**Length of Survey (ft)**
1	B3	N/A	N/A	N/A	N/A	N/A	4.4	2437
2	B2	N/A	N/A	N/A	N/A	N/A	3.4	1250
3	A2	N/A	N/A	N/A	N/A	N/A	6.3	2190
4	F3	N/A	N/A	N/A	N/A	N/A	6.0	3942
5	B4	N/A	N/A	N/A	N/A	N/A	14.8	583

Table C2-5. Summary of 1994 and 1995 LWD sample (average pieces per 100 feet by channel type), Mad River HPA.

Lindsay Creek		Recruitment Zone					In Channel	
Reach	Channel Type	Dead & Down	Dead & Standing	Perched	Live Conifer	Live Deciduous	LWD	No. of Sections
1	F5	0.9	0.5	0.1	4.9	2.9	3.6	28
Cañon Creek		**Recruitment Zone**					**In Channel**	
Reach	**Channel Type**	**Dead & Down**	**Dead & Standing**	**Perched**	**Live Conifer**	**Live Deciduous**	**LWD**	**No. of Sections**
1	B4	0.5	1.0	0.5	5.8	2.3	1.3	2
2	D4	0.5	0.3	0.8	4.1	2.6	4.9	4
3	B3	2.6	0.5	0.4	5.0	3.5	1.5	4
4	F3	1.1	0.3	0.0	6.4	2.1	0.3	8
5	A2	1.3	0.1	0.4	6.6	3.4	1.8	6
Dry Creek		**Recruitment Zone**					**In Channel**	
Reach	**Channel Type**	**Dead & Down**	**Dead & Standing**	**Perched**	**Live Conifer**	**Live Deciduous**	**LWD**	**No. of Sections**
1	B4	0.9	1.1	0.3	2.8	1.8	1.8	4
2	A3	2.0	0.5	0.0	1.5	3.5	0.5	1
3	B3	0.0	1.0	1.0	2.5	10.0	6.5	1

Table C2-6. Summary of 1994 and 1995 LWD sample (average pieces per 100 feet by channel type), North Fork Mad River HPA.

North Fork Mad River		Recruitment Zone					In Channel	
Reach	Channel Type	Dead & Down	Dead & Standing	Perched	Live Conifer	Live Deciduous	LWD	No. of Sections
1	F4	0.2	0.0	0.1	1.1	0.8	0.6	12
2	B3	1.3	0.1	0.1	4.0	1.1	0.4	4
3	F2	0.3	0.1	0.3	3.2	0.8	0.2	6
4	A2	1.8	0.0	0.1	1.0	2.5	0.0	4
5	F2	1.4	0.4	0.3	6.2	4.7	1.1	36
6	F4	1.7	1.2	0.1	7.7	3.1	1.7	6
7	F3	1.4	1.0	0.1	6.6	2.6	1.4	7
8	F4	1.3	0.4	0.2	5.7	2.9	2.2	9
Long Prairie Creek		**Recruitment Zone**					**In Channel**	
Reach	**Channel Type**	**Dead & Down**	**Dead & Standing**	**Perched**	**Live Conifer**	**Live Deciduous**	**LWD**	**No. of Sections**
1	B3	1.9	2.5	0.4	2.6	9.7	2.4	7
2	B2	3.0	0.0	1.5	0.5	5.5	1.5	1
3	B3	2.0	1.2	0.3	5.8	6.3	5.3	3
4	F3	1.3	0.0	0.0	0.0	11.8	0.0	2
5	B2	3.5	0.0	1.5	0.0	6.0	3.5	1
6	F3	2.0	0.0	1.0	4.3	3.5	0.5	2
7	B2	6.5	0.5	0.0	0.0	4.0	0.0	1

Table C2-7. Summary of 1994 and 1995 LWD sample (average pieces per 100 feet by channel type), Humboldt Bay HPA.

Salmon Creek		Recruitment Zone					In Channel	
Reach	Channel Type	Dead & Down	Dead & Standing	Perched	Live Conifer	Live Deciduous	LWD	No. of Sections
1	F3	1.3	0.3	0.4	1.9	1.8	1.8	19
2	F1	0.8	0.5	0.5	3.8	1.8	3.0	2
3	F3	4.5	0.3	0.3	5.5	0.8	16.3	2
4	F1	0.0	0.0	0.0	1.0	0.0	5.0	1
5	F3	1.9	0.3	0.3	5.7	2.3	4.5	8
6	B2	3.3	0.7	1.2	9.5	6.4	6.1	7

Table C2-8. Summary of 1994 and 1995 LWD sample (average pieces per 100 feet by channel type), Smith River HPA.

Stream	Size Classes of In-channel LWD and Wood within Riparian Recruitment Zone								
	1'-2' max dia.[a]; <20'	1'-2' max dia.[a]; >20'	2'-3' max dia.[a]; <20'	2'-3' max dia.[a]; >20'	3'-4' max dia.[a]; <20'	3'-4' max dia.[a]; >20'	>4' max dia.[a]; <20'	>4' max dia.[a]; >20'	All Size Classes
SF WINCHUCK									
Instream LWD	0.8	0.4	0.2	0.1	0.1	0.0	0.1	0.0	1.7
Riparian	0.2	4.2	0.1	0.7	0.1	0.2	0.0	0.1	5.6
Total	1.0	4.6	0.3	0.8	0.2	0.2	0.1	0.1	7.3
ROWDY CREEK									
Instream LWD	0.2	0.2	0.1	0.0	0.2	0.1	0.1	0.0	0.9
Riparian	0.3	2.1	0.0	0.8	0.0	0.2	0.0	0.1	3.5
Total	0.5	2.3	0.1	0.8	0.2	0.3	0.1	0.1	4.4
DOMINIE CREEK									
Instream LWD	1.7	0.3	0.5	0.4	0.2	0.2	0.1	0.0	3.4
Riparian	0.5	3.8	0.2	1.3	0.1	0.4	0.0	0.1	6.4
Total	2.2	4.1	0.7	1.7	0.3	0.6	0.1	0.1	9.8
WILSON CREEK									
Instream LWD	0.4	0.4	0.4	0.3	0.1	0.1	0.2	0.2	2.1
Riparian	0.4	2.8	0.4	0.9	0.1	0.2	0.3	0.2	5.3
Total	0.8	3.2	0.8	1.2	0.2	0.3	0.5	0.4	7.4
[a] = maximum diameter of LWD piece									

Table C2-9. Summary of 1994 and 1995 LWD sample (average pieces per 100 feet by channel type), Coastal Klamath HPA.

Stream	Size Classes of Inchannel LWD and Wood within Riparian Recruitment Zone								
	1'-2' max dia.[a]; <20'	1'-2' max dia.[a]; >20'	2'-3' max dia.[a]; <20'	2'-3' max dia.[a]; >20'	3'-4' max dia.[a]; <20'	3'-4' max dia.[a]; >20'	>4' max dia.[a]; <20'	>4' max dia.[a]; >20'	All Size Classes
HUNTER CREEK									
Instream LWD	0.8	0.4	0.3	0.3	0.2	0.2	0.3	0.2	2.7
Riparian	0.3	3.1	0.3	0.6	0.1	0.3	0.2	0.2	5.1
Total	1.1	3.5	0.6	0.9	0.3	0.5	0.5	0.4	7.8
TERWER									
Instream LWD	0.7	0.6	0.3	0.4	0.2	0.3	0.2	0.4	3.1
Riparian	0.6	4.5	0.3	1.1	0.1	0.5	0.2	0.4	7.7
Total	1.3	5.1	0.6	1.5	0.3	0.8	0.4	0.8	10.8
AH PAH									
Instream LWD	2.0	0.7	0.8	0.3	0.3	0.1	0.2	0.2	4.6
Riparian	1.3	4.1	0.5	1.2	0.5	0.4	0.6	0.4	9.0
Total	3.3	4.8	1.3	1.5	0.8	0.5	0.8	0.6	13.6
NORTH FORK AH PAH									
Instream LWD	2.1	0.7	1.0	0.2	0.2	0.1	0.5	0.2	5.0
Riparian	0.7	6.9	0.6	1.0	0.3	0.4	0.8	0.6	11.3
Total	2.8	7.6	1.6	1.2	0.5	0.5	1.3	0.8	16.3
SOUTH FORK AH PAH									
Instream LWD	2.6	0.3	1.0	0.5	0.3	0.3	0.3	0.3	5.6
Riparian	1.2	6.1	1.1	1.6	0.6	0.6	0.7	0.8	12.7
Total	3.8	6.4	2.1	2.1	0.9	0.9	1.0	1.1	18.3

Table C2-10. Summary of 1994 and 1995 LWD sample (average pieces per 100 feet by channel type), Blue Creek HPA.

Stream	Size Classes of Inchannel LWD and Wood within Riparian Recruitment Zone								
	1'-2' max dia.[a]; <20'	1'-2' max dia.[a]; >20'	2'-3' max dia.[a]; <20'	2'-3' max dia.[a]; >20'	3'-4' max dia.[a]; <20'	3'-4' max dia.[a]; >20'	>4' max dia.[a]; <20'	>4' max dia.[a]; >20'	All Size Classes
WEST FORK BLUE CREEK									
Instream LWD	1.4	0.9	0.4	0.1	0.2	0.1	0.1	0.0	3.2
Riparian	1.7	4.6	0.5	0.8	0.1	0.1	0.0	0.0	7.8
Total	3.1	5.5	0.9	0.9	0.3	0.2	0.1	0.0	11.0

Table C2-11. Summary of 1994 and 1995 LWD sample (average pieces per 100 feet by channel type), Little River HPA.

Stream	Size Classes of In-channel LWD and Wood within Riparian Recruitment Zone						
	1'-2' max dia.[a]; <20'	1'-2' max dia.[a]; >20'	2'-3' max dia.[a]; <20'	2'-3' max dia.[a]; >20'	>3' max dia.[a]; <20'	>3' max dia.[a]; >20'	All Size Classes
LITTLE RIVER							
Instream LWD	1.2	0.9	0.5	0.4	0.3	0.2	3.5
RAILROAD							
Instream LWD	3.0	1.4	1.9	1.0	0.4	0.3	8.0
LOWER SOUTH FORK LITTLE RIVER							
Instream LWD	3.6	1.2	1.6	0.7	0.5	0.4	8.0
UPPER SOUTH FORK LITTLE RIVER							
Instream LWD	2.8	0.8	1.2	0.4	0.5	0.2	5.9
[a] = maximum diameter of LWD piece							

Table C2-12. Summary of 1994 and 1995 LWD sample (average pieces per 100 feet by channel type), North Fork Mad River HPA.

Stream	Size Classes of Inchannel LWD and Wood within Riparian Recruitment Zone								
	1'-2' max dia.[a]; <20'	1'-2' max dia.[a]; >20'	2'-3' max dia.[a]; <20'	2'-3' max dia.[a]; >20'	3'-4' max dia.[a]; <20'	3'-4' max dia.[a]; >20'	>4' max dia.[a]; <20'	>4' max dia.[a]; >20'	All Size Classes
NF MAD RIVER									
Instream LWD	0.2	0.3	0.1	0.1	0.1	0.0	0.2	0.0	1.0
Riparian	0.2	4.1	0.1	1.2	0.1	0.3	0.2	0.1	6.3
Total	0.4	4.4	0.2	1.3	0.2	0.3	0.4	0.1	7.3
LONG PRAIRIE CREEK									
Instream LWD	1.0	0.5	0.1	0.4	0.0	0.2	0.0	0.0	2.2
Riparian	1.5	6.2	0.1	1.5	0.0	0.5	0.0	0.1	9.9
Total	2.5	6.7	0.2	1.9	0.0	0.7	0.0	0.1	12.1
[a] = maximum diameter of LWD piece									

Table C2-13. Summary of 1994 and 1995 LWD sample (average pieces per 100 feet by channel type), Mad River HPA.

Stream	Size Classes of Inchannel LWD and Wood within Riparian Recruitment Zone								
	1'-2' max dia.[a]; <20'	1'-2' max dia.[a]; >20'	2'-3' max dia.[a]; <20'	2'-3' max dia.[a]; >20'	3'-4' max dia.[a]; <20'	3'-4' max dia.[a]; >20'	>4' max dia.[a]; <20'	>4' max dia.[a]; >20'	All Size Classes
LINDSAY									
Instream LWD	1.9	0.3	0.6	0.2	0.3	0.1	0.2	0.1	3.7
Riparian	0.4	4.1	0.1	1.6	0.1	0.6	0.2	0.6	7.7
Total	2.3	4.4	0.7	1.8	0.4	0.7	0.4	0.7	11.4
DRY CREEK									
Instream LWD	0.9	0.1	0.3	0.1	0.0	0.0	0.0	0.0	1.4
Riparian	0.6	3.2	0.1	1.1	0.2	0.2	0.7	0.1	6.2
Total	1.5	3.3	0.4	1.2	0.2	0.2	0.7	0.1	7.6
CAÑON CR.									
Instream LWD	0.6	0.6	0.2	0.1	0.1	0.0	0.2	0.0	1.8
Riparian	0.9	3.8	0.1	1.7	0.2	0.3	0.1	0.1	7.2
Total	1.5	4.4	0.3	1.8	0.3	0.3	0.3	0.1	9.0

[a] = maximum diameter of LWD piece

Table C2-14. Summary of 1994 and 1995 LWD sample (average pieces per 100 feet by channel type), Humboldt Bay HPA.

Stream	Size Classes of Inchannel LWD and Wood within Riparian Recruitment Zone								
	1'-2' max dia.[a]; <20'	1'-2' max dia.[a]; >20'	2'-3' max dia.[a]; <20'	2'-3' max dia.[a]; >20'	3'-4' max dia.[a]; <20'	3'-4' max dia.[a]; >20'	>4' max dia.[a]; <20'	>4' max dia.[a]; >20'	All Size Classes
SALMON CREEK									
Instream LWD	0.8	0.8	0.5	0.3	0.4	0.4	0.4	0.4	4.0
Riparian	0.5	4.1	0.3	1.0	0.2	0.4	0.4	0.2	7.1
Total	1.3	4.9	0.8	1.3	0.6	0.8	0.8	0.6	11.1

[a] = maximum diameter of LWD piece

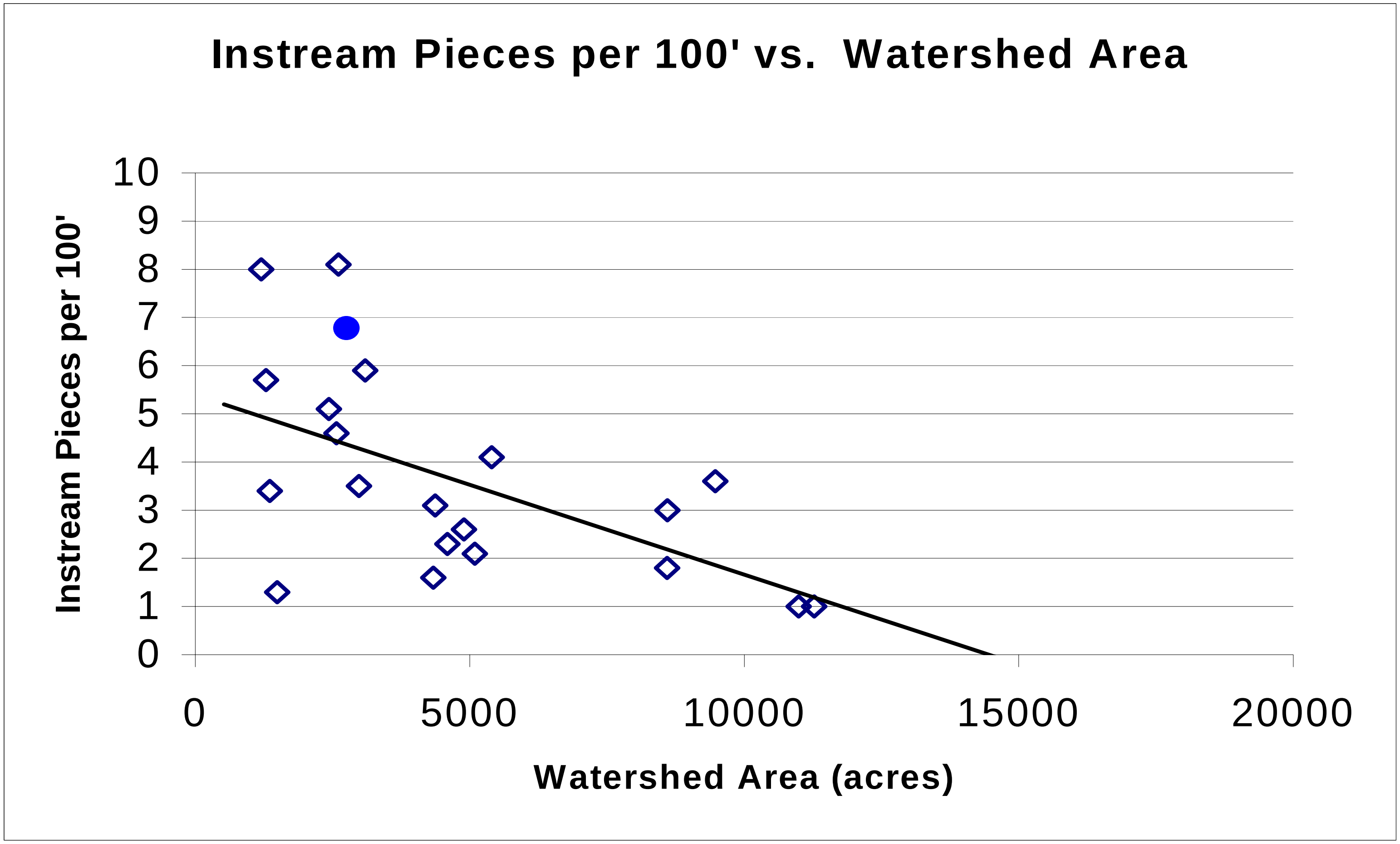

Figure C2-1. **Summary of mean number of instream LWD pieces per 100 feet of stream channel versus stream watershed area for 20 Plan Area streams. (Note: solid circle depicts Prairie Creek for reference.)**

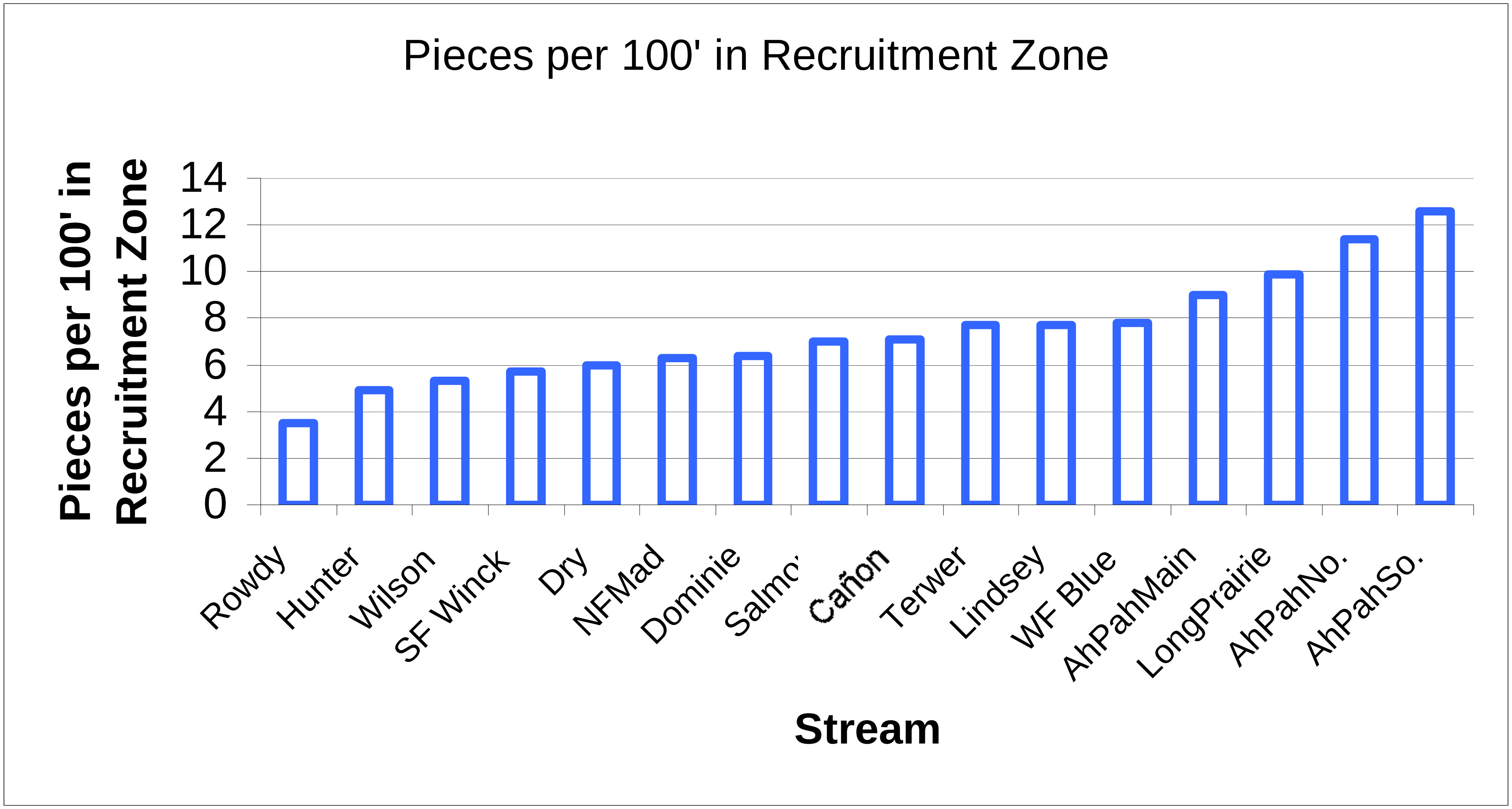

Figure C2-2. **Summary of the mean number of LWD pieces in the recruitment zone per 100 feet of stream channel for 16 Plan Area streams.**

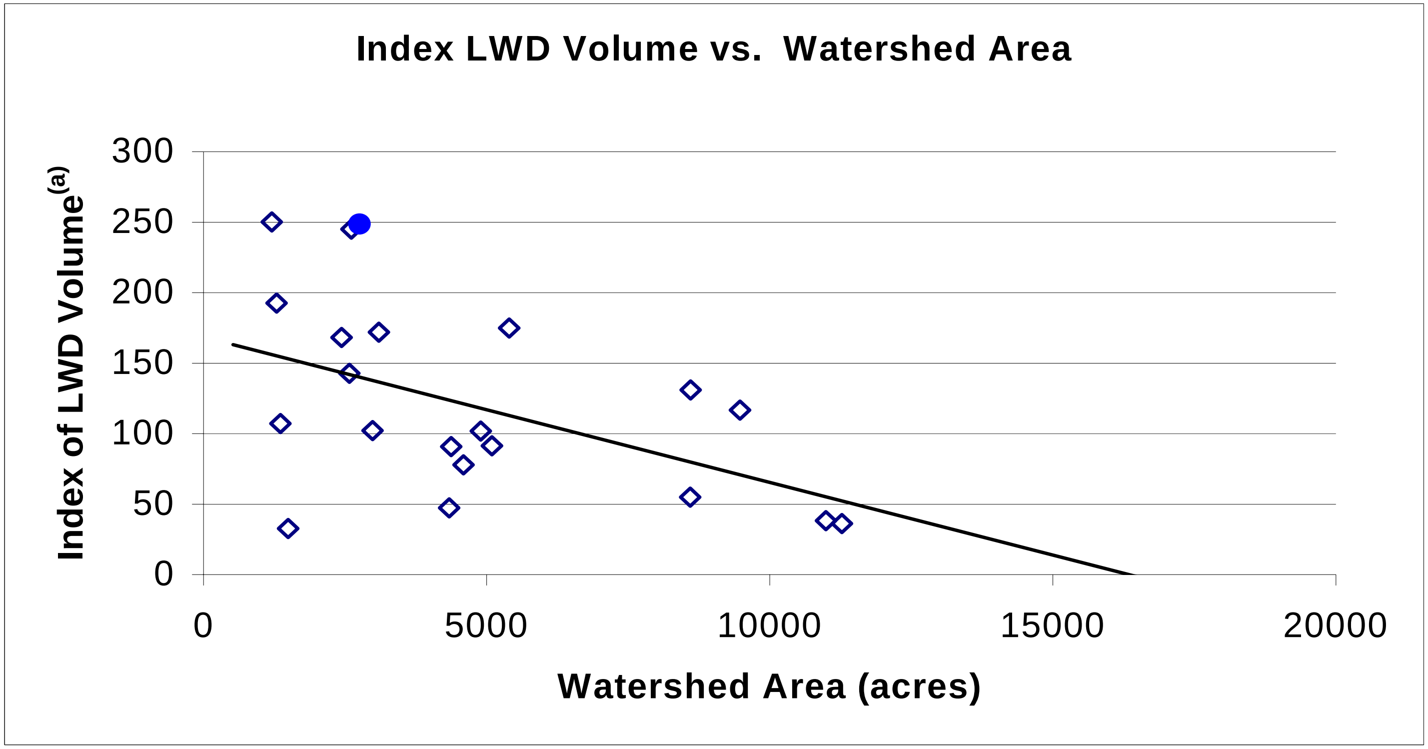

Figure C2-3. **LWD volume index versus watershed area for 20 Plan Area streams (Note: solid circle represents Prairie Creek for reference). (Index equals the maximum diameter times the mid-point of the LWD length class.)**

The results of the LWD surveys indicate that most streams surveyed had low amounts of inchannel LWD that consisted of the smallest size categories. Eleven of the sixteen streams with riparian surveys had low amounts of conifer abundance (relative to hardwoods) within the recruitment zone. These results support the conclusions drawn from the channel and habitat typing assessment: there are generally low levels of inchannel LWD available to function as shelter or to promote formation of pools in the surveyed streams. The dominant size class of inchannel LWD also parallels channel assessment descriptions of smaller diameter, alder dominated riparian zones with low numbers of large conifer (greater than 3' in diameter) as potential LWD.

As shown in Figure C2-3, an index of LWD volume for each stream surveyed was calculated and plotted against each stream's watershed area. Similar to the trend shown in Figure C2-1, (fewer pieces per 100 feet of channel with larger watershed areas) volume of LWD generally decreased with increases in watershed area (Figure C2-3).

C2.2.2 LWD Inventory Results

Results of Green Diamond's 1995 Inchannel LWD inventory are summarized in Tables C2-15 through C2-17. These tables summarize the 100% inchannel inventory displaying average pieces per 100 feet by Rosgen Channel Type and piece size category. The last two lines for each stream are the weighted average pieces per 100 feet of channel as determined by both the inventory and the 20% sample.

The results of the 1995 100% Inchannel LWD Inventory suggest that the 20% sample is comparable. CDFG's 20% sample is adequate for an estimate of average pieces per linear distance but does not address any volume or function related issues. The overall goal of the survey as designed by CDFG was to identify specific stream reaches that are in need of restoration in the form of additional LWD. To address the issues of total volume or inchannel function more detailed surveys will be needed.

C2.2.3 Prairie Creek LWD Inventory Results

The Prairie Creek inventory data is displayed in Table C2-18 as average pieces per 100 feet of channel in the various size categories. For a graphic comparison of the LWD data for Prairie Creek and the surveyed Plan Area streams, see Figures C2-1 and C2-3 above.

The section of Prairie Creek that was inventoried is a low gradient, small cobble dominated channel (Rosgen Channel Type of C4) that is considered to be a relatively undisturbed reach. Results of the Prairie Creek LWD data revealed that inchannel LWD occurred at an average of 6.8 pieces per 100 linear feet of channel for the 4.3 miles of channel inventoried (Kramer, pers. comm.)(Figure C2-1). This value exceeds all but two of the ranges calculated for any single average for the surveyed Plan Area streams (1.0 - 8.1 pieces/100'). Two tributaries in the Little River HPA, Lower South Fork and Railroad, had average piece counts at 8.1 and 8.0 pieces/100' respectively.

Table C2-15. Summary of 1995 100% in-channel LWD inventory (average pieces per 100 feet by channel type and size category), Smith River HPA.

Stream	Size Classes of In-channel LWD								
	1'-2' max dia.[a]; <20'	1'-2' max dia.[a]; >20'	2'-3' max dia.[a]; <20'	2'-3' max dia.[a]; >20'	3'-4' max dia.[a]; <20'	3'-4' max dia.[a]; >20'	>4' max dia.[a]; <20'	>4' max dia.[a]; >20'	All Size Classes
SOUTH FORK WINCHUCK RIVER									
C4	0.4	0.3	0.2	0.1	0.1	0.0	0.1	0.0	1.2
F4	0.7	0.8	0.2	0.1	0.0	0.0	0.1	0.0	1.9
C4	0.6	0.4	0.1	0.1	0.1	0.1	0.0	0.0	1.4
D3	2.3	0.4	0.0	0.0	0.2	0.2	0.0	0.0	3.2
A2	2.7	0.7	0.7	0.2	0.2	0.1	0.3	0.1	4.9
Weighted Average	0.8	0.4	0.2	0.1	0.1	0.0	0.1	0.0	1.7
20% sample	0.8	0.4	0.2	0.1	0.1	0.0	0.1	0.0	1.6
ROWDY CREEK									
D4	0.3	0.2	0.1	0.1	0.0	0.1	0.0	0.0	0.8
B3	0.1	0.1	0.1	0.0	0.0	0.0	0.1	0.0	0.4
B2	0.1	0.2	0.1	0.1	0.1	0.0	0.1	0.0	0.6
F3	0.9	0.4	0.2	0.2	0.0	0.1	0.1	0.0	2.0
Weighted Average	0.2	0.2	0.1	0.1	0.0	0.0	0.0	0.0	0.7
20% sample	0.2	0.2	0.1	0.0	0.2	0.1	0.1	0.0	0.9
DOMINIE CREEK									
F3	0.6	0.4	0.2	0.2	0.1	0.2	0.1	0.0	1.7
A3	2.6	1.0	1.3	0.7	0.4	0.5	0.1	0.1	6.6
F3	0.6	0.2	0.5	0.2	0.2	0.2	0.0	0.0	1.8
A2	2.6	0.4	0.9	0.3	0.4	0.2	0.2	0.1	5.0
Weighted Average	1.7	0.5	0.6	0.3	0.3	0.2	0.1	0.0	3.8
20% sample	1.7	0.3	0.5	0.4	0.2	0.2	0.1	0.0	3.4

[a] = maximum diameter of LWD piece

Table C2-16. Summary of 1995 100% in-channel LWD inventory (average pieces per 100 feet by channel type and size category), Coastal Klamath HPA.

Stream	Size Classes of Inchannel LWD								
	1'-2' max dia.[a]; <20'	1'-2' max dia.[a]; >20'	2'-3' max dia.[a]; <20'	2'-3' max dia.[a]; >20'	3'-4' max dia.[a]; <20'	3'-4' max dia.[a]; >20'	>4' max dia.[a]; <20'	>4' max dia.[a]; >20'	All Size Classes
AH PAH CREEK									
C4	1.7	0.7	0.4	0.2	0.1	0.1	0.2	0.1	3.4
D4	1.9	1.7	0.3	0.3	0.3	0.1	0.4	0.1	5.2
F3	2.4	0.3	0.5	0.3	0.1	0.1	0.1	0.1	4.0
A2	1.4	0.5	0.9	0.0	0.8	0.0	0.8	0.0	4.3
F4	2.5	0.3	1.3	0.0	0.6	0.1	0.3	0.1	5.2
A2	5.6	1.0	1.2	0.6	0.7	0.4	0.7	0.3	10.5
F3	3.1	0.3	1.4	0.3	0.9	0.1	0.7	0.1	6.9
Weighted Average	2.4	0.7	0.8	0.2	0.5	0.1	0.4	0.1	5.1
20% sample	2.0	0.7	0.8	0.3	0.3	0.1	0.2	0.2	4.6
NORTH FORK AH PAH CREEK									
F4	0.8	0.4	0.1	0.0	0.1	0.0	0.2	0.0	1.7
A2	3.7	0.6	0.7	0.2	0.2	0.1	0.3	0.1	5.9
B3	1.7	0.4	0.8	0.2	0.3	0.1	0.6	0.3	4.4
B2	1.5	0.9	1.3	0.1	0.2	0.1	0.4	0.3	4.9
A2	2.5	1.1	1.2	0.1	0.5	0.0	0.7	0.2	6.4
F4	2.0	0.5	0.8	0.1	0.3	0.1	0.5	0.1	4.4
Weighted Average	1.8	0.5	0.7	0.1	0.3	0.1	0.5	0.1	4.2
20% sample	2.1	0.7	1.0	0.2	0.2	0.1	0.5	0.2	5.1
SOUTH FORK AH PAH CREEK									
B4	1.2	0.6	0.5	0.3	0.1	0.2	0.2	0.1	3.1
A3	3.8	0.8	1.7	0.7	0.7	0.5	0.9	0.6	9.6
Weighted Average	2.4	0.7	1.0	0.5	0.3	0.3	0.5	0.3	6.1
20% sample	2.6	0.3	1.0	0.5	0.3	0.3	0.3	0.3	5.6

[a] = maximum diameter of LWD piece

Table C2-17. Summary of 1995 100% in-channel LWD inventory (average pieces per 100 feet by channel type and size category), Mad River HPA.

Stream	Size Classes of Inchannel LWD								
	1'-2' max dia.[a]; <20'	1'-2' max dia.[a]; >20'	2'-3' max dia.[a]; <20'	2'-3' max dia.[a]; >20'	3'-4' max dia.[a]; <20'	3'-4' max dia.[a]; >20'	>4' max dia.[a]; <20'	>4' max dia.[a]; >20'	All Size Classes
LINDSAY CREEK									
F5	1.8	0.4	0.5	0.2	0.2	0.1	0.1	0.1	3.4
20% sample	1.9	0.3	0.6	0.2	0.3	0.1	0.2	0.1	3.5

[a] = maximum diameter of LWD piece

Table C2-18. Summary of 1999 100% in-channel LWD inventory (average pieces per 100 feet by size category), Prairie Creek.

Stream	Size Classes of Inchannel LWD								
	1'-2' max dia.[a]; <20'	1'-2' max dia.[a]; >20'	2'-3' max dia.[a]; <20'	2'-3' max dia.[a]; >20'	3'-4' max dia.[a]; <20'	3'-4' max dia.[a]; >20'	>4' max dia.[a]; <20'	>4' max dia.[a]; >20'	All Size Classes
PRAIRIE CREEK									
	2.8	1.1	0.8	0.7	0.3	0.4	0.2	0.6	6.8

[a] = maximum diameter of LWD piece

Additionally, in five separate reaches within the Little River HPA and Salmon Creek, LWD tallies exceeded 6.8 pieces per 100 feet. When comparing the Prairie Creek results only to low gradient (<2%) stream reaches (Rosgen Channel Types C, D and F), five reaches in the surveyed Plan Area streams (three F3, one F4 and one C4 channel types) exceed the Prairie Creek values. These are Salmon Creek (16.3 pieces per 100') and Lower South Fork Little River (8.4, 9.4 and 10.9 pieces per 100') and Ah Pah Creek (7.0 pieces per 100'). In general, the surveyed Plan Area streams had, on average, more pieces per 100' in the higher gradient and more confined channel types. This intuitively makes sense; the smaller and steeper the stream the more likely it is for an individual LWD piece to be retained in the system.

In Prairie Creek the dominant category of inchannel LWD was in the 1' - 2' and less than 20' long" category (Table C2-18). This compares to the dominant, or co-dominant category of inchannel LWD for all but one of the surveyed Plan Area streams. The dominant inchannel category for the North Fork of the Mad River was the "1' to 2' and greater than 20' long". This difference can probably be attributed to the relatively larger size of the North Fork Mad River. In this stream an individual LWD piece less than 20 feet long would tend to be delivered through the system rather than be retained. The Prairie Creek results accurately reflect the LWD piece size for a relatively undisturbed coastal drainage. However, comparisons between Prairie Creek and many Plan Area streams may not be valid, because of differences in their morphology. Prairie Creek is a low–gradient alluvial channel in a relatively wide valley bottom, while many Plan Area streams are higher gradient in more incised channels.

Numerous factors influence the frequency, size, distribution and function of LWD including: geographic location, dominant tree species, channel width, channel gradient and drainage area. As a result, comparing LWD inventories from Green Diamond's California timberlands with data from undisturbed watersheds in other states could be inappropriate or misleading. LWD inventories from additional undisturbed watersheds including an inland, Douglas fir dominated forest, and a coastal redwood forest with steeper channel gradients than those found in Lower Prairie Creek would aid in the analysis of the existing LWD results, as these conditions are common on Green Diamond timberlands. Inventories on undisturbed watersheds of varying drainage area and channel gradient would also aid in differentiating between the many factors that influence LWD distributions

C2.3 DISCUSSION

The LWD survey results reflect the effects of past timber management practices and early habitat improvement efforts. Throughout the surveyed Plan Area streams, there were generally low amounts of LWD; and the predominate size of the existing LWD was small (primarily 1'-2' diameter pieces). The lack of large pieces of LWD (> 4' diameter and > 20' long) suggests that surveyed stream channels have been subjected to extensive channel clearing as part of past timber harvesting practices and/or early habitat improvement efforts. The relative lack of large live trees (conifers with > 4' diameters) within the recruitment zone reflects the effects of pre-FPRs management practices that removed most merchantable conifers from riparian zones adjacent to stream channels and failed to re-establish conifers in these areas. As a result, most riparian zones in sampled watersheds tend to be dominated by alder, willow, and younger conifers.

Comparisons of logged and unlogged streams or reaches provide insights into management impacts on LWD loading, recruitment rate and downstream transport. Numerous studies have compared LWD in old growth, mature second growth and recently clear-cut watersheds in Alaska, British Columbia, Washington and Oregon (Sullivan et al. 1987; Bibly and Ward 1989, 1991; Murphy and Koski 1989; Ralph et al. 1994; McHenry et al. 1998). Some studies indicated that LWD frequency was reduced in managed watersheds (Bilby and Ward 1991, McHenry et. al. 1998) and others failed to prove or detect a difference in piece counts (Ralph et al. 1994). However, every study confirmed a statistically significant reduction in sizes of LWD pieces in managed watersheds, suggesting that size and volume of LWD pieces are more important than frequency of pieces in forming and maintaining complex habitat features.

The LWD structures placed by restoration groups are often undersized (mainly in length as opposed to maximum width) for several reasons, including: 1) monetary limits per structure as required by CDFG-administered restoration funds, 2) size constraints by the cull logs available at or near a work site or donated by timber companies, and/or 3) size constraints of cull logs that restoration groups can maneuver with their equipment. Most restoration projects have also failed to mimic natural conditions, tending to locate LWD structures along channel margins with minimal amounts of wood lying within the main channel, and rarely, if ever, fully spanning the channel with large conifer.

Comparing the results of the Prairie Creek inventory with the inventories for the surveyed Plan Area streams suggests that the occurrence of larger in-channel pieces is lower in managed streams within the Plan Area than in unmanaged streams nearby. Several of the surveyed Plan Area streams had average overall piece counts per 100' within specific size categories that approached or exceeded the values seen in Prairie Creek. However, the piece lengths in these managed streams were shorter than the piece lengths in Prairie Creek, especially in similar channel types. In the 20 surveyed Plan Area streams, most of the larger diameter LWD was either: 1) old-growth root wads with little or no bole attached to them, or 2) instream restoration projects consisting of short, stubby pieces of cull logs anchored to bedrock, boulders, or riparian trees. Both of these types of LWD often provide marginal habitat compared to intact trees recruited from the riparian zone. Old-growth redwood rootwads contain fairly large volumes of wood, yet their short length provides minimal surface area for capturing and retaining additional LWD to form complex salmonid habitat. The short length of these rootwads also increases their likelihood of mobilizing during moderate storm events (as occurred during the winters of 1995-96 and 1996-97).

C2.4 CONCLUSION

LWD within Plan Area streams will be reassessed periodically during the 50-year life of the Plan with the objective of documenting increases in conifer piece frequency, size, and functionality. Improvements in the current LWD inventories and sampling designs are needed to more accurately assess the changes in volume and function of LWD debris over longer periods of time. Conditions can be expected to gradually improve as a result of current FPRs and the increased riparian standards implemented under the Plan. The hardwood dominated riparian zones now prevalent on various Plan Area streams will eventually be succeeded by redwoods and other conifers, resulting in increasing recruitment of large diameter LWD for Plan Area streams. It has been suggested (McHenry et al. 1998, Emminghamm and Hibb 1996) that without active management of riparian zones; protection of existing conifers, conifer release and/or planting that conifer succession will be extremely slow or even effectively precluded.

C2.5 REFERENCES

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Appendix C3. Long-Term Channel Monitoring

CONTENTS

Figures

Tables

C3.1 BACKGROUND

Green Diamond implemented the initial long-term monitoring program of its California watersheds in 1993. The first two years of the monitoring program was based on two U.S. Forest Service publications which address monitoring strategies of both instream and riparian conditions (Platts et al. 1983; Platts et al. 1987). At the conception of this early monitoring study, the selection of watersheds was primarily influenced by the concerns of the Regional Water Quality Control Board and the CDFG regarding possible cumulative effects of Green Diamond's activities in several basins. The primary watersheds of concern were Salmon Creek and Jacoby Creek, both tributaries to Humboldt Bay. The Salmon Creek watershed was of concern due to its highly unstable and erosive geology (Wildcat Formation) and past management practices. The Jacoby Creek watershed has sections of erosive Franciscan Formations, a diverse mix of ownership and a complex history of watershed disturbances (logging, grazing and residential development). Additional watersheds were selected to distribute the monitoring across the ownership.

The next step in designing the early monitoring program was the selection of sample stream sections within watersheds. Two approaches were utilized in selecting sampling sections:

- Paired reference (control) and test (treatment) sections; and
- A general watershed approach.

When employing the paired reference and test sections, the sections were selected on the basis of their location relative to a potential impact from a management activity (e.g., sedimentation from a timber harvest). Sections established upstream from the activity site were the reference sections and those downstream were the test sections. The data collected from the reference and test sections were compared to evaluate potential impacts. However, to make data comparable, sections above and below the management activity must be selected from stream reaches that matched according to valley bottom and riverine habitat types. Once similar stream reaches were selected, each reach was divided into 300-foot sections from which two 300-foot sections were randomly selected. A minimum of two reference and two test sections were identified for each of Green Diamond's anticipated management activities within a watershed.

Because the location of potential impacts within a watershed cannot always be identified in advance, a general watershed approach must occasionally be utilized. With this approach, the 300-foot stream sections were randomly selected throughout a watershed without identifying them as either reference and test sections. Statistically, a minimum of five to eight sections were sampled, depending on the complexity of the watershed, to insure that suitable reference and test sections would be available following future timber harvest activities. Sampling was conducted following the protocol established by Platts et al. (1983 and 1987).

These pilot projects provided valuable information regarding effective methods and response variables, and the difficulties of analyzing the resulting data. Using the information gathered in these pilot studies, a revised methodology was developed and first implemented in Cañon Creek beginning in 1995.

To fine tune the long-term monitoring methodology, Green Diamond consulted with William Trush, a watershed scientist from Humboldt State University. Trush reviewed the channel monitoring program and suggested modifying the program to reduce data collection time and improve the ability to detect changes in channel response. His review indicated that:

- Most variables measured were flow dependant and generated significant differences in channel conditions with slight changes in base summer flow;
- The systematic selection of monitoring cross sections at ten foot intervals ignored geomorphic characteristics of certain channel features and processes; and
- Flow dependant variables resulted in significant differences regardless of management activities, while systematically selected monitoring cross sections created high variance estimates.

These comments assisted Green Diamond in revising its selection of stream reaches to capture specific channel responses to significant hydrologic events (and possibly management activities) and measuring only variables that were independent of flow. This protocol was implemented on Cañon Creek (a Mad River tributary) in 1995. During 1996, Green Diamond field personnel again monitored the Cañon Creek site and established additional channel monitoring reaches on the South Fork Winchuck River (a tributary in Smith hydrographic unit), Hunter Creek (a lower Klamath River tributary), and Salmon Creek (a Humboldt Bay tributary). These surveys have continued with scheduled re-surveys every two years or after a five year flood event. Data collected on all of the monitoring sites since 1998 are scheduled for analysis in 2003. Each monitoring reach should have at least 3 years of data prior to the first analysis and updated biennially to coincide with the biennial report to the Services (see Section 6 regarding report). The purpose of that monitoring protocol was to document the recovery of Plan Area watersheds from past timber harvesting practices and to evaluate the effects of current and future harvesting practices on watershed condition and recovery. The long-term channel monitoring protocol also has potential to evaluate the effectiveness of “storm-proofing” techniques, currently in vogue, in reducing road-related erosion sources.

C3.2 METHODOLOGY

In early 1998, Green Diamond hired a statistical consultant (Trent McDonald) to assist in refining and developing methods to analyze the long-term channel monitoring data. The consultant confirmed that the data being collected was valid and rendered itself to analysis. Using the previous developed monitoring data collection methods the results were analyzed as described below.

The monitoring objective of the Class I channel monitoring project was to track long term trends in the sediment budget of Class I watercourses as evidenced by changes in channel dimensions. Initially 3 and later 9 monitoring reaches were established in 8 streams across the Plan Area. Two additional reaches were also established with a reduced protocol (thalwag profile only), because the sites did not meet the criteria necessary for doing the full protocol. The initial three streams: Cañon, Hunter, and Canyon creeks were chosen for monitoring and analysis. A section of each creek was selected for monitoring activities and field sampling was carried out on those reaches using Green Diamond's monitoring protocols as described above. Monitored sections were chosen to be the highest (closest to headwaters) depositional reach in each creek. Depositional reaches were characterized by relatively low gradient where sediment was expected to be deposited. The reasoning behind establishment of these monitoring reaches was that if changes in sediment load or other stream morphology parameters occurred anywhere in the watershed, such changes were likely to be reflected in the first depositional reach downstream. The three stream systems under study were small enough that there was only one depositional reach contained in each stream.

Three creeks in the Plan Area (Cañon Creek, Hunter Creek, and Canyon Creek) were chosen for monitoring and analysis. A section of each creek was chosen for monitoring activities and field sampling was carried out on those reaches under Green Diamond protocol. Monitored sections were chosen to be the highest (closest to headwaters) depositional reach in each creek. Depositional reaches were characterized by relatively low gradient where sediment was expected to be deposited. The reasoning behind establishment of these monitoring reaches was that if changes in sediment load or other stream morphology parameters occurred anywhere in the watershed, such changes were likely to be reflected in the first depositional reach downstream. The three stream systems under study were small enough that there was only one depositional reach contained in each stream.

Sampling occurred at Cañon Creek in 1995, 1996, and 1997. Sampling occurred in 1996 and 1997 at the other two creeks (Hunter and Canyon). Each year, thalweg elevation (defined as the height of the deepest part of the channel), bank full width, active channel width, and substrate (pebble) sizes were recorded on the monitoring reaches. Thalweg elevation residuals (see below) were analyzed for changes in variance. A change in thalweg residual variance indicates an improvement (or degradation) of pools via changes in pool depth. Bank full and active channel widths were analyzed for changes in average width. Substrate sizes were analyzed for changes in distribution.

C3.2.1 Analysis of the Thalweg

Thalweg elevation was analyzed for change in mean elevation and thalweg residuals (from a spatial polynomial regression of elevation on distance from the upper end of the reach) were analyzed for change in variance. Both sets of analyses used statistical models appropriate for correlated data. The basic data were pairs of points, (d_i, y_i), where y_i was thalweg elevation and d_i was the distance from the upper terminus of the reach to the point where y_i was measured. Because thalweg elevations were measured relatively close together (approximately every 10 feet) the measurements (i.e., the y_i) were potentially spatially correlated and did not represent independent observations. Therefore, the analyses accounted for this lack of independence by adjusting model coefficients and significance levels using a one dimensional spatial regression model (Cressie 1991; Venables and Ripley 1994). The spatial regression model estimated a one dimensional correlation function among residuals then adjusted estimates and p-values via generalized least squares regression techniques. The spatial regression techniques and the adjustment for auto-correlation is described in more detail in Attachment C3-A.

For the analysis of thalweg elevation, a regression model relating elevation of the thalweg to a cubic polynomial in distance was estimated. Included in this model was a year factor so that the interaction between year and the cubic polynomial in distance could also be estimated. In equation form and provided the reach will be monitored for three years, the regression relationship was:

$$
\begin{aligned}
E[y_i] = {} & \beta_0 + \beta_1 x_{1,i} + \beta_2 x_{2,i} \\
& + \beta_3 d_i + \beta_4 d_i^2 + \beta_5 d_i^3 \\
& + \beta_6 d_i x_{1,i} + \beta_7 d_i^2 x_{1,i} + \beta_8 d_i^3 x_{1,i} \\
& + \beta_9 d_i x_{2,i} + \beta_{10} d_i^2 x_{2,i} + \beta_{11} d_i^3 x_{2,i}
\end{aligned}
$$

where y_i was thalwag elevation measured at a distance of d_i meters from the top of the reach, $x_{1,i}$ was an indicator variable for year 1 (i.e., 1 if observation $_i$ was taken in year 1, 0 otherwise), and $x_{2,i}$ was an indicator variable for year 2 (i.e., 1 if observation $_i$ was taken in year 2, 0 otherwise). For reaches which were monitored only two years, $x_{2,i}$ and all interactions involving it were eliminated from the model (i.e., β_2, β_9, β_{10}, and β_{11} were not present in the model). These models effectively fit separate cubic polynomials in d_i each year.

The analysis for change in thalweg residual variance was a statistical test designed to detect increased (or decreased) variance in residuals which is indicative of increased (or decreased) pool depths and complexity of the reach habitat. Thalweg residuals were defined as the residuals of thalweg elevation in the above regression model; $r_{yi} = y_{yi} - \hat{y}_{yi}$, where y_{yi} was observed elevation at distance d_i in year y and $\hat{y}_{yi}$ was the predicted elevation at distance d_i in year y. The test for change in thalweg residual variance was carried out using a modified version of Levene's test (Neter et al. 1991). Absolute deviations of the residuals from their median were calculated as $d_{yi} = |r_{yi} - m_y|$, where d_{yi} was the absolute deviation associated with the i-th observation in the y-th year and m_y was the median of residuals in the y-th year. Levene's test entailed carrying out a one-way analysis of variance on the d_{yi}, with year defining the groups. Because the *ryi* were potentially (spatially) correlated, the d_{yi} were also potentially correlated and the one-way analysis of variance was adjusted using the spatial regression techniques outlined in Attachment C3-A. Variance of the original residuals was deemed significantly different across years if the (spatially adjusted) one-way analysis of variance rejected the hypothesis of equal average deviations. The distribution of thalweg residuals was also plotted as a visual interpretation aid.

C3.2.2 Analysis of Width

Both bank full and active channel widths were analyzed for changes across years. To conduct this analysis, a systematic sample of widths was computed from available data after field sampling was complete. Such a systematic sample of widths was necessary because field-sampling protocol dictated that each bank of the creek is measured separately. Consequently, width measurements were not taken completely across the creek, but rather from each bank to a center tape. Furthermore, measurements from one bank to the center tape were not necessarily in the same place as measurements to the opposite bank. Therefore width could not be computed directly from the raw data and consequently a systematic sample of widths was computed and analyzed by the following methods. The systematic sample of widths was computed by first connecting left and right bank width measurements with straight lines to form an approximate stream channel. A random starting point along the center tape was then chosen and widths (across the whole channel) were computed at regular intervals along the center tape. The number of systematic points in the sample was equal to the smaller of the two sample sizes taken on each bank. For example, if 50 measurements were taken on the left bank and 75 measurements were taken on the right bank, 50 systematic measurements of width were taken to analyze. A picture of the systematic sample of widths computed at Cañon Creek in 1996 is presented in Figure C3-1 below.

The systematic sample of widths was computed each year for each creek. Average width was analyzed using one-way analysis of variance (anova) techniques analogous to the modified Levene's test described for analysis of thalweg residual. A one-way analysis of variance (two sample t-test if only two years) was computed, with year as the grouping factor, to test for changes in mean stream width. Because measurements in the field were taken relatively close together and because spacing of the systematic sample of widths was relatively tight, computed widths were potentially correlated and consequently the analysis of variance was modified to adjust for spatial correlations using the techniques outlined in Attachment C3-A. This analysis of variance was parallel to the modified Levene's test described for analysis of thalweg residual variance.

C3.2.3 Analysis of Substrate Size

Substrate size, or pebble size, was measured at between 5 and 10 sites within each monitored reach. Each site was approximately 50 feet by 50 feet in size and consisted of sand bars, lee banks, and other rocky areas in the stream. At each site, field personnel measured the secondary axis of rocks (pebbles) which were collected by selecting one near the toe of their right foot as transects were walked around the site. Collection and measurement continued until 150 rocks were measured. All measurements were reported in millimeters and the smallest measurement was one millimeter.

The distribution of pebble size was plotted and analyzed for changes across years assuming independence of the measurements. Due to the large distances (relative to average pebble size) at which rocks were measured and the fact that several independent systematic samples were taken at each site, spatial correlations among observations were highly unlikely and consequently no adjustments for such correlation were made. The hypothesis of no change in distribution was tested using two sample Wilcoxon rank sum tests (Wilcoxon 1945, Hollander and Wolf 1973) or three sample Kruskal-Wallis tests (Lehmann 1975; Hollander and Wolf 1979), depending on the number of years data were collected from a stream. Substrate size measurements from all sites within a year were combined for testing because site to site differences in substrate size were not of interest and, if such differences existed, would tend to inflate the distribution's variance and provide a conservative analysis. Treating the systematic measurements as if they were purely random (i.e., by assuming independence) also inflates the distribution's variance and further contributes to a conservative analysis.

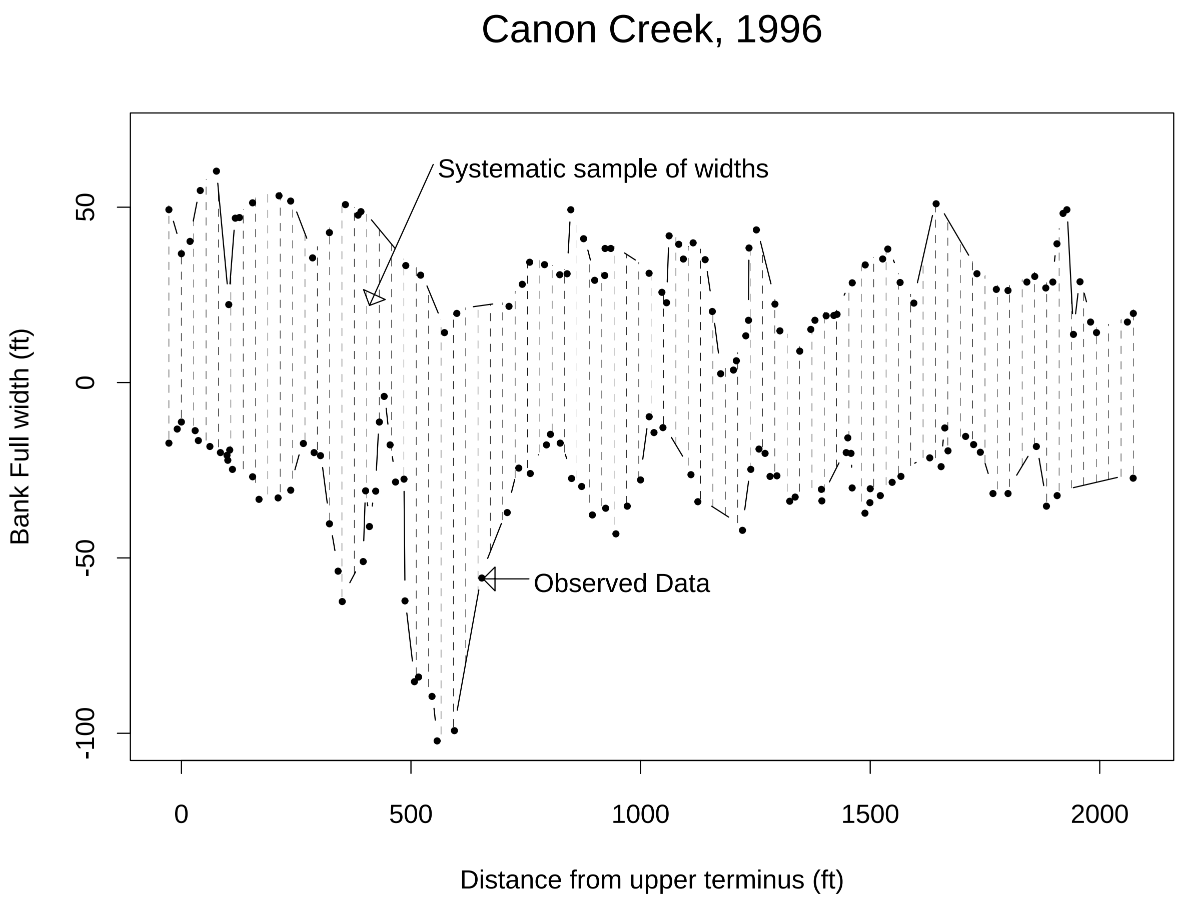

Figure C3-1. Diagram of the systematic sample of widths taken for the investigation of width (Cañon Creek 1996). This example shows bank full width at Cañon Creek in 1996. Zero in vertical dimension represents the center tape while negative numbers represent the left bank and positive numbers represent the right. Dots are observed bank full measurements with linear interpolation between each. Dashed lines show the systematic sample of widths.

Three quantiles from each substrate distribution were estimated. The 16-th, 50-th, and 84-th quantiles were estimated from each distribution to facilitate comparison with sediment movement models developed elsewhere (USEPA 2000). The 16-th quantile was defined as that point in the distribution that was greater than 16% of the observations and less than 84% of the observations. By symmetry, the 84-th quantile was defined as that point in the distribution that was greater than 84% of the observations and less than 16% of the observations. The 50-th quantile was defined similarly and corresponded to the median. The standard error of each quantile was estimated using standard bootstrap methods (Manly 1997).

C3.3 RESULTS

C3.3.1 Analysis of the Thalweg

At Cañon Creek, thalweg elevation measurements were significantly correlated with other thalweg elevations measured nearby. Correlation of thalweg residuals (i.e., residuals computed from the initial regression) within 8 feet of one another was 0.52 in 1995 (95% confidence interval 0.21 - 0.83), 0.81 in 1996 (95% confidence interval = 0.46 - 1.0), and 0.73 in 1997 (95% confidence interval = 0.52 - 0.95).

A graph of the final spatial regression model for Cañon Creek appears in Figure C3-2. There was a significant difference in overall curvature of the thalweg profile at Cañon Creek between 1995 and later years ($p<0.0001$ for 1995 vs. 1996; $p<0.0001$ for 1995 vs. 1997). The overall curvature of the thalweg profile was negative in 1995 while in 1996 and 1997 curvature was positive. Inspection of Figure C3-2 shows that the middle half (approximately) of the Cañon Creek monitoring reach remained at roughly the same elevation in all three years, but that the upper and lower quarters (approximately) were lower in 1995 and than in 1996 and 1997. No significant differences existed in the linear or cubic trends between 1995, 1996, and 1997. No significant differences existed in overall thalweg trend between 1996 and 1997 ($p=0.29$ for linear trend, $p=0.37$ for quadratic trend, $p=0.77$ for cubic trend).

Thalweg elevation measurements in Hunter Creek were significantly correlated with similar measurements taken nearby. Correlation of thalweg residuals within 8 feet of one another was 0.44 in 1996 (95% confidence interval 0.11 - 0.78), and 0.98 in 1997 (95% confidence interval 0.64 - 1.0).

A graph of the final spatial regression model for Hunter Creek appears in Figure C3-3. A marginally significant difference existed in the coefficient of the cubic trend term between 1996 and 1997 at Hunter Creek ($p=0.072$). This difference in third order trend, if deemed significant, was caused by a drop in thalweg elevation from 1996 to 1997 near the bottom third of the monitoring reach, between 1500 and 2200 feet from the upper terminus of the reach.

Thalweg elevation measurements in Canyon Creek were significantly correlated with similar measurements taken nearby. Correlation of thalweg residuals in Canyon Creek within 8 feet of one another was 0.69 in 1996 (95% confidence interval = 0.42 - 0.97), and 0.65 in 1997 (95% confidence interval = 0.43 - 0.87).

A graph of the final spatial regression model for Canyon Creek appears in Figure C3-4. No significant differences occurred in overall thalweg elevation in Canyon Creek between 1996 and 1997 (p= 0.36 for year*linear term, p=0.78 for year*quadratic term, p=0.10 for year*cubic term). Because yearly interaction was not significant, interaction was dropped from the final regression at Canyon Creek and consequently the lines in Figure C3-4 were forced to be exactly parallel. There was no difference in the parallel lines of Figure C3-4 (p=0.67).

The distributions of thalweg residual for Cañon, Hunter, and Canyon creeks appear in Figure C3-5, Figure C3-6 and Figure C3-7. In addition to standard histograms, these figures display a (Gaussian) kernel smooth density estimate for each distribution. Absolute deviations from the median, used in Levene's test, measured near one another were significantly correlated in every creek every year.

Table C3-1 contains estimates and confidence intervals for correlation between absolute deviations within 8 feet of one another. After adjustment for spatial correlation using the method outlined in Attachment C3-A, there remained a significant decrease in thalweg residual variance at Cañon creek between 1995 and latter years (p=0.0019 for 1995 vs. 1996; p=0.0013 for 1995 vs 1997).

Inspection of the histograms in Figure C3-5 confirm that there were more large negative thalweg residuals in 1995 than there were in 1996 and 1997. There was no significant difference in thalweg residual variance between 1996 and 1997 at Cañon Creek (p=0.5379). Thalweg residuals at Hunter and Canyon creeks displayed changes similar to those at Cañon Creek. Variance of thalweg residuals was higher in 1996 than 1997 at both Hunter and Canyon creeks (p=0.0465 for Hunter, p=0.0365 for Canyon). Inspection of Figure C3-6 and Figure C3-7 confirm that there were more large negative residuals in 1996 than in 1997 at both creeks.

Table C3-1. Estimated correlations among absolute thalweg residual deviations from the median measured less than 8 feet apart.

Creek	Year	Estimated Correlation	Approximate 95% confidence interval	
			Low	High
Cañon	1995	0.50	0.19	0.81
	1996	0.83	0.49	1.00
	1997	0.70	0.49	0.91
Hunter	1996	0.38	0.05	0.72
	1997	0.89	0.55	1.0
Canyon	1996	0.70	0.42	0.97
	1997	0.60	0.38	0.82

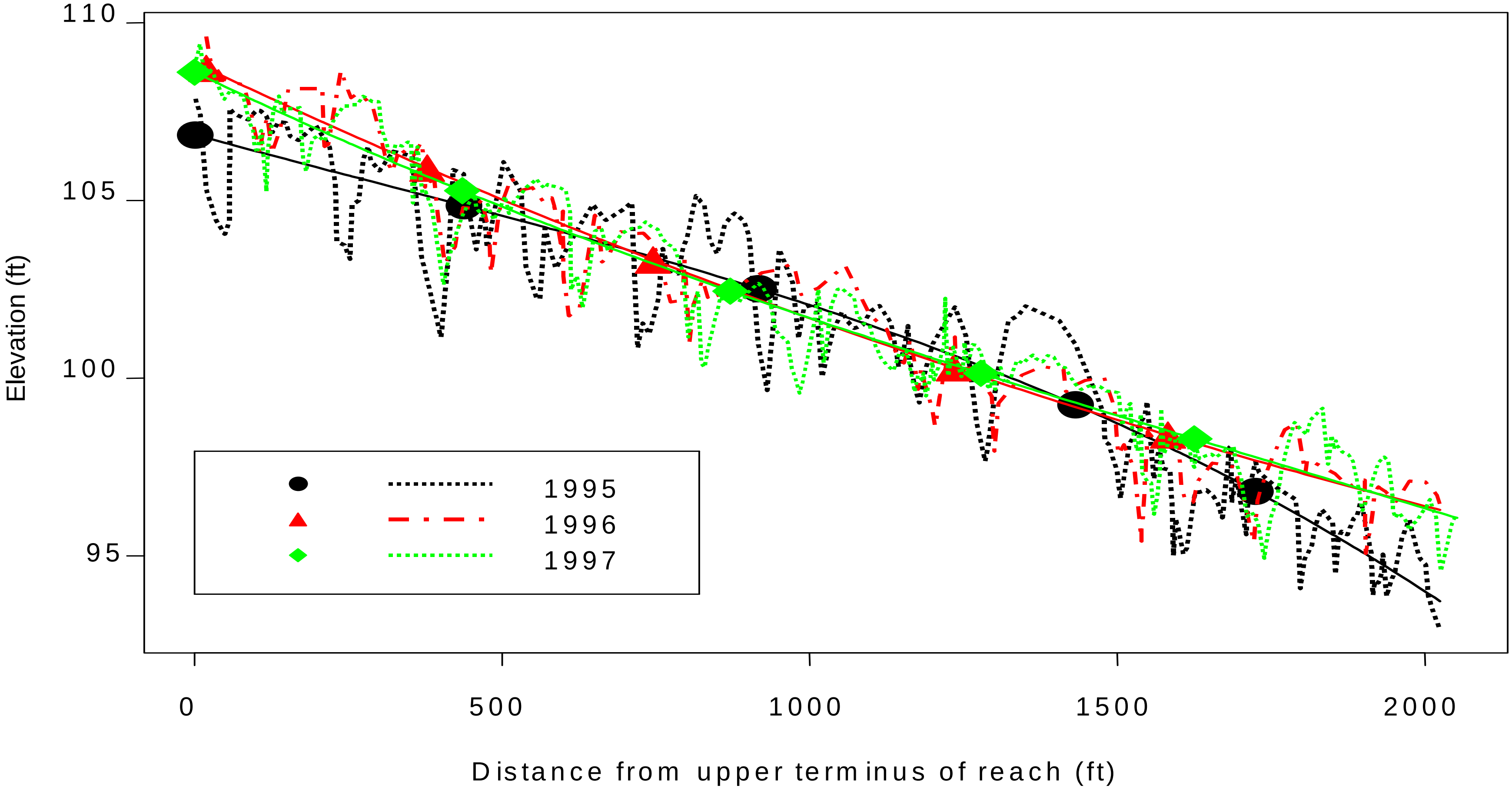

Figure C3-2. **Thalweg elevation profile for the Cañon Creek monitoring reach, 1995, 1996, and 1997. Dashed lines show measured elevations. Solid lines show trend estimated by spatial regression that adjusted for auto-correlation in residuals. Curvature (2nd derivative) was negative in 1995, positive in 1996 and 1997.**

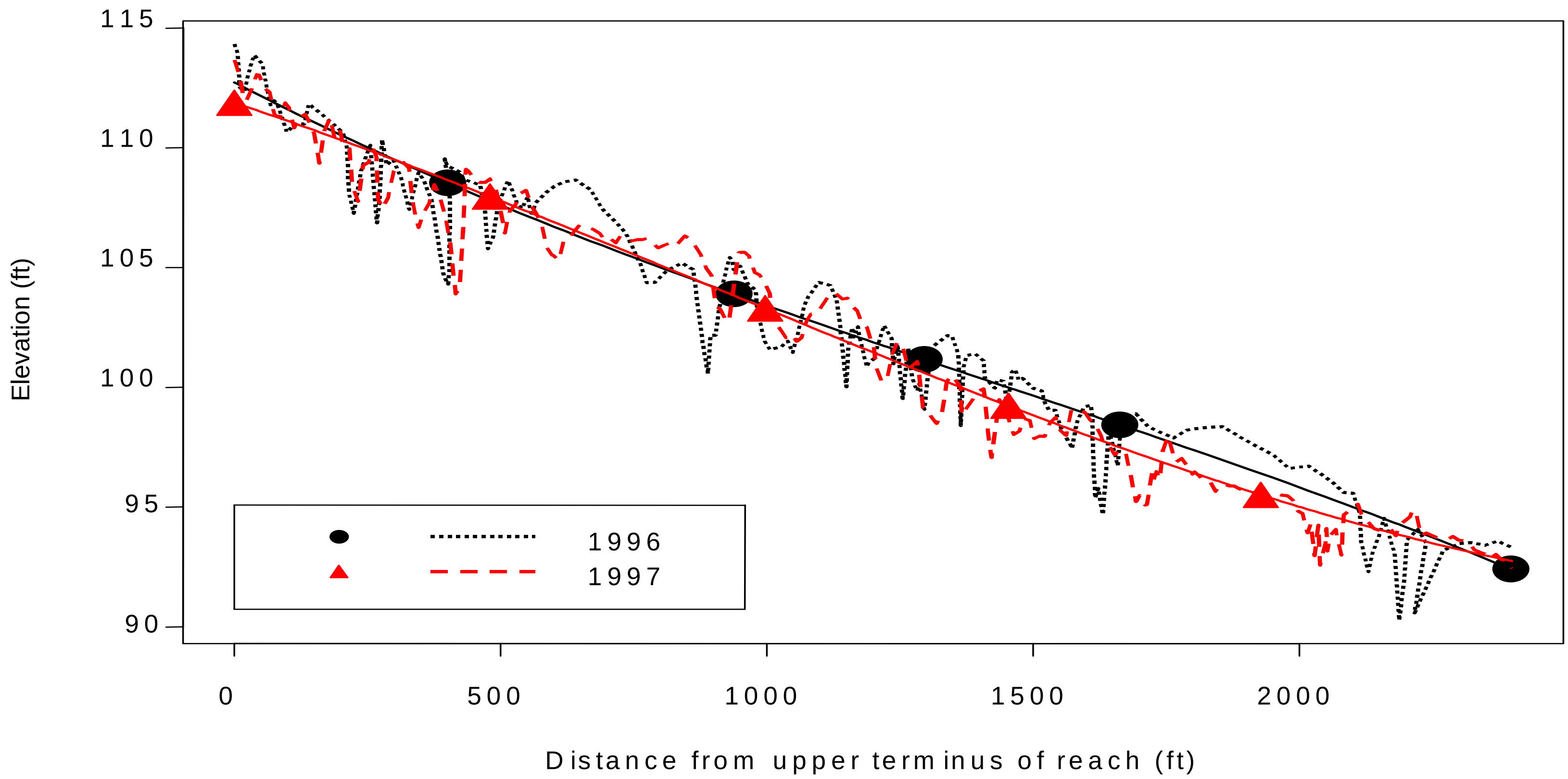

Figure C3-3. Thalweg elevation profile for the Hunter Creek monitoring reach in 1996 and 1997. Dashed lines show measured elevations. Solid lines show trend estimated by spatial regression that adjusted for auto-correlation in residuals.

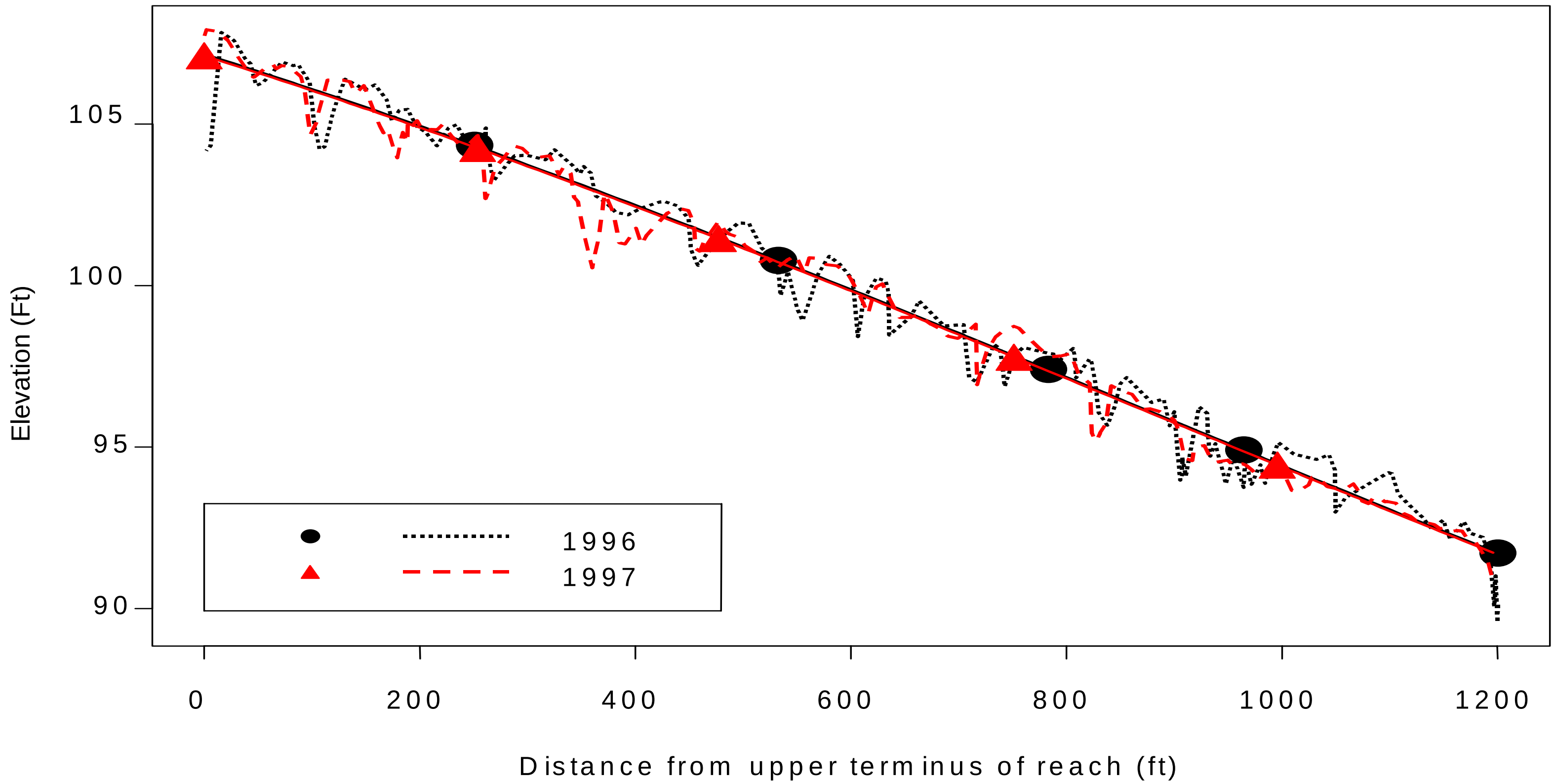

Figure C3-4. **Thalweg elevation profile for the Canyon Creek monitoring reach in 1996 and 1997. Dashed lines show measured elevations. Solid lines show trend estimated by spatial regression that adjusted for auto-correlation in residuals.**

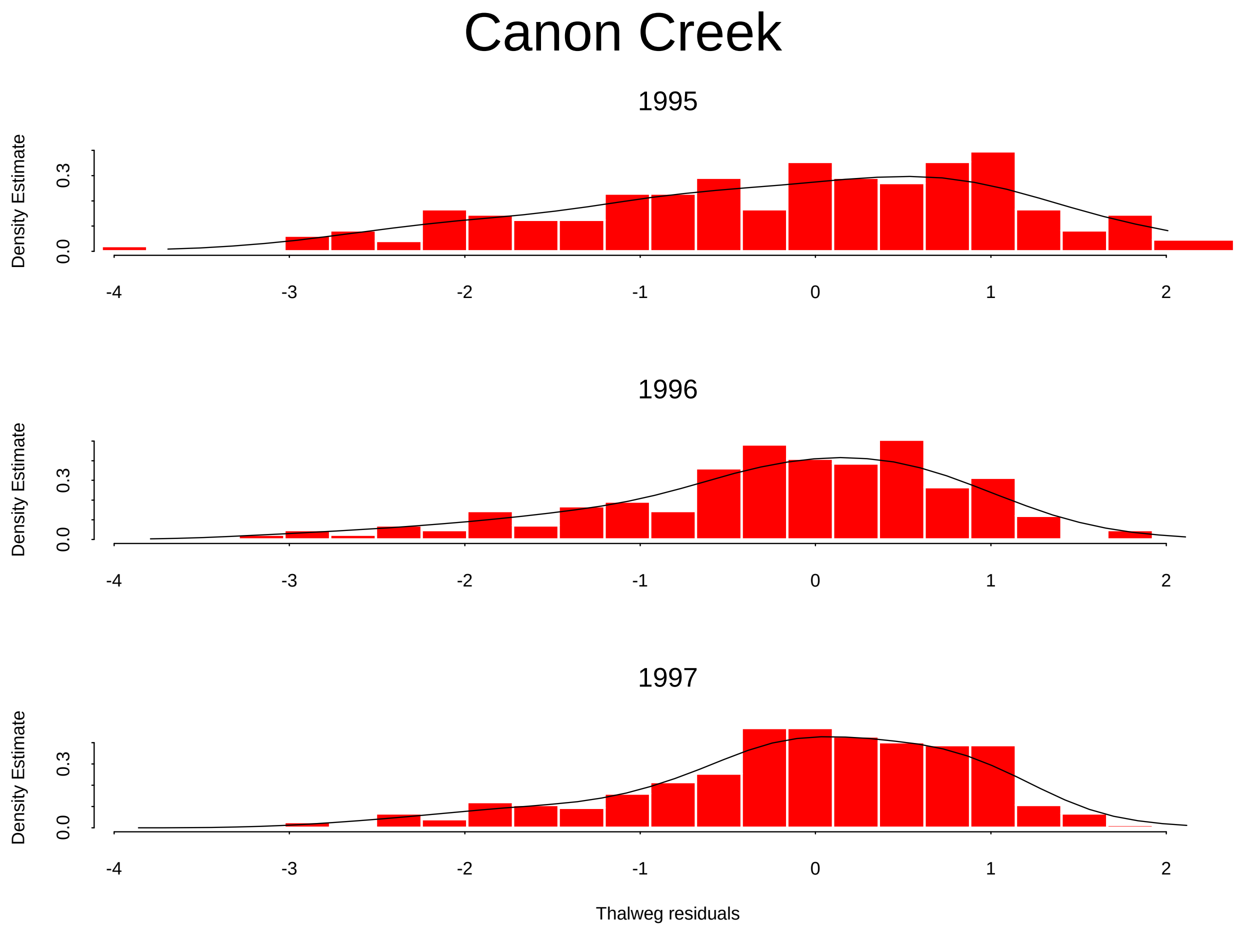

Figure C3-5. **Histograms of thalweg residuals at Cañon Creek, 1995 through 1997, used to compare variance of residuals among years. Residuals computed using models fit in Figure C3-1. Solid line is Gaussian kernel smoothed density estimate.**

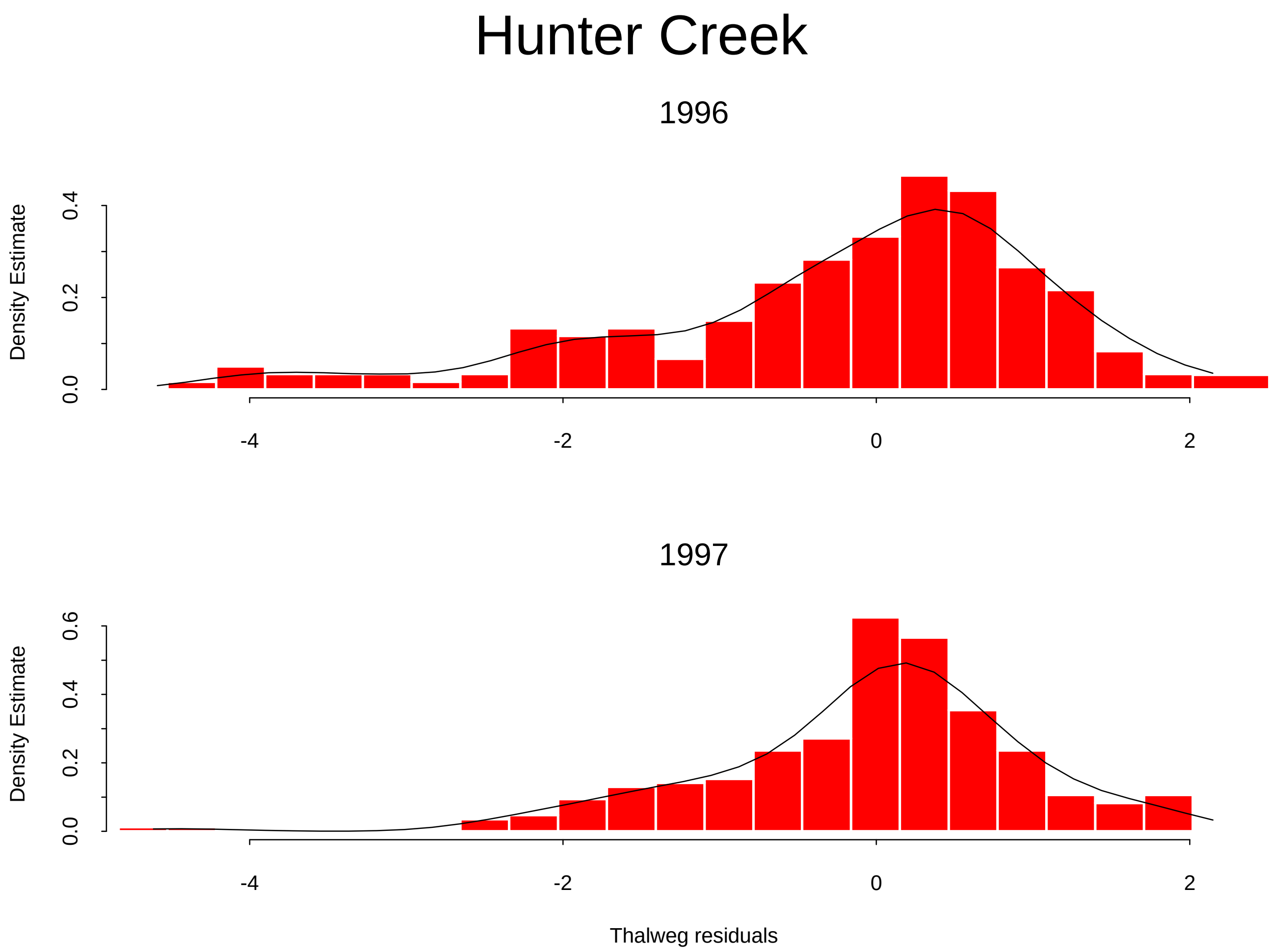

Figure C3-6. **Histograms of thalweg residuals at Hunter Creek, 1996 and 1997, used to compare variance of residuals among years. Residuals computed using models fit in Figure C3-2. Solid line is Gaussian kernel smoothed density estimate.**

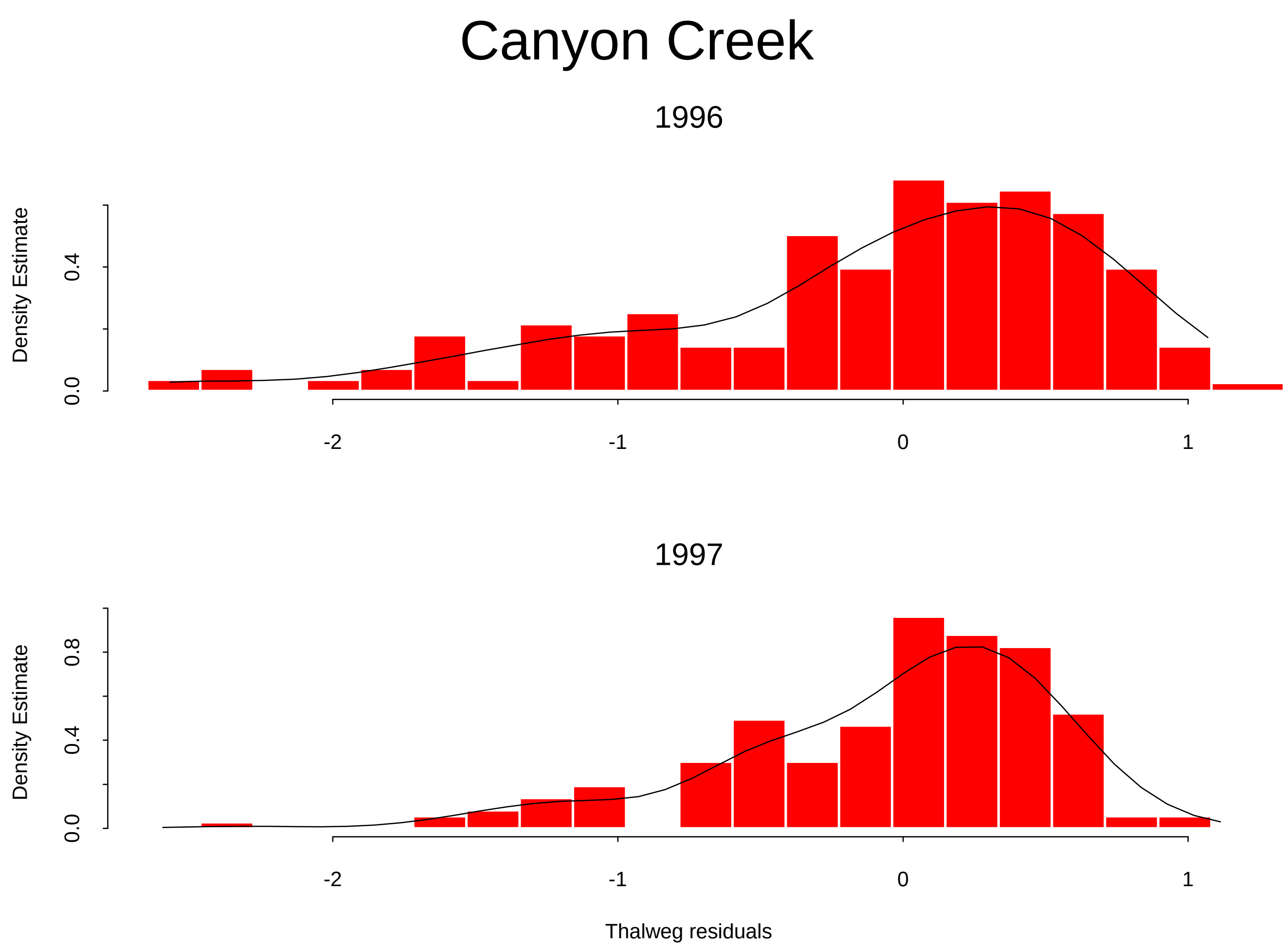

Figure C3-7. Histograms of thalweg residuals at Canyon Creek, 1996 and 1997, used to compare variance of residuals among years. Residuals computed using models fit in Figure C3-3. Solid line is Gaussian kernel smoothed density estimate.

C3.3.2 Analysis of Width

Both bankfull and active channel width measurements were significantly correlated when measured close together. For bank full width at Cañon Creek, the estimated correlation among measurements within 100 feet of one another was generally greater than 0.5 in all years and never lower than 0.32. The estimated correlation among active channel width measurements at Cañon Creek which were within 100 feet of one another was greater than 0.47 in all years and as high as 0.82 for measurements within 25 feet of one another. Similar high spatial correlations were observed in Hunter and Canyon creeks. Correlation of both bankfull and active channel widths measured within 50 to 75 feet of one another was generally greater than 0.5. Consequently, substantial adjustments were made to the estimates and p-values when correlations were accounted for.

Table C3-2 contains estimated mean bankfull and active channel widths for all years of the study. Values reported in Table C3-2 were obtained from the coefficients of the spatial regression (anova) model and standard errors are adjusted for estimated correlations. At Cañon Creek, the observed increase in mean bank full width from 1995 to 1996 was almost statistically significant at the α=0.05 level (p=0.054). Mean bank full width at Cañon Creek was significantly bigger in 1997 when compared to 1995 (p=0.015), but there was no difference in bankfull width between 1996 and 1997 (p=0.57). Active channel widths followed a pattern similar to bankfull. Active channel width at Cañon Creek increased significantly between 1995 and subsequent years (p<0.0001 for 1995 vs. 1996; p<0.0001 for 1995 vs. 1997), but remained constant between 1996 and 1997 (p=0.45 for 1996 vs. 1997). At Hunter Creek, neither bank full and active channel width changed significantly between 1996 and 1997 (p=0.90 for bankfull, p=0.88 for active channel). At Canyon Creek, the change in bankfull width between 1996 and 1997 was almost statistically significant at the α=0.05 level (p=0.057). Active channel width at Canyon Creek was not significantly different between 1996 and 1997 (p=0.25).

Table C3-2. Estimated bankfull and active channel width for all years of the study.[1]

Creek	Year	Estimated Mean Bankfull Width (ft)	Standard Error, Bankfull	Estimated Mean Active Channel Width (ft)	Standard Error, Active Channel
Cañon	1995	47.39	4.68	29.51	2.64
	1996	62.06	5.97	47.16	2.36
	1997	67.15	6.61	50.78	4.11
Hunter	1996	56.2	3.42	38.5	3.15
	1997	57.0	5.13	37.8	3.40
Canyon	1996	33.4	1.39	20.8	1.04
	1997	27.0	3.00	18.6	1.58

Note
1 Estimates and standard errors were computed from the spatial regression model that accounted for spatial correlation. All measurements in feet. Significance levels can be found in the text.

C3.3.3 Analysis of Substrate Size

Figure C3-8, Figure C3-9, and Figure C3-10 display estimates of substrate size distribution for the three monitored creeks for all years of the study. Table C3-3 contains the estimated 16-th, 50-th, and 84-th quantiles from each distribution depicted in the figures, as well as each quantile's bootstrap standard error.

Table C3- 3. Estimated quantiles of substrate distributions found in three monitored creeks.[1]

Creek	Year	16th Quantile (Standard Err.)	50th Quantile (Standard Err.)	84th Quantile (Standard Err.)
Cañon	1995	14 (0.59)	36 (0.94)	68 (1.62)
	1996	11 (0.60)	29 (0.91)	63 (1.77)
	1997	16 (1.59)	44.5 (1.91)	80 (2.29)
Hunter	1996	17 (0.85)	41 (1.69)	85 (2.60)
	1997	15 (0.76)	44 (1.55)	98 (3.36)
Canyon	1996	9 (0.73)	35 (1.22)	67 (1.58)
	1997	15 (1.25)	43.5 (1.53)	84 (2.45)

Note
1 Standard errors of each quantile computed using 1000 bootstrap iterations. All measurements in millimeters (mm). 50-th quantile is the median.

The three distributions of pebble size at Cañon Creek, depicted in Figure C3-8, were all significantly different from one another ($p<0.0001$, Kruskal-Wallis; $p<0.0001$ Wilcoxon 1995 vs. 1996; $p<0.0001$, Wilcoxon, 1995 vs. 1997; and $p<0.0001$, Wilcoxon, 1996 vs. 1997). Although marginally difficult to visualize in Figure C3-8, the tests and values in Table C3-3 indicated that, in general, the distribution of pebble size shifted to the left (smaller) from 1995 to 1996 and then shifted back to the right (larger) from 1996 to 1997. Most of the distributional differences among years at Cañon Creek can be attributed to differences in the right hand tail of the distribution, with relatively more small substrate observed in 1996.

The distribution of pebble size at Hunter Creek was marginally significantly different between 1996 and 1997 ($p=0.061$, Wilcoxon). Quantiles reported in Table C3-3 indicated that the change in distribution, although not significant at the $\alpha=0.05$ level, involved a slight increase in the relative frequency of larger pebbles in 1997, relative to 1996.

The distribution of pebble size at Canyon Creek increased from 1996 to 1997 ($p<0.0001$, Wilcoxon). Inspection of Table C3-3 and Figure C3-10 reveals that almost all of the distribution of pebble size shifted to the right (larger) in 1997 at Canyon Creek, relative to 1996.

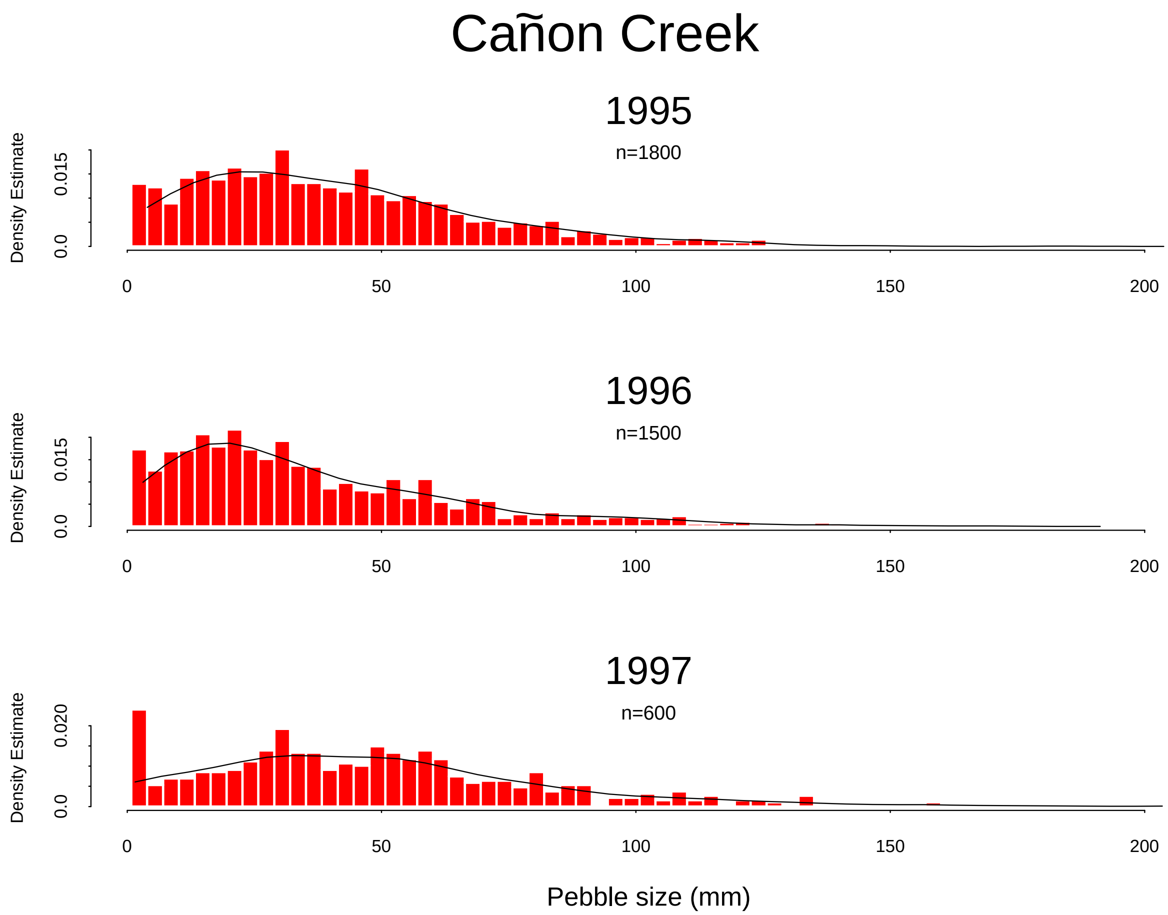

Figure C3-8. **Estimated distributions of pebble size in Cañon Creek during the study. Solid lines are Gaussian kernel smooth density estimates.**

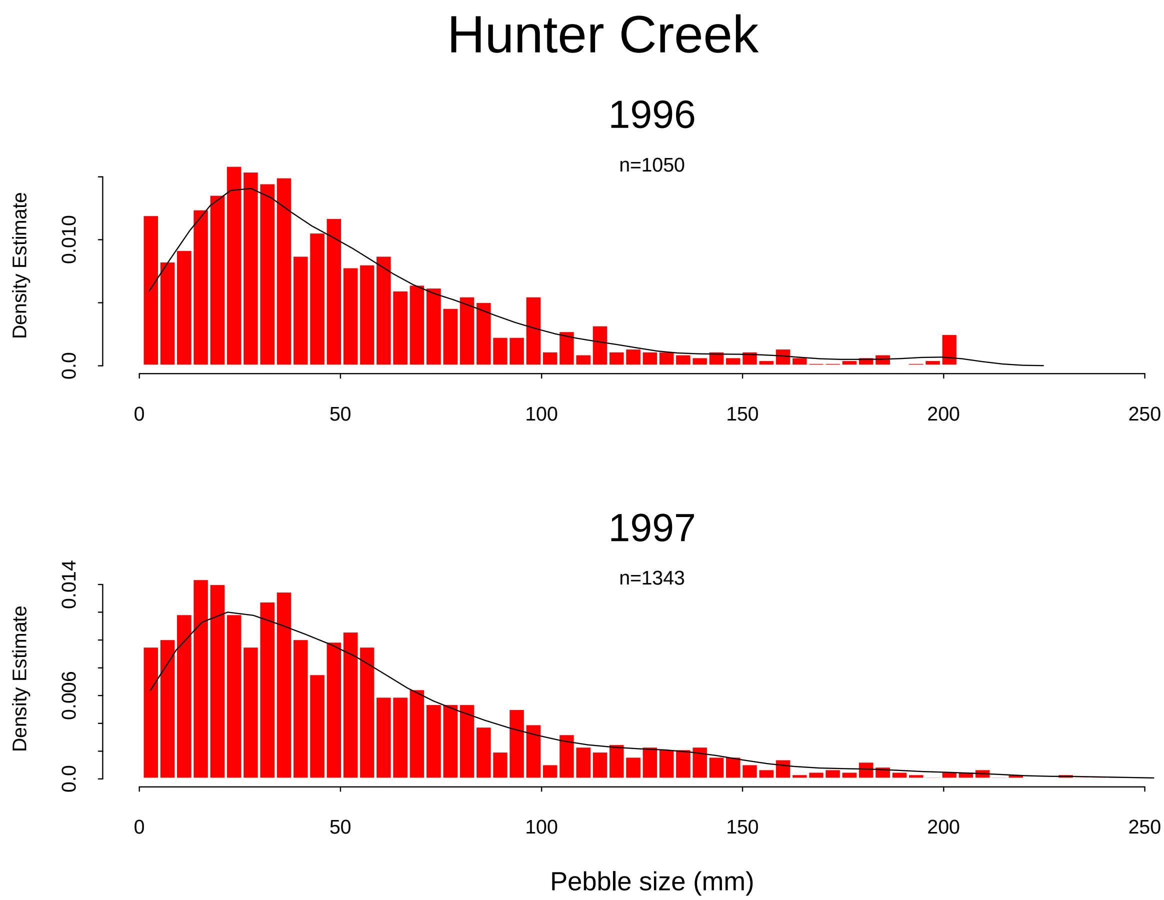

Figure C3-9. **Estimated distributions of pebble size in Hunter Creek during the study. Solid lines are Gaussian kernel smooth density estimates.**

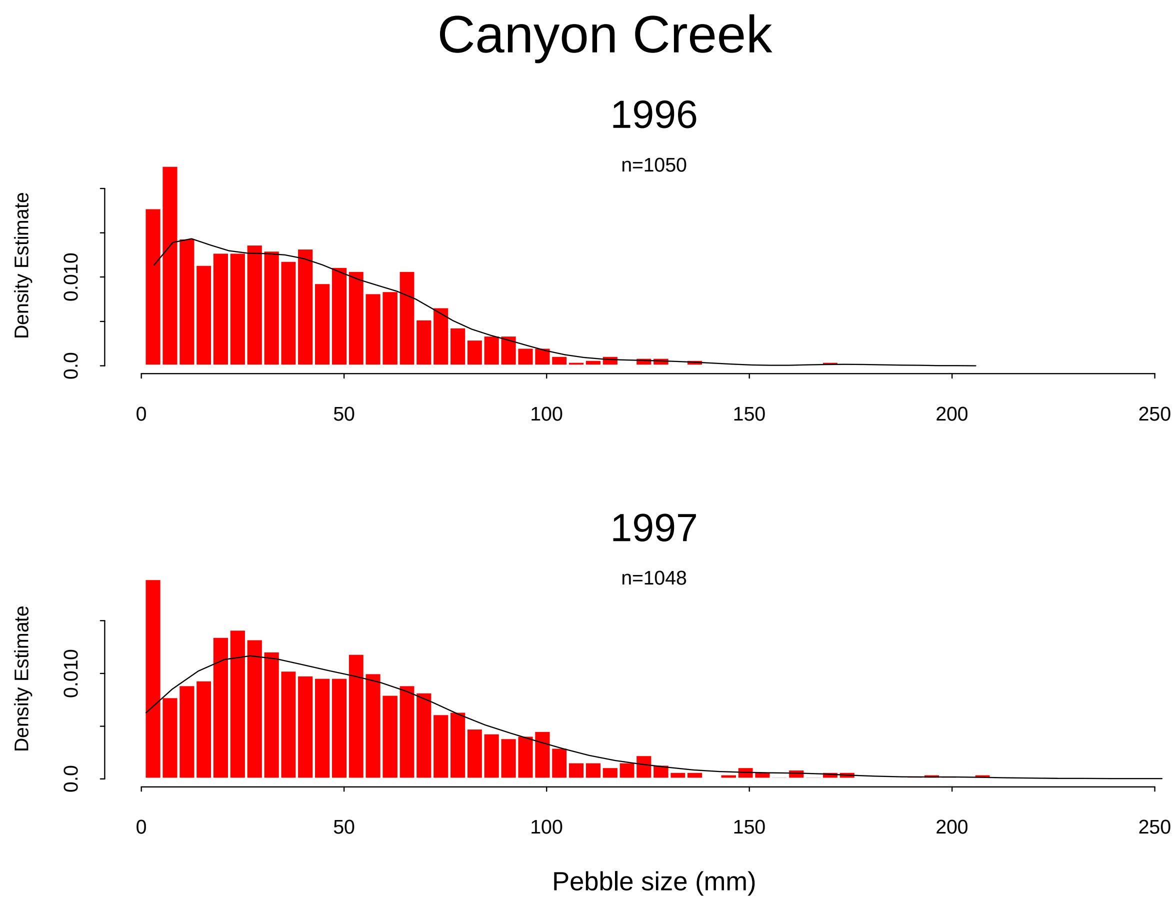

Figure C3-10. Estimated distributions of pebble size in Canyon Creek during the study. Solid lines are Gaussian kernel smooth density estimates.

As a caution when interpreting the results of this section, note that the number of pebbles measured in each creek each year was quite high (number of pebbles measured is given as >n= in Figure C3-8 through Figure C3-10). Such large sample sizes caused high statistical power to detect even relatively small differences in distributions. Small differences, although statistically significant, should be judged as to whether or not they are of any practical importance before any management decisions are made.

C3.4 DISCUSSION

The fundamental assumption associated with the long term channel monitoring is that the morphology of a depositional stream reach acts as a response surface for upslope sediment inputs. When sediment delivery increase beyond the capacity of the stream to transport it, depositional reaches will become aggraded, reduced sediment inputs will result in the opposite response. Although the morphological changes of stream reaches due to upslope sediment inputs have been well documented (Swanston 1991; Benda 1990; Benda and Dunne 1987; Hagans et al. 1986; Heede 1980), there are limitations associated with using this phenomenon for monitoring hillslope sediment production.

Quantification of some of the complex changes in channel morphology that result from changes in sediment supply can be problematic. Some changes such as the degree of sinuousity of a given stream reach generally follow predictable patterns depending on changes in the sediment load, but quantification in a statistically rigorous manner may not be possible. To deal with this potential problem, the channel monitoring protocol has been refined over time to focus on variables that respond in predictable ways and lend themselves to statistical analysis. The primary response variables that were determined to be suitable for measurement with minimum subjectivity and rigorous statistical analysis include changes in thalweg elevation and residuals, bankfull and active channel width, and substrate particle size distribution.

One of the most commonly raised concerns related to using channel morphology for monitoring is the lag times that can be associated with upslope sediment inputs and the corresponding response in the depositional reach. There is also a potential problem associated with separating natural sediment inputs from management related inputs. Both of these limitations are exacerbated with increasing distances between the upslope sediment sources and the depositional reach. As a result, the use of this monitoring approach was limited to depositional stream reaches that are closely coupled to transport reaches and potential hillslope sediment sources. Ideally, each monitoring reach is located in the watershed such that it is the first depositional reach immediately below continuously confined high gradient reaches that deliver sediment from upslope delivery sites with no capacity to store sediments in route. In reality, it is usually not possible to find the ideal monitoring reach and the selected reaches vary in how closely they are located to transport reaches and the extent to which sediments can be stored upstream of the monitoring site.

However, the response variables were found to be sensitive to mass wasting and major storm events, which have been shown to significantly change the channel dimensions. For example in Canon Creek, there was a significant decrease in the thalweg residual variance between 1995 and 1996. Between these two sampling years, there was a 10-15 year flood event (January 1996) that altered the channel morphology. The resurvey during the summer following the January 1996 flood indicated that the frequency of large deep pools decreased and the upstream and downstream ends of the monitoring reach aggraded. In this particular case, the response time was rapid in terms of showing changes in the morphology of the reach following a storm. However, Canon Creek has several miles of upstream transitional reaches that have the capacity of storing sediment, so that the aggrading of the channel did not necessarily indicate increased hillslope sediment inputs during the 1996 flood. This short coming of some of the first monitoring reaches has been recognized, and subsequent monitoring reaches have been placed so that this problem will be minimized. Although the data have not yet been analyzed, there is strong evidence that a second Hunter Creek monitoring reach located further upstream responded dramatically to a mass wasting event triggered higher up in the watershed during a November 1998 storm. The changes in the monitoring reach appeared to occur within days of the storm event. Given the differences in their placement, Green Diamond believes that the current monitoring sites have a range of response times that can vary from days to 1-2 years following a >5-year storm event. The individual response time of each monitoring site will be confirmed over time through additional monitoring.

An additional challenge associated with using channel dynamics for monitoring purposes is understanding the range of natural variability that is associated with any given stream. As a result, it likely will be necessary to continue monitoring for extended periods of time to develop a full understanding of the natural relationship between storm recurrence intervals and stream morphology. Even though it may be difficult to delineate natural variability from anthropogenic changes in the near term, Green Diamond believes that many useful insights will be gained in understanding the link between hillslope processes and channel morphology.

C3.5 CONCLUSION

This is a long term monitoring study, and therefore Green Diamond does not expect to be able to determine trends in the sediment budget of Class I watercourses for possibly 10-15 years. Threshold values for monitoring can not be established until lag times and the range of natural variability for individual watersheds or sub-basins are understood. In the interim period, Green Diamond expects to gain useful insights concerning the relationship between channel dynamics and hillslope processes within the Plan Area. By integrating data from different monitoring approaches, Green Diamond believes that channel monitoring will ultimately be a powerful tool for better understanding of the relationship between management activities and stream habitat condition for the Covered Species in the Plan Area.

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ATTACHMENT C3-A

This attachment describes the spatial regression technique used in the analysis of mean thalweg elevation, thalweg residuals, and mean channel width. This spatial regression analysis attempted to account for spatial correlations in the responses, which arise because measurements were taken close together. The technique can be described in three steps; 1) ordinary least squares parameter estimation, 2) auto-correlation modeling, and 3) weighted linear regression. Each step is described below.

Step one of the spatial regression analysis estimated a regular (Normal theory) regression of responses (i.e., thalweg elevation, thalweg residual, or channel width) onto a set of indicator variables and/or other explanatory study covariates. For example, the analysis for change in average thalweg elevation related elevation of the thalweg to a cubic polynomial of distance. The models for thalweg residual and channel width were analysis of variance (anova) models and contained indicator functions delineating the years of the study. More details about the models used for each response can be found in the main body of this report.

Step two of the spatial regression analysis estimated and modeled the auto-correlation among observed regression residuals. Estimated auto-correlations among residuals were deemed significant at various distances if an approximate 95% confidence interval surrounding Moran's *I* statistic (Moran 1950) did not contain zero. Moran's *I* was computed for relatively short lag distances, longer lag distances were ignored. If significant auto-correlation were found in the residuals, a non-linear correlation model which predicted correlation as a function of the distance between measurements was fit to the estimated correlations (see below for the form of the variance model). Auto-correlations (if significant) were modeled (spatially) within year and no (temporal) correlation was allowed across years.

If significant auto-correlations existed, a *spherical* variance model (Cressie 1991) was fit to model correlations as a function of distance. The spherical variance model had the form $v(d_{ij}) = c_1(1\text{-}1.5(d_{ij}/h_0)\text{-}0.5(d_{ij}/h_0)^3)$ if $d_{ij} \# h_0$ and 0 if $d_{ij} > h_0$ where d_{ij} was the distance between measurements $_i$ and $_j$, and c_1 and h_0 were parameters to be estimated (c_1 is commonly called the intercept and h_0 is commonly called the range). The parameters c_1 and h_0 were estimated by forming all possible statistics $z_{ij} = (r_i - \mu\, r)(r_j - \mu r)/s_r^2$, where r_i was the regression residual from the $_i$-th observation and s_r^2 was the sample variance of the residuals, and plotting the z_{ij} against d_{ij}. This graph was then smoothed using a Gaussian kernel smoother (Venables and Ripley 1994; Statistical Sciences 1995) and the spherical model was fit to the smoothed estimates using non-linear least squares estimation techniques (Statistical Sciences 1994, documentation for nlminb function). Kernel smoothing was carried out by the S-Plus function ksmooth (Statistical Sciences, 1995).

Step three of the spatial regression analysis used the estimated variance-covariance matrix derived from the variance model computed in step two as a weight matrix to re-compute coefficients, standard errors, and p-values obtained at step one. This weighted regression step is described next. Assume X was the original design matrix used in the regression model at step one which contained indicator variables and/or polynomials in distance. Assume Y was the vector of responses, and V was the estimated variance-covariance matrix obtained at step two. The re-computed vector of coefficients, $\hat{\beta}$, and variance was,

$$\hat{\beta} = (X'V^{-1}X)^{-1}X'V^{-1}Y$$
$$\mathrm{var}(\hat{\beta}) = (X'V^{-1}X)^{-1}.$$

Significance of an element in $\hat{\beta}$ was assessed by comparing the ratio of the element to its standard error to a (Student's) T distribution having *n*-*p* degrees of freedom (*n* was total number of observations, *p* was the number of columns in X). This test is commonly referred to as a Wald t-test (Venables and Ripley 1994).

Appendix C4. Assessment of Erosion and Sedimentation in Class III Watercourses: A Retrospective Study

CONTENTS

Figures

Tables

C4.1 INTRODUCTION

California forest practice rules during the period of this study (1992-1998) required that Class III watercourses (typically first order streams that do not support aquatic life) be delineated as equipment exclusion zones and that ground disturbance be minimized, but they did not require retention of existing forest canopy. Concerns have been raised that complete removal of trees from Class IIIs will result in destabilizing these headwater areas resulting in an upslope extension of the channel and increased risk of shallow rapid landslides. The mechanisms that could trigger these potential effects may not be fully mitigated by the existing forest practice regulations: loss of root strength in the soil column that could increase mass wasting, decrease bank stability and increased incident precipitation and storm runoff that could increase mass wasting and fluvial erosion processes in Class III watercourses. There is some evidence suggesting the latter from Caspar Creek (Lewis 1998). The net effect is that there could be significant increases in sediment production from watercourses even though Class I and II watercourses may have ample buffer retention. Because the majority of a channel network is made up of the first order channels, the overall impact of destabilized Class IIIs may be quite large even though increased sediment delivery in any given Class III is small. There is also the concern that if a debris torrent is triggered from one of these Class III areas, there will be no opportunity for delivering LWD into the channel below if no trees are retained in the uppermost reaches of these watercourses. The role of LWD in erosion and sedimentation processes in Class III channels is also potentially significant. LWD provides sediment storage sites, controls channel grade by preventing channel bed erosion, and deflects and concentrates stream flow thereby both protecting banks from erosion and magnifying fluvial bank erosion processes.

However, there are few empirical data available to assess the magnitude of these potential problems in northern California forestlands. To begin with, the proportion of first order streams that are designated as Class IIIs in current timber harvest plans (THPs) has not been quantified. Since any headwater channel that is judged to support “aquatic life” must be classified as a Class II, an unknown but increasingly higher proportion of first order channels are receiving protection as Class II watercourses. Although the forest practice rules have not changed, this trend has occurred primarily due to the southern torrent salamander. The transition began at Green Diamond in 1992 when its biological staff began demonstrating to the foresters that many first order channels supported torrent salamanders. The rest of the California north coast region followed suit when the torrent salamander was petitioned to be state listed in 1995. The species was not listed, but a mandatory training program to learn to identify the habitat of the salamander was instituted for all registered professional foresters that wished to submit THPs within the range of the species. Region wide, this had a dramatic effect on watercourse classification and in some areas there are few Class IIIs at the head of a Class II watercourse. The channel begins as a Class II, because it has intermittent habitat for torrent salamanders.

In addition to not knowing the extent of Class IIIs in THPs, there are no data on the changes that result in these watercourses following timber harvest. In particular, it is important to know the degree to which channel extension or head-cutting is occurring along with some quantification of the amount of sediment that is being generated from the existing channel banks due to bank erosion or channel scour. It is also important to know if destabilized Class IIIs are contributing to increases in shallow rapid landslides.

Past protection of Class III watercourses during timber harvest was a combination of both compliance and effectiveness of the forest practice rules as they were implemented through the THP process. Therefore, completed THPs were used as the basis for the selection and assessment of the condition of Class IIIs. A retrospective approach was used to randomly select completed THPs from across the ownership, and quantify the number and extent of both Class II and III watercourses that were identified by the RPF prior to harvesting. The selected watercourses were visited, and data were gathered on the physical condition of the Class III watercourse. Since this was a retrospective study and it was not possible to utilize controls, subtle changes in Class IIIs following timber harvest could not be quantified. Rather the objective was to assess the extent to which major changes occurred in Class IIIs that were responsible for substantial increases in management related sediment production. Specifically, the objectives were to: 1) collect data to characterize and describe Class III channels following clearcut harvest under the past Forest Practice Rules and Green Diamond's spotted owl HCP; and 2) explore potential relationships between key response variables that correlate strongly with sediment production (e.g. bank erosion and number of landslides) and other important stream variables. There also was the opportunity to compare pre-harvest characteristics of Class III watercourses that were assessed as part of the Little River monitoring study to a sub-set of the streams from the retrospective study that were located within or adjacent to the Little River HPA. Unfortunately, this was not a pre and post-treatment assessment of the same streams, but it did allow for general comparisons of characteristics before and after harvest.

It is important to reiterate that this was a retrospective study and comparisons to untreated control streams (i.e., unharvested Class III watercourses in advanced second growth or virgin old growth) were not possible. Therefore, conclusions from the study were limited in scope. The primary objectives were to provide a description of key variables of Class III watercourses sampled and quantify gross changes that might have occurred following clearcut timber harvesting. A stratified random sampling design was followed, so it was appropriate to draw inferences to the total sampling universe. However, since the sampling was tied to recent harvesting (1992-1998), the inferences need to be restricted to that portion of the total ownership that has experienced significant harvesting in recent years. Despite these limitations, the study has significant value simply because there is so little known about the characteristics of Class III watercourses or the impact of timber harvest on them.

C4.2 METHODS

C4.2.1 Site Selection

The Class III retrospective survey was conducted across all of Green Diamond's property with the exception of some of outlying areas (e.g. South Fork Mountain, Supply and Goose Creeks) where logistical constraints would have drastically reduced the efficiency of the project. All of Green Diamond's ownership within the Mad River was included in the study, including lands outside the HPAs. A stratified random sampling of Class III watercourses was employed throughout the remaining tracts (management units) of the ownership. All Class IIIs in completed THPs from 1992-1998 were classified as either a “run-through” or “within” (Figure C4-1). A “run-through” refers to a Class III watercourse where the beginning of the channel is outside the harvest unit, but if the channel was initiated within the boundaries of the harvest unit, it was designated “within.” The number of Class IIIs was then randomly sampled at frequency of 2:9 within streams and 1:9 run-throughs. The sampling was weighted toward within streams in order to focus on channel extension of Class IIIs. The original THP map for each selected unit was reviewed as well as aerial photos to ensure that selected units were true clearcuts. Units that had non-clearcut prescriptions (i.e. seed tree removal, selection harvest or commercial thinning) were not included in the sample. In addition, a minimum apparent channel length of 200 feet on the THP map was required to be included in the sample. However, in the field, the actual channels varied from minimums of 113 and 58 feet, and maximums of 1146 and 1295 feet for run-through and within channels, respectively.

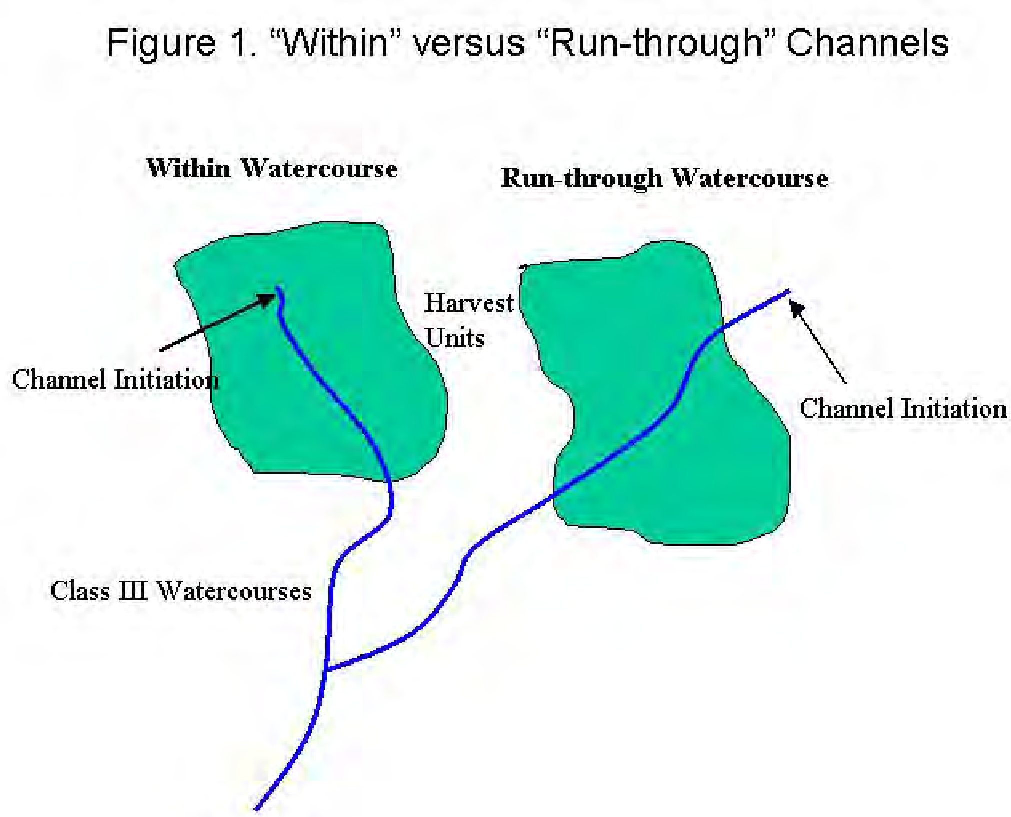


Figure C4-1. "Within" versus "run-through" channels.

Bedrock geology underlying each study site was determined based on USGS geologic maps and characterized as “consolidated” or “unconsolidated” by Oscar Huber (retired geologist, CDF). Consolidated bedrock geology included the Franciscan series (undifferentiated, melange, sandstone with siltstone, rocks and schist), Galice and ultramafic rocks. The undifferentiated Wildcat Group, Hookton and Falor Formations, Alluvium, Quaternary marine terraces and coastal plain sediment were considered unconsolidated bedrock geology.

THPs were not selected before 1992, because of a property-wide shift in the designation of Class II versus III watercourses. Prior to that year, many small intermittent channels were classified as Class IIIs that would have been designated a Class II after 1992. (This shift resulted from the recognition of southern torrent salamander habitat as noted above.) THPs were not selected after 1998 to insure that Class IIIs had experienced at least one winter of storms.

C4.2.2 Field Protocol

Before going into the field, Green Diamond delineated the Class III drainage as mapped on the original THP map. Assessment of the watercourse began at the lowest point on the channel within the THP unit. If the lowest end was within a riparian protection zone or habitat retention area (HRA), then Green Diamond began the channel measurements at the uppermost edge of the standing timber. Measurements were taken systematically up the channel at 10-foot intervals based on a random start within the first 10-foot interval. At each 10-foot sampling interval, the active channel width, maximum depth, was measured, and it was determined if there was evidence of an exposed active channel (channel bed exposed by fluvial processes). The linear length of exposed bank within 15 feet of the channel on both banks also was measured. If the exposed bank was part of an earth flow or slide, the entire limit of the exposed ground was measured. Game trails and animal burrows were not included in measurements of exposed banks, but their occurrences were noted. Watershed drainage area at the downstream end of study sites was also determined.

At every 50-foot interval, the bank angle perpendicular to the channel on the left and right banks was measured. At every 100-foot interval, the mean understory vegetation height was measured, and percent overstory canopy closure was determined using a densiometer. The channel gradient was measured with a clinometer at the beginning of the layout and at all major gradient breaks in slope throughout the remaining channel layout. Large woody debris (LWD) greater than 6 inch diameter with no minimum length requirement was measured (length and average diameter) wherever it occurred throughout the channel. It was recorded if the LWD was hardwood or conifer (if not clear which, “hardwood” was recorded, which provides a more conservative estimate of the longevity of the LWD), and it was noted if the LWD was acting as a control point. (A control point was any in-channel feature retaining sediment and/or preventing head-cutting.) The location and type of all other control points (roots, boulders, bedrock, etc.) were recorded in addition to LWD, and the size (with the exception of bedrock) and the vertical drop below the control point were measured. The area and location of any significant (generally greater than 3 feet in length) bank erosion were measured, and the predominant channel substrate, presence and flow of water, changes in predominant vegetation, and the occurrence of any aquatic vertebrates were noted.

Green Diamond photo documented the site, looking upstream at the beginning of the layout, both directions in the middle, and downstream at the end. In addition, any major gradient breaks in the channel that precluded visibility, any significant mass wasting, large scours, or other major features that affected the channel were photo documented.

The in-channel survey was continued until the Class III channel ended at a headwall, or at the harvest unit boundary, if the channel was a run-through. Green Diamond assessed the channel for evidence of head cutting by looking for evidence of recent scour or bank erosion at the initiation of the channel. In addition, Green Diamond compared the mapped initiation of the channel from the THP map relative to the current initiation of the channel. Green Diamond surveyed the associated road system within the sub-basin and sketched the drainage area onto a topographic map. Green Diamond recorded any stream piracy or diversions associated with the road system and include it in the drainage area. On the topographic map, Green Diamond recorded road failures, inner gorge slides or other larger scale sediment delivery features within the sub-basin. Data collected are summarized in Table C4-1.

An ongoing monitoring program in the Little River watershed utilizing a BACI (before-after-control-impact) experimental design allowed for a partial comparison of pre-treatment (advanced second growth with no recent timber harvesting activities) Class III watercourses to some of the post-treatment streams from this retrospective study. The same protocols described above were applied to the pre-treatment assessment of 26 Class III watercourses in the Little River, which were compared to 29 post-treatment (retrospective) watercourses located within or adjacent to the Little River watershed.

Table C4-1. Summary of continuous and categorical variables measured on surveyed Class III watercourses.[1]

Continuous	Categorical
Width and depth of active channel	Exposed active channel
Length of surveyed channel	Exposed banks
Channel gradient	Channel initiation (run-through vs. within)
Bank slope	Bedrock geology
Number of years (winters) since harvest	Type of harvest (tractor vs. cable)
Drainage area above the channel	Burn history
Height of ground vegetation	
Total canopy closure	
LWD: #, length, diameter and volume	
Bank erosion: number and area	
Slides: number and area	

Note
1 Exposed active channel and exposed banks were assessed as a categorical variable at each 10-foot sample interval, but summarized as a percentage of the total samples intervals measured. Response variables are highlighted.

C4.2.3 Data Analysis

Green Diamond selected four variables that best reflected potential sediment delivery to the lower portions of a watershed as the primary response variables for analysis. These variables were cross-sectional area (product of the active channel depth and width measurement), percent exposed active channel, frequency of sites with bank erosion and number of slides relative to channel length. Forward stepwise regression was performed using function step.glm (generalized linear model) in the computer program S-Plus. Step.glm added variables from the pool of potential explanatory (independent) variables, one at a time, until the model AIC (Akaike’s Information Criterion) would not decrease if another variable was entered. The variable chosen for inclusion at each step was the variable that provided the greatest improvement of the modeled likelihood among variables that were not yet in the model. This addition amounted to adding the variable at each step with the most significant likelihood score statistic. Significance of terms in the final model was assessed using an approximate F-test based on the drop-in-deviance likelihood ratio. GLM R^2 values were calculated, which are equivalent in interpretation (amount of the variation in the dependent variable explained by the independent variable) to R^2 values from regression based on a normal distribution.

Response variables ‘bank erosion’ and ‘number of slides’ were modeled using a Poisson regression that included an “offset” to relate the count to the length of sampled stream segment. ‘Percent exposed active channel’ was modeled using binomial regression. ‘Cross-sectional area’ of the channel was modeled using Normal regression theory, but was first transformed by computing the natural log of the variable. To meet assumptions of normality, cross-sectional area and percent exposed active channel were also transformed (natural log for area and square root for percent scour) before performing t-tests or analysis of covariance (ANCOVA). For stepwise regression, geology was treated as a categorical variable with two levels: ‘unconsolidated’ and ‘consolidated’.

C4.3 RESULTS

There were 899 THP units operated within the study area from 1992-1998. To find units that meet the criteria of having a Class III watercourse located within a clearcut block, 553 harvest units were initially selected using a stratified random sampling design. From these units, 110 Class III watercourses were identified that appeared to have met the criteria for inclusion in the survey. On field inspection, some of these Class III watercourses had to be to be eliminated (e.g. trees were retained in the Class III to meet habitat retention guidelines under Green Diamond’s spotted owl HCP), which resulted in 100 channels ultimately being assessed across Green Diamond’s ownership (Figure C4-2). Forty-seven of the channels were run-throughs (channel initiated outside the harvest unit) and 53 were within channels (initiated within the harvest unit). Because the selection of Class IIIs was dependent on recent (1992-1998) harvesting activities, the number of channels assessed per HPA was not necessarily proportional to the area of the HPA. In addition, the number of Class III watercourses associated with each unit varied across the study area. The majority of harvest units within most of the study area had no or only one Class III watercourse within or adjacent to the unit, while the majority of units had multiple Class III watercourses in the two most southerly HPAs (Table C4-2). The greatest number of channels (25) was assessed in the Mad River HPA, followed by Smith River (20), North Fork Mad River (14), Little River (13), Humboldt Bay (11), Eel River (6), the area in the Mad River that is outside the Plan Area (3), and two each for Redwood Creek, Coastal Lagoons, Coastal and Interior Klamath HPAs. Of the 100 watercourses selected to be assessed as Class IIIs based on the original THP, 16 were judged to have at least a small portion that was a Class II watercourse based on Green Diamond’s current more thorough and conservative approach to evaluating streams for the presence of headwater amphibians or their habitat.

The mean length and cross-section area of run-through channels were greater than within channels (Table C4-3), as might be expected because they were generally lower in the watershed and had greater drainage area. However, the mean cross-sectional areas were not significantly different ($t = 1.81$, d.f. = 96, $P = 0.073$) between run-through and within channels.

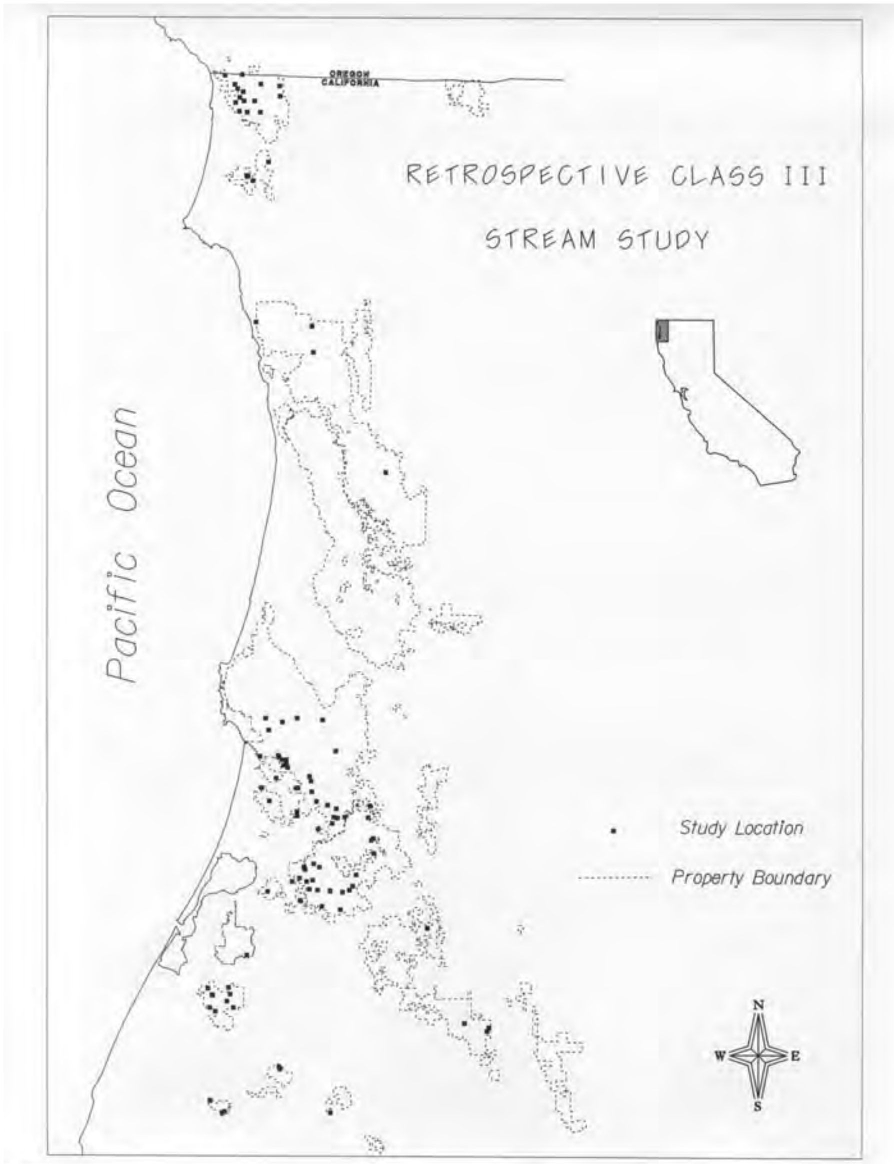


Figure C4-2. Location of Class III channels assessed on Green Diamond's ownership.

Table C4-2. Summary of harvest units operated from 1992-1998 within each Hydrographic Planning Area and the number of units with no or only one Class III watercourse within or adjacent to the harvest unit.[1]

Hydrographic Planning Area	Harvest Units	Percentage with no Class III	Percentage with one Class III
Smith River	141	36.2	24.1
Blue Creek	53	34.0	35.8
Coastal Klamath	152	38.1	31.6
Interior Klamath	145	39.3	27.6
Redwood Creek	51	62.7	21.6
Coastal Lagoon	11	27.3	27.3
Little River	38[2]	5.3[2]	15.8[2]
NF Mad River	61	23.0	29.5
Mad River	126	17.5	26.2
Humboldt Bay	42	14.3	16.7
Eel River	42	11.9	16.7
Area outside the Plan Area	64	43.8	28.1
Total	899	32.0	26.2

Notes
1 Summary includes all units whether or not there were any type of watercourses associated with the harvest unit.
2 Harvest units in this HPA were developed and operated by a previous owner.

Table C4-3. Summary of Class III watercourse characteristics.[1]

Variables	Run-through		Within		Total	
	N	mean (SE)	N	mean (SE)	N	mean (SE)
Drainage area (acres)	47	10.5 (2.48)	53	5.6 (0.66)	100	7.9 (1.24)
Channel length (ft)	47	451.5 (31.62)	53	346.1 (34.46)	100	395.6 (24.02)
Channel width (ft)	47	2.55 (0.147)	53	2.69 (0.234)	100	2.62 (0.140)
Channel depth (ft)	47	0.33 (0.029)	53	0.25 (0.002)	100	0.29 (0.019)
X-section area (ft^2)	47	0.96 (0.146)	53	0.67 (0.083)	100	0.81 (0.083)
Channel gradient (%)	47	31.5 (1.79)	53	35.2 (1.81)	100	33.4 (1.28)
Bank slope (%)	47	47.4 (2.481)	53	43.0 (2.61)	100	45.1 (1.81)
Exposed bank (%)	47	0.66 (0.113)	53	1.00 (0.343)	100	0.84 (0.189)

Note
1 Cross-sectional area of the channel represents the product of the active channel depth and width measurement. RT = run-through channels and Within = within channels.

Green Diamond conducted a forward stepwise regression analysis to determine which of the independent variables explained variation in mean channel cross-sectional area. The first variable to enter the model was drainage *area* (F = 20.80, d.f. = 1,92, P < 0.001, improvement R^2 = 0.237, model coefficient = 0.044), followed by underlying bedrock geology (F = 8.23, d.f. = 1,92, P = 0.005, improvement R^2 = 0.061, model coefficient = -0.455) indicating greater channel width in unconsolidated bedrock geology), stream gradient (F = 9.16, d.f. = 1,92, P = 0.003, improvement R^2 = 0.051, model coefficient = -0.016) and number of rock controls (F = 3.93, d.f. = 1,92, P = 0.051, improvement R^2 = 0.027, model coefficient = 0.937). The full model explained 37.5% of the variation in cross-sectional area of channels among streams. The cross-sectional area of channels with consolidated underlying geologic materials was significantly less when corrected for drainage area than channels in unconsolidated geology (consolidated area: n = 74, $\bar{x}$ =

0.61, SE = 0.048; unconsolidated area: n = 24, $\bar{x}$ = 1.41, SE = 0.273; ANCOVA: F = 13.52, d.f. = 1,95, P < 0.001). This relationship between drainage area and cross-sectional area of the active channel is illustrated in Figure C4-3.

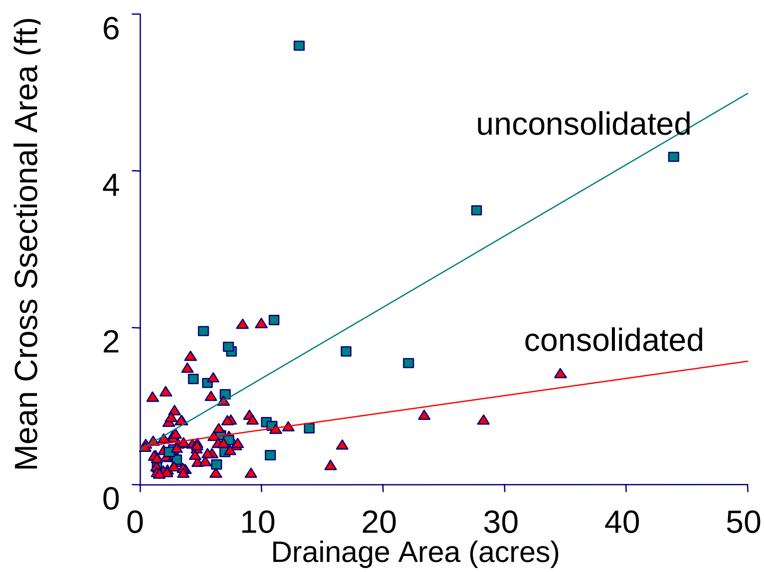
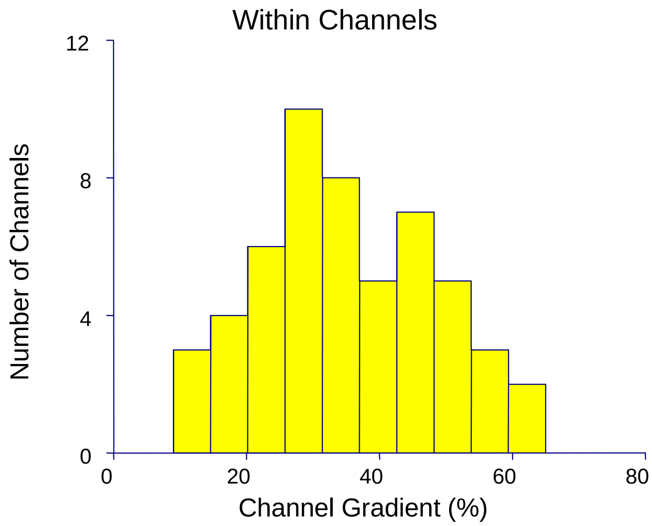


Figure C4-3. Mean cross sectional area (ft^2) of channels versus drainage area in consolidated and unconsolidated bedrock geology. Triangles represent consolidated geology and squares unconsolidated geology. Regression equation for consolidated geology: Y = 0.477 + 0.022*drainage, R^2 = 0.096; unconsolidated geology: Y = 0.447 + 0.091*drainage, R^2 = 0.409.

Consistent with being higher in slope position, within channels had somewhat higher mean stream gradient ($\bar{x}$ = 35.2, SE = 1.82) compared to run-through channels ($\bar{x}$ = 31.5, SE = 1.79), although the differences were not statistically significant (t = 1.44, d.f. = 98, P = 0.153). In addition, the distribution of stream gradients indicated that both types of Class III channels had a similar wide range of stream gradients (Figure C4-4). There was no difference in channel gradient or bank slope between consolidated and unconsolidated bedrock geologies with drainage area as the covariate (ANCOVA: stream gradient – F = 0.51, d.f. = 1,97, P = 0.478; bank slope – F= 1.02, d.f. = 1,97, P = 0.315). The mean number of LWD pieces per 100 feet of Class III channel was 4.80 (SE = 0.318), while mean volume was 226.6 (SE = 25.02) cubic feet per 100 feet of channel. However, the distribution in the number and volume of LWD (Figure C4-5) indicated that most channels had relatively low amounts with a small proportion of channels having high amounts of LWD. Of the LWD associated with these channels, 85.0% (SE = 2.59) was determined to be conifer.



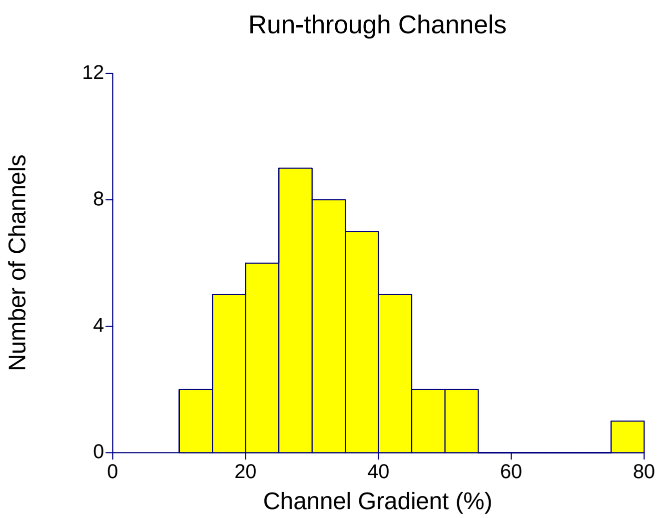
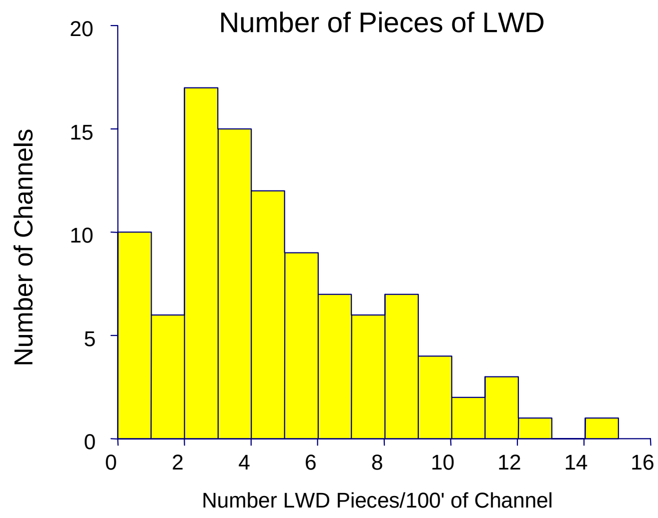


Figure C4-4. Distribution of stream gradients for "within" and "run-through" Class III watercourses.



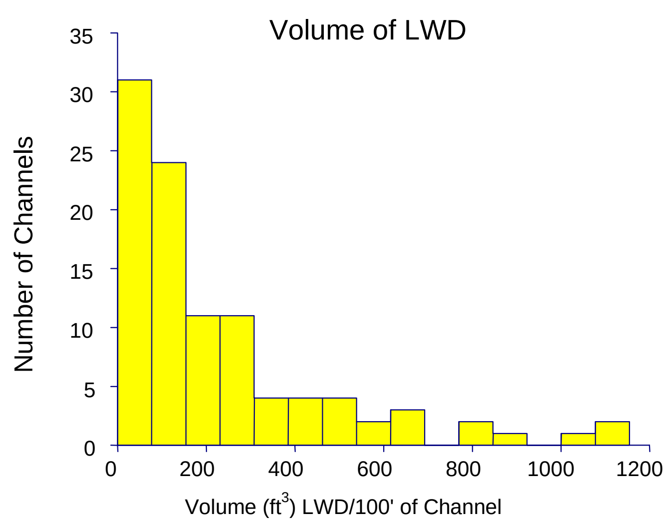


Figure C4-5. **Distribution among surveyed Class III watercourses of the number and volume of LWD per 100 feet of channel.**

The mean number of total control points per 100 feet of Class III channel was 0.93 (SE = 0.121) with most (>75%) of the controls being formed by LWD (Figure C4-6). Roots and rocks (large rock or bedrock) were particularly rare in forming control points in these Class III channels.

Mean percent exposed active channel (EAC – percent of 10-foot sample intervals with evidence of an exposed active channel) for within and run-through channels was 23.3 (SE = 2.88) and 24.6 (SE = 2.55), respectively. The difference was not statistically different (t = 1.097, d.f. = 97, P = 0.275) so the two channel types were combined for additional analysis. The distribution of mean percent EAC channel (Figure C4-7) was highly skewed to the left with most channels showing little or no EAC. Green Diamond conducted a forward stepwise regression to further explore the relationship between EAC and other independent variables measured. The first variable to enter the model was the total number of channel control points (F = 41.427, d.f. = 1,93, P < 0.001, improvement R^2 = 0.232, model coefficient = 0.474), followed by mean height of riparian ground vegetation (F = 6.75, d.f. = 1,93, P = 0.011, improvement R^2 = 0.047, model coefficient = 0.220), and underlying bedrock geology (F = 5.33, d.f. = 1,93, P = 0.023, improvement R^2 = 0.036, model coefficient = -0.498). The full model explained 31.5% of the variation in EAC of channels among streams. Green Diamond expected channel scour to be positively correlated with stream gradient, but it did not enter the stepwise regression model. To graphically explore the relationship, Green Diamond produced a scatter plot of EAC and gradient (Figure C4-8), which further illustrates the lack of correlation between these two variables.

The preponderance of LWD as channel controls and the apparent positive correlation between channel controls and EAC prompted us to graphically look at the relationship between LWD controls and EAC (Figure C4-9). Although there is considerable variation, it is apparent that there was a positive relationship between the number of LWD controls and percent EAC.

Sites along the banks of the Class III channels with bare mineral soil that were the result of undercutting or sloughing were termed bank erosion. Relative to the axis of the channel, these sites were longer (mean length = 9.6 feet, SE = 0.81) than wide (mean width = 5.3 feet, SE = 0.47). Among the 100 channels surveyed, there were 107 total sites with bank erosion. Most sites (57%) had no bank erosion, while a few streams had relatively frequent bank erosion (Figure C4-10). Green Diamond conducted a forward stepwise regression to further explore the relationship between bank erosion and other independent variables measured. The only variables to enter the model were underlying bedrock geology (F = 8.05, d.f. = 1,93, P = 0.006 improvement GLM R^2 = 0.258, model coefficient = -0.787) (greater bank erosion in unconsolidated geology), followed by total canopy closure (F = 7.75, d.f. = 1,93, P = 0.007, improvement GLM R^2 = 0.086, model coefficient = -0.030) (less bank erosion with greater canopy closure) and volume of LWD (F = 3.21, d.f. = 1,93, P = 0.077, improvement GLM R^2 = 0.026, model coefficient = 0.001) (greater bank erosion with more LWD). The full model explained 37.1% of the variation in bank erosion among streams.

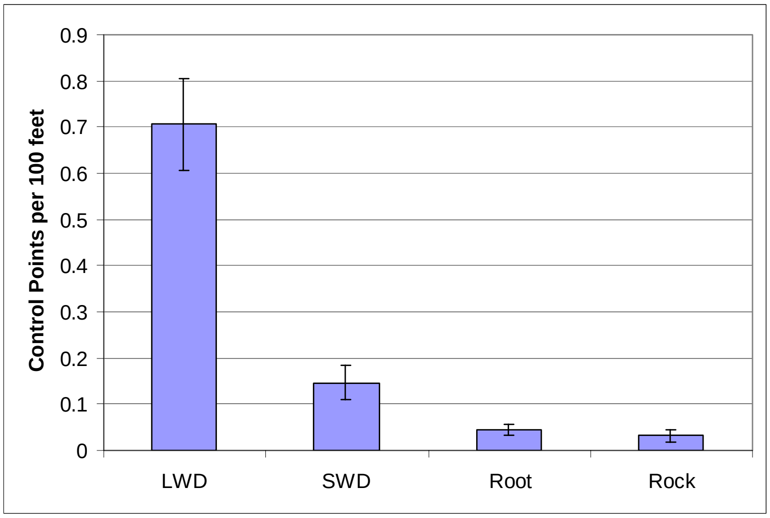


Figure C4-6. Mean number of control points per 100 feet of channel with standard error bars. LWD = control points formed from large woody debris (>6 inches), SWD = control points formed from collections of small woody debris (<6 inches), root = control points formed by tree roots and rock = control points formed from large rocks or bedrock.

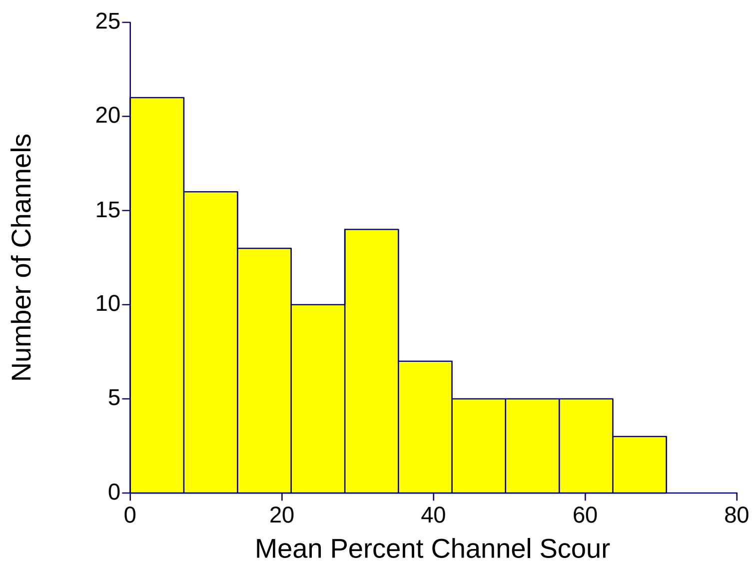


Figure C4-7. Distribution of mean percent exposed active channel (EAC) among surveyed Class III watercourses.

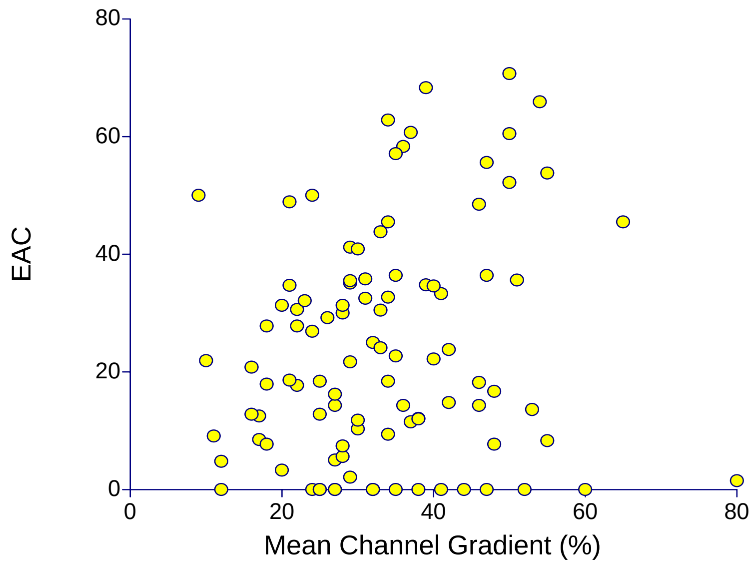


Figure C4-8. **Mean channel gradient versus mean percent exposed active channel (EAC) for individual watercourses.**

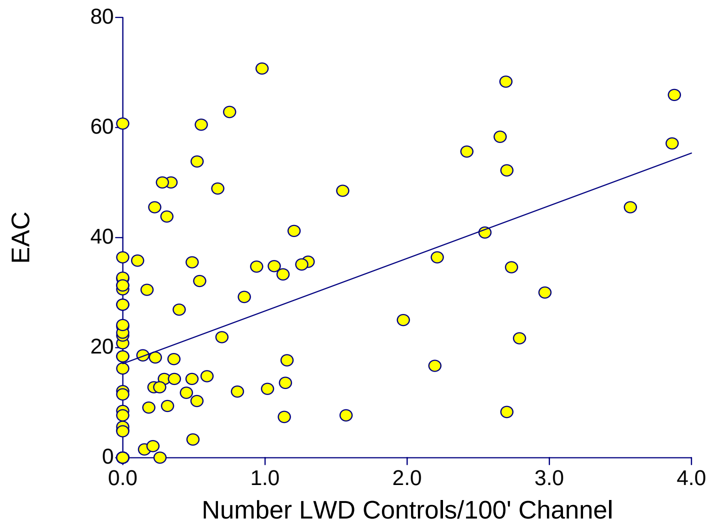


Figure C4-9. **Number of LWD control points per 100 feet of channel versus mean percent exposed active channel. Trend line is the least squares regression line. Regression equation: Y = 0.010 + 0.026*EAC, R^2 = 0.245.**

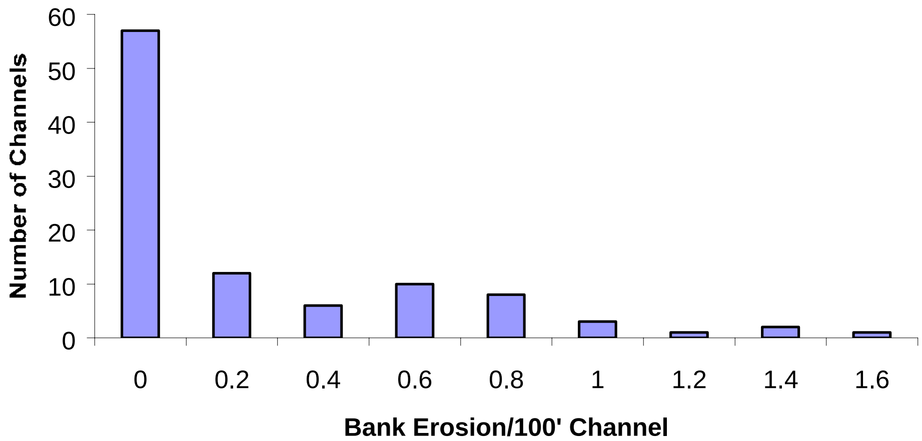


Figure C4-10. Distribution of sites with bank erosion among surveyed Class III watercourses. Each value in the figure represents the mean value from a given stream.

Twenty-four shallow rapid landslides were identified while surveying the 100 Class III watercourses. One slide was associated with a road and not included in further analysis, while all of the rest of the slides were associated with an inner gorge or steep streamside slope. There were no debris torrents associated with any of the channels surveyed. The distribution of landslides among surveyed channels (Figure C4-11) indicated that most (85%) had no slides with a few of the channels accounting for the majority of the slides. The cumulative frequency distribution of the length (maximum head scarp distance) of the landslides indicated that 80% of the slides were located within less than 20 feet of the channel (Figure C4-12). The results of a forward stepwise regression analysis of the relationship between landslides (number/100 feet of channel) and other independent variables measured indicated that the first variable to enter the models was stream gradient (F = 7.17, d.f. = 1,91, P = 0. 009, improvement GLM R^2 = 0.350, model coefficient = 0.027). This was followed by mean height of ground vegetation (F = 30.15, d.f. = 1,91, P < 0.001, improvement GLM R^2 = 0.093, model coefficient = -1.128), mean bank slope (F = 25.74, d.f. = 1,91, P < 0.001, improvement GLM R^2 = 0.072, model coefficient = 0.054), number of LWD controls (F = 14.56, d.f. = 1,91, P < 0.001, improvement GLM R^2 = 0.051, model coefficient = 0.473) and years since harvest (F = 14.57, d.f. = 1,91, P < 0.001, improvement GLM R^2 = 0.071, model coefficient = 0.322). The full model explained 63.6% of the variation in the number of slides among streams.

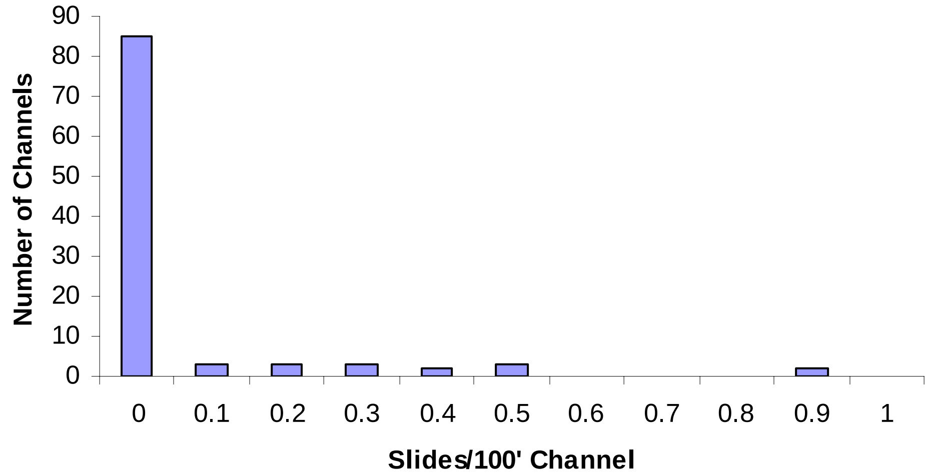


Figure C4-11. Distribution of landslides among surveyed class III watercourses. Each value in the figure represents the mean value from a given stream.

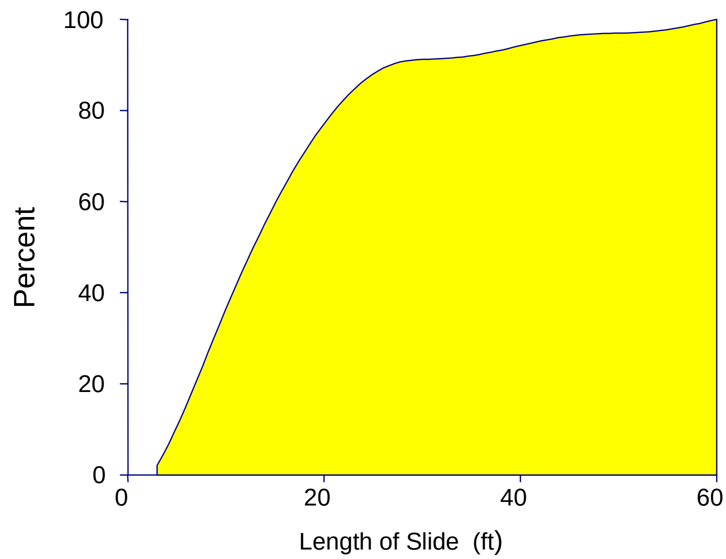


Figure C4-12. Cumulative frequency distribution of the length (maximum head scarp distance) of 23 inner gorge or steep streamside slope landslides associated with surveyed Class III watercourses.

Bank erosion or slides at the initiation of Class III watercourses are evidence of head cutting or channel extension. In the 53 within channels where this could be assessed, the only channel extension or head cutting observed was due to runoff from roads. This occurred in both in within and run-through channels and was typically associated with improper road drainage. There was no direct evidence for head cutting or channel extension due to hillslope processes. There was also no evidence of channel extension based on the mapped initiation of the channel in the THP map, but these maps were not considered very precise.

C4.3.1 Comparisons with Pre-treatment Steams

There were 26 Class III watercourses that were assessed as part of the Little River monitoring program. These were compared to 29 Class III watercourses in or adjacent to the Little River HPA that were assessed as part of this retrospective study. Although these streams were spatially and temporally separated, most characteristics were similar (Table C4-4).

Using ANCOVA with drainage area as a covariate, cross-sectional area and percent EAC (square root transformed) for pre and post-treatment streams were not significantly different (Cross-sectional area: F = 0.31, d.f. = 1,49, P = 0.583; Percent scour: F = 2.72, d.f. = 1,52 P = 0.105).

Table C4-4. Summary of pre- and post-treatment Class III watercourse characteristics.

Variables	Pre-treatment		Post-treatment	
	N	mean (SE)	N	mean (SE)
Drainage area (acres)	26	8.0 (1.40)	29	8.7 (3.60)
Active channel length (ft)	26	374.9 (51.81)	29	405.2 (50.54)
Active channel width (ft)	24	2.05 (0.156)	28	2.42 (0.231)
Active channel depth (ft)	24	0.28 (0.024)	28	0.26 (0.106)
Cross-sectional area (ft^2)	24	0.567 (0.063)	28	0.617 (0.063)
Channel gradient (%)	24	28.5 (2.10)	29	30.4 (2.19)
Bank slope	26	16.8 (1.21)	29	21.8 (1.41)
Percent exposed active channel	26	15.0 (2.47)	29	27.7 (4.26)
Bank erosion sites/100 ft	26	0.46 (0.127)	29	0.33 (0.084)
Slides/100 ft	26	0.03 (0.033)	29	0.05 (0.034)

C4.4 DISCUSSION

C4.4.1 Limitations

The preceding data are retrospective in nature and do not provide comparisons to untreated control streams (i.e. unharvested Class III watersheds in advanced second growth or virgin old growth.) Therefore, it is important to identify the type of conclusions that one should expect to be able to draw from the data. Most of the data were descriptive in nature, which allowed us to create an “image” of the characteristics of Class III watercourses sampled. Green Diamond followed a stratified random sampling design, so it was appropriate to draw inferences to the total sampling universe. However, since the sampling was tied to recent harvesting (1992-1998), the inferences should be restricted to that portion of the total ownership that has experienced significant harvesting in recent years. In addition to descriptive characterizations of these watercourses, the objective was to assess the extent to which major changes occurred in Class IIIs that were responsible for substantial increases in management related sediment production. Caution must always be used when attempting to establish treatment effects or cause and effect relationships using a retrospective study design, but this type of study can be useful in identifying major or gross changes that occurred in Class III watercourses following clearcut timber harvest. It should be noted that most knowledge concerning the impact of timber harvest on geologic or hydrological processes comes from studies that were retrospective in nature. Before-after-control-impact (BACI) experiments (Skalski and Robson 1992; McDonald et al. 2000)) are the only approach to definitively assess the impact of a treatment on a response variable, and there have been few studies that utilize such an experimental approach on landscape level geologic or hydrologic processes.

Despite these limitations, the pre-treatment data set from the Little River HPA indicates that there were not gross differences between treated and untreated control streams for this HPA. This suggests that the results of the retrospective study may be interpreted with greater confidence than might otherwise be possible for a retrospective study. However, it is also recognized that conclusions from this one region may not hold for other HPAs with steeper topography or unconsolidated geology.

C4.4.2 Channel Size

An expected feature of these first order channels associated with Class III watercourses was that they were generally steep with an overall mean channel gradient of 33.4%. However, there was also considerable variation in gradient with a range from 9-80%. The size of the active channel was also quite small with a mean cross-sectional area (product of the channel depth and width measurement) of 0.81 ft^2, which can also be represented by a mean volume (volume of substrate that was transported to produce the existing channel) of 8.07 ft^3/100 feet of channel. In addition, this was a maximum estimate since Green Diamond only measured the maximum depth of the channel at each 10-foot sampling interval. It was also important to note the influence that geology had on the size of Class III channels. Channels with unconsolidated underlying geology (i.e. most of the channels in the Humboldt Bay and Eel River HPAs), had channels approximately twice the cross-sectional area than channels in consolidated geology. Qualitative field observations further support that Class III watercourses were much larger in areas with unconsolidated geology. The suggestion that underlying geology is an important determinant of the size and hydrologic response of Class III watercourses is generally consistent with findings from the Freshwater Watershed Analysis. In Freshwater, Class III channels draining the extremely weak Wildcat Group enlarged significantly following initial harvest, while Class III watercourses in Franciscan Formation sandstones did not. Recent harvest, however, did not appear to have dramatic effects on Class III channels in either of the major bedrock formations (Freshwater Watershed Analysis, Stream Channel Module).

C4.4.3 Exposed Active Channel and Control Points

Observations of EAC can be interpreted as an indicator of fluvial erosion or deposition. The fact that the percentage of the bed showing EAC was correlated with control points suggests that fluvial erosion and deposition processes as expressed by EAC were associated with control points

Green Diamond has no information by which to judge the relative merits of the number of control points per unit length of stream channel identified from this study. A high proportion of control points were made up of LWD, but it was of interest to note that even collections of small woody debris (SWD) could serve as control points in these channels. LWD and SWD in the channel created plunge pools that were responsible for streambed scour immediately below the control point. Evidence for this was provided by the fact that the total number of control points was the first variable to enter the regression model (with a positive coefficient) with EAC as the dependent variable. It is generally thought that although control points may cause scour in short waterfalls immediately below the control point, they prevent overall channel down-cutting. Control points may also correlate with the abundance of roughness elements that cause lateral scour. With a retrospective study, Green Diamond was not able to detect subtle changes in mean channel bed elevation, and apparently, there were sufficient control points in all streams to prevent any major “unraveling” of the channels.

One of the potential effects of harvest is an increase in peak storm runoff in Class III channels. The potential for channel bed erosion (down cutting) is limited by erosion-resistant elements of the channel bed. Roots and rocks (large rock or bedrock) rarely formed control points. LWD was the dominant channel element forming control points in these Class III channels. This is consistent with the conceptualization of Class III channels as ephemeral streams with low sediment transport capacity; these would be expected to be colluvial channels with weak fluvial sorting of hillslope material and relatively fine bed texture. The fact that EAC occurred in only 25% of 10-foot channel measurements also demonstrates that fluvial processes were spatially intermittent in these Class III channels. Consequently, few bedrock or coarse sediment exposures in the channel bed may be expected and proportionately more might be expected in Class II channels or larger Class III channels as suggested by the stepwise regression for channel cross-section area.

The abundance of LWD is significant in relation to the frequency of control points. Green Diamond has no data on the amount or distribution of LWD in Class II watercourses for comparison, but LWD surveys from the smallest Class I watercourses produced a mean of 5-6 pieces per 100 feet of channel in comparison to 4.8 for the Class III watercourses. However, these comparisons may not be appropriate, because the LWD surveys were conducted following different protocols. Green Diamond saw no evidence of transport of LWD in Class III watercourses. LWD was primarily composed of conifer in these Class III channels, which was generally not the case for Class I watercourses. However, this was consistent with the general observation of relatively few hardwoods such as red alder in upslope positions, while alder was a predominant component in many Class I watercourses.

Sites with bank erosion (bare mineral soil on the bank of the channel that was the result of undercutting or sloughing) were generally not large (about 50 ft^2) and did not occur in most channels. Relatively few channels were responsible for most of the bank erosion reported (Figure C4-10). Underlying bedrock geology (more bank erosion in unconsolidated geology), total canopy closure (less bank erosion with greater total canopy) and volume of LWD (more bank erosion with greater of amounts of LWD) were the only dependent variables that entered a stepwise regression analysis of bank erosion versus all appropriate independent variables measured. Increases in bank erosion in unconsolidated geology were expected, as was a decrease in bank erosion with increases in total canopy. (Canopy closure was coming from the regrowth of shrubs and trees since the streams were all in clearcuts with no tree retention.) However, the positive relationship between bank erosion and LWD was not as intuitive. Presumably, LWD directs flow into the banks of the channel thus increasing the sites with bank erosion.

C4.4.4 Slides and Debris Flows

There were relatively few total slides associated with these Class III watercourses and most of the slides occurred in just a few of the channels. In addition, the maximum head scarp distance for 80% of the slides was only 20 feet. It was also notable that there were no debris flows associated with any of these channels even though some had mean stream gradients as high as 80%. Number of LWD control points per 100 feet of channel (positive coefficient), stream gradient (positive coefficient), mean height of ground vegetation (negative coefficient), bank slope (positive coefficient), and number of years since harvest (positive coefficient) were the dependent variables that entered a stepwise regression analysis of the number of landslides versus all appropriate independent variables measured. The positive association between landslides and stream gradient as well as bank slope was predictable, given the importance of slope angle in slope stability. These two variables explained over 40% of the variation in landslides among streams and accounted for over two-thirds of the variation explained by the full regression model. A negative association with ground vegetation might be expected due to increased root strength, but this variable only explained 9% of the variation in the model. Positive correlation between years since harvest and landslide frequency may also be explained relative to root strength (initially declining following harvest), but the variable only explained 7% of the variation in the model making further speculation unwarranted. The potential reason for the positive association between inner gorge landslides and LWD control points was not so intuitive. Green Diamond believes that the apparent association was most likely created by landslides bringing LWD into the channel, and not that LWD in the channel had any direct effect on the rate of landslides. However, once again the variable contributed so little (5%) to explaining variation in the model that conclusions are unwarranted.

C4.5 CONCLUSIONS

This study suggests that there were no gross short-term effects of timber harvest on erosion in and near Class III channels for the period 1992-1998. There were few sites that experienced extensive bank erosion and less than 25% of 10-foot channel intervals contained exposed active channel (EAC). Furthermore, in the 100 sites examined, there were no debris flows. This is significant in that there were several potential triggering storms in 1996 and 1998 and there was above average (generally 120-140% of normal) total rainfall in all years except 1992 and 1994. In addition, 53% of the streams surveyed were harvested from 1996-1998 when the potential effects of increased incident precipitation (caused by reduced forest canopy) on soil erosion should have been greatest immediately following harvest. However, there is an expected lag effect of approximately 5 to 20 years associated with reduced root strength (Zeimer 1981; Sidle 1992), and a concomitant increased rate of landsliding (Sidle et al. 1985, p. 73-76). It may therefore be concluded that under the recent regime of harvest practices, Class III channels were not responding to harvest in the short-term by unraveling and causing the potential for major increases in sedimentation downstream. However, these results do not rule out the possibility that there were increases in sediment production from more subtle and chronic sources, or that a longer period of study might reveal changes not recognized in this investigation. The tendency for most of the sediment production from Class IIIs to be limited to a relatively few streams, particularly in regions with unconsolidated geology, suggest that effective mitigation can be provided by site specific geologic review where conditions warrant.

Since there were no controls, this study was not capable of assessing whether the observed erosion indicators differ significantly from either virgin old growth or advanced second growth forest stand conditions. In particular, it provides no clear evidence regarding whether predicted increases in peak runoff have induced significant increases in rates of fluvial erosion. This study was very similar to the retrospective study of the impact of timber harvest on water temperature in Class II watercourses (see Appendix C5), in that, potential short-term impacts of timber harvest were too subtle to be readily detected with a retrospective study design. That led to a BACI experimental design for Class II water temperature (see Appendix C5), and the BACI design has also recently been initiated for sediment production of Class III watercourses. The initial data set from the Little River HPA suggests that control-treatment comparisons may not show significant effects in that region.

The landslides recorded in this study that delivered sediment to Class III watercourses were associated with steeper stream gradients and bank slopes, shorter vegetation (a combination of silvicultural treatment, site preparation and time since harvest) greater time interval since harvest and more LWD in the channel. These findings were consistent with expectations regarding known triggering mechanisms for landslides (Sidle et al. 1985). The dominant predictor of landslide potential was the slope of the stream and its banks. Collectively it explained over 40% of the variation in landslides among streams and accounted for over two-thirds of the variation explained by the full regression model. However, it was much more difficult to determine potential management effects from this study. To begin with, the two variables that had management implications (height of ground vegetation and time since harvest) collectively only explained a small fraction of the variation of slides among streams. In addition, the height of ground vegetation could represent the influence of multiple management factors. Moreover, height of ground vegetation, had the opposite model coefficient as the direct measurement of time since harvest. It is likely that this retrospective study design is not capable of detecting management effects on landsliding. A more effective study design would include control streams, before-after data or both (BACI experiment).

Without reference or control streams for comparison, it was not possible to assess the quantity of LWD in Class III watercourses in the study area. However, LWD was the predominate element in the formation of channel bed grade control points. In addition, LWD was positively correlated with exposed active channels and bank erosion and, in some cases, with slides. Hence, there was evidence that LWD interacts with fluvial processes in Class III watercourses, but it was not possible to predict the impact of changes in the volume of LWD in Class III watercourses from this study.

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Appendix C5. Water Temperature Monitoring

CONTENTS

Figures

Tables

C5.1 GENERAL WATER TEMPERATURE MONITORING

C5.1.1 1994-1995 Water Temperature Monitoring Program

C5.1.1.1 Objectives

- Document diurnal and seasonal temperature fluctuations;
- Determine maxima and duration of daily peak water temperatures; and
- Identify stream reaches with temperatures that may exceed the thresholds of juvenile salmonids (especially coho salmon).

C5.1.1.2 Methods

Water temperatures were recorded with HOBO® (Onset Computer Corp.) temperature recorders. These devices automatically recorded temperatures at specified time intervals and were left in use for extended periods (up to six months). Two different models where deployed in 1994 and 1995; the HOBO® HTI –05/37^{o}C with an accuracy of +/- 0.2^{o}C and the HOBO®® HTI –37/46^{o}C with an accuracy of +/-0.5^{o}C. No attempt at calibration was made during the first two years of temperature monitoring. The manufacturer's specifications where well within the expected requirements of the temperature monitoring. Each thermograph is capable of recording approximately 1800 data points. The length of deployment depends on the selected recording interval. A recorder launched at a 0.8 hr interval will have to be downloaded and restarted within 45 days and thus runs a risk of missing a peak temperature while the recorder is out of the water. An interval of 1.2 hours records 20 temperatures per day and will last 90 days until the memory is full. The hottest three months of the year (July, August and September) were targeted as the summer monitoring window. To test the assumption that a 1.2 hour interval was enough to catch the entire diurnal range in 1994 three thermographs were launched at an interval of 0.6 hours and placed "piggy-back" on thermographs launched at 1.2 hour intervals. A third data set at 2.4 hours was created by deleting every other record in the 1.2 hr. data set. The 1.2 hour interval accurately represents average temperatures but has the potential to miss the absolute extremes by up to two or three tenths of a degree. Since this is within the accuracy of the thermograph (+/- 0.2^{o}C) it was determined for practical reasons (i.e. deployment length of 90 days) that 1.2 hours was adequate.

The HOBO®s were typically deployed in the upper, middle and lower reaches of the larger streams with fewer HOBO®s in smaller streams. Actual site selection often depended upon property ownership and access issues. In larger streams the lowest monitoring site in the watershed would frequently be just inside Green Diamond's property boundary. The placement of each HOBO® was in the thalweg of a riffle or the head of a pool where water was mixed (to avoid thermal gradients). The HOBO®s were started between mid- June and early July and recorded temperatures throughout the summer months. They were removed between late September to early November. Time intervals of either 1.2 or 0.8 hours were used to accurately capture both diurnal temperature fluctuations and daily maximum temperatures. During the summer of 1994, 40 HOBO® temperature recorders were placed in fish bearing stream reaches distributed throughout Green Diamond's California property in areas that reflect a wide variety of stream conditions. In 1995, 28 Class I reaches where monitored (Table C5-1).

Table C5-1. Watersheds and number of reaches in 1994-1995 temperature monitoring program.

Watershed	No. of Reaches Monitored in 1994	No. of Reaches Monitored in 1995
South Fork Winchuck River	2	1
Rowdy Creek	2	1
South Fork Rowdy	2	0
Dominie Creek	2	0
Wilson Creek	3	1
Hunter Creek	2	2
Turwar Creek	3	3
McGarvey Creek	2	0
Blue Creek	1	1
Potato Patch Creek	1	1
West Fork Blue Creek	2	1
Ah Pah Creek	0	2
Bear Creek	1	3
Tectah	0	2
Tully	0	1
Roach	0	1
Pecwan Creek	1	3
Coyote Creek	1	0
Lindsay Creek	1	1
North Fork Mad River	3	1
Long Prairie Creek	1	0
Dry Creek	1	0
Cañon Creek	3	1
Maple Creek	1	0
Boulder Creek	1	1
Jacoby Creek	2	0
Salmon Creek	2	1

C5.1.1.3 Results

The 1994-95 monitoring effectively documented both diurnal (the difference between daily maximum and minimum temperatures) and seasonal temperature variations. Green Diamond calculated maximum weekly average temperatures (MWAT) for the 1994-1996 data as defined by the 1997 document Aquatic properly functioning condition matrix, a.k.a. the "Inter-Agency Matrix" (NMFS 1997). MWATs were generated by identifying the 7-day interval with the peak temperature and then calculating a mean temperature from all the data points recorded by the HOBO® device. For example, because Green Diamond has set their HOBO®s to record temperatures at 1.2 hour intervals (20 recordings for a 24-hour period), a MWAT would be the average of 140 data points for the hottest 7-day interval of the monitoring period. The MWAT for that creek was to be compared to established MWAT thresholds for a specific life stage and species. The MWAT threshold of 17.4^{o}C for Coho summer rearing habitat was suggested in the "Inter-Agency Matrix" document. The temperature data indicated that on a Plan Area scale summer water temperatures were probably not limiting summer rearing habitat for salmonids. Of the 68 monitoring sites in 1994 and 95, 94% were below the suggested MWAT threshold of 17.4^{o}C. The four sites that exceeded the MWAT of 17.4^{o}C were all large order streams with watersheds more than 15,000 acres upstream of the recorder. (See Tables C5-2 through C5-12 for a complete summary of Green Diamond's Class I and Class II summer temperature monitoring). Green Diamond believes that the single MWAT threshold value failed to account for natural variations in water temperature due to geographic location and drainage area of the monitored sub-basin. Also, depending on the method used to test the upper incipient lethal temperature of juvenile salmonids, a critical MWAT can range from 16.8^{o}C to 18.4^{o}C (Armour 1991; Thomas et al. 1986; Becker and Genoway 1979).

Following the 1994-5 temperature monitoring seasons improvements to the temperature monitoring protocol included collecting information relating to riparian canopy closure, channel morphology, flow and drainage area above the location of HOBO®s. Temperature monitoring was continued annually in selected stream reaches, either those that exhibit excessive temperatures or those of special biological significance (extremely diverse or abundant salmonid populations). In 1995 Green Diamond conducted some experimental Class II temperature monitoring which was formalized and expanded in 1996.

C5.1.2 Water Temperature Monitoring Program (1996 to the Present)

C5.1.2.1 Objectives

- Document the highest:

  (a) 7DMAVG (highest 7-day moving average of all recorded temperatures),

  (b) 7DMMX (highest 7-day moving average of the maximum daily temperatures),

  (c) seasonal temperature fluctuations for each site for both Class I and Class II watercourses.

- Identify stream reaches with temperatures that have the potential to exceed the monitoring thresholds relative to the drainage area above the monitoring site for both Class I and Class II watercourses. (To account for the relationship between water temperature and drainage area, water temperature was regressed on the square root of watershed drainage area at locations known to support populations of southern torrent salamanders, tailed frogs or coho salmon throughout Green Diamond's ownership in the HPAs.

- Directly assess the effects of timber harvest on water temperatures in Class II watercourses (Before, After, Control, Impact [BACI] experiments).

One of the major changes in the monitoring protocols occurred in the analysis of the data. Initially the analysis of the MWAT was a manual search through the data file to find the seasonal peak and then it was assumed that the encompassing seven-day period would provide the highest average temperature. This process was automated in 1996 with an Excel Macro that actually calculated the average for every 7-day period and then selected the highest average as the critical metric.

Table C5-2. Summer water temperature monitoring summary, Smith River HPA.

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
Goose (high)	13020101	1	1999	1:12	14.0	8/26	15.1	8/25	16.2	7/13	12.8	297.6
Goose (high)	13020101	1	2000	1:12	14.8	8/2	16.2	8/2	16.7	8/8	13.9	297.6
Wilson (low)	14010801	1	1994	0:48	12.2	7/4	13.7	7/4	13.9	6/27	11.4	7930.0
Goose (low)	14020201	1	1998	1:12	17.0	7/25	19.0	7/25	19.8	7/26	15.9	14752.0
Goose Low	14020201	1	1999	1:12	16.0	8/26	17.7	8/25	19.0	7/13	14.6	14752.0
Goose Trib	14020202	1	1998	1:12	15.4	7/25	16.8	7/24	17.8	7/26	14.6	4197.0
Goose Trib	14020202	1	1999	1:12	14.8	8/26	16.0	8/25	16.5	7/13	13.3	4197.0
Goose Mid	14022601	1	1999	1:12	14.8	8/26	15.8	8/25	16.1	8/25	14.1	663.6
Goose Mid	14022601	1	2000	1:12	15.5	8/2	16.8	8/2	17.3	8/1	14.7	663.6
Goose, East Fork	14022602	1	1999	1:12	13.2	8/26	13.7	8/25	13.9	8/25	12.6	1114.1
Goose, East Fork	14022602	1	2000	1:12	13.9	8/2	14.5	8/2	14.9	8/3	13.2	1114.1
Wilson Trib.	14510401	2	1996	1:12	12.1	8/30	12.5	8/30	12.7	8/30	11.9	679.1
Wilson (high)	15012901	1	1994	0:48	13.6	8/15	14.2	8/16	14.5	8/13	13.3	2146.0
Wilson (mid)	15013201	1	1994	0:48	13.4	8/16	14.0	8/16	14.2	8/13	13.1	3961.0
Wilson (mid)	15013201	1	1995	0:48	13.8	8/4	14.5	7/30	14.8	7/31	13.3	3961.0
Wilson (mid)	15013201	1	1996	1:12	13.8	8/30	16.1	8/30	16.5	8/30	12.9	3961.0
Wilson (mid)	15013201	1	1997	1:12	14.3	9/3	15.3	9/2	15.4	8/27	13.7	3961.0
Wilson (mid)	15013201	1	1998	0:08	13.8	8/15	14.4	8/13	14.8	7/26	13.4	3961.0
Wilson (mid)	15013201	1	1999	1:12	13.7	8/27	14.1	8/27	14.2	8/26	13.6	3961.0
Wilson (mid)	15013201	1	2000	1:12	13.7	9/22	14.5	7/31	14.6	7/29	13.3	3961.0
Goose (really low)	15023501	1	1997	1:12	17.3	8/9	19.6	8/9	20.6	8/7	16.1	22067.7
Little Mill	17010701	1	1998	1:12	13.5	8/13	14.2	8/13	14.7	7/26	13.0	2274.0
Little Mill	17010701	1	1999	1:12	13.5	8/27	14.0	8/25	14.2	8/26	13.6	2274.0
Sultan	17011901	1	1997	1:12	15.2	8/5	16.9	8/7	17.7	8/7	13.6	1281.0
Sultan	17011901	1	1999	1:12	13.7	8/26	14.2	8/25	14.5	8/26	13.7	1281.0
Peacock	17012901	1	1998	1:12	13.2	9/15	14.0	8/13	14.5	7/26	12.7	846.0
Peacock	17012901	1	2000	1:12	14.0	9/16	14.6	9/16	14.5	6/28	12.6	846.0
Campsix	17512501	1	1998	1:12	13.2	8/13	13.5	8/13	13.8	8/12	13.0	233.0
Campsix	17512501	1	1999	1:12	13.3	8/27	13.5	8/27	13.7	8/26	13.3	233.0
SF Winchuck Trib #1	18010601	2	1995	1:12	13.4	8/2	14.3	8/2	14.6	7/31	13.1	13.7
SF Winchuck Trib #2	18010602	2	1995	1:12	13.2	8/3	13.6	8/2	13.9	7/31	13.3	24.6
Rowdy trib R1700	18010901	2	1999	1:12	13.5	8/25	13.8	8/25	14.2	8/27	13.3	124.2
Rowdy trib R1700	18010901	2	2000	1:12	13.1	6/27	13.7	6/26	14.3	6/27	13.3	124.2

Table C5-2 Continued. Summer water temperature monitoring summary, Smith River HPA.

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
Rowdy (high)	18011601	1	1994	1:12	14.8	8/15	16.3	8/15	16.5	7/14	13.1	7667.0
Rowdy (high)	18011601	1	1999	1:12	14.9	8/26	15.8	7/12	16.6	7/12	13.3	7667.0
Rowdy (high)	18011601	1	2000	1:12	15.6	9/19	16.9	9/19	17.8	9/19	15.8	7667.0
Ravine	18011701	1	1998	0:08	13.6	9/15	14.3	9/3	15.0	9/13	13.8	803.0
Ravine	18011701	1	2000	1:12	14.3	9/19	14.8	9/19	15.6	9/19	14.8	803.0
Rowdy trib. #3	18011801	2	1995	1:12	12.9	8/2	13.2	8/2	13.6	7/31	13.1	72.5
Rowdy trib. #4	18011901	2	1995	1:12	14.6	9/21	17.6	9/21	19.8	9/20	13.4	260.4
Rowdy Trib	18012001	2	1996	1:12	12.7	8/31	13.3	8/31	13.4	8/30	12.0	296.5
Rowdy (low)	18013001	1	1994	1:12	16.4	8/14	19.0	8/15	19.8	7/14	12.8	12587.0
Rowdy (low)	18013001	1	1995	1:12	16.6	8/3	19.3	8/3	19.4	8/19	13.7	12587.0
Rowdy (low)	18013001	1	1996	1:12	16.2	7/28	18.6	7/27	19.4	7/21	14.2	12587.0
Rowdy (low)	18013001	1	1997	1:12	16.5	8/5	19.1	8/5	19.9	8/7	14.8	12587.0
Rowdy (low)	18013001	1	1999	1:12	15.5	8/27	17.4	7/12	18.1	7/12	13.4	12587.0
Rowdy (low)	18013001	1	2000	1:12	15.6	7/31	17.4	7/30	18.3	8/2	14.3	12587.0
SF Rowdy (low)	18013002	1	1994	1:12	13.7	8/14	14.9	8/15	15.2	9/20	12.5	2573.0
SF Rowdy (low)	18013002	1	1997	1:12	14.3	8/27	15.6	8/5	16.1	8/7	13.3	2573.0
Savoy	18013003	1	1998	0:08	14.0	8/13	15.0	8/13	15.4	9/12	12.9	2573.0
Savoy	18013003	1	1999	1:12	13.7	8/27	14.4	8/25	14.6	8/22	13.1	2573.0
SF Rowdy (high)	18013004	1	1998	0:08	14.1	8/13	15.1	8/13	15.5	9/12	13.2	1552.0
SF Rowdy (high)	18013004	1	1999	1:12	13.7	8/27	14.4	8/24	14.6	7/12	12.1	1552.0
Savoy (high)	18013201	1	1994	1:12	13.1	8/16	13.8	8/15	14.0	8/16	12.5	2264.1
SF Winchuck River (high)	18510101	1	1994	1:12	13.2	8/16	13.8	8/15	14.3	9/21	12.5	1193.1
SF Winchuck River (high)	18510101	1	1999	1:12	13.4	8/25	13.8	8/25	14.4	7/12	12.9	1193.1
Gilbert	18510401	1	1997	1:12	14.6	9/3	15.5	9/2	15.9	9/5	13.9	1506.7
Gilbert	18510401	1	1999	1:12	13.2	8/27	13.8	8/24	14.1	8/21	12.7	1506.7
D2010CD	18511101	2	1996	1:12	12.2	10/9	12.5	10/8	13.1	10/8	12.5	10.5
D2010 CD	18511101	2	1997	1:12	12.3	8/7	12.8	8/7	13.7	8/7	12.8	10.5
D2010 CD	18511101	2	1998	1:12	12.6	9/3	13.1	9/3	13.9	9/3	12.9	10.5
D2010 CD	18511101	2	1999	1:12	12.1	8/25	12.5	8/25	13.4	7/12	11.2	10.5
D2010 CD	18511101	2	2000	1:12	10.9	9/19	11.0	9/19	11.2	9/19	10.9	10.5
D2010CU	18511102	2	1996	1:12	10.9	10/10	10.9	10/7	11.1	9/14	10.8	1.6

Table C5-2 Continued. Summer water temperature monitoring summary, Smith River HPA.

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
D2010 CU	18511102	2	1997	1:12	11.0	9/14	11.2	9/14	11.4	8/7	11.1	1.6
D2010 CU	18511102	2	1998	1:12	11.9	9/11	12.1	9/11	12.0	9/2	11.9	1.6
D2010 CU	18511102	2	1999	1:12	10.6	8/27	10.7	9/10	10.8	8/26	10.8	1.6
D2010TD	18511103	2	1996	1:12	12.4	10/8	12.7	10/8	13.7	10/8	12.2	37.3
D2010 TD	18511103	2	1997	1:12	13.1	8/6	13.8	8/6	15.2	8/7	13.4	37.3
D2010 TD	18511103	2	1998	1:12	13.4	9/3	14.0	9/3	14.8	9/3	13.7	37.3
D2010 TD	18511103	2	1999	1:12	12.6	8/25	12.9	8/25	13.7	7/12	12.0	37.3
D2010 TD	18511103	2	2000	1:12	13.3	9/18	13.7	9/18	14.5	9/19	14.0	37.3
D2010TU	18511104	2	1996	1:12	11.6	10/10	11.7	10/10	12.0	10/8	12.0	7.3
D2010 TU	18511104	2	1997	1:12	11.6	9/28	11.7	8/8	12.0	8/7	12.0	7.3
D2010 TU	18511104	2	1998	1:12	12.2	9/4	12.4	9/4	12.8	9/3	12.5	7.3
D2010 TU	18511104	2	1999	1:12	11.2	8/26	11.3	8/25	11.7	8/27	11.2	7.3
D2010 TU	18511104	2	2000	1:12	11.9	9/19	12.1	9/19	12.5	9/19	12.3	7.3
D1120TD	18511105	2	1996	1:12	13.0	10/8	13.5	10/8	14.6	10/8	13.1	71.5
D1120 TD	18511105	2	1997	1:12	13.1	9/3	13.5	9/8	14.3	8/8	12.5	71.5
D1120 TD	18511105	2	1998	1:12	13.2	9/12	13.7	9/12	14.3	9/12	14.0	71.5
D1120 TD	18511105	2	1999	1:12	12.5	8/25	12.8	8/25	13.3	8/27	12.5	71.5
D1120 TD	18511105	2	2000	1:12	14.7	9/18	15.5	9/18	16.8	9/19	15.4	71.5
Dominie (high)	18511201	1	1994	1:12	11.9	9/19	12.1	7/10	12.5	9/20	12.0	394.5
D1120TU	18511202	2	1996	1:12	12.2	10/8	12.5	10/7	13.4	10/8	12.0	14.4
D1120 TU	18511202	2	1997	1:12	12.1	9/26	12.5	9/26	13.1	8/7	12.2	14.4
D1120 TU	18511202	2	1998	1:12	12.4	9/4	12.8	9/12	13.3	9/12	12.9	14.4
D1120 TU	18511202	2	1999	1:12	11.9	8/25	12.1	8/25	12.5	8/27	12.0	14.4
D1120 TU	18511202	2	2000	1:12	12.6	9/19	12.9	9/19	13.6	9/19	13.2	14.4
D1120CU	18511203	2	1996	1:12	13.0	10/8	13.4	10/8	14.6	10/8	13.1	17.7
D1120 CU	18511203	2	1997	1:12	12.7	9/8	13.1	8/6	14.0	8/7	13.4	17.7
D1120 CU	18511203	2	1998	1:12	13.4	9/4	13.8	9/12	14.3	9/3	13.9	17.7
D1120 CU	18511203	2	1999	1:12	12.7	8/25	13.1	8/25	13.7	8/26	13.3	17.7
D1120 CU	18511203	2	2000	1:12	14.0	9/18	14.5	9/19	15.6	9/19	14.8	17.7
D1120CD	18511204	2	1996	1:12	12.5	10/10	12.9	8/31	13.7	10/8	12.5	33.7
D1120 CD	18511204	2	1997	1:12	12.9	9/3	13.2	9/3	13.7	8/7	13.1	33.7
D1120 CD	18511204	2	1998	1:12	13.1	9/12	13.7	9/12	14.5	9/12	13.7	33.7
D1120 CD	18511204	2	1999	1:12	12.3	8/25	12.7	8/25	13.1	8/27	12.2	33.7
D1120 CD	18511204	2	2000	1:12	14.1	9/19	14.7	9/19	15.9	9/20	13.2	33.7
Dom Trib. #1	18511401	2	1995	1:12	13.3	7/30	14.0	8/1	14.6	7/31	13.1	40.8
Dom Trib. #2	18511402	2	1995	1:12	14.4	8/2	18.5	8/3	20.7	7/31	12.8	15.2
Dom Trib. #3	18511403	2	1995	1:12	13.9	9/22	14.8	9/21	16.9	9/19	14.8	38.7

Table C5-2 Continued. Summer water temperature monitoring summary, Smith River HPA.

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
Dom Trib. #4	18511404	2	1995	1:12	13.7	8/1	14.5	8/1	15.8	7/31	13.7	4.7
Dominie trib	18511405	2	1996	1:12	13.3	8/31	14.0	8/31	14.3	8/30	12.5	347.9
Dominie (low)	18512301	1	1994	1:12	14.3	8/15	16.0	8/16	16.2	8/13	13.1	2254.0
Dominie (low)	18512301	1	1997	1:12	14.7	9/3	15.8	9/3	16.4	9/5	13.6	2254.0
Dominie (low)	18512301	1	1998	0:08	14.2	8/13	15.4	8/13	15.7	7/26	13.3	2254.0
SF Winchuck River (low)	19513301	1	1994	1:12	14.5	8/15	16.0	8/16	16.5	8/31	12.2	5891.0
SF Winchuck River (low)	19513301	1	1995	1:12	14.7	8/3	16.1	8/3	16.9	8/19	13.4	5891.0
SF Winchuck River (low)	19513301	1	1996	1:12	14.8	8/31	16.5	8/30	17.5	9/1	13.4	5891.0
SF Winchuck River (low)	19513301	1	1997	1:12	15.5	9/3	16.9	8/5	17.7	8/7	14.5	5891.0
SF Winchuck River (low)	19513301	1	1998	0:08	14.7	8/14	16.6	9/14	18.4	9/11	12.2	5891.0
SF Winchuck River (low)	19513301	1	1999	1:12	14.1	8/16	15.1	8/23	15.8	6/22	12.5	5891.0
SF Winchuck River (low)	19513301	1	2000	1:12	15.3	9/18	16.5	9/18	17.8	9/20	14.1	5891.0

Table C5-3. Summer water temperature monitoring summary, Coastal Klamath HPA.

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
Little Surpur	11020201	1	1996	1:12	15.3	7/28	16.2	7/28	16.4	7/30	14.6	1601.0
Little Surpur	11020201	1	1998	1:12	14.8	7/25	15.3	7/24	15.8	7/26	14.6	1601.0
Little Surpur	11020201	1	1999	1:12	15.1	8/26	16.2	8/26	16.8	9/11	12.2	1601.0
Surpur	11020301	1	1996	0:30	14.6	7/28	15.8	7/28	16.1	7/30	13.9	3236.6
Surpur	11020301	1	1997	1:12	14.4	9/2	15.6	8/6	16.1	8/8	14.0	3236.6
Surpur	11020301	1	1999	1:12	14.0	8/26	14.7	8/25	14.9	8/24	12.8	3236.6
Tectah (old)	11021201	1	1995	0:48	16.2	8/4	18.2	8/3	18.6	8/4	15.1	11413.0
Tectah (low)	11021202	1	1995	1:12	16.3	8/4	18.5	8/3	19.1	8/4	15.2	11413.0
Tectah (low)	11021202	1	1997	1:12	16.3	8/16	16.6	8/15	16.7	8/8	15.9	11413.0
Tectah (low)	11021202	1	1998	1:12	16.9	7/25	18.7	7/25	19.4	7/26	16.1	11413.0
Tectah (low)	11021202	1	1999	1:12	16.2	8/26	17.4	7/11	18.1	7/12	14.6	11413.0
Tectah (low)	11021202	1	2000	1:12	17.3	8/4	18.8	8/1	19.5	6/28	15.4	11413.0
Tectah Trib. (class II)	11021301	2	1996	1:12	13.1	8/26	13.4	8/25	13.9	8/24	13.1	189.5
Tectah (mid)	11023401	1	1995	0:48	15.1	8/4	16.5	8/3	17.0	8/4	14.8	6892.5
Tectah (mid)	11023401	1	1997	1:00	15.6	8/6	17.1	8/6	17.9	8/8	14.9	6892.5
Tectah (mid)	11023401	1	1999	1:00	15.4	8/26	16.5	8/25	16.7	7/13	13.9	6892.5
McGarvey (high)	12010201	1	1994	0:48	12.8	8/17	13.3	8/16	13.4	8/13	12.8	1337.4
NF Ah Pah Trib. (161_up)	12020901	2	1996	1:12	13.3	7/29	13.6	7/28	13.9	7/30	13.1	616.7
NF Ah Pah Trib. (161_up)	12020901	2	1997	1:12	13.4	9/3	13.5	9/1	13.6	8/7	13.3	616.7
Ah Pah (North Fork)	12021601	1	1995	1:12	14.8	8/4	15.8	8/3	16.2	8/4	14.3	670.0
Ah Pah (North Fork)	12021602	1	1996	0:30	15.0	7/28	16.1	7/28	16.4	7/30	14.1	670.0
Ah Pah (North Fork)	12021602	1	1997	0:30	14.7	8/6	15.5	8/6	16.1	8/8	14.2	670.0
NF Ah Pah Trib. (161_lo)	12021603	2	1996	1:12	13.9	7/28	14.5	7/28	14.8	7/30	13.3	669.7
NF Ah Pah Trib. (161_lo)	12021603	2	1997	1:12	14.0	9/3	14.5	8/6	14.8	8/7	13.7	669.7
Ah Pah (South Fork)	12022101	1	1995	1:12	14.2	8/4	15.5	8/2	15.9	8/4	13.4	1501.0
Ah Pah (South Fork)	12022101	1	1996	0:30	14.0	7/30	14.8	7/28	14.9	7/29	13.8	1501.0
Ah Pah (South Fork)	12022101	1	1997	0:30	14.0	9/4	14.2	9/4	14.2	8/7	13.7	1501.0
Ah Pah (South Fork)	12022101	1	1999	1:00	13.4	8/27	13.7	8/27	13.8	8/28	13.5	1501.0
Ah Pah (Middle Fork)	12022103	1	1996	0:30	15.2	7/31	15.6	8/25	15.4	7/29	15.0	3068.0
Ah Pah (Middle Fork)	12022103	1	1997	0:30	15.9	8/6	18.3	8/6	18.7	8/7	14.9	3068.0

Table C5-3 Continued. Summer water temperature monitoring summary, Coastal Klamath HPA.

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
Bear Creek Trib	12022401	2	1996	1:12	13.4	8/18	13.7	8/18	14.1	8/24	13.1	435.2
Bear (Klamath)	12022501	1	1995	0:48	14.1	8/4	14.9	8/3	15.1	8/4	13.9	2659.0
Bear (Klamath)	12022601	1	1994	0:48	13.7	8/17	14.4	8/16	14.5	8/13	13.3	5343.0
Bear (Klamath)	12022601	1	1995	0:48	14.4	8/4	15.2	8/3	15.4	8/4	13.9	5343.0
Bear (Klamath)	12022601	1	1996	0:30	14.9	7/28	16.0	7/28	16.2	7/29	14.2	5343.0
Bear (Klamath)	12022601	1	1996	1:12	14.9	7/28	15.9	7/28	16.2	7/30	14.2	5343.0
Bear (Klamath)	12022601	1	1997	0:30	14.8	8/9	15.7	8/6	16.0	8/8	14.3	5343.0
Bear (Klamath)	12022601	1	1999	1:00	14.1	8/26	14.8	8/25	15.0	8/26	14.0	5343.0
Bear, South Fork(Klamath)	12033101	1	1995	0:48	13.2	8/4	13.7	8/3	14.0	8/4	13.1	1216.5
Hunter	13010402	1	1995	0:30	12.2	6/23	13.6	6/22	14.2	6/20	11.2	13710.7
Hunter	13010402	1	1999	0:30	12.2	7/14	13.9	7/14	14.3	6/22	10.9	13710.7
McGarvey (low)	13012401	1	1994	0:48	13.4	7/20	14.3	7/19	14.5	7/7	12.8	4808.0
McGarvey (low)	13012501	1	1996	0:30	14.6	7/28	15.7	7/27	15.9	7/29	14.0	4808.0
McGarvey (low)	13012501	1	1999	0:30	15.0	8/26	16.4	8/25	16.8	8/25	14.6	4808.0
Turwar (low)	13020501	1	1994	0:48	17.6	8/16	19.7	8/16	19.9	8/14	16.1	16746.0
Turwar (low)	13020501	1	1995	0:48	16.9	8/4	18.7	8/2	19.1	7/16	15.1	16746.0
Turwar (low)	13020501	1	1996	1:12	17.2	7/28	18.9	7/27	19.3	7/29	15.8	16746.0
Turwar (low)	13020501	1	1997	1:12	17.4	8/6	19.1	8/5	19.6	8/7	16.2	16746.0
Turwar (low)	13020501	1	1998	1:12	17.0	8/15	18.4	8/13	19.0	7/26	15.8	16746.0
Turwar (low)	13020501	1	1999	1:12	16.6	8/25	18.6	7/12	19.1	7/13	14.8	16746.0
Turwar (low)	13020501	1	2000	1:12	17.2	8/1	19.3	8/1	19.7	8/1	16.0	16746.0
Tarup	13022901	1	1996	0:30	13.6	7/28	14.2	7/28	14.3	7/29	13.4	3098.0
Omagaar	13023201	1	1996	0:30	13.5	7/28	14.0	7/28	14.2	7/29	13.1	857.2
Hunter (mid)	14010201	1	1994	0:48	13.5	8/16	14.2	8/16	14.3	8/13	13.3	3197.6
Hunter (mid)	14010201	1	1999	1:00	14.3	8/27	14.9	8/27	15.1	8/22	13.8	3197.6
Hunter (mid)	14010201	1	1999	1:12	14.1	8/27	14.5	8/25	14.6	8/22	13.7	3197.6
Kurowitz	14010202	1	1999	1:12	14.5	8/26	15.3	7/12	15.9	7/12	12.9	864.9
Kurowitz	14010202	1	2000	1:12	15.2	9/19	16.4	7/31	17.0	9/19	15.3	864.9
Hunter Trib.	14011101	2	1996	1:12	13.1	8/30	15.5	8/31	15.9	9/1	11.4	608.2
Hunter (low)	14011102	1	1995	0:48	14.9	8/4	17.0	8/2	17.2	7/31	13.6	5701.0
Hunter (low)	14011102	1	1996	1:12	14.9	7/28	16.8	7/28	17.0	7/29	13.9	5701.0
Hunter (low)	14011102	1	1997	1:12	15.3	8/6	17.6	8/5	18.0	8/7	14.2	5701.0
Hunter (low)	14011102	1	1998	0:08	15.2	8/14	17.0	8/13	17.5	7/26	14.2	5701.0
Hunter (low)	14011102	1	2000	1:12	15.6	7/31	18.2	7/31	18.5	8/1	14.4	5701.0
Mynot	14013501	1	1999	1:12	13.4	8/26	14.0	8/22	14.3	6/22	11.4	516.7
Turwar (high)	14020601	1	1994	0:48	13.2	7/31	13.7	7/31	13.7	7/31	12.8	1317.0
Turwar (high)	14020601	1	1995	0:48	14.4	8/4	14.9	8/3	15.4	8/4	14.6	1317.0

Table C5-3 Continued. Summer water temperature monitoring summary, Coastal Klamath HPA.

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
Turwar Trib. (class II)	14020602	2	1996	1:12	14.3	8/14	14.8	8/14	14.9	8/15	14.0	369.1
Turwar (mid)	14022101	1	1994	0:48	17.0	8/15	19.2	7/16	19.6	7/18	15.0	7606.0
Turwar (mid)	14022101	1	1995	0:48	16.8	8/4	19.1	8/3	19.4	8/4	15.6	7606.0
Turwar	14022102	1	1997	1:00	16.4	8/6	17.9	7/21	18.7	7/19	15.0	8238.0
Turwar	14022102	1	1999	1:00	16.2	8/25	17.9	8/23	18.6	7/12	14.4	8238.0
SF Turwar	14022901	1	2000	1:12	14.4	9/19	15.2	9/19	16.3	9/19	13.9	5091.2
Hunter (high)	15013501	1	1994	0:48	14.0	8/16	14.7	8/16	14.8	7/18	12.6	2163.2
Hunter (high)	15013501	1	1995	0:48	14.3	8/7	15.5	8/7	15.8	8/7	13.1	2163.2

Table C5-4. Summer water temperature monitoring summary, Blue Creek HPA.

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
Blue (West Fork)	12020101	1	1994	0:48	14.2	8/16	15.8	8/16	15.9	8/13	13.4	8616.0
Blue (West Fork)	12020101	1	1996	1:12	15.3	7/28	16.8	7/28	17.0	7/30	14.5	8616.0
Blue (West Fork)	12020101	1	1997	1:00	15.2	8/6	16.9	8/6	17.3	8/7	14.5	8616.0
Blue (West Fork)	12020101	1	1999	1:00	15.1	8/26	16.5	8/23	16.8	8/22	14.3	8616.0
Blue	12021101	1	1994	0:48	17.2	8/15	19.8	7/16	20.2	7/18	14.8	78520.0
Blue	12021101	1	1995	0:48	17.3	8/4	19.9	8/3	20.6	8/4	15.6	78520.0
Blue	12021101	1	1997	0:30	18.0	8/6	20.9	8/6	21.4	8/7	16.1	78520.0
Blue	12021101	1	1999	0:30	17.1	8/23	19.5	8/23	20.1	8/22	15.6	78520.0
Slide	12021401	1	1997	1:00	14.3	8/6	16.1	8/6	16.4	8/7	13.6	78520.0
Slide	12021401	1	1999	1:00	13.5	8/25	15.4	7/12	16.0	7/12	12.4	78520.0
Nickowitz	12030301	1	1996	0:30	14.4	7/28	15.3	7/28	15.5	7/30	13.6	9693.0
Nickowitz	12030301	1	1997	1:00	14.2	8/7	15.2	8/6	15.7	8/8	13.8	9693.0
Nickowitz	12030301	1	1999	1:00	13.7	8/26	14.4	8/25	14.7	7/13	12.7	9693.0
Coyote (Blue Cr.)	12031701	2	1996	1:12	11.5	8/26	12.0	8/25	12.8	8/24	11.3	435.2
Dandy	13020801	1	2000	1:12	13.3	9/16	13.8	9/16	13.7	9/13	12.9	1244.3
Blue (West Fork)	13022301	1	1994	0:48	12.9	7/19	13.5	7/19	13.7	7/17	12.3	1389.0
Potato Patch (185_up)	13022501	2	1996	1:12	14.1	7/28	15.6	7/28	15.8	7/25	13.1	482.5
Potato Patch (185_up)	13022501	2	1997	1:12	15.1	8/10	16.7	8/9	17.3	8/8	14.6	482.5
Potato Patch	13023601	1	1994	0:48	14.0	8/11	14.5	8/10	14.6	8/9	13.9	1782.0
Potato Patch	13023601	1	2000	1:12	14.5	9/20	14.9	9/19	15.6	9/20	14.3	1782.0
Blue	13033401	1	1996	0:30	17.1	7/28	18.7	7/28	19.0	7/30	15.5	31753.1
Blue	13033401	1	1997	1:00	16.7	8/9	18.2	8/6	18.7	8/8	15.7	31753.1
Blue	13033401	1	1999	1:00	15.5	8/26	16.5	8/25	17.2	7/13	13.8	31753.1
Crescent City Fork	13033402	1	1996	0:30	15.2	7/28	16.5	7/28	16.6	7/28	14.1	14343.1
Crescent City Fork	13033402	1	1997	1:00	14.7	8/9	15.9	8/6	16.5	8/8	14.2	14343.1
Crescent City Fork	13033402	1	1999	1:00	14.2	8/26	15.0	8/25	15.8	7/13	13.0	14343.1
Potato Patch (185_lo)	13033801	2	1996	1:12	15.4	7/28	16.0	7/28	16.2	7/28	15.0	1079.0
Potato Patch (185_lo)	13033801	2	1997	1:12	15.3	8/10	16.0	8/9	16.7	8/8	15.1	1079.0

Table C5-5. Summer water temperature monitoring summary, Interior Klamath HPA.

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
Tully (high)	09030301	1	1999	1:12	14.5	8/26	15.4	8/26	16.2	7/13	13.4	1096.2
Tully	09030301	1	2000	1:12	14.9	8/3	16.1	8/3	16.4	8/2	14.2	1096.2
Pine	09040501	1	1999	1:00	17.4	8/26	18.2	8/25	19.3	7/13	16.3	31200.7
Mettah (high)	10021001	1	1999	1:12	14.0	8/26	14.6	8/26	14.8	8/28	13.9	362.6
Mettah (high)	10021001	1	2000	1:12	14.1	8/5	15.0	8/4	15.3	8/3	13.7	362.6
Roach (upper)	10022501	2	1997	1:12	16.4	8/9	18.6	8/6	19.6	8/8	15.3	10808.1
Cappell	10030301	1	1996	0:30	16.4	7/28	17.3	7/28	17.5	7/28	15.3	5253.1
Roach	10030801	1	1995	0:48	18.2	7/22	20.3	7/22	21.5	7/24	17.8	18613.0
Roach	10030801	1	1996	0:30	20.1	7/28	22.1	7/28	22.4	7/27	19.1	18613.0
Morek	10030901	1	1996	1:12	14.7	7/28	15.5	7/28	15.6	7/26	14.3	2561.9
Waukell (past Tectah)	10032301	2	1996	1:12	12.9	8/13	13.2	8/13	13.4	8/13	12.5	153.9
Tully (low)	10032501	1	1995	0:48	16.1	8/4	17.1	8/4	18.0	8/5	15.3	11085.0
Tully (low)	10032501	1	1997	1:00	16.6	8/9	17.8	8/9	18.5	8/8	16.2	11085.0
Johnson (188_lo)	11022401	2	1996	1:12	13.7	7/28	14.1	7/28	14.2	7/26	13.4	907.7
Johnson (188_lo)	11022401	2	1997	1:12	13.2	8/8	13.7	8/7	14.2	8/8	13.3	907.7
Johnson (188_up)	11022402	2	1996	1:12	13.3	7/28	13.7	7/28	13.9	7/30	12.8	770.9
Johnson (188_up)	11022402	2	1997	1:12	12.8	8/7	13.2	8/7	13.9	8/8	12.9	770.9
Johnson	11022403	1	1997	1:12	13.2	8/27	13.3	8/24	13.4	9/17	13.1	1940.5
Mettah (low)	11023601	1	1996	1:12	16.1	7/28	17.2	7/28	17.3	7/27	15.6	6180.5
Clirliah Trib	11023602	2	1996	1:12	13.6	8/26	13.9	8/25	14.2	8/24	13.6	259.5
Pecwan, West Fork	11030901	1	1995	0:48	12.9	8/4	13.4	8/3	14.2	8/4	13.3	7473.8
Pecwan, West Fork	11030901	1	1999	1:12	12.5	8/26	12.8	8/26	13.1	8/26	12.6	7473.8
Pecwan, West Fork	11030901	1	2000	1:12	12.4	8/3	12.8	6/26	13.3	8/2	12.3	7473.8
Pecwan, East Fork	11031501	1	1995	0:48	12.7	8/4	13.2	8/3	14.2	8/4	12.9	6585.0
Pecwan, East Fork	11031501	1	1999	1:12	12.7	8/26	13.0	8/26	13.4	8/26	12.9	6585.0
Pecwan, East Fork	11031501	1	2000	1:12	12.6	8/4	13.1	6/26	13.6	6/28	12.3	6585.0
Pecwan	11031701	1	1994	0:48	14.1	7/20	15.0	7/19	15.3	7/17	13.3	17594.0
Pecwan	11031701	1	1995	0:48	14.4	8/7	15.6	8/7	17.8	8/4	14.6	17594.0
Pecwan	11031701	1	1996	0:30	15.7	7/28	17.0	7/28	17.3	7/30	14.8	17594.0
Pecwan	11031701	1	1999	1:00	14.7	8/26	16.0	8/26	16.3	8/26	14.4	17594.0

Table C5-6. Summer water temperature monitoring summary, Redwood Creek HPA.

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
Lake Prairie	05041901	2	1997	2:24	15.4	8/9	17.5	8/9	18.5	8/7	14.6	973.5
Lake Prairie	05041901	2	1998	2:30	15.6	7/21	17.1	7/21	17.8	7/21	14.9	973.5
Lake Prairie	05041901	2	1999	1:12	14.4	8/26	15.9	7/12	17.1	7/13	13.6	973.5
Lake Prairie	05041901	2	2000	0:36	15.1	8/2	16.4	6/26	17.0	6/28	13.5	973.5
Redwood at Miñon	05042001	1	2000	1:12	17.7	8/5	20.6	8/5	20.9	8/3	15.9	18416.6
Pardee	05043201	2	1996	2:24	14.4	7/28	15.0	7/27	15.2	7/27	14.3	1579.0
Pardee	05043201	1	1997	2:24	13.6	8/10	14.1	8/10	14.8	8/8	13.3	1579.0
Pardee	05043201	2	1998	2:30	14.1	7/21	14.7	7/22	15.2	7/22	14.3	1579.0
Pardee	05043201	2	1999	1:12	13.2	8/26	13.6	8/26	14.3	7/13	12.3	1579.0
Pardee	05043201	1	2000	0:36	9.9	5/24	10.7	5/23	11.3	5/21	8.9	1579.0
Lupton	06031501	1	1997	1:12	14.9	8/10	15.9	8/10	16.7	8/8	14.5	2862.0
Lupton	06031501	1	1998	1:12	15.2	9/4	16.0	9/4	16.5	7/26	14.5	2862.0
Lord Ellis	07033301	2	1996	1:12	12.7	8/26	12.9	8/26	13.3	8/24	12.8	371.7
Coyote (Rdwd)	08020201	1	1994	1:12	16.0	8/16	16.9	8/14	17.4	8/13	15.5	5025.0
Coyote (Rdwd)	08020201	1	1999	1:12	17.1	8/26	18.5	7/12	19.7	7/12	15.9	5025.0
Coyote (Rdwd)	08020201	1	2000	1:12	17.8	8/1	19.2	8/1	19.9	6/28	16.1	5025.0
Redwood at Panther	08021301	1	1999	1:12	20.9	7/13	22.9	7/12	24.0	7/12	19.8	15688.1
Redwood at Panther	08021301	1	2000	1:12	22.0	8/1	23.9	8/1	24.7	6/27	19.6	15688.1
Panther (Rdwd)	08021401	1	1998	0:08	14.6	9/4	15.3	7/25	15.9	7/26	14.4	3814.0
Panther (Rdwd)	08021401	1	1999	1:12	14.5	8/26	14.9	8/26	15.0	8/24	13.8	3814.0
Panther (Rdwd)	08021401	1	2000	1:12	14.4	8/4	15.1	8/2	15.4	8/2	14.0	3814.0
Panther (Rdwd)	08021402	1	1994	2:00	13.1	7/22	13.5	7/20	13.6	7/17	12.2	3814.0
Panther (Rdwd)	08021402	1	1995	2:00	14.2	8/4	14.8	8/4	15.2	8/5	13.6	3814.0
Panther Rhva 2	08021601	2	2000	1:12	12.7	9/20	13.0	9/20	13.7	9/20	12.3	58.1
Panther Rhva 3	08022102	2	2000	1:12	12.8	9/20	13.1	9/20	13.8	9/20	12.7	75.5
Panther O-6	08022201	2	1999	1:12	13.6	8/26	13.8	8/26	14.1	8/27	13.7	455.6
North Fork Dolly Varden	08023601	2	1996	2:24	14.5	7/28	14.9	7/28	15.2	7/30	14.0	1069.0
North Fork Dolly Varden	08023601	2	1996	2:24	12.5	10/10	12.7	10/8	13.1	10/8	12.2	1069.0
North Fork Dolly Varden	08023601	2	1997	2:24	13.9	7/27	14.3	7/26	14.6	7/28	13.7	1069.0
North Fork Dolly Varden	08023601	2	1997	2:24	14.4	8/12	14.7	8/12	14.8	8/13	14.0	1069.0
North Fork Dolly Varden	08023601	2	1998	2:30	14.7	8/30	15.1	8/30	15.2	7/22	14.6	1069.0
North Fork Dolly Varden	08023601	2	1999	1:12	13.9	8/26	14.1	8/26	14.3	7/13	13.1	1069.0
South Fork Dolly Varden	08023602	2	1996	2:24	14.8	7/28	15.2	7/28	15.5	7/29	14.6	597.2
South Fork Dolly Varden	08023602	2	1998	2:30	14.7	8/30	15.1	8/30	15.5	7/23	13.7	597.2
South Fork Dolly Varden	08023602	2	1999	1:12	13.9	8/26	14.2	8/26	14.5	7/12	13.6	597.2
South Fork Dolly Varden	08023602	2	2000	1:12	13.8	8/3	14.3	6/27	15.1	6/27	13.8	597.2
Coyote Trib (Redwood Cr.)	09033101	2	1996	1:12	14.4	8/26	15.1	8/25	16.1	8/24	14.2	879.1

Table C5-7. Summer water temperature monitoring summary, Coastal Lagoons HPA.

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
WindyTD	08010601	2	1999	1:12	11.8	8/27	11.9	8/26	12.1	8/27	11.9	34.3
WindyTD	08010601	2	2000	1:12	12.1	9/19	12.2	9/19	12.3	9/19	12.1	34.3
WindyCD	08010602	2	1999	1:12	11.7	8/27	11.9	8/27	12.0	8/27	11.9	45.6
WindyCD	08010602	2	2000	1:12	12.2	9/19	12.4	9/18	12.8	9/19	12.5	45.6
WindyTU	08010701	2	1999	1:12	10.6	8/27	10.6	8/27	10.8	8/27	10.6	26.9
WindyTU	08010701	2	2000	1:12	11.3	9/25	11.3	9/23	11.3	9/20	11.2	26.9
WindyCU	08010702	2	1999	1:12	11.9	8/27	12.0	8/27	12.2	8/27	12.0	33.7
WindyCU	08010702	2	2000	1:12	12.4	9/19	12.6	9/20	12.9	9/19	12.7	33.7
Maple (mid)	08010801	1	1994	2:00	15.0	8/19	15.8	8/19	16.1	8/21	14.3	1687.7
Maple (mid)	08010801	1	1996	2:00	14.9	7/28	15.5	7/27	15.8	7/29	14.2	1687.7
Maple (mid)	08010801	1	1999	1:12	15.3	8/26	15.8	8/26	16.1	8/29	14.2	1687.7
Maple (mid)	08010801	1	2000	1:12	15.3	7/31	15.9	7/31	16.5	6/27	14.5	1687.7
M-Line	08010802	1	1994	2:00	13.9	8/19	14.6	8/21	14.6	8/20	13.4	361.7
M-Line	08010802	1	1995	2:00	14.2	8/3	15.5	8/3	15.8	8/1	13.2	361.7
M-Line	08010802	1	1999	1:12	14.1	8/26	14.5	8/26	14.8	8/26	14.2	361.7
M-Line	08010802	1	2000	1:12	13.7	7/31	14.4	7/31	14.8	8/1	13.3	361.7
Maple (high)	08011201	1	1994	2:00	14.0	8/19	14.5	8/21	14.8	8/19	14.0	2639.2
Maple (high)	08011201	1	1995	2:00	14.9	8/4	15.8	8/3	16.2	8/1	14.1	2639.2
Maple (high)	08011201	1	1996	2:00	15.2	7/28	16.1	7/27	16.2	7/27	14.6	2639.2
Clear	08011202	1	1997	2:00	14.1	9/3	14.8	7/25	15.2	8/8	13.7	1864.2
Clear	08011202	1	2000	1:12	14.1	8/1	15.3	7/31	16.0	6/28	13.2	1864.2
Beach	08011501	1	1994	2:00	13.9	8/19	14.6	8/21	14.6	8/19	13.6	469.1
Beach	08011501	1	1995	2:00	14.3	8/4	14.7	8/3	14.9	8/1	14.0	469.1
Beach	08011501	1	1999	1:12	14.8	8/26	15.6	8/25	16.0	8/26	14.7	469.1
Beach	08011501	1	2000	1:12	14.8	8/1	15.6	7/31	15.9	8/1	14.3	469.1
Beach	08011501	1	2000	1:12	14.8	8/1	15.6	7/31	15.9	8/1	14.3	469.1
Luffenholtz	08012901	1	1996	2:00	12.8	7/28	13.3	7/28	13.8	7/29	12.1	1688.9
Luffenholtz	08012901	1	1997	2:00	13.5	9/3	14.0	9/3	14.4	8/27	12.6	1688.9
M1CU	08020701	2	1999	1:12	13.2	8/26	13.5	8/26	14.1	8/27	13.2	179.3
M1CU	08020701	2	2000	1:12	13.5	9/18	13.8	9/18	14.9	9/20	13.0	179.3
M1CD	08020702	2	1999	1:12	13.5	8/26	13.9	8/26	14.8	8/27	13.4	193.3
M1CD	08020702	2	2000	1:12	13.9	9/17	14.4	9/18	15.6	9/20	12.8	193.3
M1TU	08021701	2	1999	1:12	13.2	8/26	13.4	8/26	14.0	8/27	13.3	70.0
M1TU	08021701	2	2000	1:12	13.9	9/18	14.3	9/20	15.2	9/20	13.2	70.0
M1TD	08021702	2	1999	1:12	14.1	8/26	14.6	8/25	15.3	8/27	14.1	79.1
M1TD	08021702	2	2000	1:12	14.7	9/17	15.1	9/18	16.3	6/27	14.2	79.1
M1TU2	08021703	2	1999	1:12	13.6	8/26	13.9	8/26	14.6	8/27	13.6	59.4
M1TU2	08021703	2	2000	1:12	14.2	9/20	14.6	9/20	15.4	9/20	13.8	59.4
M1TD2	08021704	2	1999	1:12	14.0	8/26	14.4	8/26	15.2	8/27	14.1	65.4

Table C5-7 Continued. Summer water temperature monitoring summary, Coastal Lagoons HPA.

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
M1TD2	08021704	2	2000	1:12	14.7	9/17	15.3	9/18	16.5	9/20	13.4	65.4
Mill Cr. (LP)	08511301	1	1997	2:00	13.8	9/3	14.0	9/3	14.2	9/4	13.7	617.3
McDonald	09010501	1	1996	2:00	12.6	8/29	13.6	8/30	14.0	8/30	12.6	3346.4
McDonald	09010501	1	1997	2:00	14.9	9/3	15.7	9/3	16.0	8/24	14.4	3346.4
McDonald	09010501	1	2000	1:12	13.3	9/17	14.1	9/18	14.7	9/19	12.9	3346.4
Pitcher	09012001	1	1996	2:00	13.6	8/30	15.0	8/31	15.6	8/30	13.6	3358.4
Pitcher	09012001	1	1997	2:00	14.9	9/3	15.9	9/3	16.2	9/1	13.7	3358.4
Pitcher	09012001	1	1999	1:12	13.7	8/27	14.3	8/27	14.7	8/29	13.1	3358.4
Pitcher	09012001	1	2000	1:12	13.3	7/31	13.8	9/17	14.2	9/19	12.6	3358.4
NF Maple Trib Fline	09012701	2	1999	1:12	13.0	8/27	14.5	7/21	13.5	8/27	13.2	249.5
Maple,NF (lower)	09012901	1	1994	2:00	14.4	8/13	15.3	8/5	15.8	8/13	14.5	6467.0
Maple,NF (lower)	09012901	1	1995	2:00	14.9	7/29	16.2	7/29	16.7	7/27	14.2	6467.0
Maple,NF (lower)	09012901	1	1996	2:00	14.8	7/27	15.7	7/27	16.2	7/27	14.4	6467.0
Maple,NF (lower)	09012901	1	1998	1:12	15.0	8/14	15.8	8/14	16.2	7/19	14.3	6467.0
Maple,NF (lower)	09012901	1	1999	1:12	15.3	8/27	16.0	8/23	16.5	8/29	14.8	6467.0
Maple,NF (lower)	09012901	1	2000	1:12	15.1	7/31	15.8	7/31	16.2	8/1	14.6	6467.0
Maple (low)	09012902	1	1994	2:00	15.3	8/16	16.2	8/6	16.7	8/3	15.0	16797.0
Maple (low)	09012902	1	1996	2:00	15.4	7/27	17.0	7/12	17.4	7/13	14.1	16797.0
Maple (low)	09012902	1	1998	1:12	15.8	8/14	17.5	7/16	18.4	7/19	14.9	16797.0
Maple (low)	09012902	1	1999	1:12	16.1	8/24	18.4	8/23	19.1	8/21	14.5	16797.0
Maple (low)	09012902	1	2000	1:12	16.5	7/31	18.7	7/31	19.6	8/1	15.4	16797.0
Maple,NF (upper)	09013401	1	1996	2:00	13.2	7/31	13.3	7/31	13.4	7/29	13.2	3460.3
Maple,NF (upper)	09013401	1	1997	2:00	14.1	8/27	14.7	8/22	15.3	8/7	13.3	3460.3
NF Maple Trib F8	09013401	2	2000	1:12	13.5	9/17	13.8	9/17	14.5	9/19	13.3	273.0
McDonald, NF	10012901	1	1997	2:00	13.5	9/3	13.8	9/3	14.0	8/29	12.5	1273.5
McDonald, NF	10012901	1	2000	1:12	13.3	9/17	13.6	9/17	14.1	9/19	12.7	1273.5

Table C5-8. Summer water temperature monitoring summary, Little River HPA.

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
Freeman	07010301	1	2000	1:12	13.4	7/31	14.1	7/31	14.3	8/1	13.0	1083.0
Little River (low)	07010801	1	1994	2:00	16.8	8/5	18.8	8/5	19.3	8/3	16.2	26011.0
Little River (low)	07010801	1	1995	2:00	16.8	8/4	18.6	7/29	19.3	7/31	15.6	26011.0
Little River (low)	07010801	1	1996	2:00	17.4	7/28	19.4	7/31	20.1	7/28	16.6	26011.0
Little River (low)	07010801	1	1998	1:12	17.0	8/15	19.0	8/15	19.8	7/19	16.2	26011.0
Little River (low)	07010801	1	1999	1:12	16.9	8/23	18.7	7/14	19.6	8/21	16.4	26011.0
Little River (low)	07010801	1	2000	1:12	17.4	7/31	19.2	7/30	20.2	7/28	16.5	26011.0
Carson	07011001	1	1997	1:12	14.9	8/27	15.4	8/27	15.8	7/18	13.7	2440.0
Carson	07011002	1	1998	1:12	14.7	8/14	15.6	7/16	16.2	7/18	14.0	2440.0
Carson	07011002	1	1999	1:12	14.8	8/24	15.3	8/23	16.0	6/22	13.2	2440.0
Carson	07011002	1	2000	1:12	14.8	7/31	15.3	7/31	15.7	7/28	14.4	2440.0
M155CD	07011201	2	1999	1:12	12.4	8/27	12.5	8/27	12.6	8/27	12.5	44.1
M155CD	07011201	2	2000	1:12	12.3	8/2	12.4	8/1	12.6	8/1	12.4	44.1
M155CU	07011202	2	1999	1:12	13.2	8/27	13.5	8/27	13.7	8/26	13.4	34.5
M155CU	07011202	2	2000	1:12	13.9	9/18	14.5	9/18	15.7	9/19	13.8	34.5
MitsuiCU	07011301	2	1996	1:12	12.8	8/30	13.0	8/30	13.4	10/8	12.8	60.4
MitsuiCU	07011301	2	1997	1:12	14.1	9/3	14.2	9/3	14.3	8/26	14.0	60.4
MitsuiCU	07011301	2	1998	1:12	12.8	8/14	13.0	8/14	13.3	7/26	12.8	60.4
MitsuiCU	07011301	2	1999	1:12	12.9	8/26	13.1	8/26	13.3	8/26	12.8	60.4
MitsuiCU	07011301	2	2000	1:12	13.2	9/18	13.5	9/18	14.1	9/19	13.3	60.4
MitsuiCD	07011401	1	1996	1:12	12.4	7/30	12.6	8/30	12.8	7/30	12.2	97.9
MitsuiCD	07011401	1	1997	1:12	13.7	9/3	14.4	9/3	14.6	9/4	13.7	97.9
MitsuiCD	07011401	1	1998	1:12	12.8	8/15	13.0	8/14	13.1	8/15	12.6	97.9
MitsuiCD	07011401	1	1999	1:12	13.1	8/27	13.6	8/26	14.0	8/29	12.8	97.9
MitsuiCD	07011401	1	2000	1:12	12.9	8/1	13.3	7/31	13.6	9/19	12.7	97.9
MitsuiTD	07011402	2	1996	1:12	11.8	8/31	12.0	8/30	12.0	8/27	11.4	63.0
MitsuiTD	07011402	2	1997	1:12	12.9	9/3	13.0	9/5	13.1	9/3	12.8	63.0
MitsuiTD	07011402	2	1998	1:12	12.3	8/16	12.4	8/14	12.5	8/12	12.2	63.0
MitsuiTD	07011402	2	1999	1:12	12.3	8/27	12.4	8/27	12.5	8/26	12.3	63.0
MitsuiTD	07011402	2	2000	1:12	12.5	9/19	12.7	9/19	13.0	9/18	12.7	63.0
MitsuiTU	07011403	2	1996	1:12	12.1	8/30	12.2	8/30	12.5	9/15	12.0	47.0
MitsuiTU	07011403	2	1997	1:12	14.0	8/27	14.1	8/26	14.6	8/26	14.0	47.0
MitsuiTU	07011403	2	1998	1:12	14.3	7/24	14.4	7/23	14.6	7/26	14.3	47.0
MitsuiTU	07011403	2	1999	1:12	13.6	8/27	13.6	8/27	13.7	8/27	13.6	47.0
MitsuiTU	07011403	2	2000	1:12	13.4	8/3	13.4	8/2	13.6	8/1	13.4	47.0

Table C5-8 Continued. Summer water temperature monitoring summary, Little River HPA.

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
Little River, Upper SF	07020601	1	1994	2:00	14.5	8/19	15.9	8/16	16.2	8/3	14.0	3619.0
Little River, Upper SF	07020601	1	1995	2:00	14.7	8/3	16.5	8/3	17.0	7/31	13.7	3619.0
Little River, Upper SF	07020601	1	1998	1:12	15.0	8/14	16.5	7/20	16.8	7/18	13.7	3619.0
Little River, Upper SF	07020601	1	1999	0:36	14.8	8/27	15.2	8/27	15.6	8/29	14.5	3619.0
Little River, Upper SF	07020601	1	2000	1:12	15.3	7/31	16.5	7/31	16.8	8/1	14.6	3619.0
Little River Headwaters	07021401	2	1996	2:24	12.7	7/28	13.3	7/28	13.4	7/28	12.5	468.0
Little River Headwaters	07021401	2	1997	2:24	12.1	8/10	12.4	8/9	12.9	8/7	12.3	468.0
Little River Headwaters	07021401	2	1998	2:30	12.4	8/30	12.7	8/30	13.1	8/29	12.2	468.0
Little River Headwaters	07021401	2	1999	1:12	11.6	8/26	11.8	8/26	12.2	8/26	11.9	468.0
Little River Headwaters	07021401	2	2000	1:12	11.8	8/3	12.1	8/2	12.4	8/2	12.0	468.0
M155TD	07021801	2	1999	1:12	12.2	8/27	12.4	8/27	12.6	8/29	11.4	26.5
M155TD	07021801	2	2000	1:12	12.4	9/18	12.7	9/18	13.3	9/19	12.5	26.5
M155TU	07021802	2	1999	1:12	12.8	8/27	13.2	8/27	13.6	8/27	12.7	21.2
M155TU	07021802	2	2000	1:12	14.1	9/17	14.9	9/17	16.3	9/19	14.0	21.2
Railroad	08013401	1	1994	2:00	14.4	8/19	15.7	8/19	15.9	8/19	13.3	1721.0
Railroad	08013401	1	1995	2:00	14.4	7/29	15.6	7/29	15.9	7/31	13.4	1721.0
Railroad	08013401	1	1998	1:12	14.6	8/14	15.5	8/13	15.9	7/19	14.0	1721.0
Railroad	08013401	1	1999	0:36	15.0	8/27	16.8	8/23	17.5	8/29	14.4	1721.0
Railroad	08013401	1	2000	1:12	15.2	7/31	16.3	7/31	16.6	8/1	14.7	1721.0
Little River, Lower SF	08013601	1	1994	2:00	14.6	7/24	16.3	8/5	16.9	8/3	14.5	3452.0
Little River, Lower SF	08013601	1	1995	2:00	15.2	7/30	16.7	8/3	17.2	8/1	14.0	3452.0
Little River, Lower SF	08013601	1	1998	1:12	15.9	7/23	17.4	7/23	18.1	7/26	15.2	3452.0
Little River, Lower SF	08013601	1	1999	0:36	15.6	8/27	16.5	8/23	17.2	8/22	14.5	3452.0
Little River, Lower SF	08013601	1	2000	1:12	16.1	7/31	18.0	7/31	18.5	8/1	15.2	3452.0
Little River (mid)	08013602	1	1994	1:36	15.2	7/30	16.4	7/29	16.9	7/31	14.4	13176.3
Little River (mid)	08013602	1	1996	2:00	16.0	7/28	17.5	7/28	17.9	7/29	14.8	13176.3
Little River (mid)	08013602	1	1999	1:12	15.5	8/27	16.2	8/27	16.6	8/29	15.3	13176.3
Little River (mid)	08013602	1	2000	1:12	15.8	7/31	17.0	7/31	17.4	8/1	15.0	13176.3
Danielle	08013603	1	2000	1:12	14.2	7/31	16.0	7/31	16.4	8/1	13.4	479.2
Heightman	08013604	1	2000	1:12	13.6	8/1	14.0	7/31	14.3	8/1	13.4	688.3
Little River (upper)	08022901	1	1994	2:00	13.4	8/21	14.2	8/21	14.5	8/19	13.3	8755.0
Little River (upper)	08022901	1	1995	2:00	14.0	8/3	15.2	8/3	15.8	7/31	13.3	8755.0
Little River (upper)	08022901	1	1996	2:00	14.1	7/28	15.3	7/27	15.8	7/30	12.6	8755.0
Little River (upper)	08022901	1	1999	1:12	14.1	8/27	14.7	8/27	15.3	8/29	13.1	8755.0
Little River (upper)	08022901	1	2000	1:12	14.3	9/18	15.1	9/18	16.1	9/19	13.9	8755.0
Little River (up98)	08023101	1	1998	1:12	15.3	8/14	17.0	8/14	17.4	7/26	14.3	9557.0
C-Line	08023201	1	2000	1:12	13.7	9/18	14.1	9/18	15.0	9/19	13.4	788.0

Table C5-9. Summary water temperature monitoring summary, Mad River HPA.

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
Boulder Trib	04030301	2	1996	1:12	13.2	8/31	13.8	8/31	14.7	8/24	12.7	190.0
Boulder	04030501	1	1994	1:12	16.6	8/17	19.6	8/17	20.1	8/14	14.0	11617.1
Boulder	04030501	1	1995	1:12	16.7	8/3	18.3	8/3	18.8	7/16	15.2	11617.1
Boulder	04030501	1	1997	1:12	18.1	8/9	20.8	8/6	21.6	8/7	15.8	11617.1
Boulder	04030501	1	1998	1:12	17.7	7/25	20.1	8/13	21.2	7/26	16.4	11617.1
Boulder	04030501	1	1999	1:12	17.2	8/26	19.6	7/12	20.6	7/13	14.4	11617.1
Goodman Prairie	04032902	1	1999	1:12	15.2	8/26	15.9	8/26	16.3	8/29	14.1	938.0
Goodman Prairie	04032902	1	2000	1:12	15.3	9/18	16.3	9/18	17.0	9/19	15.0	938.0
Graham	04033501	2	1996	1:12	12.6	8/17	13.4	8/17	13.6	8/17	11.4	723.0
Cañon (high)	05020101	1	1994	1:12	14.9	8/17	15.9	8/16	16.2	8/14	14.0	6421.0
Cañon (high)	05020101	1	1999	1:12	16.0	8/26	17.1	8/26	17.8	8/29	14.6	6421.0
Black Dog Treatment/5300	05020701	2	1999	1:12	11.8	8/27	12.0	8/27	12.4	8/27	11.9	92.0
Dry	05020801	1	1994	1:12	12.1	8/17	13.1	6/26	13.4	6/23	10.2	1601.0
Dry	05020801	1	1999	1:12	13.2	8/27	14.2	6/24	15.2	6/22	11.3	1601.0
Black Dog	05020802	2	1996	2:24	13.0	7/28	13.5	7/27	13.7	7/28	12.8	503.0
Black Dog	05020802	2	1997	2:24	14.2	9/3	14.4	9/3	14.6	8/26	14.0	503.0
Black Dog	05020802	2	1998	2:30	13.3	7/19	13.8	7/19	14.0	7/16	12.8	503.0
Black Dog	05020802	2	1999	1:12	13.3	8/26	13.6	8/26	13.9	8/26	13.4	503.0
Cañon (low)	05021001	1	1994	1:12	16.7	8/17	18.5	7/16	19.1	7/18	14.3	9869.0
Cañon (low)	05021001	1	1995	1:12	16.9	8/4	18.4	7/29	19.4	7/16	15.5	9869.0
Cañon (low)	05021001	1	1996	1:12	17.7	7/28	19.4	7/6	19.9	7/6	15.1	9869.0
Cañon (low)	05021001	1	1997	1:12	18.8	7/17	21.6	7/17	22.1	7/15	16.7	9869.0
Cañon (low)	05021001	1	1998	0:08	18.5	7/24	20.6	7/18	21.2	7/19	17.0	9869.0
Cañon (low)	05021001	1	1999	1:12	17.6	8/24	18.9	8/24	20.0	6/22	14.7	9869.0
Cañon (low)	05021001	1	2000	1:12	18.2	8/1	20.0	6/26	21.1	6/27	16.1	9869.0
Cañon (mid)	05021201	1	1994	1:12	15.8	8/17	18.0	8/17	18.4	8/14	13.1	8620.0
Cañon (mid)	05021201	1	1999	1:12	16.8	8/27	17.4	8/24	17.8	7/11	15.3	8620.0
Cañon (mid)	05021201	1	2000	1:12	17.9	8/1	19.6	8/1	20.3	8/1	16.6	8620.0
Green Diamond	05021401	1	1997	1:12	15.3	9/3	16.4	7/21	17.0	8/7	13.9	226.5
Green Diamond	05021401	1	1999	1:12	15.1	8/27	16.2	8/26	16.8	8/29	14.0	226.5
Mad River Trib. #1	05021601	2	1995	1:12	12.8	8/3	13.3	8/2	13.7	7/31	12.5	30.5
Mad River Trib. #2	05021602	2	1995	1:12	14.1	7/30	15.1	7/30	15.5	7/16	13.1	38.5
Mad River Trib. #3	05021603	2	1995	1:12	12.6	8/4	12.8	8/2	12.8	7/27	12.2	38.4
Mad River Trib. #4	05021604	2	1995	1:12	14.0	8/3	15.3	7/29	16.5	7/31	13.4	74.5
6001CD	05021605	2	1996	1:12	12.0	7/28	12.1	7/28	12.2	7/28	12.0	26.5
6001CD	05021605	2	1997	1:12	12.8	9/3	13.0	8/27	13.1	8/26	12.8	26.5
6001CD	05021605	2	1998	1:12	12.3	8/14	12.5	7/23	12.8	7/26	12.2	26.5
6001CD	05021605	2	1999	1:12	12.1	8/27	12.3	8/26	12.5	8/26	12.3	26.5

Table C5-9 Continued. Summer water temperature monitoring summary, Mad River HPA.

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
6001CD	05021605	2	2000	1:12	12.6	9/18	12.8	9/18	13.0	9/18	12.7	26.5
6001CU	05021606	2	1996	1:12	11.6	8/30	11.7	8/30	11.9	7/29	11.6	10.2
6001CU	05021606	2	1997	1:12	12.7	9/3	12.8	9/3	12.8	8/29	12.2	10.2
6001CU	05021606	2	1998	1:12	12.1	9/5	12.2	9/3	12.3	7/26	12.0	10.2
6001CU	05021606	2	1999	1:12	12.5	8/27	12.7	8/27	12.9	8/26	12.6	10.2
6001CU	05021606	2	2000	1:12	12.8	9/18	13.1	9/18	13.7	9/18	13.0	10.2
6001TD	05021607	2	1996	1:12	12.7	7/28	13.5	7/27	13.8	7/28	12.4	62.5
6001TD	05021607	2	1997	1:12	13.3	9/3	13.6	9/1	13.9	8/7	12.4	62.5
6001TD	05021607	2	1998	1:12	13.6	8/14	14.9	8/13	15.4	7/26	13.1	62.5
6001TD	05021607	2	1999	1:12	13.4	8/26	14.1	8/24	14.3	8/22	12.5	62.5
6001TD	05021607	2	2000	1:12	13.4	9/17	14.4	7/31	14.7	6/27	12.2	62.5
5410CU	05021608	2	1996	1:12	12.3	7/28	13.1	8/30	13.3	7/29	11.9	854.9
5410CU	05021608	2	1997	1:12	12.6	9/4	13.1	9/4	13.4	9/6	11.6	854.9
5410CU	05021608	2	1998	1:12	12.0	9/5	12.7	9/5	12.9	9/4	11.4	854.9
5410CU	05021608	2	1999	1:12	11.6	8/27	11.8	8/26	12.0	8/29	11.2	854.9
5410TD	05021701	2	1996	1:12	12.2	8/29	12.5	8/30	12.7	8/30	12.2	365.0
5410TD	05021701	2	1998	1:12	12.9	8/14	13.1	8/4	13.3	7/26	12.6	365.0
5410TD	05021701	2	1999	1:12	12.5	8/27	12.7	8/23	12.8	8/21	12.2	365.0
5410TD	05021701	2	2000	1:12	12.7	9/17	13.0	7/31	13.2	9/19	12.7	365.0
5410TU	05021702	2	1996	1:12	12.5	7/28	12.7	7/28	13.0	7/30	12.0	187.9
5410TU	05021702	2	1997	1:12	13.8	9/3	13.9	9/1	14.1	8/29	13.3	187.9
5410TU	05021702	2	1998	1:12	13.1	9/5	13.5	9/3	14.0	9/3	13.4	187.9
5410TU	05021702	2	1999	1:12	13.4	8/26	14.0	8/26	14.3	8/26	14.0	187.9
5410TU	05021702	2	2000	1:12	13.1	9/18	13.5	6/27	14.3	6/27	13.3	187.9
5410CD	05021703	2	1996	1:12	13.6	7/28	14.3	7/28	14.5	7/28	13.3	885.8
5410CD	05021703	2	1997	1:12	14.0	9/4	14.3	9/4	14.5	8/7	13.4	885.8
5410CD	05021703	2	1998	1:12	13.3	8/14	14.2	7/24	14.5	7/26	12.9	885.8
5410CD	05021703	2	1999	1:12	13.3	8/27	13.6	8/27	13.9	8/29	12.6	885.8
5410CD	05021703	2	2000	1:12	13.3	8/1	14.0	7/31	14.3	8/1	12.9	885.8
6001TU	05022101	2	1996	1:12	12.3	7/31	12.5	8/30	12.7	8/12	12.2	43.9
6001TU	05022101	2	1997	1:12	12.6	9/14	12.7	9/14	12.8	9/17	12.5	43.9
6001TU	05022101	2	1998	1:12	12.9	7/25	12.9	7/22	12.9	7/19	12.8	43.9
6001TU	05022101	2	1999	1:12	12.3	8/26	12.8	8/24	13.3	8/26	12.3	43.9
6001TU	05022101	2	2000	1:12	12.4	9/17	13.2	9/18	14.0	9/18	12.2	43.9
Mad River Trib. #5	05022201	2	1995	1:12	13.7	7/30	14.1	8/2	14.6	8/1	13.1	356.7
Mad River Trib. #6	05022202	2	1995	1:12	13.6	7/30	14.0	7/30	14.3	7/31	13.4	242.4
Mad River Trib. #7	05022203	2	1995	1:12	13.5	7/30	13.9	7/29	14.6	7/31	13.4	149.9
Devil	05022301	1	1997	1:12	14.5	9/6	14.5	9/4	14.6	9/6	14.5	1447.6
Devil	05022301	1	1999	1:12	14.5	8/27	15.0	8/27	15.2	8/26	14.6	1447.6

Table C5-9 Continued. Summer water temperature monitoring summary, Mad River HPA.

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
Cañon (class II)	05030801	2	1996	1:12	12.4	8/17	12.9	8/17	13.3	8/24	12.0	260.9
Maple (Mad)	05033001	1	1994	1:12	14.1	8/17	15.5	8/16	15.9	8/14	12.0	7496.0
Maple (Mad)	05033001	1	1996	1:12	16.8	7/28	20.4	7/28	21.1	7/30	14.5	7496.0
Maple (Mad)	05033001	1	1997	1:12	16.4	8/9	19.0	8/6	19.6	8/7	14.6	7496.0
Maple (Mad)	05033001	1	1999	1:12	15.7	8/26	17.6	8/23	18.4	8/29	14.1	7496.0
Mill Cr (Mck.)	06010401	1	1997	1:12	13.3	8/27	13.7	8/27	14.0	9/17	12.3	704.6
Lindsay	06011101	1	1994	1:12	15.9	8/18	16.8	8/17	17.1	8/19	15.2	8811.0
Lindsay	06011101	1	1995	1:12	15.9	7/29	17.1	7/29	17.8	7/16	15.2	8811.0
Lindsay	06011101	1	1996	1:12	15.9	7/28	16.6	7/28	17.2	7/30	15.0	8811.0
Lindsay	06011101	1	1997	1:12	16.1	7/18	16.9	7/17	17.3	7/7	14.8	8811.0
Lindsay	06011101	1	1998	0:08	15.8	8/14	16.8	8/13	17.4	7/15	14.5	8811.0
Lindsay	06011101	1	1999	1:12	15.3	8/24	16.2	8/24	17.1	8/29	14.9	8811.0
Lindsay	06011101	1	2000	1:12	15.4	7/31	16.0	7/31	16.3	8/1	15.0	8811.0

Table C5-10. Summer water temperature monitoring summary, North Fork Mad River HPA.

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
Mule	06020301	1	1996	1:12	13.0	8/30	15.1	8/31	15.6	8/31	11.6	338.7
Mule	06020301	1	2000	1:12	13.4	8/2	14.0	8/1	14.3	8/1	12.9	338.7
Jackson	06020302	1	1998	1:12	13.6	7/25	13.9	7/25	14.2	7/26	13.6	511.0
Jackson	06020302	1	1999	1:12	13.2	8/27	13.6	8/26	13.7	8/29	12.5	511.0
Denman	06020303	1	2000	1:12	15.3	9/17	17.0	9/17	18.1	9/18	14.2	878.2
Long Prairie	06021101	1	1994	1:12	14.2	8/17	15.4	8/17	15.5	8/14	12.8	6231.0
Long Prairie	06021101	1	1998	0:08	14.9	7/24	16.0	7/24	16.6	7/26	14.4	6231.0
Long Prairie	06021101	1	1999	1:12	14.8	8/26	15.6	8/26	16.1	8/29	13.4	6231.0
Gossinta	06021102	1	2000	1:12	15.0	9/17	17.9	9/19	19.7	9/18	13.8	730.9
Pollock	06021401	1	1996	2:24	13.7	7/29	14.0	7/28	14.3	7/29	13.7	1060.3
Pollock	06021401	1	1997	2:24	14.5	9/3	14.9	9/3	15.1	8/7	13.7	1060.3
Pollock	06021401	1	1998	2:30	13.9	8/10	14.6	8/9	15.2	7/23	13.4	1060.3
Pollock	06021401	1	1999	1:12	13.6	8/27	13.8	8/27	13.9	8/29	13.3	1060.3
Bald Mountain	06021402	1	1999	1:12	14.2	8/26	14.7	8/26	14.9	8/29	13.4	3008
Poverty	06021501	1	1999	1:12	14.1	8/26	15.7	8/26	16.4	8/29	10.5	404.4
Jiggs	06022201	2	1996	2:24	12.9	7/28	13.3	7/27	13.4	7/25	12.2	664.9
Jiggs	06022201	2	1997	2:24	13.7	9/3	14.5	8/5	14.6	8/7	13.1	664.9
Jiggs	06022201	2	1998	2:30	12.9	8/10	13.4	7/19	13.7	7/22	12.5	664.9
Jiggs	06022201	2	1999	1:12	13.4	8/26	14.5	8/25	14.8	8/26	13.1	664.9
NF Mad (middle)	06022301	1	1994	1:12	17.1	8/17	18.7	8/17	18.8	8/14	15.5	23462.9
NF Mad (middle)	06022301	1	1999	1:12	17.3	8/26	19.0	7/12	19.6	7/13	14.5	23462.9
NF Mad (middle)	06022301	1	2000	1:12	17.3	8/1	19.8	7/31	20.2	8/1	15.6	23462.9
Jiggs Upper	06022601	2	1999	1:12	12.9	8/26	13.1	8/26	13.7	8/27	13.0	421.5
Sullivan Gulch	06022801	1	1997	1:12	15.2	9/3	15.9	7/16	16.3	7/18	13.9	1536.0
Sullivan Gulch	06022801	1	1999	1:12	14.6	8/27	15.1	8/24	15.9	6/22	12.5	1536.0
Sullivan Gulch	06022801	1	2000	1:12	14.9	7/31	15.6	6/17	16.2	6/27	13.5	1536.0
NF Mad (lower)	06022802	1	1994	1:12	17.7	7/17	20.3	8/16	20.5	8/14	15.5	27634.0
NF Mad (lower)	06022802	1	1995	1:12	18.4	8/3	20.7	8/2	21.5	7/16	16.2	27634.0
NF Mad (lower)	06022802	1	1996	1:12	19.7	7/28	21.4	7/28	21.9	7/30	18.1	27634.0
NF Mad (lower)	06022802	1	1997	1:12	19.5	7/17	22.4	8/5	23.2	8/7	17.2	27634.0
NF Mad (lower)	06022802	1	1998	1:12	18.9	7/23	21.8	8/13	22.6	8/13	17.4	27634.0
NF Mad (lower)	06022802	1	1999	1:12	17.8	8/23	20.4	7/14	21.2	8/22	16.2	27634.0
NF Mad (lower)	06022802	1	2000	1:12	19.0	8/1	21.1	8/1	22.0	6/27	16.5	27634.0

Table C5-10 Continued. Summer water temperature monitoring summary, North Fork Mad River HPA.

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
NF Mad (site 1a)	06022803	1	1998	1:12	18.8	8/14	22.3	8/13	23.1	8/12	16.6	26613.0
Watek	06023201	1	1996	1:12	12.1	8/28	13.4	8/30	13.7	8/31	10.5	615.8
East Fork North Fork Mad	07022201	2	1996	2:24	14.1	7/28	14.4	7/28	14.6	7/30	13.7	153.7
Canyon (class II)	07022601	2	1996	2:24	14.6	7/28	15.1	7/28	15.2	7/14	14.0	847.5
Canyon (class II)	07022601	2	1997	2:24	14.0	8/10	14.4	8/10	14.8	8/8	13.9	847.5
Canyon (class II)	07022601	2	1998	2:30	14.2	8/30	14.6	7/21	15.5	7/22	13.7	847.5
Canyon (class II)	07022601	2	1999	1:12	13.8	8/26	14.2	8/26	14.6	7/13	12.3	847.5
Canyon (class II)	07022601	2	2000	1:12	13.8	6/27	14.7	6/27	15.8	6/28	13.2	847.5
Canyon RHVA 1	07022701	2	2000	1:12	13.4	9/19	13.7	9/19	14.4	9/20	13.1	28.1
Canyon RHVA 2	07022702	2	2000	1:12	12.0	9/20	12.3	9/20	12.9	9/20	11.8	95.4
NF Mad (upper)	07022801	1	1994	1:12	13.9	8/17	14.9	8/17	15.2	8/20	12.5	5252.6
NF Mad (upper)	07022801	1	1999	1:12	14.5	8/26	15.2	8/26	15.6	8/29	13.3	5252.6
NF Mad (upper)	07022801	1	2000	1:12	14.7	8/1	15.3	8/1	15.9	6/28	13.7	5252.6
Canyon	07022802	1	1997	1:12	14.4	8/9	15.3	8/9	15.9	8/7	13.7	1870.2
Canyon	07022802	1	1998	1:12	14.0	9/4	15.1	9/3	15.6	9/3	13.4	1870.2
Canyon	07022802	1	1999	1:12	14.0	8/26	14.6	8/26	14.8	8/26	14.0	1870.2
East Fork of North Fork	07022803	1	2000	1:12	13.5	8/2	14.1	8/1	14.6	6/28	12.8	1276.6
Krueger	07023401	1	2000	1:12	14.1	8/2	14.5	8/1	14.8	8/1	13.8	708.9
Railroad (NF Mad)	07023402	1	2000	1:12	13.4	8/3	13.5	8/1	13.7	8/1	13.3	545.3

Table C5-11. Summer water temperature monitoring summary, Humboldt Bay HPA.

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
Salmon (high)	03011801	1	1994	1:12	13.7	8/19	14.1	8/16	14.3	8/19	13.4	3294.3
Salmon (high)	03011801	1	1999	1:12	15.2	8/24	16.7	8/24	17.3	8/22	14.3	3294.3
Salmon (high)	03011801	1	2000	1:12	15.1	7/31	16.0	7/31	16.7	6/27	13.7	3294.3
Salmon (low)	03510901	1	1995	1:12	16.0	7/16	17.2	7/15	18.1	7/16	15.5	7858.0
Salmon (low)	03510901	1	1996	1:12	16.1	7/28	17.3	7/28	17.7	7/28	15.9	7858.0
Salmon (low)	03510901	1	1997	1:12	18.1	8/5	20.5	8/5	20.9	8/3	15.9	7858.0
Salmon (low)	03510901	1	1998	0:08	17.4	7/17	19.3	7/16	20.1	7/15	16.0	7858.0
Salmon (low)	03510901	1	1999	1:12	16.6	8/23	17.7	8/22	18.5	6/22	14.6	7858.0
Salmon (low)	03510901	1	2000	1:12	16.7	7/29	17.8	7/29	18.9	6/27	15.2	7858.0
Salmon (mid)	03511001	1	1994	1:12	15.8	8/19	16.4	8/17	16.8	8/14	14.6	6979.0
Ryan (upper)	04011801	1	1994	2:00	14.7	8/19	15.7	8/20	16.1	8/19	14.6	1154.6
Ryan (upper)	04011801	1	1995	2:00	15.2	7/26	15.9	7/16	16.8	7/27	15.2	1154.6
Ryan (upper)	04011801	1	1997	2:00	14.9	7/18	15.3	7/22	15.9	7/24	13.7	1154.6
Ryan (upper)	04011801	1	1999	1:12	14.9	8/24	15.4	8/24	15.8	8/22	14.3	1154.6
Henderson	04510101	1	1997	2:00	14.2	9/4	14.5	9/4	14.8	9/5	14.2	922.6
Henderson	04510101	1	1999	1:12	13.5	8/27	13.7	8/26	14.0	8/26	13.3	922.6
Henderson	04510101	1	2000	1:12	13.4	7/31	13.5	7/31	13.8	7/31	13.5	922.6
Guptil	04511201	1	1997	2:00	14.9	9/3	15.5	8/26	16.1	8/25	14.9	1146.2
Guptil	04511201	1	1999	1:12	14.2	8/27	14.4	8/27	14.8	8/29	13.9	1146.2
Guptil	04511201	1	2000	1:12	13.8	7/31	14.2	7/31	14.3	7/31	13.9	1146.2
Bear Ryan	04511202	1	1999	1:12	13.5	8/27	13.7	8/27	14.0	8/29	13.3	719.2
Bear Ryan	04511202	1	2000	1:12	13.5	7/31	13.8	7/31	14.1	8/1	13.1	719.2
Ryan, SF	04511302	1	1997	2:00	14.9	7/18	16.5	7/20	17.0	7/18	13.4	1799.2
Ryan, SF	04511302	1	2000	1:12	14.6	7/31	15.1	7/31	15.4	8/1	14.1	1799.2
Morrison	05011401	1	1997	1:12	14.8	9/3	15.3	9/3	15.8	7/18	13.1	575.0
Morrison	05011401	1	1998	1:12	14.1	8/14	15.2	8/13	15.4	8/12	13.6	575.0
Morrison	05011401	1	1999	1:12	13.9	8/27	14.4	8/23	14.6	8/22	13.1	575.0
Morrison	05011401	1	2000	1:12	13.8	9/18	14.3	7/31	14.7	8/1	13.5	575.0
Rocky	05011501	1	1999	1:12	12.4	8/27	12.4	8/27	12.5	8/22	12.2	465.8
Jacoby (low)	05012401	1	1994	1:12	13.5	8/17	14.7	8/16	14.9	8/14	11.7	4345.0
Jacoby (high)	05023001	1	1994	1:12	12.4	8/17	13.9	8/16	14.3	8/14	10.8	1128.4
Ryan (low)	05513601	1	1997	2:00	15.6	9/3	16.3	9/3	16.9	9/5	14.8	7341.1
Ryan (low)	05513601	1	1998	1:12	15.1	8/13	15.7	8/13	16.1	8/13	14.5	7341.1
Ryan (low)	05513601	1	1999	1:12	14.8	8/24	15.3	8/24	15.9	8/22	14.3	7341.1
Ryan (low)	05513601	1	2000	1:12	14.8	7/31	15.3	7/31	15.6	7/28	15.0	7341.1

Table C5-12. Summer water temperature monitoring summary, Eel River HPA.

Stream Name	Site ID	Class	Year	Interval	7DMAVG	Mid Date (7DMAVG)	7DMMX	Mid Date (7DMMX)	Max	Max Date	Min after Max	Area (acres)
Slater	01510101	1	1997	1:12	14.4	7/8	14.9	7/8	15.3	7/8	14.5	1133.0
Slater	01510101	1	1998	0:08	14.9	7/20	15.5	7/18	15.8	7/18	14.7	1133.0
Slater	01510101	1	1999	1:12	14.1	8/27	14.5	8/12	14.8	8/29	13.4	1133.0
Wilson (VanD)	02012301	1	1997	1:12	15.3	9/3	15.9	9/3	16.3	9/4	15.3	686.2
Wilson (VanD)	02012301	1	1998	0:08	14.3	8/13	15.1	8/13	15.5	8/12	13.6	686.2
Wilson (VanD)	02012301	1	1999	1:12	14.3	8/24	15.0	8/23	15.6	8/29	13.3	686.2
Cuddeback	02012302	1	1998	0:08	14.2	8/13	14.9	8/13	15.2	8/12	13.8	558.0
Cuddeback	02012302	1	1999	1:12	14.0	8/27	14.4	8/23	14.9	8/29	13.4	558.0
Fielder	02012501	1	1999	1:12	14.0	8/26	14.4	8/23	14.6	8/22	13.7	109.5
Fielder	02012501	1	2000	1:12	13.8	7/31	14.2	7/31	14.3	7/29	13.7	109.5
Stevens	02023501	1	1999	1:12	16.4	8/26	19.3	8/23	20.5	8/29	14.6	506.9
Stevens	02023501	1	2000	1:12	16.6	7/31	18.4	7/31	19.0	7/29	15.9	506.9

The nomenclature changed as well. The term MWAT (Maximum Weekly Average Temperature) is a specific threshold determined for a particular life stage and species (Armour 1991). MWAT is a fixed value for a specific species, not a field measurement that varies by stream. The more appropriate term is 7DMAVG (Seven-Day Moving Average) which is the highest average temperature out of all possible seven consecutive days. The 7DMAVG may or may not include the absolute maximum temperature or the 7DMMX recorded during the season. The maximum temperature often occurs later in the fall during low flow conditions that coincide with the loss of deciduous canopy and a reduced coastal marine layer influence. During this time of year the daily peaks may be high but the daily average, due to overnight cooling, will be less than the mid summer peaks.

C5.1.2.2 Methods

Green Diamond continues to use Onset Computer Corporation's temperature data loggers although the HOBO® models are being phased out for a variety of reasons. The reliability of the HOBO® models came into question when calibration of the units began to occur annually. Even with regular maintenance and battery exchanges the thermographs failed more frequently as they aged. Advances in memory capacity and battery life provided for a new model know as a TidbiT®. The TidbiT® has the same accuracy as the HOBO® HTI –05/37^{o}C, 3 years more battery life, almost 18 times more memory and it is water proof. Every thermograph is calibrated (see Appendix D) to confirm its reliability. Individual recorders with identical measurements are used in Paired Watershed BACI experiments (see Objectives and Methods-Class II Paired Watershed Streams below). With the introduction of the TidbiT® the length of deployment became less of a concern yet the primary monitoring window remained from July through September. Early attempts at modifying the recording interval to capture as much data as the thermograph was capable of only produced huge files that were difficult to analyze. For instance a Tidbit® launched at 8-minute intervals (0.13 hours) will record 180 records per day and last 180 days before the memory is full. Analysis again confirmed that an interval of 1.2 hours would capture the necessary details of the diurnal extremes. The recording interval was kept at 1.2 hours.

In addition to the Class I monitoring Green Diamond began a program of Class II monitoring in headwater streams known to have populations of Tailed Frogs or Torrent Salamanders. All of the methods apply to both classes of streams with a few exceptions. Due to the small size of many of the Class II watercourses the actual placement of the recorders tended to be in deeper water in order to avoid the possibility of late summer dewatering. Also, the Class II sites were frequently associated with other biological monitoring and thus are not necessarily at the lowest point in the sub-watershed.

Other site-specific variables are collected at every temperature-monitoring site or measured from maps, aerial photos or GIS. The inclusion of specific variables will help in the interpretation of the thermograph data. These variables currently include canopy closure, stream aspect, channel dimensions, flow and watershed area. Green Diamond has cooperated extensively during this period with the Forest Science Project's "*Regional Assessment of Stream Temperatures Across Northern California and Their Relationship to Various Landscape – Level and Site – Specific Attributes*". The previous list of variables and more were collected for and contributed to the FSP for inclusion in the regional temperature analysis.

Green Diamond has also acquired temperature profiles from other agencies and landowners that have worked within or near the HPAs. Louisiana Pacific (LP) monitored temperature in several Class I watercourses across their ownership in Humboldt County. When Green Diamond purchased the LP property in 1998, it also acquired these data files along with site location maps dating back to 1994. Green Diamond and LP were active participants in the Fish, Farm, and Forest Community effort to establish standardized monitoring methods in order to conduct regional temperature evaluations such as the FSP's "*Regional Assessment of Stream Temperatures Across Northern California and Their Relationship to Various Landscape – Level and Site – Specific Attributes*". LP's methods were comparable to Green Diamond's and as a result their historic data has been assimilated into the database. Many of the LP sites have become some of Green Diamond's annual monitoring stations. The Yurok Tribal Fisheries Program (YTFP) has extensively monitored the tributaries as well as the main stem of the lower Klamath River. This is a coordinated effort to make the best use of respective resources and avoid repetitive monitoring of specific sites. The YTFP and Green Diamond share the same monitoring methods and thus resulting data files for the Klamath area. Several agencies such as the California Conservation Corp, California Department of Fish and Game, National Park Service, the United States Fish and Wildlife Service and the US Forest Service have all monitored stream temperature on or near Green Diamond Property. Unfortunately most of these monitoring efforts are not coordinated with Green Diamond or potentially have different methods and protocols. As a result these data must be evaluated on a case by case basis as to whether or not to include them in the database.

C5.1.2.3 Results

At the end of the year 2000, Green Diamond has recorded and/or collected 400 temperature profiles in approximately 108 Class I watercourses and 210 temperature profiles in approximately 70 Class II watercourses. All of these profiles have been processed to calculate the 7DMAVG, 7DMMX, absolute maximum, and the minimum following the maximum temperatures as well as the associated dates of occurrence. Various attributes have been collected for many of these monitoring stations, specifically watershed area. Temperature monitoring data are summarized and shown in Tables C5-2 through 12.

C5.1.2.4 Discussion

The monitoring window from mid-June through mid-September generally captures the seasonal peak 7DMAVG temperature. On occasion 7DMAVG temperatures in late September and early October were documented. In several stream reaches, maximum water temperatures occurred in late September (upper Dominie Creek, lower Savoy Creek, and Upper South Fork Winchuck River) [Smith River HPA]. These late occurring maximum temperatures were probably affected in part by diminishing stream flow, since the photoperiod of maximum daylight hours and sun angle had occurred two months earlier. Also, the geography of coastal northern California may promote the late occurrence of maximum stream temperatures. A dense band of marine fog that often extends up coastal stream courses is common during June and July. By mid-August this marine layer starts to break up, and the rest of the late summer/early fall is generally clear and warm prior to the onset of fall and winter rains. Finally, the deciduous habit of alders and willows in riparian areas may influence late peak temperatures.

Of the 400 Class I records for the period 1994 to 2000, 375 (93.8%) were at or below the "Inter-agency Matrix" suggested MWAT threshold of 17.4°C. Green Diamond believes that the single MWAT threshold value fails to account for natural variations in water temperature due to geographic location, climatic factors and drainage area of the monitored sub-basin. Also, depending on the method used to test the upper incipient lethal temperature of juvenile salmonids, a critical MWAT can range from 16.8°C to 18.4°C (Armour 1991; Thomas et al. 1986; Becker and Genoway 1979). Stream and watershed specific factors create a wide variation in processes that affect water temperatures (Beschta et al. 1987). The relationship of water temperature and watershed area was examined to help account for the observed natural variation in water temperature. The data suggests that water temperature was positively associated with watershed area and was relatively predictable for watershed areas up to 10,000 acres. Above 10,000 acres, the temperature variation increased probably in response to the complex interacting physical factors (Beschta et al. 1987).

Of the 25 records that were above the suggested MWAT threshold, 17 had watershed areas of more than 10,000 acres above the monitoring site. The 8 records that exceeded the 17.4°C threshold and had watershed areas less than 10,000 acres occurred in 6 different streams. The higher temperatures appear to be caused by either variations in climatic factors or by a flood event that set back the riparian vegetation. For example, in the winter of 1995/1996 Cañon Creek experienced a flood that removed the riparian canopy in the lower reaches of the stream. Prior to the flood in 1994 and 1995 this reach had 7DMAVG temperatures of 16.7°C and 16.9°C, respectively. For the last 5 years following the flood, the 7DMAVG temperatures have exceeded 17.4°C. With the loss of the streamside vegetation, there was a greater proportion of the stream surface exposed to direct solar radiation. Low discharge in this lower reach also exacerbates the high stream temperatures. However, the general trend since the flood has been a gradual recovery of the riparian canopy and a decrease of the highest 7DMAVG stream temperatures.

C5.1.2.5 Conclusions

Green Diamond believes that a single threshold value fails to accurately represent the natural variation found in water temperature between sites. For this reason, future water temperatures will be evaluated based on the yellow and red light thresholds described in Section 6.3. The expected temperature for a site will be based on its watershed size rather than a generic threshold value applied equally to all streams.

C5.1.3 References

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C5.2 CLASS II PAIRED WATERSHED TEMPERATURE MONITORING

C5.2.1 Retrospective Study

C5.2.1.1 Objectives and Methods

The first study was a retrospective study of water temperature conducted during the summer of 1995. For this study, groups of small headwater streams in close proximity with similar flow, aspect, and geology were selected. One group of streams were direct tributaries of the Mad River, while the other streams within Green Diamond's ownership were tributaries of Rowdy and Dominie Creeks in the Smith River watershed (Table C5-13). The streams differed in that some flowed through areas that had been recently harvested by clearcutting (cut) on both sides of the stream with Green Diamond's riparian buffers (standard state regulated widths but minimum 70% total canopy retention) left along the streams, while the other streams (uncut) were located in intact stands of second growth. One stream had only been harvested on one side (1/2 cut), but it was included with the cut group for analysis. In an attempt to see if there was a coastal effect in the results, Green Diamond also collaborated with the Hoopa Tribal Forestry to conduct the same type of study on similar sized streams within the Hoopa Reservation. A wide variety of silvicultural practices and riparian buffers have been implemented on the Hoopa Reservation over the years, so they selected sites that most resembled Green Diamond's silviculture and riparian leave standards. HOBO thermographs were placed in a total of 11 cut streams and 10 uncut streams. However, two of the HOBOs in cut streams were placed in reaches that went dry during the study, and one of the HOBOs in an uncut stream was removed by some unknown person during the study. The restrictions of finding comparable sites within the Hoopa Reservation limited the interior area to only three cut and two uncut streams (Table C5-13).

Table C5-13. List of uncut and cut tributaries with watershed area (acres), stream orientation (aspect in ^{o}), adjacent stand age (years for uncut, feet for cut), and cover type (RW=redwood, DF=Douglas-fir), mean and mean maximum water temperature (^{o}C) with standard deviations.[1]

Uncut	Area	Aspect	Adjacent Stand	Mean Temp	Mean Max.
MR #4	74	46	70, RW	13.2 (1.05)	14.7 (0.73)
MR #5	338	19	70, RW	12.8 (0.60)	13.7 (0.38)
MR #7	160	344	70, RW	12.5 (0.63)	13.6 (0.46)
Rowdy #2	28	291	35-40, RW	12.7 (0.39)	13.1 (0.50)
Rowdy #3	78	159	35-40, RW	12.1 (0.45)	12.6 (0.55)
Dominie #3	46	345	45-50, RW	12.9 (0.91)	14.4 (1.01)
Dominie #4	7	210	45-50, RW	12.9 (0.79)	14.0 (1.00)
Hoopa #1	28	30	35-40, DF	13.5 (0.57)	14.1 (0.82)
Hoopa #6	338	100	*10-15/OG, DF	12.2 (1.23)	13.3 (1.46)
Cut	**Area**	**Aspect**	**Adjacent Stand**	**Mean Temp**	**Mean Max.**
MR #1	28	39	1400	12.4 (0.42)	13.0 (0.31)
MR #2	46	24	1900	13.2 (0.73)	14.7 (0.44)
MR #3	38	15	**1100/70, DF	12.2 (0.23)	12.6 (0.21)
MR #6	234	6	2700	12.8 (0.56)	13.7 (0.33)
Rowdy #1	22	255	1200	12.5 (0.64)	13.4 (0.83)
Dominie #1	37	298	1000	12.5 (0.62)	13.3 (0.74)
Hoopa #2	46	22	1500	13.3 (1.45)	14.6 (1.82)
Hoopa #3	38	107	1000	11.8 (1.01)	12.9 (1.14)
Hoopa #5	234	80	600	11.1 (0.55)	11.6 (0.70)

Notes

1 For cut tributaries, all variables are the same except that the adjacent stand description is replaced with the length (feet) of clearcut on both sides of the stream. Cover types of the riparian buffers of the cut tributaries were presumed to be the same as the corresponding uncut tributaries.

* West side was 10-15 year old second growth and the east side was old growth.

** West side was clearcut and the east side had 70 year old second growth.

In all cases, HOBOs were placed at the lower end of the cut unit, or in the same respective location on the uncut streams. Prior to placement, the HOBOs for each region were tested in a water bath to insure that they were all giving readings that were within the manufactures specified limits (plus or minus 0.2^0 C) relative to each other. However, they were not calibrated to a known standard (ice bath) to insure that the readings were accurate. For each region, the seven consecutive warmest days of the season were selected and the mean maximum and overall mean water temperatures for the period were calculated. Differences between means and variances of the two groups of streams were tested using a two-sample t-test (NCSS 1997).

C5.2.1.2 Results

Visual inspection of HOBO data output from the two groups of streams did not reveal any consistent trends. The coldest streams with the least daily variation appeared to be Mad River #3 (1/2 cut), Rowdy #3 (uncut) and Hoopa #5 (cut). The warmest streams with the greatest daily extremes in temperature were Mad River #4 (uncut), Dominie #3 (uncut), Dominie #4 (uncut) and Hoopa # 2 (cut). In general, a visual ranking of all of the streams would indicate that prior timber harvesting did not correlate well with either the mean values or amount of variation in stream temperatures. Analysis of the data also indicated that there was no significant difference between the mean maximum (t = 0.74, d.f. =16, P = 0.471) or overall mean (t = 1.34, d.f. = 16, P = 0.199) temperatures for the cut and uncut groups (see below).

Stream Groups	N	Mean Temp (^{0}C)	S.E.	Mean Max. (^{0}C)	S.E.
Uncut	9	13.51	0.192	14.19	0.283
Cut	9	13.11	0.227	13.85	0.352

There were too few streams available to make a meaningful comparison of uncut and cut streams in the more interior Hoopa Reservation, but a comparison was made between all coastal and all interior (Hoopa) streams. The temperatures of the five Hoopa streams (mean max. = 14.25; overall mean = 13.33) were similar to the 13 coastal streams (mean max. = 13.93; overall mean = 13.30), with no significant difference (mean max: t = 0.68, d.f. = 16, P = 0.508; overall mean: t = 0.94, d.f. = 16, P = 0.363).

This retrospective comparison of stream temperatures in cut versus uncut streams provided evidence that timber harvest was not having a substantial impact on stream temperature. Increasing the sample size of the two groups would have increased confidence in the conclusion that as a group, streams with riparian buffers on Green Diamond's ownership were not warmer than streams that were flowing through uncut areas. However, it did not permit a comparison of more subtle changes in stream temperature following timber harvesting. Since the inherent differences in stream temperatures between the two groups of streams was not known prior to harvesting, it was not possible to directly assess the changes that might have occurred. Due to the fundamental limitations of a retrospective study, Green Diamond concluded that continuing these comparisons between cut and uncut streams would provide little additional information and discontinued the study.

C5.2.2 Before-After-Control-Impact (BACI) Water Temperature Study

C5.2.2.1 Objectives and Methods

In summer 1996, Green Diamond initiated a monitoring program in non-fish bearing (Class II) watercourses to assess the adequacy of riparian buffers in maintaining water temperatures following timber harvest. Streams in areas where timber harvest was planned were identified and paired with streams in close proximity that had similar size, aspect, and streambed geology. The objective of this study was to examine the impact of timber harvest on water temperature in small Class II watercourses by comparing maximum temperature differentials between fixed upper and lower points of selected stream reaches. These temperature differentials were measured on matched pairs of streams, one member of which was scheduled for timber harvest, while the other was to be left undisturbed. The paired stream design was adopted to control for confounding factors that can influence water temperature such as ground water inputs and microclimatic factors. Measurements were initiated in both streams of a pair at least one year prior to timber harvest. These data represent a before-after-control-impact (BACI) (Green 1979; Stewart-Oaten et al. 1986; Skalski and Robson 1992) observational study. While observational studies cannot infer cause and effect relationships, BACI studies represent the best available setup for detecting changes after disturbance. Monitoring of the stream pairs is scheduled to continue at least three years after harvest, or until the temperature profile of the two streams return to the pre-treatment pattern. However, the data reported here only represent a preliminary assessment of data collected from 1996-1998. Analysis of 1999 and 2000 data is currently in progress.

For each pair of streams, the stream located in a future harvest unit was designated as the “treatment” stream, while the other stream was designated as the “control” stream. Two remote temperature data loggers were placed in the treatment stream at the upstream and downstream edges of the harvest unit. Another pair of temperature recording devices was placed in the control stream at locations that were similar in stream spacing (distance apart) and watershed position relative to the treatment stream. Treatments consisted of clearcuts placed on both sides of the stream with standard forest practice buffer widths (50-75 feet) and 70% total canopy retention. Each steam pair is referred to as a *site.*

The five sites selected in 1996 include:

- One pair in the headwaters of Dominie Creek (D1120) in the Smith River HPA ;
- One pair of tributaries to the South Fork Winchuck River (D1120 in the Smith River HPA ;
- One pair in the headwater tributaries of the Little River (Mitsui) in the Little River HPA;
- One pair off the mainstem Mad River in the Mad River HPA; and
- One pair in the headwater tributaries of Dominie Creek in the Mad River HPA.

In 1999, three pairs were added to the study:

- Two pairs of tributaries to Maple Creek (Windy Point and M1) in the Mad River HPA; and

- One pair of tributaries to the Lower South Fork Little River (M155) in the Little River HPA.

Timber harvest at Mitsui and D2010 took place in winter 1996/1997. Timber harvest at 6001 and 5410 took place in winter 1997/1998. As of winter 1999/2000, timber harvest had not yet occurred at D1120. Timber harvest at Mitsui and D2010 took place in winter 1996/1997. Timber harvest at 6001 and 5410 took place in winter 1997/1998. The Maple Creek units where harvested in winter 1999/2000. As of winter 1999/2000, timber harvest had not occurred at D1120 or the Lower South Fork unit.

The study is still in its data collection phase on pairs where the treatment site was harvested after 1999 or has yet to be harvested. However, a preliminary analysis has been conducted of data from the four pairs harvested before 1999 (Mitsui, D2010, 6001, and 5410).

As indicated in Table C5-14, mean length of control and treatment reaches on Mitsui, D2010, 6001, and 5410 was 1069.2 feet (SE = 515.71) and 1210.2 feet (SE = 650.63), respectively. Mean percent canopy closure following timber harvest was 79.8 (SE = 5.27) and 75.2 (SE = 3.70) for control and treatment reaches, respectively, but the difference was not statistically significantly ($P < 0.05$) using a one-tailed paired t-test ($t = 1.73$, d.f. = 3, $P = 0.091$).

The upstream and downstream placement of temperature recording devices allowed measurement of the temperature differential across the treatment area and an assessment of the extent to which water temperature changed as it flowed through the treatment area. Interest was primarily in quantifying increases in water temperature as it flowed through the treatment area compared to similar measurements in the control stream reach.

Temperature recording devices were calibrated prior to deployment. For calibration, all data loggers (mostly HOBOs initially and later TidbiTs) were placed in an ice bath and temperature readings were taken after three hours. Pairs of data recorders for upstream and downstream deployment on the same stream were formed by pairing instruments with identical readings after three hours in the ice bath. The manufacturer's specification limit was 0.2 ^{o}C . All recorders were programmed to record temperature (^{o}C) every 1.2 hours or 20 times every 24 hours. For this analysis, data were recorded on five pairs of streams.

Table C5-14. Initial five pairs in the Class II BACI study, with stream reach length, mean canopy closure throughout the reach, and aspect.

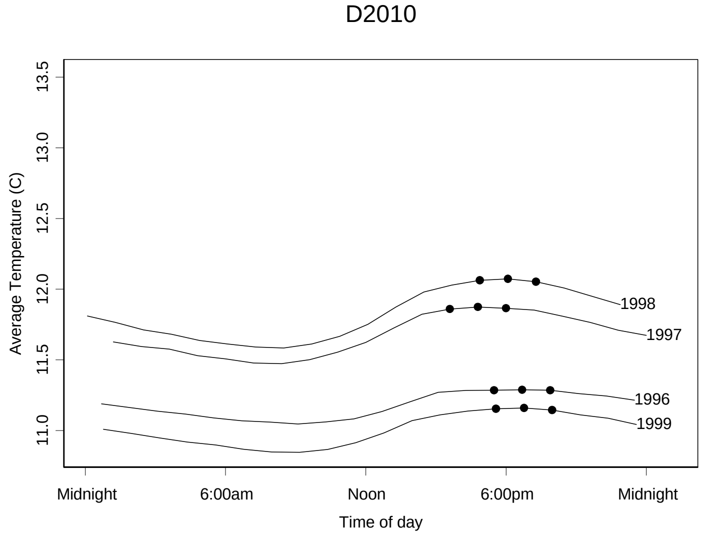
Stream (Drainage)	Type of Treatment	Reach Length (ft)	Canopy Closure (%)	Aspect (^{o})
5410 (Dry Creek)-	Control	1755	81	320
5410 (Dry Creek)-	Harvested	2090	73	0
6001 (Mad River)+	Control	541	74	10
6001 (Mad River)+	Harvested	764	69	55
Mitsui (Little River)-	Control	856	70	285
Mitsui (Little River)-	Harvested	1312	73	330
D1120 (Dominie Creek)	Control	1605	95	185
D1120 (Dominie Creek)	Scheduled for harvest	*1625	95	200
D2010 (SF Winchuck)+	Control	1125	94	345
D2010 (SF Winchuck)+	Harvested	675	86	350

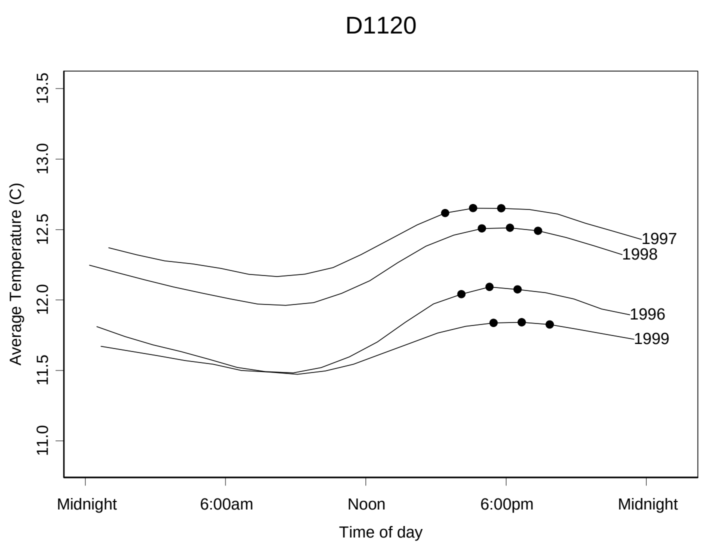
Note
*Asterisks on the reach length for the D1120 indicate the expected length of stream that will be adjacent to the scheduled harvest.

Data loggers were deployed in all streams by early summer each year and collected after 15 September. For analysis, attention was restricted to the time period 1 August to 15 September. This time period is generally the warmest time of year in Northern California. Upstream and downstream temperatures collected on a single stream were matched according to the time of day they were recorded and the difference between them (downstream - upstream) was calculated every 1.2 hours. To identify a response variable that quantified the amount of heat gain produced in the treatment area, intra-day temperature profiles were computed that identified the warmest time of day for each stream each year. The three temperature readings closest to the warmest time of day for each stream were defined to be the *maximum temperature window*. The intra-day temperature profiles used to define the maximum temperature window and, consequently, the daily maximum temperature differences appear in Figure C5-1. In Figure C5-1, values from all four temperature probes (i.e., the upstream and downstream probes on both the treatment and control streams) were averaged every 1.2 hours to arrive at an estimate of overall average water temperature. The three readings that defined the maximum temperature window for each stream each year have been plotted as circles in Figure C5-1. Across streams and years, the maximum temperature window varied from 2:00 pm to 9:07 pm. The warmest time of day for the five study sites was, on average, 5:45 pm.

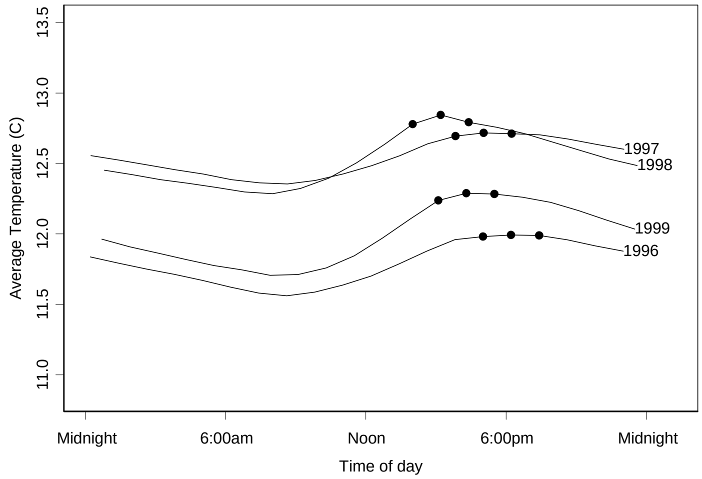
The maximum downstream – upstream temperature difference that occurred within the daily maximum temperature window was computed and used as the response variable in the BACI analysis. For example, suppose that the three temperature readings nearest to the warmest time of day at a stream occurred at 5:00 pm, 6:12 pm, and 7:24 pm. For each day between 1 August and 15 September, the difference between the downstream and upstream probe at 5:00 pm, 6:12 pm, and 7:24 pm was computed. The maximum of these three differences was used as the response variable in the BACI analysis for that particular day. One maximum difference was computed for each day.

Figure C5-1. Initial five study sites shown below with smoothed daily water temperature profiles computed from the mean of all four temperature probes (i.e. upstream and downstream from the treatment and control streams). Dots show recordings defining the daily maximum temperature window for each site.

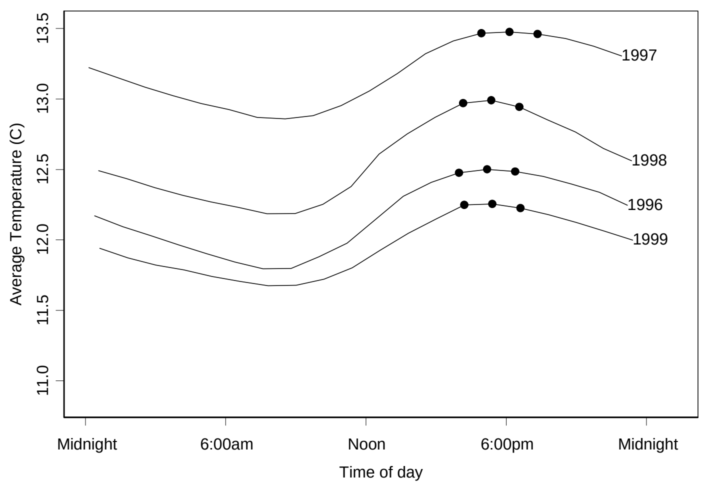


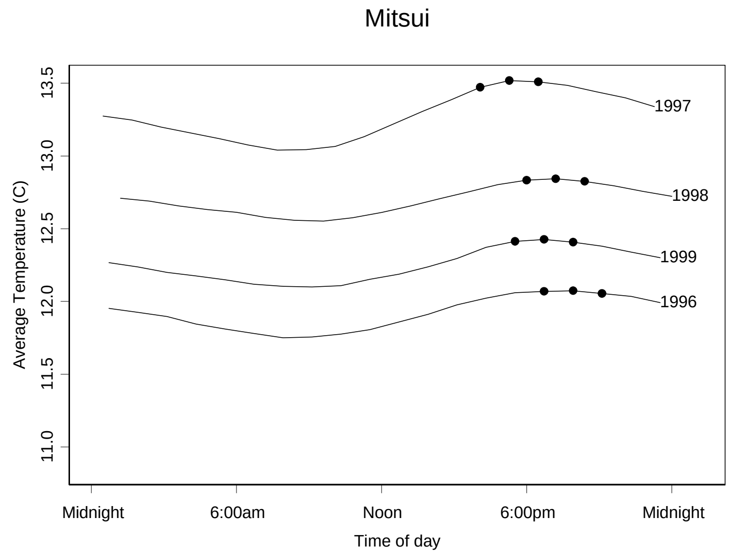


6001


Average Temperature (C)
13.5
13.0
12.5
12.0
11.5
11.0
1997
1998
1999
1996
Midnight
6:00am
Noon
6:00pm
Midnight
Time of day

5410


Average Temperature (C)
13.5
13.0
12.5
12.0
11.5
11.0
1997
1998
1996
1999
Midnight
6:00am
Noon
6:00pm
Midnight
Time of day



Given the serial nature of the daily temperature recordings, the data were assessed for temporal auto-correlation. Significant auto-correlation existed in the yearly time series of maximum temperature differentials at each site. Where significant auto-correlation was found, error estimates were adjusted to correct for the estimated auto-correlations. (See Attachment A below for details.)

The statistical analysis used to assess harvest impacts was a modified BACI analysis. The modification was made necessary due to the estimated auto-correlations in the daily temperature recordings. BACI analyses assess the lack of parallelness in response profiles through time. This lack of parallelness was measured by the treatment by time (year) interaction from an analysis of variance (ANOVA) with time as one factor and treatment as the other. The BACI analysis allows the overall level of responses to be different between control and treated sites both before and after treatment, but requires the after treatment *difference* in control and treated responses to be the same as the before treatment *difference* in control and treated responses. If the after treatment difference in responses is different from the before treatment difference in responses, the BACI analysis will show that there was significant change in treatment areas after application. Differences between sites in the direction and magnitude of temperature changes after harvest became apparent upon plotting of the data. Given the variability in which individual streams responded to the treatment, each site was analyzed separately and no statistical inference to other sites was possible. Additional information on the use of ANOVA in the BACI estimation process can be found in McDonald et al. (2000). Additional details specific to this study can be found in Attachment A below.

C5.2.2.2 Results of Preliminary Analysis

Significant auto-correlation existed in the yearly time series of maximum temperature differential at each site. Estimated correlation of maximum temperature differential values that were one day apart ranged from 0.49 at D1120 to 0.81 at 5410. Auto-correlation at D2010, D1120, and 6001 was negligible between values separated by more than 5 days. Auto-correlation at 5410 and Mitsui was negligible between values separated by 13 or more days.

Table C5-15 contains estimated mean maximum temperature difference and standard errors between the downstream and upstream temperature probes for all streams each year of the study. Means and standard errors in Table C5-15 were estimated from the BACI model adjusting for auto-correlation. Positive values indicate that the average maximum downstream temperature was warmer than the upstream temperature, while negative numbers indicate the reverse. Average heating or cooling between the upstream and downstream probes was variable.

Table C5-16 contains estimated average maximum temperature differences before and after timber harvest. (D1120 is missing from Table C5-16, because it had not yet been harvested.) After harvest, D2010 and 6001 experienced an increase in the maximum temperature differential, while Mitsui and 5410 experienced a decrease relative to their control streams. The 95% confidence intervals for the increases at D2010 and 6001, and decreases at Mitsui and 5410 did not include zero and therefore should be considered “significantly” different from zero.

D1120 was not harvested during the course of data collection and provided a check of the appropriateness of BACI analysis. Under similar conditions, the BACI analysis hypothesizes that the profile of temperature responses through time on the treatment and control streams should, within statistical error, be parallel to one another. Figure C5-2 plots the estimated profile of average maximum temperature differential across years for D1120. Assuming a hypothetical harvest occurred in winter 1996/1997, the estimated change in maximum temperature differential on the hypothetical treatment stream was 0.013^{o}C with approximate 95% confidence interval of -0.149^{o}C to 0.175^{o}C. Applying the same hypothetical treatment to the following year, the estimated change in maximum temperature differential on the hypothetical treatment stream was -0.082^{o}C with approximate 95% confidence interval of -0.223^{o}C to 0.058^{o}C. The profiles plotted in Figure C5-2 are parallel within the limits of statistical error, because the associated confidence intervals contain zero.

Plots of the estimated mean maximum downstream-upstream differences from Table C5-15 were plotted in Figure C5-3 below along with the average maximum temperature differential expected by the BACI analysis had there been no harvest. With no treatment effect, the expected mean treatment profiles were parallel to the control stream profile.

Table C5- 15. Yearly estimated mean maximum downstream-upstream temperature differences of the initial five sites. [1]

Mean Maximum Downstream-Upstream Temperature Difference, °C			
Stream	**Year**	**Treatment Stream (SE)**	**Control Stream (SE)**
D2010	1996	0.839 (0.101)	0.991 (0.101)
	1997	1.601 (0.101)	1.436 (0.101)
	1998	1.705 (0.101)	1.029 (0.101)
	1999	1.288 (0.101)	1.234 (0.101)
D1120	1996	0.952 (0.051)	0.175 (0.051)
	1997	1.300 (0.051)	0.393 (0.051)
	1998	0.977 (0.051)	0.136 (0.051)
	1999	0.764 (0.051)	0.176 (0.051)
6001	1996	0.392 (0.087)	0.240 (0.087)
	1997	0.787 (0.087)	0.293 (0.083)
	1998	1.484 (0.087)	0.226 (0.083)
	1999	1.227 (0.088)	-0.243 (0.088)
5410	1996	0.316 (0.099)	1.227 (0.099)
	1998	-0.026 (0.095)	1.423 (0.095)
	1999	-0.041 (0.101)	1.480 (0.101)
Mitsui	1996	-0.146 (0.125)	-0.071 (0.125)
	1997	-0.928 (0.125)	0.135 (0.125)
	1998	-1.294 (0.125)	0.007 (0.125)

Note
1 All measurements in Celsius. Standard errors estimated from BACI model.

Table C5-16. Estimated average maximum temperature differences before and after harvest on four sites where harvesting occurred prior to 1999. [1]

Stream	Harvest Period	Estimated Average Maximum Temperature Difference, °C		Estimated Change After Harvest, °C (SE)	Approximate 95% Confidence Interval on Increase
		Treatment (SE)	Control (SE)		
D2010	Before	0.756 (0.098)	0.898 (0.098)	0.497 (0.16)	0.182 to 0.811
	After	1.515 (0.057)	1.16 (0.057)		
6001	Before	0.535 (0.061)	0.139 (0.061)	1.044 (0.123)	0.803 to 1.286
	After	1.323 (0.062)	-0.117 (0.062)		
5410	Before	0.178 (0.139)	0.486 (0.139)	-1.372 (0.239)	-1.84 to -0.904
	After	-0.368 (0.096)	1.312 (0.096)		
Mitsui	Before	-0.214 (0.129)	-0.222 (0.129)	-1.31 (0.224)	-1.748 to -0.871
	After	-1.28 (0.091)	0.022 (0.091)		

Note
1 Values of change after harvest (Column 5) quantify the lack of parallelism in temperature differential profiles and are equal to the interaction effects in the BACI ANOVA. For example, at D2010 estimated change after harvest equaled 0.497 = (1.515-0.756)-(1.16-0.898). Positive numbers for change after harvest indicate heating of the treatment section after harvest relative to the control section. Negative numbers indicate cooling of the treatment section after harvest relative to the control section.

Figure C5-2. Estimated means at D1120 where no harvest has occurred. Hollow circles and dashed line indicate perfect parallelness between treatment and control streams. Filled circles show actual estimates.

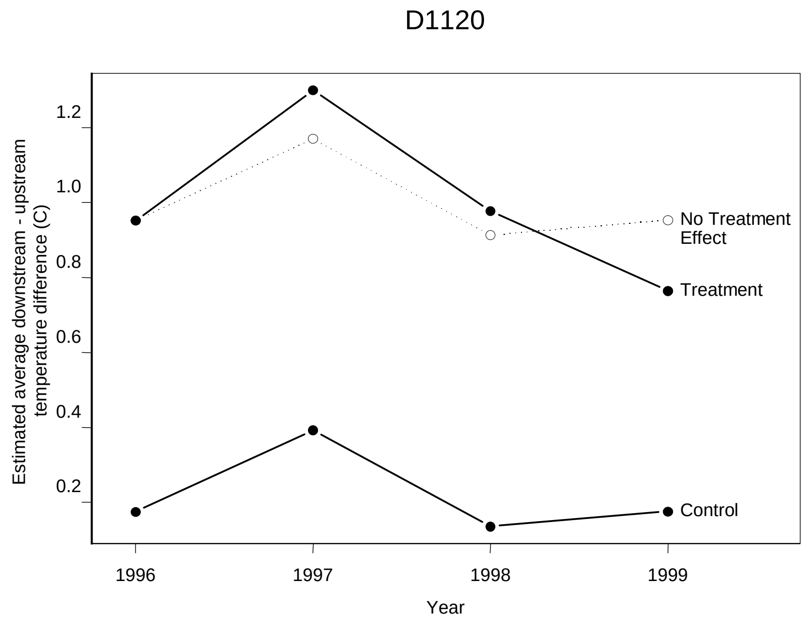
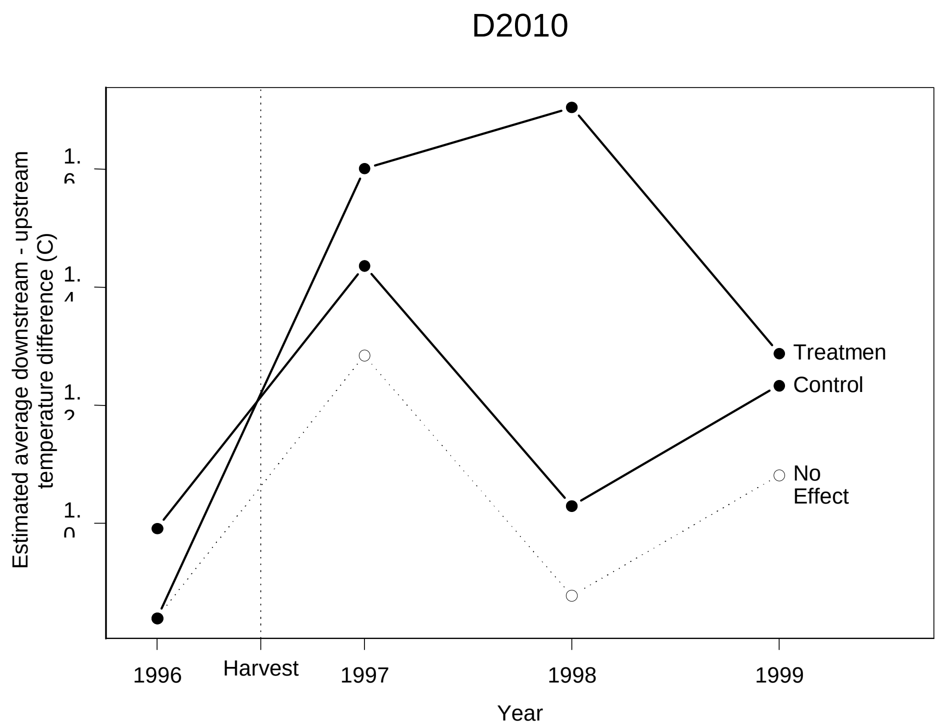
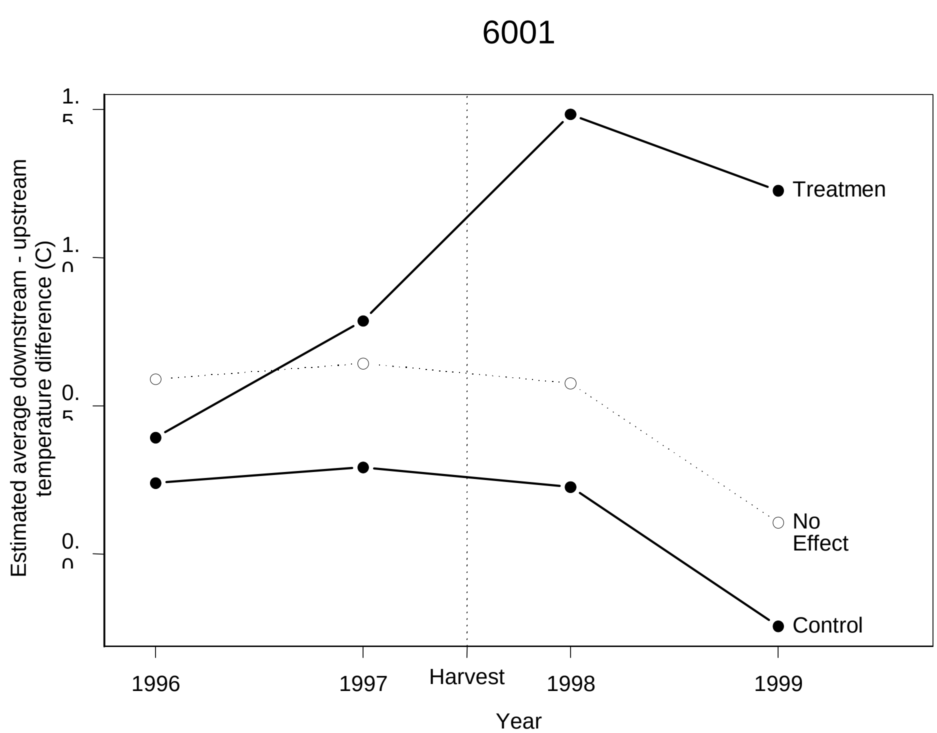
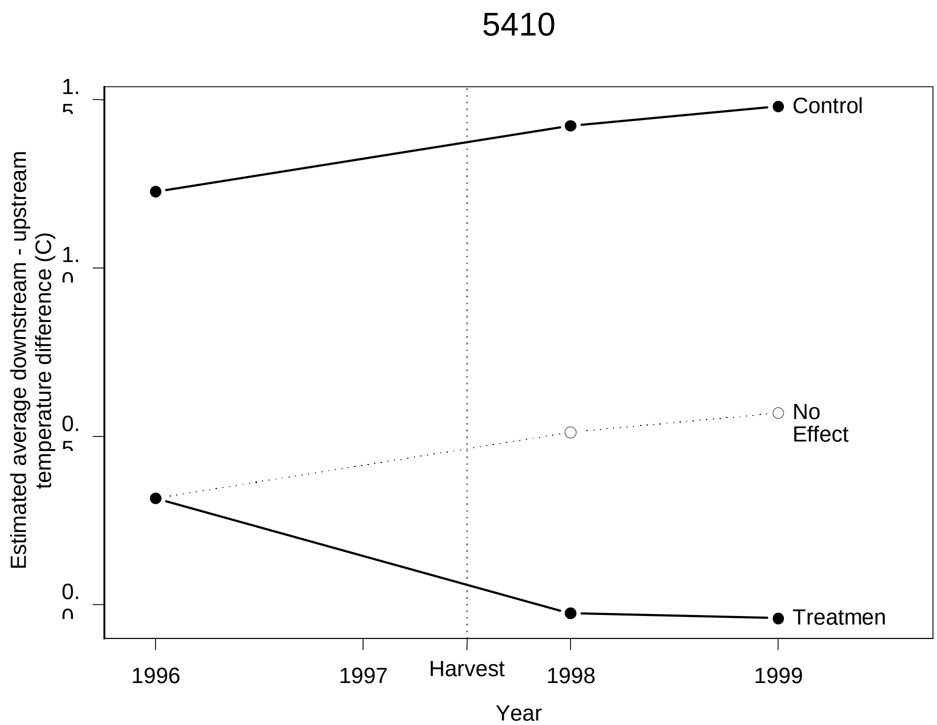


Figure C5-3. Estimated means before and after harvest from the BACI model adjusted for auto-correlation. Filled circles show actual estimates, while hollow circles show locations of treatment means under the hypothesis of no treatment effect. Monitoring stopped in 1998 at Mitsui, because timber surrounding the control stream was harvested during winter 1998.



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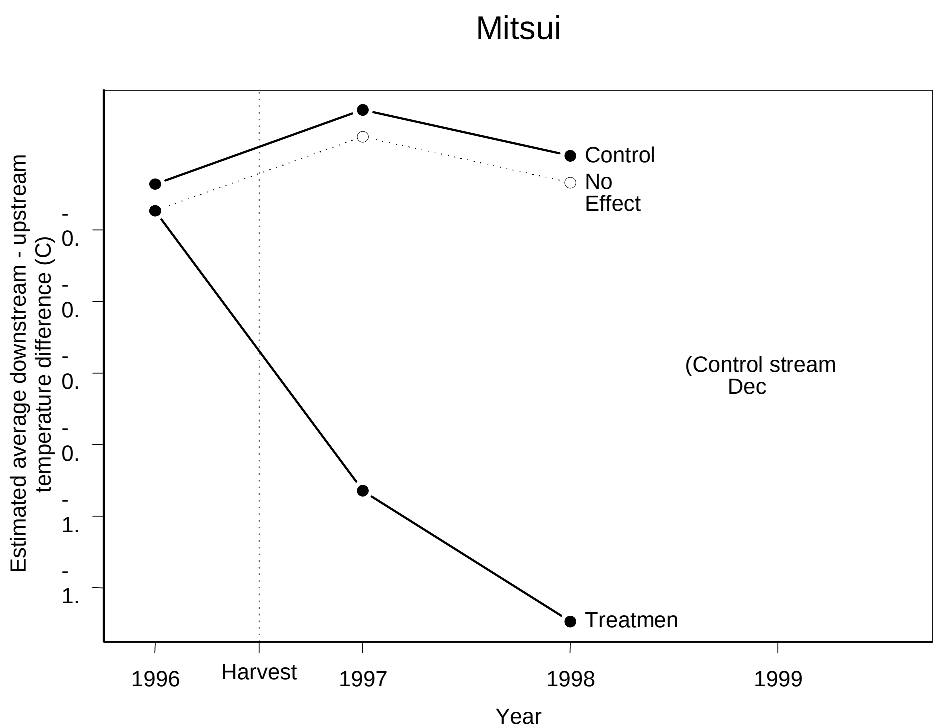
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C5.2.2.3 Discussion

The impacts of timber harvest on water temperature on small Class II watercourses were assessed at the warmest time of day during the warmest time of the year. This was done to insure the maximum test of the effectiveness of riparian buffers in mitigating the potential impacts of increased water temperatures following clearcut timber harvest adjacent to a watercourse. In addition, the assessment was focused on the warmest time of the year, since it is believed that the Covered Species are most likely to be impacted by increases in water temperature that may cause water temperature to exceed some biological threshold. It is also important to note that the retention standards on the riparian buffers were significantly less than what is being proposed in the AHCP. The riparian buffers all followed standard forest practice widths, but with Green Diamond's minimum 70% total canopy retention (retention standard created by Green Diamond's NSO HCP).

Empirical data and theoretical models of water temperature profiles indicate that water temperature generally increases in most watersheds as water flows downstream during the warmest times of the year (Beschta et al. 1987). Increases in the water temperature are the result of multiple factors, but typically most of the increased thermal energy of the water results from the air temperature being elevated relative to the water temperature. The rate of increase is largely a function of the temperature differential between air and water. Therefore, if air temperature increases in the riparian areas following timber harvest, one would predict an increase in the rate at which water temperature warms as it flows downstream through the harvested area.

The thermal profiles of the monitored streams indicated that the changes in water temperature as it flowed downstream was a rather complex process and did not always fit the pattern of increased warming as water flowed downstream. As noted in Table C5-16, mean water temperature decreased rather than increased as it flowed downstream during at least one year in four of the ten streams. Monitoring reaches were selected to insure that tributaries did not enter within the sample reach, so these decreases were most likely due to ground water inputs or changes in the microclimate within the stream reach.

Fortunately, this study was designed using a BACI approach, which controlled for unexpected patterns in the thermal profiles of either the treatment or control streams. All that was necessary for a valid experiment was for the relationship between treatment and control streams to remain constant through time minus a treatment effect. The results from the D1120 (Figure C5-2) provided support that this assumption was valid.

The data from this study are preliminary, but already it is apparent that the response of water temperature to timber harvest in small headwater streams is complex. All of the treatment streams showed a significant change in water temperature relative to the controls streams following timber harvest, but in two of the sites, the treatment streams were warmer while the other two were colder. There are no other data to help provide clues as to why these sites responded in opposite directions to timber harvest, but Green Diamond speculates that it may be due to altered hydrology. Clearcutting adjacent to a stream should increase the amount of water that is retained in the soil for a few years following harvest primarily due to a reduction of evapotranspiration water losses. If some treatment streams had groundwater inputs while others did not, it would be possible that the increased groundwater could result in relatively cooler water temperatures following harvest in those treatment streams with groundwater inputs. Those treatment streams without significant groundwater inputs would have the greater potential to experience increases in water temperature following harvest. If this pattern persists in additional monitored sites, one would conclude that the cumulative effect of timber harvest on water temperature in small Class II watercourses within a watershed should net to zero.

The retrospective study of water temperature did not allow us to assess changes in water temperature following timber harvest, but the results were consistent with the observations of the BACI study. Cut and uncut streams varied in terms of which streams were colder and there was no statistical difference in the mean values for the streams.

It is also important to note that the magnitude of the differences following harvest, regardless their direction, were quite small (about 0.5 to 1.4^{o}C) even thought the streams were being analyzed during the annual extremes in elevated water temperatures. In addition, the peaks in water temperature only lasted a few hours in the late afternoon and early evening. Green Diamond believes that it is unlikely that the magnitude of these temperatures would have a biological impact on any of the Covered Species given the 7DMMX reported for most of the Class II watercourses within the Plan Area. (See Water Temperature Monitoring above.)

C5.2.2.4 Conclusions

The Class II water temperature monitoring is in the early phases of a long term study that will include additional sites along with additional post-harvest monitoring on the existing sites. As such, these data should be considered preliminary. However, pre-AHCP mitigation measures associated with small Class II watercourses appear to prevent large magnitude changes in water temperature following timber harvest. Presumably, the increased protection measures afforded Class II watercourses in the AHCP will further reduce the likelihood of temperature impacts due to timber harvest. Green Diamond believes that the small magnitude and reversed direction of the temperature changes following timber harvest will not result in any direct or cumulative biological impacts on any of the Covered Species.

C5.2.3 References

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C5.2.4 Attachment A to BACI Class II Temperature Monitoring

This attachment describes estimation of the BACI model and correction for auto-correlation in the data. The analysis is described in three steps; 1) ordinary least squares parameter estimation, 2) auto-correlation modeling, and 3) weighted linear regression.

C5.2.4.1 Ordinary Least Squares Parameter Estimation

Step one of the analysis fit a Normal theory regression model to indicator variables delineating treatment and control observations and before and after observations. Let x_{ti} be an indicator variable whose value was 1 if observation i came from the treatment stream, 0 otherwise. Let x_{97i} be an indicator variable whose value was 1 if observation i was collected in 1997, 0 otherwise. Similarly, let x_{98i} be an indicator function with value 1 if observation i was collected in 1998 and let x_{99i} be an indicator function with value 1 if observation i was collected in 1999. Step one of the analysis fit the regression model,

$$\mathbf{E[y_i]} = \beta_0 + \beta_1 \mathbf{x_{ti}} + \beta_2 \mathbf{x}_{97i} + \beta_3 \mathbf{x}_{98i} + \beta_4 \mathbf{x}_{99i} + \beta_5 \mathbf{x_{ti}} \mathbf{x}_{97i} + \beta_6 \mathbf{x_{ti}} \mathbf{x}_{98i} + \beta_7 \mathbf{x_{ti}} \mathbf{x}_{99i}$$

where y_i was the maximum difference between downstream and upstream temperature readings on day i that occurred during the maximum temperature window.

Estimates of the overall before-after control-impact interaction (i.e., the difference of differences in means) were computed using contrasts of coefficients in the model (McDonald et al., 2000). For example, the overall BACI contrast for a pair of streams harvested in winter 1996/1997 was,

$$\mathbf{BACI}_{96} = -1/3\beta_5 - 1/3\beta_6 - 1/3\beta_7$$

The overall BACI contrast for a pair of streams harvested in winter 1997/1998 was,

$$\mathbf{BACI}_{97} = 1/2\beta_5 - 1/2\beta_6 - 1/2\beta_7$$

Let μ_{BT} be the mean response on the treatment stream before treatment. Let μ_{BC} be the mean response on the control stream before treatment. Let μ_{AT} be the mean response on the treatment stream after treatment, and let μ_{AC} be the mean response on the control stream after treatment. The BACI contrasts listed above both estimate,

$$(\mu_{BT} - \mu_{BC}) - (\mu_{AT} - \mu_{AC})$$

The negative of these BACI contrasts appear in column 5 of Table C5-16 above.

C5.2.4.2 Auto-correlation Modeling

Step two of the analysis assessed and modeled auto-correlations among residuals of the regression fit during step one. No auto-correlations were checked among residuals from different streams or different years. Auto-correlations among residuals from different stream or years were assumed to be zero. If significant auto-correlation were found in the residuals of the regression model, a non-linear variance model was fit to the correlations and an estimated residual variance-covariance matrix was constructed. The variance model used at this step was of such a form that non-singularity of the resulting variance-covariance matrix was assured.

The significance of auto-correlations among residuals of the original model were assessed using Moran's I (Moran, 1950) statistic at various separations in time (time lags). If a (Bonferronni corrected) 95% confidence interval surrounding Moran's I did not overlap zero, the auto-correlation was deemed significant.

Provided significant auto-correlations existed, a *spherical* correlation model was fit to observed correlations. The spherical variance model was fit by forming all possible pairs of residuals and calculating the statistics $z_{ij} = (r_i - \mu_r)(r_j - \mu_r)/s_r^2$, where r_i was the model residual from the *i*-th observation and s_r^2 the sample variance of the residuals. The z_{ij} were then plotted against the time between observation *i* and observation *j* to form a correlation scatter gram. The correlation scatter gram was then smoothed using a Gaussian kernel smoother (Venables and Ripley, 1994; Statistical Sciences, 1995). The spherical correlation model was fit to the smoothed correlation scatter gram using non-linear least squares techniques. The spherical correlation model contained two parameters and had the form,

$$v(d_{ij}) = \begin{Bmatrix} c_1\left(1 - \frac{3}{2}\frac{d_{ij}}{h_0} + \frac{1}{2}\left(\frac{d_{ij}}{h_0}\right)^3\right) \text{ if } 0 \le \mathrm{d_{ij}} \le \mathrm{h_0} \\ 0 \text{ if } \mathrm{d_{ij}} > \mathrm{h_0} \end{Bmatrix}$$

where d_{ij} was the time between observation i and j. Based on the significance of auto-correlations beyond 20 days, only d_{ij} less than 20 days were considered when fitting the spherical model.

C5.2.4.3 Weighted Linear Regression

Step three of the analysis used the estimated residual variance-covariance matrix from step 2 as a weight matrix to recompute the coefficients of the regression model obtained at step one. Standard errors for coefficients and contrasts were also recomputed using elements of the estimated variance-covariance matrix as weights. Specifically, if **X** was the design matrix containing the indicator variables used in the regression model at step one, **Y** was the vector of responses, and **V** was the estimated residual variance-covariance matrix obtained at step two, then the recomputed vector of coefficients, $\hat{\beta}$, and variances were,

$$\hat{\beta} = (X'V^{-1}X)^{-1}X'V^{-1}Y$$
$$\mathrm{var}(\hat{\beta}) = (X'V^{-1}X)^{-1}.$$

In this study, interest was in the BACI contrasts defined above. Variance of the BACI contrasts were computed as,

$$\mathrm{var}(\mathrm{BACI}) = \mathrm{var}(\mathrm{x}\hat{\beta}) = \mathrm{x}(X'V^{-1}X)^{-1}x'$$

where **x** was the vector of constants defining the contrast.

Appendix C6. Fish Presence/Absence Surveys

CONTENTS

C6.1 INTRODUCTION AND PURPOSE

Fish presence/absence surveys are ongoing across the Plan Area. The purpose of the presence / absence (P/A) survey is to positively identify a stream reach of interest as a Class I (fish bearing) or Class II (non-fish bearing) watercourse. These surveys are primarily employed in association with a proposed Timber Harvesting Plan (THP) and are intended to assist the RPF with a proper identification of watercourse reaches within the proposed THP. However, the P/A Survey may on occasion be used to identify watercourse reaches not associated with a THP. Both situations will serve to help Green Diamond to better understand and manage for the public trust resources located within the Plan Area.

A key assumption of these surveys is that it is specifically understood that only the presence of fish species can be absolutely proven. Absence of fish can only be inferred from a lack of presence.

C6.2 METHODOLOGY

C6.2.1 Materials

- Appropriate Safety Equipment
- Backpack Electrofisher
- Dip Nets
- Maps and/or aerial photos of area

C6.2.2 Methods

The watercourse reach of interest shall be searched in an upstream direction whenever reasonable. The electrofisher settings shall be adjusted to the least harmful, yet effective setting possible (begin with P-16). Electrofishing will occur in appropriate salmonid habitat such as slower water and pools.

If fish are observed; capture the first few fish in order to identify to species and then release immediately. Continue working upstream, once fish are observed in a pool discontinue shocking and proceed to the next appropriate salmonid habitat. Continue until the reach of interest is covered or 1000' past the last observed salmonid.

If no fish are observed; confirm that the electrofisher unit is working properly. Search for an amphibian species, usually a Pacific giant salamander (*Dicamptodon ensatus*), and observe its behavior during shocking (shock the water within 3 feet of the amphibian, not the organism itself). If the amphibian responds to the electrofishing, then continue working upstream searching for fish. If the organism does not respond, double-check the settings and all connections on the electrofisher unit. Confirm that the warning beeper is working. Re-shock the pool and observe the amphibian. If there is still no response, increase the electrofisher units' settings to I-5 at 300 volts. Re-shock. If there is still no response, discontinue electrofishing and troubleshoot the electrofishing unit. If the amphibian responds, continue working upstream searching for salmonids until the reach of interest has been covered or 1000’ past the last observed salmonid or known Class I watercourse.

C6.2.3 Follow-up

Once presence or absence has been determined this information will be reported to the Plan Coordinator. A map showing the exact location of electrofishing activities and a summary of field notes describing what was found during the survey will be provided to the Plan Coordinator. All information will also be recorded on the Fish and Herp base maps to update the map records.

C6.3 RESULTS AND DISCUSSION

The presence/absence survey information will be entered into Forest Resources Information System (FRIS) database and the results appropriately incorporated into the Timber Harvest Plan (THP). A series of GIS based (FRIS) maps will be continuously updated with information obtained from the presence/absence surveys. The maps and database provides current information on the distribution of fish on a property wide basis. The current fish distribution maps and tables for each HPA are presented in Section 7.

C6.4 CONCLUSIONS

A presence/absence survey is a valuable technique to establish Class I watercourse determinations and fish species distributions across the Plan Area on a site-specific basis. The extent of anadromy for streams is generally known across the Plan Area with the exception of the actual extent for each individual species. The presence/absence surveys are primarily used to delineate the extent of resident populations in low order Class I watercourses.

Appendix C7. Summer Juvenile Salmonid Population Estimates

CONTENTS

Tables

C7.1 INTRODUCTION

In 1995, data collection on the summer populations of juvenile coho salmon and 1+ and older steelhead was initiated in three Plan Area streams: South Fork of the Winchuck River (Smith River HPA), Wilson Creek (Smith River HPA), and Cañon Creek (Mad River HPA). Since 1995, data collection has occurred annually on these three original creeks for chinook salmon, and cutthroat trout in addition to coho salmon and steelhead. Four more creeks were added in 1998: Hunter Creek (Coastal Klamath HPA); Lower South Fork Little River, Railroad Creek, and Upper South Fork Little River (all Little River HPA). Sullivan Gulch (North Fork Mad River HPA) was added to the program in 1999. The purpose of these population surveys is to estimate and monitor summer populations of young-of-the-year coho salmon, chinook salmon, steelhead and cutthroat trout. Dive counts estimate salmonid population size during summer low flow periods (August-September). These fish represent the population of juvenile salmonids that will be shortly out-migrating or over-wintering in Plan Area streams.

C7.2 METHODS

The 1995 effort was part of a pilot study to test and refine a sampling methodology developed by Drs. Scott Overton and David Hankin in conjunction with funding through the Fish, Farm and Forest Communities Forum (FFFC). Juvenile salmonid population sampling has evolved since the program's inception in 1995. The population estimate methodology was based on the Hankin and Reeves (1988) two-phase survey design, with the most recent modifications being incorporated from Hankin (1999). These changes have been adopted to improve statistical validity, reduce variance, increase efficiency in the field, and reduce electrofishing effort. The current protocol is especially appropriate for small streams containing special status species where injury and mortality are a concern from a federal Endangered Species Act "take" stand-point.

The current protocol allows for increased use of diver counts for estimating the abundance of juvenile salmonids in streams. This approach reduces the need for electrofishing and related possible mortality of special status species (e.g. coho salmon).

The first phase of the current sampling design classifies habitat units into riffles, runs, pools, and deep pools, measures dimensions of each unit, and then randomly selects a fraction of units in each habitat class for phase 1 sampling (employing the Adaptive Sequential Independent Sampling [ASIS] method [Hankin 1999]). ASIS is used in first and second phase unit selection permitting habitat mapping and unit selection decisions to be made in the field. Phase 1 sampling consists of diving each selected unit to obtain an initial count of salmonids within the sampling unit. Riffle segments are electrofished as diving cannot be conducted in riffles. A subset of the sampled units is then randomly selected for calibration using the ASIS method. The mode of calibration (2nd phase sampling) is determined by the number of individuals counted in each unit. If the initial dive count is less than 20 individuals (of a given species), calibration is conducted by Method of Bounded Counts (Robson and Whitlock 1964). The Method of Bounded Counts (MBC) is utilized to calibrate dive counts when the unit population size is small ($n<20$), producing a substantial reduction in electrofishing effort. If the initial dive count of the target species exceeds 20 fish, calibration is made by four-pass removal electrofishing method. Calibration within deep-pool stratums is made only by MBC, as electrofishing is inefficient in this habitat stratum. In riffles selected for calibration, a 2 to 3 pass-removal electrofishing method is the mode of calibration.

If the method of bounded counts is the mode of calibration the 3 additional diver counts are made immediately following the 1 phase dive counts. If the 2nd phase sampling is conducted by the 4 pass-removal electrofishing method the electrofishing is conducted within no more than 2 days following phase 1 sampling. The methods employed for sample selection and estimation, the ASIS methodology, and phase 2 calibration methods are those of Hankin (1999). Additional discussion of the applicability and assumptions of the population estimation methodology employed by Green Diamond are found in Hankin (1999).

This protocol has also been slightly modified from previous years to provide more consistency between individual crews and from year to year. In the past, the difference between a deep pool and a shallow pool was based on processional judgment on whether or not the habitat mapping crew thought it possible to effectively electrofish a particular unit. If a pool was considered to be too complex; i.e. too much large woody debris (LWD), small woody debris (SWD), or deep undercut banks, it was classified as a deep pool and only calibrated by repeated dive counts.

Since 1999, pools less than 1.1 meters in depth are considered shallow pools and pools greater than or equal to 1.1 meters in depth are considered deep pools regardless of cover. This provided better consistency between crews, allowing comparisons of population estimates between different streams, crews, and property owners. The reduction in total number of deep pools and the corresponding increase in shallow pools is a result of this protocol change and not in the quality or quantity of available habitat. Green Diamond believes that this change to the protocol has also provided a much better estimate due to the increased number of calibrated shallow pools. The complexity of the pool does not appear to influence the ability to effectively electrofish those units.

C7.3 RESULTS

The summarized results of the summer juvenile population estimates for the 8 Plan Area streams are presented in Tables C7-1 through C7-4. The summer juvenile population estimates and the (+/-) 95% confidence interval (C.I.) for coho salmon for the years 1995 through 2000 are shown in Table C7-1. Table C7-2 summarizes the summer juvenile population estimates and (+/-) C.I.s for steelhead for the years 1995 through 2000. Tables C7-3 and C7-4 provide summaries of juvenile summer population estimates and corresponding (+/-) 95% C.I.s for cutthroat trout and chinook salmon respectively, for the years 1996 through 2000.

C7.4 DISCUSSION

C7.4.1 Methodology Effectiveness

The modified Hankin and Reeves juvenile sampling protocol has worked well for estimating juvenile coho salmon and 1+ steelhead populations. Consideration early in the development of the protocol was also given to cutthroat and chinook. Including cutthroat and chinook as species accounted for in the survey methodology has presented some complications, which are apparent looking at data collected from 1995 to 2000.

Table C7-1. Summer juvenile coho population estimates in eight Plan Areas streams, 1995-2000.

Stream	Year	Habitat	Population Estimate	95% C.I. (+/-)
SF Winchuck River	1995	DP, SP, Run, Riffle	Unable to be estimated	
	1996	DP	32	23
		SP, Run, Riffle	4*	n/a
			Total 36	
	1997	DP	156*	n/a
		SP, Run, Riffle	331	140
			Total 487	
	1998	DP	33	7
		SP, Run, Riffle	0	0
			Total 33	
	1999	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
	2000	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
Wilson Creek	1995	DP, SP, Run, Riffle	1370†	212
	1996	DP	357	116
		SP, Run, Riffle	164	123
			Total 521	
	1997	DP	209*	n/a
		SP, Run, Riffle	27*	n/a
			Total 236	
	1998	DP	355	108
		SP, Run, Riffle	25	22
			Total 380	
	1999	DP	0	0
		SP, Run, Riffle	19	21
			Total 19	
	2000	DP	21	18
		SP, Run, Riffle	23	23
			Total 44	
Hunter Creek	1998	DP	317	122
		SP, Run, Riffle	81	88
			Total 398	
	1999	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
	2000	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
Railroad Creek (Little River)	1998	DP	85	34
		SP, Run, Riffle	164	84
			Total 249	
	1999	DP	0	0
		SP, Run, Riffle	339	64
			Total 339	
	2000	DP	14*	n/a
		SP, Run, Riffle	162	79
			Total 176	

Table C7-1 Continued. **Summer juvenile coho population estimates in eight Plan Areas streams, 1995-2000.**

Stream	Year	Habitat	Population Estimate	95% C.I. (+/-)
Lower SF Little River	1998	DP	2,397	282
		SP, Run, Riffle	1,213	312
			Total 3,610	
	1999	DP	1,774	253
		SP, Run, Riffle	6,129	883
			Total 7,903	
	2000	DP	1,403	232
		SP, Run, Riffle	3,364	761
			Total 4,767	
Upper SF Little River	1998	DP	265	101
		SP, Run, Riffle	473	186
			Total 738	
	1999	DP	182	134
		SP, Run, Riffle	1,048	484
			Total 1,230	
	2000	DP	68	89
		SP, Run, Riffle	275	83
			Total 343	
Sullivan Gulch	1999	DP	147	30
		SP, Run, Riffle	636	265
			Total 783	
	2000	DP	10*	n/a
		SP, Run, Riffle	41	37
			Total 51	
Cañon Creek	1995	DP, SP, Run, Riffle	919†	377
	1996	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
	1997	DP	20*	n/a
		SP, Run, Riffle	23	36
			Total 43	
	1998		Not Estimate Made	
	1999	DP	231	101
		SP, Run, Riffle	179	89
			Total 410	
	2000	DP	160	47
		SP, Run, Riffle	123	38
			Total 283	

Notes
* Units not calibrated or no fish observed in calibration units making an estimate impossible. These numbers are a sum of fish observed in non-calibrated units.
† Estimate from Chris Moyer's thesis work.

Table C7-2. Summer juvenile steelhead population estimates in eight Plan Area streams, 1995-2000.

Stream	Year	Habitat	Population Estimate	95% C.I. (+/-)
SF Winchuck River	1995	DP, SP, Run, Riffle	932†	332
	1996	DP	1,092	145
		SP, Run, Riffle	822	150
			Total 1,914	
	1997	DP	237*	n/a
		SP, Run, Riffle	619	230
			Total 856	
	1998	DP	1,459	189
		SP, Run, Riffle	1,069	206
			Total 2,528	
	1999	DP	327	71
		SP, Run, Riffle	768	101
			Total 1,095	
	2000	DP	1,205	175
		SP, Run, Riffle	2,028	463
			Total 3,233	
Wilson Creek	1995	DP, SP, Run, Riffle	1,041†	253
	1996	DP	909	189
		SP, Run, Riffle	960	348
			Total 1,869	
	1997	DP	146*	n/a
		SP, Run, Riffle	100	21
			Total 246	
	1998	DP	875	177
		SP, Run, Riffle	544	96
			Total 1,419	
	1999	DP	331	153
		SP, Run, Riffle	410	124
			Total 741	
	2000	DP	365	149
		SP, Run, Riffle	932	148
			Total 1,297	
Hunter Creek	1998	DP	1,012	351
		SP, Run, Riffle	790	154
			Total 1,802	
	1999	DP	130	42
		SP, Run, Riffle	745	123
			Total 875	
	2000	DP	815	270
		SP, Run, Riffle	1,206	394
			Total 2,021	
Railroad Creek (Little River)	1998	DP	35	54
		SP, Run, Riffle	80	44
			Total 115	
	1999	DP	12	9
		SP, Run, Riffle	64	24
			Total 76	
	2000	DP	5*	n/a
		SP, Run, Riffle	72	35
			Total 77	

Table C7-2 Continued. **Summer juvenile steelhead population estimates in eight Plan Areas streams, 1995-2000.**

Stream	Year	Habitat	Population Estimate	95% C.I. (+/-)
Lower SF Little River	1998	DP	176	61
		SP, Run, Riffle	54	31
			Total 230	
	1999	DP	56	20
		SP, Run, Riffle	157	42
			Total 213	
	2000	DP	23	19
		SP, Run, Riffle	39	17
			Total 62	
Upper SF Little River	1998	DP	132	28
		SP, Run, Riffle	218	55
			Total 350	
	1999	DP	50	11
		SP, Run, Riffle	168	66
			Total 218	
	2000	DP	16	28
		SP, Run, Riffle	236	55
			Total 252	
Sullivan Gulch	1999	DP	10	4
		SP, Run, Riffle	7	8
			Total 17	
	2000	DP	2*	n/a
		SP, Run, Riffle	55	21
			Total 57	
Cañon Creek	1995	DP, SP, Run, Riffle	1,041†	253
	1996	DP	359	99
		SP, Run, Riffle	317	69
			Total 676	
	1997	DP	90	n/a
		SP, Run, Riffle	508	106
			Total 598	
	1998		No Estimate made	
	1999	DP	197	53
		SP, Run, Riffle	375	121
			Total 572	
	2000	DP	348	70
		SP, Run, Riffle	585	93
			Total 933	

Notes
* Units not calibrated or no fish observed in calibration units making an estimate impossible. These numbers are a sum of fish observed in non-calibrated units.
† Estimate from Chris Moyer’s thesis work.

Table C7-3. Summer juvenile coastal cutthroat trout population estimates in eight Plan Area streams, 1995-2000.

Stream	Year	Habitat	Population Estimate	95% C.I. (+/-)
SF Winchuck River	1995	DP, SP, Run, Riffle	No Estimate Made	
	1996	DP	299	56
		SP, Run, Riffle	131	25
			Total 430	
	1997	DP	56*	n/a
		SP, Run, Riffle	331	140
			Total 487	
	1998	DP	283	67
		SP, Run, Riffle	194	39
			Total 477	
	1999	DP	115	32
		SP, Run, Riffle	265	66
			Total 380	
	2000	DP	172	50
		SP, Run, Riffle	302	123
			Total 474	
Wilson Creek	1995	DP, SP, Run, Riffle	No Estimate Made	
	1996	DP	120	47
		SP, Run, Riffle	38	16
			Total 158	
	1997	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
	1998	DP	27	19
		SP, Run, Riffle	3	4
			Total 30	
	1999	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
	2000	DP	15	15
		SP, Run, Riffle	0	0
			Total 15	
Hunter Creek	1998	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
	1999	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
	2000	DP	35	25
		SP, Run, Riffle	15	10
			Total 50	
Railroad Creek (Little River)	1998	DP	0	0
		SP, Run, Riffle	10	6
			Total 10	
	1999	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
	2000	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	

Table C7-3 Continued. **Summer juvenile coastal cutthroat trout population estimates in eight Plan Areas streams, 1995-2000.**

Stream	Year	Habitat	Population Estimate	95% C.I. (+/-)
Lower SF Little River	1998	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
	1999	DP	0	0
		SP, Run, Riffle	82	22
			Total 82	
	2000	DP	1*	n/a
		SP, Run, Riffle	18†	17
			Total 19	
Upper SF Little River	1998	DP	1*	n/a
		SP, Run, Riffle	6	7
			Total 7	
	1999	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
	2000	DP	0	0
		SP, Run, Riffle	4	13
			Total 4	
Sullivan Gulch	1999	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
	2000	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
Cañon Creek	1995	DP, SP, Run, Riffle	No Estimate Made	
	1996	DP	13	13
		SP, Run, Riffle	0	0
			Total 13	
	1997	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
	1998		No Estimate Made	
	1999	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
	2000	DP	17	11
		SP, Run, Riffle	4	4
			Total 21	

Notes
* Units not calibrated or no fish observed in calibration units making an estimate impossible. These numbers are a sum of fish observed in non-calibrated units.
† Estimate made using data from electro-fishing

Table C7-4. Summer juvenile chinook population estimates in eight Plan Area streams, 1995-2000.

Stream	Year	Habitat	Population Estimate	95% C.I. (+/-)
SF Winchuck River	1995	DP, SP, Run, Riffle	No Estimate Made	
	1996	DP	313	101
		SP, Run, Riffle	35	13
			Total 348	
	1997	DP	12*	n/a
		SP, Run, Riffle	85	17
			Total 97	
	1998	DP	688	232
		SP, Run, Riffle	220	163
			Total 908	
	1999	DP	496	208
		SP, Run, Riffle	899	156
			Total 1,395	
	2000	DP	66	26
		SP, Run, Riffle	42	30
			Total 108	
Wilson Creek	1995	DP, SP, Run, Riffle	No Estimate Made	
	1996	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
	1997	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
	1998	DP	3*	n/a
		SP, Run, Riffle	8	13
			Total 11	
	1999	DP	1*	n/a
		SP, Run, Riffle	0	0
			Total 1	
	2000	DP	0	0
		SP, Run, Riffle	1*	n/a
			Total 1	
Hunter Creek	1998	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
	1999	DP	30	37
		SP, Run, Riffle	26	34
			Total 56	
	2000	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
Railroad Creek (Little River)	1998	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
	1999	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
	2000	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	

Table C7-4 Continued. Summer juvenile chinook population estimates in eight Plan Areas streams, 1995-2000.

Stream	Year	Habitat	Population Estimate	95% C.I. (+/-)
Lower SF Little River	1998	DP	4*	n/a
		SP, Run, Riffle	0	0
			Total 4	
	1999	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
	2000	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
Upper SF Little River	1998	DP	0	0
		SP, Run, Riffle	0	0
			Total 0	
	1999	DP	0	0
		SP, Run, Riffle	2*	n/a
			Total 2	
	2000	DP	0	0
		SP, Run, Riffle	6	19
			Total 6	
Sullivan Gulch	1999	DP	2	2
		SP, Run, Riffle	1*	n/a
			Total 3	
	2000	DP	4*	n/a
		SP, Run, Riffle	8	10
			Total 12	
Cañon Creek	1995	DP, SP, Run, Riffle	No Estimate Made	
	1996	DP	23	37
		SP, Run, Riffle	0	0
			Total 23	
	1997	DP	8*	n/a
		SP, Run, Riffle	8	18
			Total 16	
	1998		No Estimate Made	
	1999	DP	249	208
		SP, Run, Riffle	89	48
			Total 338	
	2000	DP	28	15
		SP, Run, Riffle	44	46
			Total 72	

Note
* Units not calibrated or no fish observed in calibration units making an estimate impossible. These numbers are a sum of fish observed in non-calibrated units.

Juvenile population estimates within Plan Area streams continue to include estimates for juvenile chinook (0+) and 1+ cutthroat. Chinook population estimates are relatively small compared to coho and steelhead. In the Plan Area, the majority of the chinook out-migrate before summer low flow conditions are reached, making it difficult to sample a closed population.

Cutthroat greater than 1+ years of age are included in the population estimate, although small populations and species migration patterns may complicate the estimation methodology. Both cutthroat and steelhead can sometimes be difficult to distinguish as young of the year or 1+ fish. Generally, when cutthroat reach a size greater than 120mm, they are easily distinguished from steelhead. By inaccurately distinguishing between “trout” life history stages, the methodology may underscore year class population size and may potentially underestimate or overestimate steelhead and/or cutthroat populations within Plan Area streams that contain sizeable runs of either species. A second concern for estimating cutthroat populations can be drawn from juvenile out-migration trapping results obtained from the Little River drainage. As seen during juvenile out-migrant trapping, a large number of parr and pre-smolting cutthroat are observed moving through the traps during late winter and fall. Steelhead of similar age classes are also observed moving through the traps. The summer population estimates, only include those cutthroat or steelhead that remain in the streams throughout the year. It is possible that the “trout” population is underestimated because a large proportion of the population left the system during winter and fall prior to conducting the summer population estimate. A third concern when applying this methodology to “trout” is the approachability of the species through diver observation. Unlike coho salmon, “trout” are skittish and hide as a diver approaches, making counts difficult and identification sometimes impossible. During Phase 2 calibration, this can affect MBC, which relies on a surveyor’s ability to observe the same fish on subsequent dives.

C7.4.2 Population Size

Juvenile coho population estimates from the Plan Area vary from stream to stream and year to year. In data sets that span a period of five years, juvenile coho population estimates vary widely; increasing in some streams and decreasing in others. Overall, Plan Area streams north of Redwood Creek show a downward progression in coho populations (Table C7-1). Data collected from streams south of Redwood Creek show relatively stable or increasing populations. Studies within these streams have not occurred long enough to infer trends; however, factors such as low winter flows and poor ocean conditions can contribute to poor adult escapement. This observation is supported by spawning surveys that occur within Plan Area streams, which documented little to no returning adult coho. These observations do not always hold true as is discussed under the Spawning Survey section of Appendix C, however, it can help to explain population estimates that observed no coho salmon in some north Plan Area streams (S.F. Winchuck and Hunter Creek).

Steelhead estimates indicate stable or increasing populations both north and south of Redwood Creek (Table C7-2). Juvenile populations within streams north of Redwood Creek tend to show the highest population estimates. Within these streams, habitat conditions may be more suited for this species that has behaviors adapted for swift flowing, higher gradient watercourses, with reduced velocity refuge.

Juvenile cutthroat populations tend to show very limited numbers within Plan Area streams, other than the SF Winchuck. However, presence/absence surveys indicate that cutthroat are widely dispersed across the Plan Area. Cutthroat trout populations tend to decrease south of Redwood Creek and disappear from state records south of the Eel River (Gerstung 1997). Populations of cutthroat trout that often prefer low velocity habitats, may out compete coho within areas like the S.F. Winchuck.

Juvenile chinook salmon tend to out-migrate from Plan Area streams prior to June. The juvenile dive counts take place in the months of August and September during summer low flow. Residual populations of chinook salmon counted during the summer dives demonstrate species presence, but cannot be used for population estimates due to their early season out-migration patterns.

C7.4.3 Summer Habitat Preference

During summer low flows, pool habitat is the preferred habitat type for all species (Tables C7-1 through C7-4), specifically deep pools. Species competition within this habitat type becomes apparent in high production years or in small streams with limited pool habitat available. Other habitat types such as runs and shallow pools are well utilized by all species. Depending on the amount of available habitat during high production years, juvenile coho salmon can be found distributed in all habitat types including riffles. This is likely a result of fully seeded habitats, where intraspecific competition causes redistribution among available habitat types even into “less desirable” rearing habitats such as riffles. In lower production years, such as 2000, coho salmon may be out competed by steelhead or cutthroat trout for deep pool habitat.

C7.5 CONCLUSIONS

Using this protocol to estimate juvenile chinook populations is not recommended, but may work for more northern populations (British Columbia and Alaska) that over-winter in freshwater. It is also not well suited for cutthroat trout due to their limited numbers within Plan Area streams and their tendency to move downstream of survey reaches prior to summer low flows. Overall, juvenile population sampling using the modified Hankin and Reeves survey methodology is very useful for estimating juvenile coho populations, and appears to be well suited for 1+ steelhead trout, although significant numbers of steelhead can be observed moving downstream prior to summer surveys. Juvenile coho are generally unafraid of divers and are very approachable. Identification is simple, using both physical attributes and their distinct behavior as key identifiers. Steelhead are skittish and not often seen during subsequent Phase 2 calibration dives, never-the-less 95% C.I. indicate limited variation among population estimates for this species.

Juvenile coho populations within the Little River watershed appear stable and well seeded in all three-survey years, and in the majority of Little River tributaries. Population estimates north of Little River may reflect habitat conditions more suitable for steelhead, however many other factors including adult escapement and interspecific competition could account for the observed estimates. Steelhead 1+ juveniles appear to be distributed in sizable numbers in all surveyed Plan Area streams. While changes (positive or negative) in summer population estimates is clearly of interest, it remains unclear what, if any, changes can be related to management. Currently, population trends cannot be inferred from available data for any of the species, however these estimates may help determine relationships between coho populations in different streams throughout the Plan Area, and the climactic and/or habitat conditions which affect summer population size, when combined with other monitoring efforts.

C7.6 REFERENCES

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Appendix C8. Out-Migrant Smolt Trapping

CONTENTS

Figures

Tables

C8.1 INTRODUCTION

Juvenile salmonid out-migrant (emigrant) smolt trapping has been conducted on several Plan Area streams since 1999. The out-migrant trapping project is designed to monitor the abundance, size, and timing of out-migrating smolts, and to look for long term trends in any or all of these variables. This trapping program is conducted to obtain annual population estimates on emigrating salmonid smolts (coho salmon, chinook salmon, steelhead trout and coastal cutthroat trout). The results of the out-migrant trapping are used in conjunction with the summer population monitoring to estimate overwinter survival in those streams monitored. The juvenile out-migrant trapping also helps to identify factors affecting smolt emigration timing, and establish baseline and long-term trend data on the abundance of juvenile salmonid populations in the watersheds monitored.

During March through July, 1999 Green Diamond conducted juvenile out-migrant trapping for salmonids on the Lower South Fork of the Little River (LSFLR), Upper South Fork of the Little River (USFLR) and Railroad Creek (RRC). These three creeks are all located in the Little River drainage and in the Little River HPA. During March through June, 2000 Green Diamond again conducted juvenile out-migrant trapping for salmonids on the LSFLR, USFLR and RRC as well as adding Carson Creek (CC) to the monitoring program. Like the other three creeks, Carson Creek is located in the Little River drainage.

C8.2 METHODS

C8.2.1 Trapping

Trapping was conducted using a V-notch weir, pipe, and a live-box to capture the juvenile salmonids (Figure C8-1). A second box was attached to the primary box to reduce in-trap predation. Fine mesh screen separated the entrances between the two boxes to serve as a barrier to separate larger fish from the smaller fish. Additional rock cover was provided within the live boxes to serve as refugia for young of the year (YOY) fish. The weirs were constructed with fence posts and wooden pallets. A weir overflow was constructed to provide adult fish passage upstream. The pipe emptied out onto a McBane ramp that dissipated the velocity of the water and gently guided the fish into the box trap. Inside the trap there is a V-shaped panel which creates a large slack water area in the box. This provides an area where the fish can be protected from the stream's current. In 1999, the trap was operated 24 hours a day and checked daily each morning. In 2000, the traps were checked twice daily, in the morning and in the evening. During periods when significant numbers of out-migrants were captured, the trap was checked more frequently as needed. The captured juvenile fish were anesthetized with Alka Seltzer™, identified, measured (fork length) and most were immediately released below the weir.

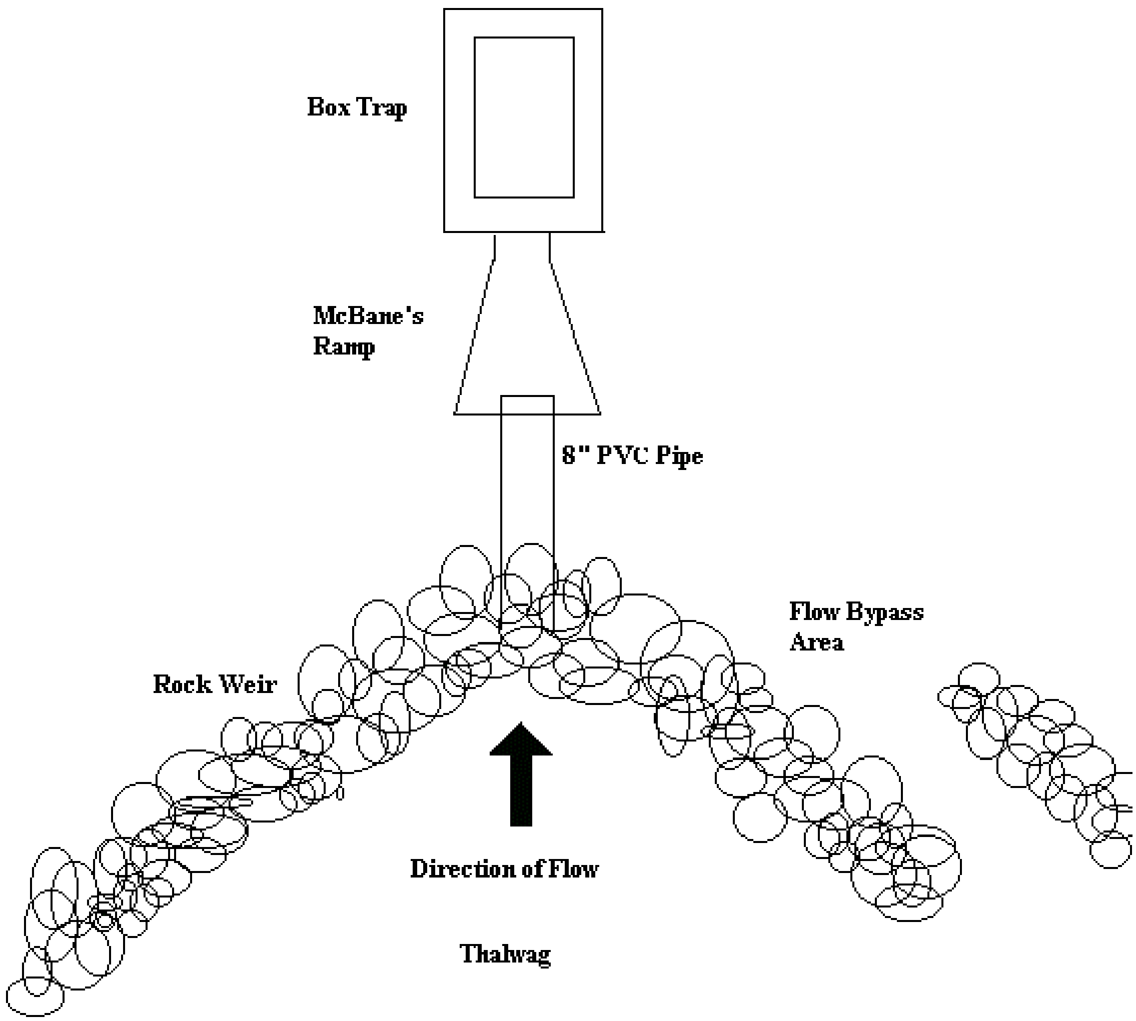

Figure C8-1. Out-migrant fish trapping system (not shown to scale),

Steelhead and cutthroat trout one year or older had their stomachs pumped (gastric lavage) to determine if predation had possibly occurred in the live-box. A subsample of all smolted salmonids were fin clipped (caudal) and released upstream of the weir to determine the trap efficiency. The fin clipped smolts were held in separate live box to determine any possible mortality associated with handling and marking the fish. The smolts were released during the evening trap check period. Recaptured fish from the trap efficiency tests were not used again for subsequent efficiency tests. All caudal fin clip samples from juvenile coho salmon were collected and stored in individual coin envelopes. The samples were air-dried on filter paper and sent to the Bodega Marine Laboratory, University of California. That institution is conducting a study on the genetic variation and population structure of coho salmon in California. Green Diamond is also sending tissue samples from coho carcasses collected during adult escapement surveys to the UC Bodega Bay Marine Laboratory for genetic analyses.

Trap efficiency was calculated by using only species that were actively leaving the drainage on their seaward migration (defined as smolts). Smolts received a fin clip. Four different clips were used throughout the trapping season to test trap efficiency. The easiest clips to identify are caudal fin clips. They were released upstream of the weir in the evening. This allowed the fish ample recovery time and allows for checking for possible mortality from the clipping and handling of them.

C8.2.2 Stomach Pumping (Gastric Lavage)

1+ and older cutthroat and steelhead underwent a stomach pumping procedure to determine predation in the live box. No adult run-back steelhead underwent the pumping procedure. In 1999 the size of the fish that under went gastric lavage ranged from 62-341mm in length. In 2000 the size of the fish ranged from 62-332mm in length. Anesthetized fish were pumped by inserting a small tube down their throat and into their stomach. Water was then pumped into their stomach through the tube. Once the stomach is filled with water, the stomach contents spill out. The contents were then processed. Items were identified to species if possible. After identification the contents were stored in zip-lock bags and preserved with isopropyl alcohol. The pumped fish were placed in a recovery bucket and monitored for approximately one hour prior to their release downstream of the weir. Any stomach pumped smolts were held in a live trap and released during the evening trap check.

C8.3 RESULTS

C8.3.1 Drainage Area and Length of Streams Trapped

A summary of the 1999 project stream drainage area and lengths of utilized habitat above the traps is provided in Table C8-1. In 1999, Green Diamond also quantified habitat conditions in these three streams to assess the survival of juvenile populations in varying freshwater habitats. A summary of the stream drainage and length of utilized habitat above the out-migrant trap during the 2000 out-migrant-trapping project is provided in Table C8-2.

Table C8-1. Drainage area and length of utilized habitat above the trap location for each creek in the 1999 out-migrant trapping study.

	USFLR	LSFLR	Railroad Creek
Drainage area (sq. miles)	5.70	5.31	2.75
Length of available habitat (miles)	1.50	2.16	1.21

Table C8-2. Drainage area and length of utilized habitat above the trap location for each creek in the 2000 out-migrant trapping study.

	USFLR	LSFLR	Railroad Creek	Carson Creek
Drainage area (sq. miles)	5.70	5.31	2.75	3.81
Length of available habitat (miles)	1.1	2.2	0.5	≈2.0

C8.3.2 Population Estimates

Out-migrant smolt population estimates were generated using a preliminary version of software for analysis of stratified mark-recapture data (Bjorkstedt, 2000). The summary of the smolt out-migrant population estimates and their 95% confidence intervals are shown in Table C8-3.

Table C8-3. Summary of the out-migrant population estimated for the years 1999 and 2000.

Coho	**1999**		**2000**	
	Estimate	95% CI	Estimate	95% CI
LSFLR	293	± 21	1,682	± 60
USFLR	27	± 13	147	± 25
Carson Ck	Did Not Trap		1,802	± 30
Railroad Ck	21	± 4	68	± 1
Steelhead	**1999**		**2000**	
	Estimate	95% CI	Estimate	95% CI
LSFLR	103	± 27	46	± 43
USFLR	50	± 7	72	± 3
Carson Ck	Did Not Trap		12	± 3
Railroad Ck	46	± 16	14	± 1
Cutthroat	**1999**		**2000**	
	Estimate	95% CI	Estimate	95% CI
LSFLR	108	± 28	22	± 4
USFLR	35	± 10	13	± 7
Carson Ck	Did Not Trap		60	± 6
Railroad Ck	50	± 5	23	± 1

C8.3.3 Over Wintering Survival

Overwintering survival is calculated by dividing the winter population by the summer population. One of the key assumptions with overwintering survival is that none of the fish in the summer population estimate migrate prior to the downstream migrant trapping being installed. The summer and winter population estimates are shown in Table C8-4.

Table C8-4. Summary of the summer and winter population estimates for the tributaries of the Little River for 1998-1999 and 1999-2000.

Stream	Coho (YOY) Summer Population	Coho Smolts Winter Population	Overwintering Survival Estimate	Drainage Area (Sq. miles)	Miles of Habitat	Summer Fish/Mile	Winter Fish/Mile
	1998	**1999**					
USFLR	738	27	3.7%	5.70	1.5	492	18
LSFLR	3,610	293	8.1%	5.31	2.2	1,641	133
RR Ck	249	21	8.4%	2.75	1.2	208	18
	1999	**2000**					
USFLR	1,230	147	12.0%	5.70	1.1	1,118	134
LSFLR	7,903	1,682	21.3%	5.31	2.2	3,592	765
RR Ck	339	69	20.4%	2.75	0.5	678	138
Carson Ck	NA	1,802	NA	3.81	≈2.0	NA	901

C8.3.4 Species Composition

In 1999 juvenile out-migration trapping captured several different fish (and amphibian) species within the Little River drainage (Table C8-5). The majority of the fish captured were in the genus *Oncorhynchus*. However, there was incidental capture of non-target species mostly lamprey and amphibians. Table C8-6 summarizes the total number of salmonid smolts that were captured and recaptured for all streams in 1999. From these results the Lower South Fork Little River was the most productive coho stream of those trapped in 1999. Trapped fish were identified to species when possible. Due to the similarities between YOY steelhead and YOY cutthroat trout these were grouped into the trout category. All coho, chinook and trout (YOY cutthroat and steelhead) were YOY fish, while all steelhead and cutthroat trout were 1+ fish or older. Some of the 1+ steelhead were determined to be run-back steelhead returning to the ocean. The total numbers of all salmonids trapped in 1999 are summarized below (Table C8-7). The USFLR and LSFLR produced significant numbers of trout and chinook in addition to coho salmon. Table C8-8 summarizes the total number of salmonid smolts that were captured and recaptured for all streams in 2000. From these results Carson Creek was the most productive coho stream trapped.

In 2000 adult cutthroat trout were defined as fish >200mm and not showing signs of smoltification. The total numbers of all salmonids captured in 2000 are shown in Table C8-9. The Lower South Fork Little River and Carson Creek were the most productive coho streams, while both the Upper South Fork Little River and Lower South Fork Little River produced significant numbers of trout and chinook.

There were some mortalities associated with the trapping process. The summary of the trapping mortality in 1999 and 2000 are provided in Tables C8-10 and C8-11 respectively. This summary also includes the mortalities associated with the stomach pumping (gastric lavage) procedure Tables C8-12 and C8-13. Improvements in trap design and trapping procedures were made throughout the trapping season in an effort to reduce these mortalities.

Table C8-5. Species captures during out-migrant trapping in the Little River drainage in 1999 and 2000.

Common Name	Scientific Name	1999	2000
Coho Salmon	*Oncorhynchus kisutch*	X	X
Chinook Salmon	*Oncorhynchus tshawytscha*	X	X
Steelhead	*Oncorhynchus mykiss*	X	X
Coastal Cutthroat Trout	*Oncorhynchus clarki clarki*	X	X
Pacific Lamprey	*Lamperta tridentata*	X	X
Western Brook Lamprey	*Lamperta richardsoni*	X	X
Pacific Giant Salamander	*Dicamptodon ensatus*	X	X
Tailed Frog	*Ascaphus truei*	X	X
Prickly Sculpin	*Cottus asper*		X
Three-Spined Stickleback	*Gasterosteus aculeatus*		X

Table C8-6. Trapping totals for clipped and recaptured smolts in 1999.

Stream	Clipped Smolts			Recaptured Smolts		
	Coho	Steelhead	Cutthroat	Coho	Steelhead	Cutthroat
LSFLR	220	36	40	187	13	19
USFLR	15	30	18	9	20	10
Railroad Ck	15	18	35	12	10	25
Total	250	84	93	208	43	54

Table C8-7. Trapping totals for unclipped fish in 1999.

Stream	Unclipped Fish				
	Coho	Steelhead	Cutthroat	Trout	Chinook
LSFLR	3,543	454	57	10,435	5,812
USFLR	599	778	112	14,503	4,133
Railroad Creek	422	281	88	4,131	0
Total	4,564	1,513	257	29,069	9,945

Table C8-8. Trapping totals for captured and recaptured smolts in 2000.

Stream	Captured Smolts			Recaptured Smolts		
	Coho	Steelhead	Cutthroat	Coho	Steelhead	Cutthroat
LSFLR	1,104	34	15	522	19	13
USFLR	100	57	7	72	42	5
Carson Ck	1,408	7	43	612	4	31
Railroad Ck	63	12	21	56	10	16
Total	2,675	110	86	1,262	75	65

Table C8-9. Trapping totals for unclipped fish in 2000.

Stream	Unclipped Fish				
	Coho (YOY)	Steelhead	Cutthroat	Trout (YOY)	Chinook (YOY)
LSFLR	1,911	509[a]	50[e]	4,911	3,680
USFLR	140	960[b]	31[f]	5,451	5,277
Carson Ck	131	504[c]	296[g]	1,481	874
RRC	763	850[d]	44[h]	1,228	0
Total	2,945	2,823	421	13,071	9,831

Notes

[a] 16 of steelhead were adult
[b] 17 of steelhead were adult
[c] 2 of steelhead were adult
[d] 6 of steelhead were adult
[e] 7 of cutthroat were adult
[f] 4 of cutthroat were adult
[g] 23 of cutthroat were adult
[h] 6 of cutthroat were adult

Table C8-10. 1999 in-trap mortality.

Stream	In-Trap Mortality				
	Coho (YOY)	Steelhead (1+)	Cutthroat (1+)	Trout (YOY)	Chinook (YOY)
LSFLR	23	4	0	155	56
USFLR	3	1	1[a]	318	58
Railroad Creek	3	2	1[a]	157	0
Total	29	7	2	630	114

Note

[a] These fish were killed in the gastric lavage procedure

Table C8-11. 2000 in-trap mortality.

Stream	In-Trap Mortality					
	Coho Smolts	Coho (YOY)	Steelhead (1+)	Cutthroat (1+)	Trout (YOY)	Chinook (YOY)
LSFLR	4	7	5	0	77	23
USFLR	1	1	5	0	105	74
CC	8	7	4	1	46	19
RRC	1	3	2	0	24	0
Total	14	18	16	1	252	116

Table C8-12. 1999 predation mortality determined from gut contents from stomach pumping.

Stream	Predation Mortality				
	Coho (YOY)	Steelhead (1+)	Trout (YOY)	Chinook (YOY)	Salmonids (YOY)
LSFLR	112	6	934	361	105
USFLR	30	3	1,731	329	119
Railroad Creek	82	1	1,162	0	50
Total	224	10	3,827	690	274

Table C8-13. 2000 predation mortality determined from gut contents from stomach pumping.

Stream	Predation Mortality					
	Coho (Smolt)	Coho (YOY)	Steelhead (1+)	Trout (YOY)	Chinook (YOY)	Salmonids (YOY)
LSFLR	9	89	5	157	133	84
USFLR	1	4	2	578	265	153
Carson Ck	15	11	9	141	244	49
Rail Road Ck	2	39	7	212	0	44
Total	27	143	23	1,088	642	330

Contents from stomach pumping conducted during the 1999 and 2000 trapping program were identified to species if possible (Tables C8-12 and C8-13). Some of the items were digested to a point to which species could not be determined but fish were positively identified as juvenile salmonids. All preyed on coho, chinook, trout and salmonids were young of the year (YOY) fish. The preyed on steelhead were 1+ fish. Some of the other contents identified from stomach pumping from the 1999 trapping included: aquatic invertebrates, salmonid eggs, Pacific giant salamanders, tailed frog tadpoles and one mouse.

C8.3.5 Size and Condition

Salmonid growth increases at varying rates depending on the abundance of aquatic insects and plant life during critical rearing periods (Murphy and Meehan 1991). Size can also be influenced by density related competition. The fork lengths of the first 20 YOY coho (fork length) were measured to the nearest mm. The remaining individuals were counted but not measured. All smolts were measured. Table C8-14 shows the range of coho lengths measured in 1999 as well as their average length by age for each stream. All fish handled appeared to be in good condition and length of YOY fish increased steadily as the trapping season progressed. Table C8-15 shows the range of coho lengths and weights as well as their average length by age for each stream as measured in 2000. All fish handled appeared to be in good condition and lengths and weights of YOY fish increased steadily as the trapping season progressed.

Table C8-14. Average and range of lengths (mm) of coho salmon in USFLR, LSFLR, and Railroad Creel in 1999.

Length	USFLR		LSFLR		Railroad Ck	
	Smolts	YOY	Smolts	YOY	Smolts	YOY
Range (mm)	96-114	34-50	81-136	32-58	98-124	31-49
Avg. Length (mm)	102.9	37.5	104.6	39.3	110.6	37.9

Table C8-15. Average range of lengths and weights of coho salmon trapped in USFLR, LSFLR, Carson Creek, and Railroad Creek in 2000.

	USFLR		**LSFLR**		**Carson Creek**		**Railroad Ck**	
	Smolts	YOY	Smolts	YOY	Smolts	YOY	Smolts	YOY
Length Range (mm)	80-120	34-59	65-139	29-64	68-135	28-51	78-115	31-69
Avg. Length (mm)	103.0	46.9	94.3	42.5	97.9	40.5	96.0	45.5
Weight Range (gms)	5.8-22.4	0.4-2.1	3.3-27.7	0.1-2.9	3.4-24.0	0.4-1.6	5.3-16.3	0.2-2.3
Avg. Weight (gms)	12.7	1.1	9.9	0.7	10.3	0.7	9.8	0.9

C8.3.6 Migration Timing

In 1999 the migration of coho smolts began in April and continued into June (Figure C8-2). Factors that affect the timing of migration include the size of the fish, flow conditions, water temperature, dissolved oxygen levels, day length, and availability of food (Shapovalov and Taft 1954). The peak days of migration within USFLR and LSFLR were determined to be May 19, 1999 and April 24 within Railroad Creek. The peak period of migration lasted from the last week of April to the end of May. Migration tapered off after approximately May 3rd. This slow down coincided with a rain storm event. Figure C8-3 shows the flow of the Little River during the period of smolt migration.

In 2000, migration of coho smolts began in March and continued into June (Figure C8-4). The migration peak for Carson Creek and LSFLR occurred on April 4th 2000 and on April 14th 2000 respectively. The LSFLR had an additional peak on April 26th 2000. There were no significant peaks on Railroad Creek and USFLR in 2000. There were two periods approximately April 17th and May 11th when migration tapered off, coinciding with a storm event. Figure C8-5 shows the flow of the Little River during the period of coho smolt out-migration.

C8.4 DISCUSSION

Lower trapping efficiency is experienced during peak flow events. As shown on Figure C8-2 reduced numbers of fish are trapped during peak flow events (Figure C8-3). In 1999 a large number of smolts were trapped just prior to a peak event on May 3rd and large number were again trapped a few days after that peak event. Green Diamond believes that there are a large number of fish emigrating from the streams during these peak events.

During 1999 there was some mortality associated with the trapping. These losses were reduced by continually improving the trapping methodology and trapping equipment throughout the trapping season. It was determined that on trapping days where there was high volumes of debris loading into the traps there was a corresponding higher trap mortality. To reduce this mortality, an extra screen to catch and filter out debris was added to the traps. Traps were checked and debris was cleaned out in the evening as well as mornings on rainy or windy days. This effort helped to reduce mortality and was continued in 2000. During 1999 some mortalities were observed when fish were stranded onto dry portions of the McBane ramp. A plastic splash shield was installed that immediately solved this stranding problem.

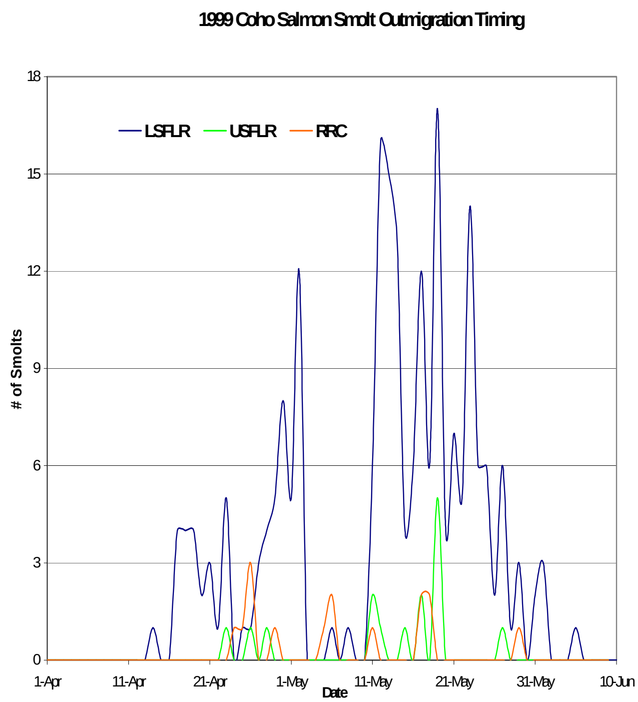

Figure C8-2. Migration timing for smolts for the 1999 trapping study in Little River.

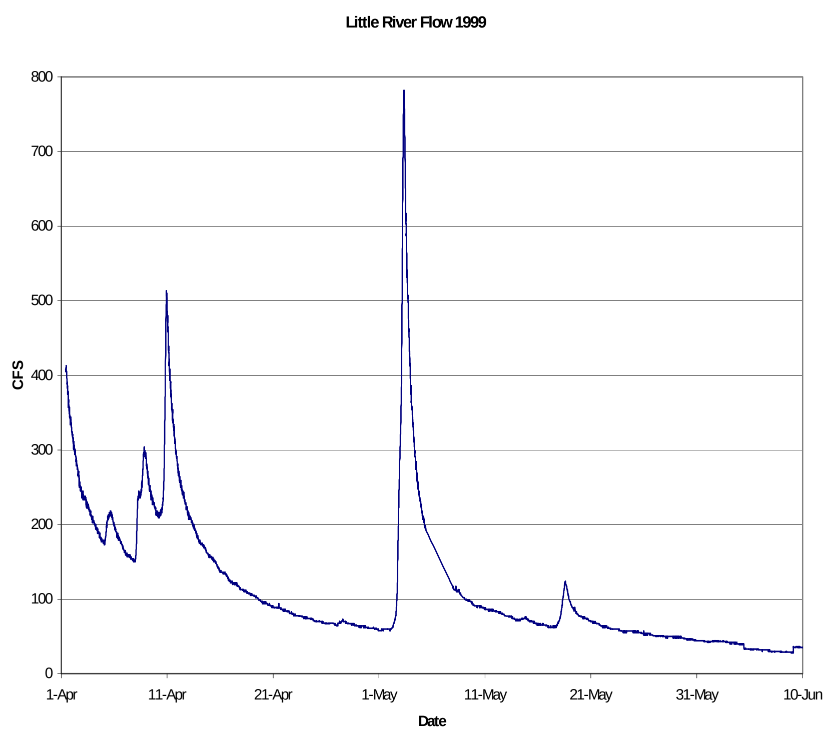

Figure C8-3. Little River flow (CFS) during 1999 peak smolt out-migration.

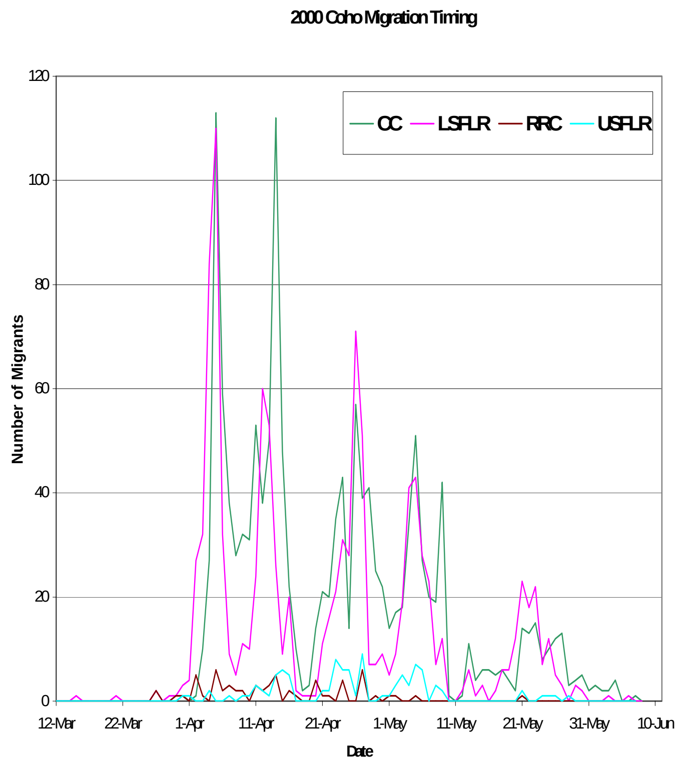

Figure C8-4. Migration timing for smolts for the 2000 trapping study in the Little River.

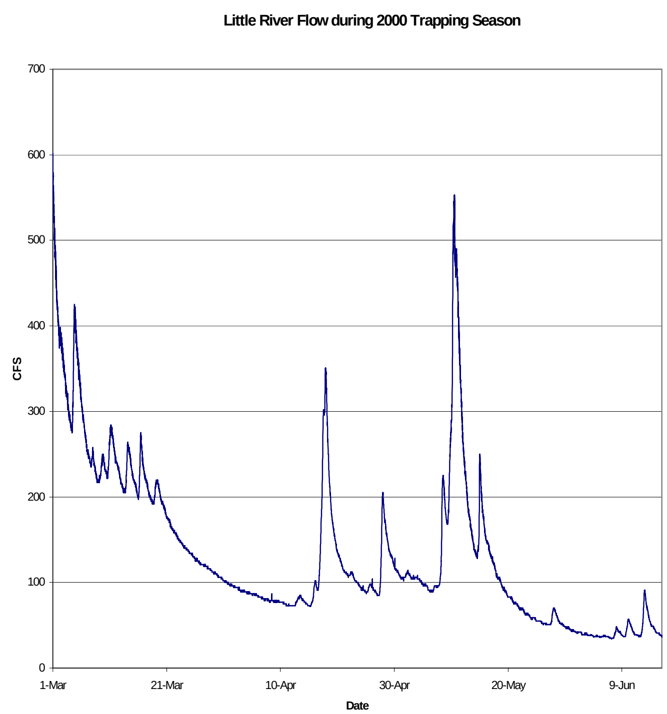

Figure C8-5. Little River flows (CFS) during peak out-migration in 2000.

Most of the trapping mortality was from loss of YOY fish. No smolts of any species were lost during trapping in 1999. There were only 2 mortalities from the stomach pumping procedure in 1999 (Table C8-12). Both of these mortalities were from improper insertion of the pumping tube while a new employee was learning the procedure.

During the 2000 trapping project lower sampling efficiency during peak flow events were again noted. Reduced numbers of fish (Figure C8-4) illustrates this during peak flow events (Figure C8-5). A good number of smolts were trapped just prior to peak events and a few days after the peak event. It is likely that a large number of fish leave the systems on these peak events because the creeks are confined channels with no flood plains. During these events the fish may be flushed out when the flows and velocities are high and the traps are relatively inefficient.

Over wintering survival rates were better in 2000 than in 1999. The increased survival rates may be higher due to the smolts leaving during the peak flow events. To determine an over-winter survival rate from the summer and winter population estimates it must be assumed that there is a closed population. This is not necessarily true. The first few days of trapping in 2000, in the LSFLR, several pre-smolt coho salmon were captured. From this observation, it appears that a portion of the coho began emigrating prior to the installation of the weir. Early pre-smolt migration violates the closed population assumption prior to pre-weir installation. In the future, trapping weirs will be installed earlier in the season to determine if a significant portion of the population begins emigration prior to the completion of smoltification. If a large number of coho are actively dispersing downstream during the winter rather than actively emigrating following smoltification during the spring, this would account for a relatively low over-winter survival rate. At the present time the survival rate of fish that disperse downstream as pre-smolts during the winter months is unknown.

In 2000, there were large numbers of mortalities associated with predation during trapping. In an effort to minimize predation during trapping, extensive refuge (cover) was provided for the YOY fish as they moved to the rear trapbox. The provision of this cover will exclude the predatory fish while provide refuge to the smaller YOY fish. Green Diamond is continuing its effort to reduce in trap predation by working with a graduate student from Humboldt State University (HSU) to develop an improved live trap box design. The student is conducting experiments to see if differently designed boxes that have different mesh separating devises help reduce predation mortality. Students from HSU are also looking at predation outside of the live boxes to determine how many of the prey items were eaten prior to being trapped. At the present time it is unknown whether the fish that are evaluated for predation are consuming their prey prior to entering the trap or while in the trap. In many cases the advanced stages of deterioration of the material within the stomach of the predatory fish indicates that it is likely that the preyed upon fish were consumed prior to being trapped.

There was also some continuing mortality associated with the trap design in 2000. Improvements were incorporated throughout the trapping season. On days where there were high amounts of organic debris loading in the traps, an increased mortality is expected. There was also some continuing mortality of fish stranding themselves onto a dry portion of the McBane ramp or into a debris deposit after coming out of the pipe. To reduce this mortality two new design elements were developed. An extra screen, to catch and filter out debris, and plastic sheeting on the McBane ramp, which prevented debris accumulation during lower flow conditions, were added to each trap. Also, the traps were checked in the evening on rainy or windy days to clean out the debris on the filter screen and inside the box trap. This combination of efforts significantly reduced mortality of YOY fish and thus will be continued and fine tuned in the future.

In 2000, from the stomach analysis, it was determined that the most common prey was YOY trout. This was followed, in order, by: YOY chinook salmon, YOY un-identified salmonids, YOY coho salmon, coho salmon smolts and 1+ steelhead. The coho smolts and 1+ steelhead were eaten by large predatory cutthroat trout. It was determined that during trapping, prey consumption followed the same order as fish abundance. The most abundant fish (YOY trout) were also most commonly recovered from the stomachs of the fish that were pumped. The only exception to this was the unidentifiable YOY salmonids. This finding suggests that there was no prey item preferred and actively selected over another.

C8.5 CONCLUSION

The use of out-migrant trapping is an excellent tool for collecting downstream migrants and is Green Diamond's best opportunity to collect information pertaining to coho production in the Little River drainage. The use of a box trap, McBane ramp, pipes and weir trapping system efficiently trap streams during low and normal flow. The out-migrant trapping program is in its preliminary stages and it is too early to determine population trends.

C8.6 REFERENCES

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Appendix C9. Spawning Surveys

CONTENTS

C9.1 METHODS

Green Diamond's staff does not attempt to generate any form of formal population or escapement estimates from the spawning surveys conducted. Due to the limitations of time, water conditions, and weather these surveys tend to be opportunistic rather than at fixed time intervals or fixed reaches. The purpose of these spawning surveys is to determine habitat use and relative numbers of spawners of all species as well as watershed conditions during the winter months. In general, the entire anadromous reach accessible to coho salmon is surveyed. In long anadromous reaches within one stream, the survey may be broken up into sub-reaches that tend to be based on accessibility and/or time available for the survey. Because of these constraints the surveys are somewhat inconsistent from year to year. Sub-reaches within one watershed may or may not be surveyed on the same day or by the same crew. Within each HPA a general description of the sub-reaches for each stream for which spawner surveys have been conduct are provided.

The following list indicates all streams by their Hydrographic Planning Area (HPA) for which spawning surveys have been conducted since 1995:

Stream	HPA
• Maple Creek	Coastal Lagoons
• North Fork Maple Creek	Coastal Lagoons
• Pitcher Creek	Coastal Lagoons
• Cañon Creek	Mad River
• Carson Creek	Little River
• Danielle Creek	Little River
• Little River	Little River
• Upper South Fork Little River	Little River
• Lower South Fork Little River	Little River
• North Fork Mad River	North Fork Mad River
• Railroad Creek	Little River
• Rowdy Creek	Smith River
• Salmon Creek	Humboldt Bay
• Savoy Creek	Smith River
• South Fork Rowdy Creek	Smith River
• South Fork Winchuck River	Smith River
• Sullivan Gulch	North Fork Mad River
• Wilson Creek	Smith River

C9.2 RESULTS

C9.2.1 Smith River HPA

Spawning surveys have been conducted on five streams within the Smith River HPA during the period of 1998 through 2000. The summaries of the results of these surveys follow.

C9.2.1.1 South Fork Winchuck River

The survey reach extends from the confluence of South Fork and mainstem Winchuck upstream approximately four miles to the end of the W1100 road.

C9.2.1.1.1 <u>1998-1999 Spawning Surveys</u>

Two spawning surveys were conducted on South Fork Winchuck River during 1998-1999: December 10, 1998 and January 8, 1999. The results of these surveys are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
46 Chinook	21 Chinook	7 Chinook
1 Steelhead	29 Unknown	
2 Unknown		

C9.2.1.2 Rowdy Creek

The two Rowdy Creek spawning survey reaches extend from the county bridge on Rowdy Creek Road upstream 13,000 feet to the R1400 bridge and then an additional 7,600 feet upstream to the confluence of Rowdy and Copper Creeks.

C9.2.1.2.1 <u>1998-1999 Spawning Surveys</u>

One spawning survey was conducted on December 15, 1998 on Rowdy Creek during 1998-1999. The results of this survey are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
11 Chinook	4 Chinook	None Observed
	3 Unknown	

C9.2.1.3 Savoy Creek

The spawning reach extends from the confluence with South Fork Rowdy upstream 3,100 feet to the anadromous barrier.

C9.2.1.3.1 <u>1999-2000 Spawning Surveys</u>

Two spawning surveys were conducted on December 3rd and 21st, 1999 on Savoy Creek during 1999-2000. The results of these surveys are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
55 Chinook	27 Chinook	18 Chinook
	13 Unknown	

C9.2.1.3.2 <u>1998-1999 Spawning Surveys</u>

One spawning surveys was conducted on December 16, 1999 on Savoy Creek during 1998-1999. The results of this survey are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
20 Chinook	13 Chinook	1Chinook
	3 Unknown	

C9.2.1.4 South Fork Rowdy Creek

The survey reach extends from the confluence with Rowdy Creek upstream 4,000 feet to the confluence with Savoy Creek. It continues upstream from Savoy Creek an additional 3,500 feet to the anadromous barrier.

C9.2.1.4.1 <u>1999-2000 Spawning Surveys:</u>

Two spawning surveys were conducted on December 7th and 21st, 1999 on South Fork Rowdy Creek during 1999-2000. The results of these surveys are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
53 Chinook	20 Chinook	15 Chinook
2 Unknown	18 Unknown	

C9.2.1.4.2 <u>1998-1999 Spawning Surveys</u>

One spawning surveys was conducted on December 16, 1999 on South Fork Rowdy Creek during 1998-1999. The results of this survey are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
20 Chinook	11 Chinook	4 Chinook
	5 Unknown	1 Unknown

C9.2.1.5 Wilson Creek

The survey reach extends from the Pacific Ocean upstream 5,000 feet to the 1st W10 bridge and then 23,000 feet up to the last W10 bridge.

C9.2.1.5.1 <u>1999-2000 Spawning Surveys:</u>

One spawning surveys was conducted on December 16, 1999 on Wilson Creek during 1999-2000. The results of this survey are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
None Observed	1 Unknown	None Observed

C9.2.2 Coastal Lagoons HPA

Spawning surveys have been conducted on three streams within the Coastal Lagoons HPA during the period 0f 1998 through 2000. The summaries of the results of these surveys follow.

C9.2.2.1 Maple Creek

The spawning survey reach extends from the confluence with North Fork Maple Creek to the gauging station for 4,500 feet. The reach continues for an additional 12,000 feet upstream of the gauging station.

C9.2.2.1.1 1999-2000 Spawning Surveys

One spawning survey was conducted on February 9, 2000 on Maple Creek, tributary to Big Lagoon during 1999-2000. The results of these surveys are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
None Observed	None Observed	None Observed

C9.2.2.1.2 1998-1999 Spawning Surveys

Two spawning surveys were conducted on December 16, 1999 and January 8, 2000 on Maple Creek during 1998-1999. The results of these surveys are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
None Observed	None Observed	None Observed

C9.2.2.2 North Fork Maple Creek

The survey reach extents from the confluence with Maple Creek to the F-4 bridge, approximately 4,500 feet. It continues upstream an additional 2,600 feet to the anadromous barrier.

C9.2.2.2.1 1999-2000 Spawning Surveys

One spawning survey was conducted on February 9, 2000 on North Fork Maple Creek during 1999-2000. The results of these surveys are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
None Observed	4 Unknown	None Observed

C9.2.2.2.2 1998-1999 Spawning Surveys

One spawning survey was conducted on December 16, 1999 and January 8, 2000 on North Fork Maple Creek during 1998-1999. The results of this survey are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
None Observed	None Observed	None Observed

C9.2.2.3 Pitcher Creek

Pitcher Creek is surveyed from the confluence with Maple Creek upstream to the anadromous barrier, just past the F-2 road bridge, for a total distance of 4,200 feet.

C9.2.2.3.1 1999-2000 Spawning Surveys

One spawning survey was conducted on April 10, 2000 on Pitcher Creek during 1999-2000. The results of these surveys are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
None Observed	12 Unknown	None Observed

C9.2.2.3.2 1998-1999 Spawning Surveys

One spawning survey was conducted on January 8, 1999 on Pitcher Creek during 1998-1999. The results of this survey are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
None Observed	None Observed	None Observed

C9.2.3 Little River HPA

Spawning surveys have been conducted on six streams within Little River HPA during the period of 1998 through 2000. The summaries of the results of these surveys follow.

C9.2.3.1 Carson Creek

Carson Creek is surveyed from its confluence with mainstem Little River to the bridge on the M-140 road, a total of 5,000 feet.

C9.2.3.1.1 1998-1999 Spawning Surveys

Two spawning surveys were conducted on December 17, 1998 and January 8, 1999 on Carson Creek, during 1998-1999. The results of these surveys are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
None Observed	6 Unknown	1 Chinook 2 Unknown

C9.2.3.2 Danielle Creek

The survey reach extends from the confluence with mainstem Little River upstream approximately 2,500 feet.

C9.2.3.2.1 <u>1998-1999 Spawning Surveys</u>

One spawning survey was conducted on December 9, 1998 on Danielle Creek during 1998-1999. The results of these surveys are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
None Observed	None Observed	None Observed

C9.2.3.3 Little River

Due to the length and depth of Little River, only two reaches totaling approximately 15,500 feet have been regularly surveyed. This reach extends from the confluence of Carson Creek to the mouth of Railroad Creek for a distance of 7,500 feet and from the mouth of Lower South Fork Little River to the mouth of Upper South Fork Little River for a distance of an additional 8,000 feet.

C9.2.3.3.1 <u>1999-2000 Spawning Surveys</u>

Seven spawning surveys were conducted on December 16th, 20th, 30th, 1999 and February 7th. March 3rd and 17th, and April 2, 2000 on Little River during 1999-2000. The results of these surveys are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
45 Chinook	15 Chinook	21 Chinook
21 Steelhead	8 Steelhead	1 Steelhead
	106 Unknown	1 Coho
		2 Unknown

C9.2.3.3.2 <u>1998-1999 Spawning Surveys</u>

One spawning survey was conducted during December 29 through 30, 1998 on Little River during 1998-1999. The results of this survey are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
66 Chinook	39 Chinook	17 Chinook
1 Coho	15 Unknown	1 Unknown
6 Unknown		

C9.2.3.4 Upper South Fork Little River

The spawning survey reach extends from the confluence with mainstem Little River upstream 5,000 feet to the V-Line bridge and then continues upstream an additional 2,300 feet to the anadromous barrier.

C9.2.3.4.1 <u>1999-2000 Spawning Surveys</u>

Two spawning surveys were conducted on December 13, 1999 and February 7, 2000 on Upper South Fork Little River during 1999-2000. The results of these surveys are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
None Observed	4 Unknown	4 Chinook

C9.2.3.4.2 <u>1998-1999 Spawning Surveys</u>

Two spawning surveys were conducted on December 9, 1998 and January 29, 1999 on Upper South Fork Little River during 1998-1999. The results of this survey are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
13 Chinook	2 Chinook	None Observed
4 Unknown	2 Unknown	

C9.2.3.5 Lower South Fork Little River

The spawning survey reach on Lower South Fork Little River extends from the confluence with mainstem Little River upstream 9,400 feet to the anadromous barrier.

C9.2.3.5.1 <u>1999-2000 Spawning Surveys</u>

Three spawning surveys were conducted on December 16, 1999, February 4th and March 24th, 2000 on Lower South Fork Little River during 1999-2000. The results of these surveys are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
1 Chinook	51 Unknown	6 Chinook
1 Steelhead		2 Coho

C9.2.3.5.2 <u>1998-1999 Spawning Surveys</u>

Two spawning surveys were conducted on December 17, 1998 and January 29, 1999 on Lower South Fork Little River during 1998-1999. The results of this survey are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
6 Chinook	3 Chinook	1 Unknown
18 Coho	12 Coho	
2 Steelhead	1 Steelhead	
4 Unknown	48 Unknown	

C9.2.3.6 Railroad Creek

The spawning reach extends from the confluence with mainstem Little River upstream to the anadromous barrier approximately for a total of approximately 5,000 feet in length.

C9.2.3.6.1 <u>1999-2000 Spawning Surveys</u>

One spawning survey was conducted on February 7, 2000 on Railroad Creek during 1999-2000. The result of this survey is shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
None Observed	9 Unknown	None Observed

C9.2.3.6.2 <u>1998-1999 Spawning Surveys</u>

One spawning survey was conducted on December 9, 1998 on Railroad Creek during 1998-1999. The result of this survey is shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
None Observed	None Observed	None Observed

C9.2.4 Mad River HPA

Spawning surveys have been conducted on one stream, Cañon Creek within the Mad River HPA during the period of 1998 through 2000. The summaries of the results of these surveys follow.

C9.2.4.1 Cañon Creek

The spawning survey reach for Cañon Creek extends from the confluence with the Mad River upstream 9,200 feet to the 4000 bridge. It then continues the 4000 bridge to the anadromous barrier, an additional 6,000 feet.

C9.2.4.1.1 <u>1999-2000 Spawning Surveys</u>

A total of nine surveys were conducted during the winter of 1999-2000. The dates of the surveys are November 11th, 19th, 22nd, and 30th, December 6th, 15th, and 27th, 1999; January 5th and February 8th, 2000. The results of these surveys are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
202 Chinook	73 Chinook	66 Chinook
1 Coho	3 Steelhead	1 Coho
12 Steelhead	65 Unknown	10 Steelhead
4 Unknown		2 Unknown

C9.2.4.1.2 <u>1998-1999 Spawning Survey</u>

Two surveys were conducted during the winter of 1998-1999. These were December 12th, 1998 and January 4th, 1999. The results of these surveys are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
66 Chinook	32 Chinook	6 Chinook
	30 Unknown	

C9.2.4.1.3 <u>1997-1998 Spawning Survey</u>

Two surveys were conducted during the winter of 1997-1998. These were conducted on December 6th and 29th, 1997. The results of these surveys are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
30 Chinook	20 Chinook	22 Chinook
3 Steelhead	2 Steelhead	1 Coho
2 Unknown	81 Unknown	

C9.2.4.1.4 <u>1996-1997 Spawning Survey</u>

One survey was conducted during the winter of 1996-1997. This survey was conducted during December 17th through 19th, 1996. The results of these surveys are shown below.

Live Fish Observed	Redds Observed	Carcasses Observed
110 Chinook	42 Chinook	7 Chinook
4 Coho	1 Coho	1 Coho
3 Unknown	4 Unknown	1 Unknown

C9.2.4.1.5 <u>1995-1996 Spawning Survey</u>

One survey was conducted during the winter of 1995-1996, on December 10th, 1995. The results of these surveys are shown below

Live Fish Observed	Redds Observed	Carcasses Observed
73 Chinook	27 Chinook	4 Chinook
3 Coho	1 Coho	
	3 Unknown	

C9.2.5 North Fork Mad River HPA

Spawning surveys have been conducted on two streams, North Fork Mad River and Sullivan Gulch within the North Fork Mad River HPA during the period of 1996 through 2000.

C9.2.5.1 North Fork Mad River

The spawning survey reach of NF Mad River extends form the confluence with Mad River upstream 11,500 feet to the county bridge at Korbel. The reach continues upstream from the county bridge at Korbel upstream 9,600 feet to the anadromous barrier, just downstream of the first bridge on the K&K road.

C9.2.5.1.1 *Spawning Survey 1999-2000*

One spawning survey was conducted on NF Mad River during the winter of 1999-2000. The survey date was December 29, 1999. The summaries of the results of this survey follow.

Live Fish Observed	Redds Observed	Carcasses Observed
76 Chinook	42 Chinook	21 Chinook
3 Steelhead	65 Unknown	7 Unknown
3 Unknown		

C9.2.5.1.2 *Spawning Survey 1998-1999*

Two spawning surveys were conducted on NF Mad River during the winter of 1998-1999. These survey dates were December 11th and 21st, 1998. The summaries of the results of these surveys follow.

Live Fish observed	Redds Observed	Carcasses Observed
42 Chinook	15 Chinook	28 Chinook
1 Steelhead	47 Unknown	5 Unknown
4 Unknown		

C9.2.5.1.3 *Spawning Survey 1997-1998*

Two spawning surveys were conducted on NF Mad River during the winter of 1997-1998. The survey dates were December 5th and 31st, 1997. The summaries of the results of these surveys follow.

Live Fish Observed	Redds Observed	Carcasses Observed
121 Chinook	65 Chinook	61 Chinook
3 Coho	18 Unknown	1 Unknown
4 Unknown		

C9.2.5.1.4 *Spawning Survey 1996-1997*

Two spawning surveys were conducted on the NF Mad River during the winter of 1996-1997. The survey dates were December 2, 1996 and January 16, 1997. The summaries of the results of these surveys follow.

Live Fish Observed	Redds Observed	Carcasses Observed
214 Chinook	213 Chinook	293 Chinook
5 Unknown	7 Unknown	2 Steelhead
		20 Unknown

C9.2.5.2 Sullivan Gulch

The spawning survey reach on Sullivan Gulch extends from the confluence with North Fork of the Mad River upstream to the anadromous barrier. This is a total distance of approximately 2,600 feet.

C9.2.5.2.1 <u>Spawning Survey 1999-2000</u>

Four spawning surveys were conducted on Sullivan during the winter of 1999-2000. The survey dates were December 10th and 15th, 1999, January 21st, and February 2nd, 2000. The summaries of the results of this survey follow.

Live Fish Observed	Redds Observed	Carcasses Observed
25 Chinook	9 Chinook	4 Chinook
	13 Unknown	2 Coho
		1 Unknown

C9.2.5.2.2 <u>Spawning Survey 1998-1999</u>

Two spawning surveys were conducted on Sullivan Gulch during the winter of 1998-1999. These survey dates were December 11th and 28th, 1998. The summaries of the results of these surveys follow.

Live Fish Observed	Redds Observed	Carcasses Observed
12 Chinook	7 Chinook	None Observed
1 Coho	14 Unknown	None Observed

C9.2.5.2.3 <u>Spawning Survey 1997-1998</u>

One spawning survey was conducted on Sullivan Gulch during the winter of 1997-1998. The survey date was December 21st, 1997. The summaries of the results of these surveys follow.

Live Fish Observed	Redds Observed	Carcasses Observed
1 Coho	1 Coho	None Observed
1 Unknown	10 Unknown	

C9.2.5.2.4 <u>Spawning Survey 1996-1997</u>

One spawning survey was conducted on Sullivan Gulch during the winter of 1996-1997. The survey date was January 9, 1997. The summaries of the results of these surveys follow.

Live Fish Observed	Redds Observed	Carcasses Observed
220 Chinook	108 Chinook	102 Chinook
5 Steelhead	2 Steelhead	18 Unknown
1 Coho		

C9.2.6 Humboldt Bay HPA

Spawning surveys have been conducted on one stream, Salmon Creek, within the Humboldt Bay HPA once during the period of 1995 through 2000.

C9.2.6.1 Salmon Creek

Spawning surveys were conducted from the County Bridge on Tompkins Hill Road upstream 8,000 feet to the second temperature recording station, just downstream of the road F-1400 bridge. Additional spot checks were made near the Walsh bridge approximately 14,000 feet upstream.

C9.2.6.1.1 <u>Spawning Survey 1998-1999</u>

One spawning survey was conducted on Salmon Creek during the winter of 1998-1999. The survey date was January 12, 1999. The summaries of the results of this survey follow.

Live Fish Observed	Redds Observed	Carcasses Observed
None Observed	7 Unknown	None Observed

C9.3 DISCUSSION

Chinook and coho relative abundance within the HPAs have fluctuated since monitoring began in 1995. The Smith River HPA, which includes South Fork Winchuck River, Rowdy Creek, Savoy Creek, South Fork Rowdy Creek and Wilson Creek, has been monitored for adult returns since 1998. Spawning surveys within these streams is sporadic, and often only conducted once in a season. Based on observed returns, no coho were seen during surveys in this HPA. Chinook were fairly common and easily distinguished during surveys. Based on late season results, it appears an adequate number of adult chinook annually escape in this HPA. Although spawning surveys have not detected adult coho, juvenile dive counts and electrofishing within these streams frequently find coho. Their numbers, however, are very low, which may factor into low observed escapement numbers. Steelhead are often seen during late winter surveys in small numbers, however juvenile population estimates within this HPA indicate that adult escapement may be much higher.

The Coastal Lagoon HPA which includes spawning survey reaches on North Fork Maple Creek, Maple Creek and Pitcher Creek are streams that are subject to irregular entry by returning salmonids. These systems are regulated by high flow events that allow for the breaching of the sand spit, which would otherwise block the entry of salmonids into their natal streams. Based on spawning survey results since 1998, it is unclear whether adequate adult escapement is received in these streams due to the timing of when the lagoon breaches. Numerous adult cutthroat trout were incidentally observed in the lower reaches of Maple Creek during a training session of the summer population estimate protocol in 1999. It is not known if the adult cutthroat were either anadromous or “lagoon run”. “Lagoon run” fish may utilize the lagoon in the same way anadromous fish utilize the ocean. Age 0+ and 1+ chinook as well as two 18-inch chinook (also possibly “lagoon run” chinook) were observed during the training session. Age 1+ coho were seen in Pitcher Creek during summer 1999, however no 0+ coho were observed in the system. This indicates that the timing of when the lagoon breaches plays an important role in determining if, when or what species enter the Maple Creek system. The absence of 0+ coho during the summer of 1999 indicates that Big Lagoon did not breach during the 1998/1999 coho run, but the presence of 1+ coho indicates that adults were able to enter during the 1997/1998 spawning season. During the formal spawning surveys only redds of unknown species have been found late in the survey season. It is likely these redds where created by anadromous or “lagoon run” cutthroat or by steelhead that were able to enter the lagoon during high winter flow. All four covered salmonid species have been observed in the Coastal Lagoon HPA, however cutthroat is the only species that have been seen in the adult form.

The Little River HPA is currently the most actively surveyed HPA for adult escapement. Surveys are conducted on six streams, which include Carson Creek, Danielle Creek, main stem Little River, Upper South Fork Little River, Lower South Fork Little River and Railroad Creek. Surveys on these streams have only been conducted since 1998, since the acquisition of the Louisiana Pacific land holdings. The main stem Little River has the highest totals of both redds, live fish and carcasses. The second largest counts have been observed on Lower South Fork Little River. The majority of spawning activity appears to be by chinook, however coho and steelhead are occasionally observed during surveys. Although these surveys would indicate very little spawning activity by these species, they are extremely abundant during summer juvenile dive counts and out-migrant trapping, indicating a fair number of adults are not observed during escapement surveys. This is often a result of survey limitations due to high flows, which often reduce visibility and flush carcasses. Survey frequency and timing are important, but even with the increased surveys adult salmonids will be missed, making it very difficult to rely on adult counts as an intricate component of the monitoring program.

Cañon Creek is currently the only stream surveyed in the Mad River HPA. Survey frequency, spacing and duration have helped to make it the most well monitored creek for adult escapement. Chinook are the most common species observed, followed by steelhead and coho salmon, respectively.

The North Fork Mad River HPA consists of two survey streams, Sullivan Gulch and North Fork Mad River. Chinook are the most frequently recorded species in North Fork Mad River, followed by steelhead and coho, respectively. Chinook salmon escapement appears robust, with only one to two surveys each season recording large adult returns. Steelhead are fairly common in early winter surveys, but the majority of survey dates in late December are probably too early to record significant numbers. Coho are infrequently observed; however, this is likely a factor of water visibility and survey timing. Sullivan Gulch, has been surveyed since 1996. Limited numbers of chinook, coho and steelhead have been observed. Chinook are the most frequently recorded salmonid, but steelhead may also make up a significant component of the survey if conducted later in the year. Based on juvenile population estimates, however, coho also make up a significant portion of the adult run, although they are rarely observed during spawning surveys.

Spawning surveys in the Humboldt Bay HPA are only conducted in Salmon Creek. Surveys were first conducted in 1998, with only seven redds being identified. Limited winter access into the watershed and visibility generally prevents effective survey coverage of the stream. Also, near the mouth of Salmon Creek, a tide gate may limit upstream migration into the watershed.

C9.4 CONCLUSIONS

Salmonid escapement surveys have helped to show that returning adult populations are using the majority of anadromous habitat available in monitored HPA streams. Opportunistic surveys looking at chinook and coho escapement may be helpful in examining age structure, sex ratios, migration timing and hatchery infiltration, however the number of HPA streams, high flows and water visibility limit the utility of the surveys to draw definitive conclusions for adult escapement estimates. Similar information would be helpful for cutthroat and steelhead adults within Plan Area streams, but only limited data can be collected on these species due to variations in their life history patterns, high flows, water conditions and the basic behaviors of the adult fish. Other survey methods such as summer juvenile fish population monitoring and out-migrant trapping are more reliable and consistent approaches to monitor population trends. The spawning surveys may help develop an understanding marine survival, however a much more intensive survey methodology would need to be employed such as adult traps installed across the ownership which would also be best combined with the monitoring of other freshwater life history stages.

Appendix C10. Mad River Steelhead Surveys

CONTENTS

C10.1 METHODS

Comprehensive dive counts of adult summer steelhead in the Mad River were conducted since 1994. These surveys were made in response to sharp declines in summer steelhead counts within index reaches surveyed annually by U.S. Forest Service personnel upstream of Green Diamond's Mad River property. Counts of both adult (over 16") and half-pounder (12"-16") sized summer steelhead were made. If possible, the presence or absence of an adipose fin was noted on all adult summer steelhead because all summer steelhead produced by Mad River Hatchery have an adipose fin clip.

The snorkel surveys were organized by California Trout and were a cooperative effort involving personnel from California Trout, Green Diamond, CDFG, USFWS, U.S.D.A. Forest Service, Coastal Stream Restoration, Trinity River Associates, Douglas Parkinson and Associates, Natural Resources Management Corporation, and Redwood Community Action Agency.

The portion of the survey identified as the Green Diamond reach extends from Deer Creek to the Department of Fish and Game's Mad River Hatchery. This segment consists of eight reaches for a total of approximately 36 miles of the Mad River:

- Reach 1: Deer Creek to Humbug Creek, 4.0 miles
- Reach 2: Humbug Creek to Big Bend, 4.6 miles
- Reach 3: Big Bend to Goodman Prairie, 4.3 miles
- Reach 4: Goodman Prairie to Church Camp, 3.7 miles
- Reach 5: Church Camp to Butler Valley Ranch, 5.8 miles
- Reach 6: Butler Valley Ranch to 4510, 3.7 miles
- Reach 7: 4510 road crossing to 4090 road crossing, 5.0 miles
- Reach 8: 4090 road crossing to Mad River Fish Hatchery, 4.7 miles

Since 1982 the U.S. Forest Service has surveyed 2 Index reaches upstream of the Green Diamond property from Ruth Dam downstream to Deer Creek. Since 1994 CDFG has surveyed the following reaches of the Mad River upstream of the Green Diamond property:

- Reach 1:
  (1994-1998): Deer Creek to Humbug Creek, 4.0 miles
- Reach 2:
  (1994, 1997, 1998): Humbug to Big Bend, 4.6 miles
  (1995) Humbug to Wilson, 2.8 miles
  (1995): Humbug to Swing Bridge, 6.2 miles

California Department of Fish and Game annually surveys the Mad River in the reach downstream of Green Diamond property from the Mad River Hatchery to Kadle Hole near the Highway 299 bridge.

C10.2 RESULTS

The 1994 snorkel surveys were conducted on August 26th and September 27-28, 1994 and covered a total of 59.8 miles of channel between Nelson Flat and the Mad River Hatchery. A total of 306 adult steelhead (265 with adipose fins, 3 with adipose clips, and 38 unknowns) and 172 half-pounder (67 with adipose fins, 0 with adipose clips, and 105 unknowns) were observed (Table C10-1). Nearly half the adult summer steelhead (141) were congregated in two pools. These pools were located below large falls (10-15 feet) over boulders that were probably low flow barriers and most of the steelhead observed below these falls were scarred and bruised. These barriers probably influenced the low fish counts in the Forest Service index reach (only 19 adult summer steelhead in 24 miles of channel) and illustrated the need for more complete surveys.

Table C10-1. Total number of summer steelhead observed in snorkeling dives on the Mad River, 1994-2000.

Year	Forest Service Reaches (Ruth Dam to Deer Creek)	Green Diamond Reaches (Deer Creek to Hatchery)		CDFG Reaches (Hatchery to Kadle Hole)		All Reaches
	(in consistent or in-complete)	Adults (clips)	½ pounders (clips)	Adults (clips)	½pounders (clips)	Total (clips)
1994	18	287 (3)	172 (0)	0 (0)	0 (0)	477 (3)
1995	41	501 (6)	10 (0)	27 (0)	11 (0)	552 (6)
1996	5	422 (41)	26 (0)	88 (0)	0 (0)	541 (41)
1997	5	225 (2)	12 (0)	54 (2)	0 (0)	296 (4)
1998	13	176 (0)	12 (0)	12 (0)	8 (0)	221 (0)
1999	No Survey	78 (0)	15 (0)	7 (0)	10 (0)	110 (0)
2000	No Survey	80 (0)	54 (0)	45 (15)	7 (0)	186 (15)

The 1995 snorkel surveys were conducted between August 24th and 26th and covered a total of 72.8 miles of channel, from Matthews Dam downstream to the Highway 299 bridge. A total of 569 adult steelhead (400 with adipose fins, 6 with adipose clips, and 163 unknowns) and 21 half-pounders (4 with adipose fins, 0 with adipose clips, and 17 unknowns) were observed (Table C10-1). Most of the adult summer steelhead (479 fish) was congregated in the upper two reaches, with large numbers of fish in several pools immediately below the upper falls. Only 40 adult summer steelhead were observed in the nearly 30 miles of channel surveyed above the upper falls.

The 1996 snorkel surveys were primarily conducted on August 26th-27th (with reach #4 completed on September 3rd) and covered the entire river from Matthews Dam downstream to the Highway 299 bridge. A total of 515 adult steelhead (408 with adipose fins, 41 with adipose clips, and 66 unknowns) and 26 half-pounders (12 with adipose fins and 14 unknowns) were observed (Table C10-1). Most of the adult summer steelhead (305 fish) was congregated in the two reaches downstream of the falls, with large numbers of fish in several pools immediately below the falls. Only five adult summer steelhead were observed in the nearly 30 miles of channel surveyed above the falls.

The 1997 snorkel surveys determined that a total of only 288 adult steelhead (284 with adipose fins, 4 with adipose clips) and 12 half-pounders (none with adipose fin clips) were observed (Table C10-1). The 1998 snorkel surveys resulted in steelhead counts of 201 adults (87 with unclipped adipose fins and at least 89 unknown) and 20 half-pounders (all with unclipped adipose fins).

In 1999, the US Forest Service reaches were not surveyed so the total number of steelhead observed were from Deer Creek to Kadle Hole and included the Green Diamond and CDFG reaches. The Green Diamond reaches were snorkeled on August 25th (reaches 1-5) and 26th (reaches 6-8). In 1999 only a total 85 adult steelhead were observed within the surveyed area. Of these 85, only seven adults were confirmed have been adipose fin clipped. In addition 25 half-pounders were observed within these reaches (Table C10-1), none of which were confirmed to have been ad fin clipped.

In 2000, the US Forest Service reaches were not surveyed so the total number of steelhead observed were from Deer Creek to Kadle Hole and included the Green Diamond and CDFG reaches. The Green Diamond reaches were snorkeled on August 31st (reaches 1-5, 8) and September 1st (reaches 6-7). The CDFG reaches were surveyed on August 25th. In 2000 only a total 80 adult steelhead were observed within the surveyed area. Sixteen of these adults were unknown as to whether they were adipose fin clipped or not and 15 were observed with adipose clips. An additional 54 half-pounders were observed within these reaches (Table C10-1).

C10.3 DISCUSSION

The Mad River summer steelhead dives revealed the importance of conducting complete surveys, as opposed to making basin-wide estimates from index reaches. Prior to 1994 information about Mad River summer steelhead was derived solely from the numbers observed within the Forest Service index reach (above the falls). Until recently some biologists considered the Mad River wild summer steelhead population in danger of extinction. However the 1994-2000 results indicate that the Mad River sustains one of the larger known populations in California, especially considering that dive surveys actually provide a minimal count of only the fish actually observed by divers.

Figure C10-1 summarizes the total numbers of summer steelhead observed in the Mad River for the years 1994-2000. From this information it appears that there is a trend that, since the 1996 dive survey, there has been a decline in the total number of summer steelhead in the Mad River (Figure C10-1). This maybe a result of many factors including differing water-year types, habitat conditions, spawning and rearing success as well as ocean and climatic conditions in the years prior to these surveys.

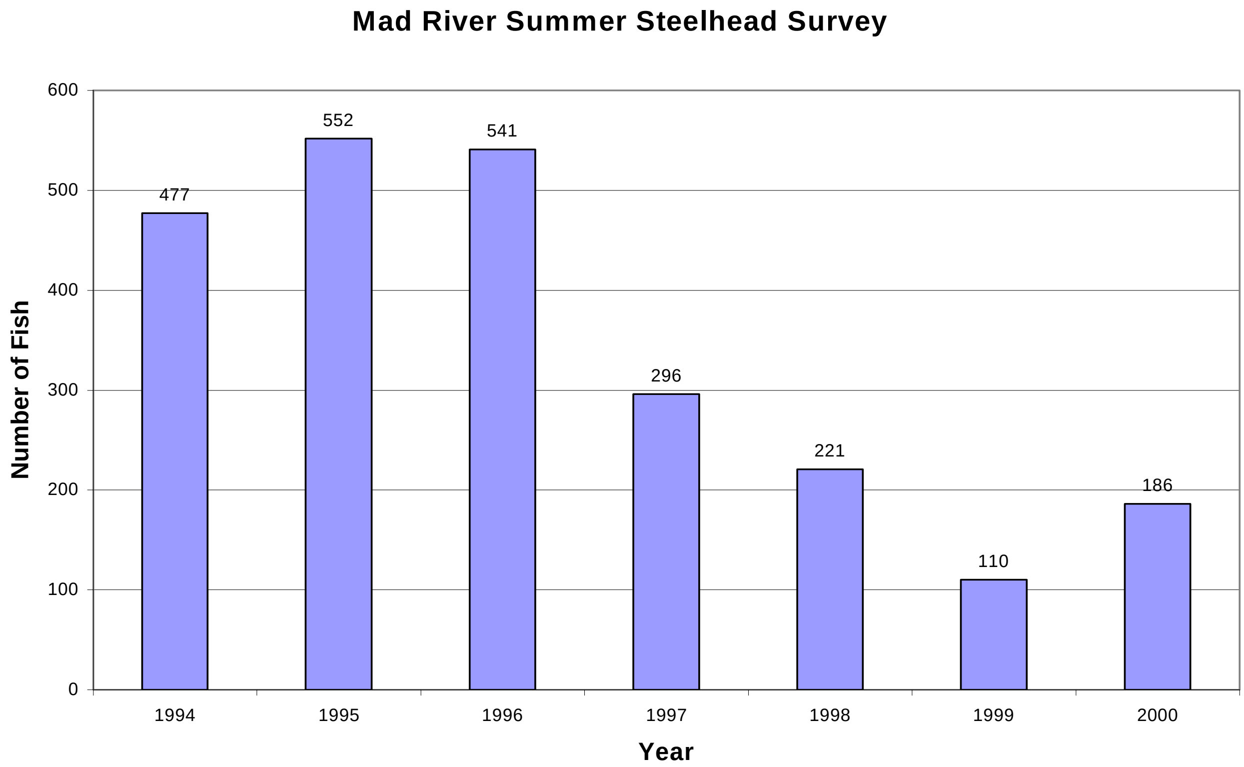

Figure C10- 1. Summary of the total number of Mad River summer steelhead observed (1994-2000).

C10.4 CONCLUSIONS

By conducting the 100 percent surveys annually, the best data for tracking long-term population trends of Mad River summer steelhead will be obtained. The survey results have already resulted in changes in steelhead management by CDFG to better protect this population of wild summer steelhead. Fishing regulations were recently modified to reduce potential impacts from sport fishing by extending the catch-and-release section and prohibiting all fishing within the channel reach where most of the adults are observed. CDFG also terminated its summer steelhead program at the Mad River Hatchery to eliminate the potential for genetic and/or disease exchange from the non-native hatchery fish to the native population.

Appendix C11. Headwater Amphibian Studies and Monitoring

CONTENTS

Tables

C11.1 STUDIES PUBLISHED IN "JOURNAL OF HERPETOLOGY"

- **Distribution and Habitat of *Rhyacotriton variegatus* in Managed, Young Growth Forests in North Coastal California**

- **Distribution and Habitat of *Ascaphus truei* in Streams in Managed, Young Growth Forests in North Coastal California (manuscript as it appeared in the *Journal of Herpetology*)**

Journal of Herpetology, Vol. 30, No. 2, pp. 184-191, 1996

Distribution and Habitat of *Rhyacotriton variegatus* in Managed, Young Growth Forests in North Coastal California

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ABSTRACT.—We examined the distribution and habitat of *Rhyacotriton variegatus* in streams of managed forests in north coastal California. We found 1475 salamanders from 220 streams from 1990-1994 through surveys of randomly selected first and second order streams and incidental searches. Of 71 headwater streams randomly selected to relate landscape variables to the presence/absence of *R. variegatus*, 57 (80.3%) contained salamanders. Geological formation was the only landscape variable that predicted the presence of *R. variegatus* in a stepwise logistic regression model. A second survey was conducted to determine which habitat variables of stream reaches were related to the presence/absence of *R. variegatus*. Thirty-one of 64 stream reaches contained salamanders and stream slope (gradient) was the only variable of 20 measured that entered a stepwise logistic regression model to predict the presence of *R. variegatus*. Pairwise comparisons indicated that reaches with salamanders had significantly higher slope, more small boulders, and less sand. No other variables, including canopy closure and water temperatures, were significant. An additional survey to further define the microhabitat for *R. variegatus* showed that abundance was positively related to stream slope and that this species was found more often than expected in high gradient riffles. The preferred substrate was gravel with smaller amounts of silt/clay, sand, and cobble. We discuss the past and future impacts of timber harvest on this species in north coastal California.

Results of studies in Douglas-fir (*Pseudotsuga menziesii*) dominated forests in the Pacific Northwest suggest that some amphibians are associated with old growth forests (Carey, 1989; Welsh, 1990; Welsh and Lind, 1991) and are sensitive to timber harvest (Bury and Corn, 1988a; Welsh and Lind, 1988; Corn and Bury, 1989; Bury et al., 1991). Torrent salamanders (*Rhyacotriton* spp.) are among stream amphibians that have been reported to be most at risk in the Douglas-fir zone. It has been suggested that local extinction can occur after clearcutting (Bury and Corn, 1988b; Corn and Bury, 1989) and that recolonization may take decades because torrent salamanders have limited dispersal abilities, small home ranges (Nussbaum and Tait, 1977), and are closely tied to cool headwaters and seeps (Nussbaum et al., 1983; Stebbins, 1985).

The southern torrent salamander (*Rhyacotriton variegatus*) is the most southerly distributed of the four species of the genus (Good and Wake, 1992). These salamanders have an aquatic larval stage, lasting perhaps 2-4 yr (Nussbaum and Tait, 1977). Transformed individuals live in the same microhabitats as the larvae. Subadults and adults are occasionally found under objects adjacent to streams and splash zones, but seldom more that 1 m from flowing water (Nussbaum and Tait, 1977). *Rhyacotriton* spp. are sensitive to timber harvest presumably because they require cool microhabitats with clean gravel and cobble (Nussbaum et al., 1983; Corn and Bury, 1989). Timber harvesting may increase deposition of fine sediments and remove canopy cover resulting in elevated temperatures.

Only one study has focused on the relationships between amphibians and logging in the redwood (*Sequioa sempervirens*) zone of north coastal California. Bury (1983) compared one clearcut and one old growth site on four study areas in western Humboldt and Del Norte Counties. He found a slight reduction in the number of species, number of individuals, and the biomass of salamanders in logged compared to old growth sites. The southern torrent salamander (*Rhyacotriton variegatus*; = *R. olympicus* of Bury) was found only in old growth sites, suggesting they are sensitive to timber harvest in this region. However, only one rivulet per site was searched and a total of two specimens of the species was captured.

We conducted a more detailed study at three hierarchical levels of survey to determine the distribution and relative abundance of *R. variegatus* in relation to major landscape variables, to correlate the presence/absence of this species with stream reach habitat variables, and to determine selected microhabitat components associated with sites utilized by *R. variegatus*.

MATERIALS AND METHODS

Study Area.—Our study was conducted on about 1500 km² of private timber lands located west of the crest of the Coast Range in western

Del Norte and Humboldt Counties, northwestern California. Most of this property lies within 32 km of the coast, but extends up to 85 km inland in places. The study area is located mostly within the north coast redwood zone (Mayer, 1988) where fog is common. Near the coast, mean summer and winter temperatures are about 18 C and 5 C, respectively, whereas extremes of 38 C in summer and −1 C in winter are not uncommon 48 km from the coast. Precipitation ranges from 102 to 254 cm annually, with 90% falling from October through April (Elford, 1974).

Coast redwood and Douglas-fir are the co-dominant conifers over most of the study area, but Douglas-fir is more prevalent at higher, drier locations. Hardwoods, such as tanoak (*Lithocarpus densiflorus*), red alder (*Alnus rubra*), Pacific madrone (*Arbutus menziesii*), and California Bay (*Umbelluaria californica*) also are major stand components. Common species along the watercourses surveyed include red alder, big leaf maple (*Acer macrophyllum*), and willows (*Salix* spp.).

Timber harvesting in the north coast area began in the late 1800s when entire drainages were clearcut in a continuum of operations that migrated inland from the coast. In the 1940s, virgin stands in our study area were selectively cut 1–4 times to remove the best redwood and Douglas-fir. Since the late 1960s, even-aged management has been used that involves relatively small clearcuts (average about 24 ha) followed by prompt artificial regeneration. About 97% of the study area consists of 0–80 yr old second and third growth forests, with the following stand age distribution: seedling/shrub (0–9 yr), 13%; sapling/poletimber (10–20 yr), 16%; small sawtimber (21–60 yr), 60%; and large sawtimber (61 + yr), 11%. Prior to 1973, no prescribed protection was given to streams in areas being harvested. Since 1973, California law has required leaving variable-width buffers along streams supporting fish or other aquatic life.

Landscape Surveys.—In 1992 we began systematic surveys of amphibians on the study area by using a stratified random sampling design to select up to four sections per township from U.S. Geological Survey maps. The number of sample sections per township was reduced if the study area was not located in the entire township. Sampling was designed to insure selection of one section per ¼ township (9 sections). Each section chosen had to include at least one half study area and have road access.

We selected 71 sections for a presence/absence survey of *Rhyacotriton variegatus*. We sampled the first headwater stream encountered along the major road through the section. Based on aerial photographs and direct observations of stream flow, the starting point for each survey was selected to ensure a minimum of 200 m of searchable length of stream but no more than about 500 m to the beginning of the wetted channel. If no *R. variegatus* was found, the entire stream reach to the beginning of the wetted channel was walked and all suitable habitat searched, with the greatest effort expended in the best habitat. If *R. variegatus* was found, we recorded distance from the starting point (m, with a hip chain), and the search was continued for a measured distance of 20–30 m to get an estimate of relative abundance. Life history category (larva or transformed) and sex of adults (inspection of cloacal lips, enlarged and squared in males) were recorded for a portion of the animals collected. Forest age of the stream drainage, cover type, stream aspect, elevation, and stream protection history were taken from a G.I.S. data base, aerial photographs (1:12,000 scale), and U.S. Geological Survey topographic maps (1:48,000 scale). Cover types were grouped into redwood, Douglas-fir, redwood/Douglas-fir mix, and hardwoods. Stream protection history was determined by the year of logging and grouped into early (pre-1974 California Forest Protection Act), intermediate (1975 through 1989), or current (1990 to present). The geological formation in which the watershed occurred was taken from U.S. Geological Survey topographic maps overlain by State of California, Department of Forestry geology maps and photographic interpretations (O. Huber, pers. comm.). Thirteen geological formations were identified but grouped into two categories, consolidated and unconsolidated, based on formation age and particle type formed following decomposition. The geologically younger category included unconsolidated marine deposits that decompose into silt and sand whereas the other group was composed of older consolidated formations that form boulders, cobbles, and gravel.

Stream Reach Surveys.—To determine which stream reach habitat variables predict presence of *Rhyacotriton variegatus*, we used the same sampling protocol to select an additional 37 first-order and 27 second-order streams with flows less than 10,500 cm^3/s within the contiguous portion of the study area.

Fixed stream reaches were located 10 m above the roadway or culvert. We established cross-stream transects (Platts et al., 1983) at 5 m intervals starting 2.5 m above the lower end of the transect. We established 10 transects in each sample stream unless physical features (waterfalls, log jams, stream going underground) prevented this. In one stream, only 25 m (5 transects) was sampled, but in all other streams a minimum of 30 m (6 transects) was sampled. We searched streams with a viewing box, where possible, and turned the substrate in search of animals. For each salamander captured, the distance from the lower end of the stream reach and the substrate type where the animal was found (boulder, gravel, cobble, sand) were recorded.

Habitat variables measured (in cm) at each transect included the amount of living vegetative overhang (total linear length up to chest height), small organic debris (SOD, total linear length of leaves, twigs, and sticks <10.2 cm in diameter on the substrate), and logs (total linear length of dead woody material >10.2 cm in diameter occurring over or in the stream but not on the stream bottom). Substrate was classified by measuring along each transect the amount of mud/silt, sand, gravel, cobble, small boulder, large boulder, and large organic debris in cm (Platts et al., 1983). The slope of the stream at each transect was measured by placing the center of a 1 m rod on the transect line parallel to the flow of water and at the stream surface and recording the slope, in degrees, from a clinometer. Canopy closure was estimated by taking readings with a densiometer in each cardinal direction in the center of the stream at the first, fifth, and tenth transects and converting to percent canopy cover.

To reduce the effects of seasonal variation, we estimated stream flow and measured temperature, pH, and conductivity during August and September. Stream flow (cm^3/sec) was estimated by measuring stream depth (cm) at ¼, ½, and ¾ intervals across the stream (Platts et al., 1983), stream width (cm) at this point, and timing the surface speed of a small floating object for three trials. Temperature was taken to the nearest 0.1 C with a Schultheis quick recording thermometer. Conductivity (ms) and pH were taken by an Oakton water test kit. Cover type and aspect were determined from maps.

Microhabitat Surveys.—We conducted a final study to further investigate the relationship between stream slope and presence of *Rhyacotriton variegatus* and to better quantify the microhabitat of this species. For this study, 14 streams (not sampled above) known to have salamanders from incidental sightings were randomly selected for sampling. The headwater portions of these streams were partitioned into low (0–5°), medium (6–10°), and high gradient (>15°) reaches. If available, two 10 meter reaches of each slope were sampled. Due to obstacles in the stream, some reaches had to be shortened to a minimum of 5 m and not all slope categories were available in all streams. The length (m, hip chain) and slope (1 m rod and clinometer, in degrees) of each reach were recorded as noted above. Aspect (compass) and stream temperature (Schultheis quick recording thermometer, 0.1 C) were recorded in the field. A water sample was taken to the laboratory and pH determined with a Beckman 40 pH meter and recorded to the nearest 0.01. Canopy closure was estimated for each reach with a densiometer read at the four cardinal directions and converted to percent canopy cover. For each reach sampled, at least five habitat point samples were taken. The point samples were collected where *R. variegatus* was located or, if no animals were found, at the best available habitat at 2 m intervals, starting at a randomly selected point. Each sample point was assigned to one of four habitat types; cascade, high gradient riffle, low gradient riffle or pool (modified from Platts et al., 1983), because they were the only habitat types readily distinguished in a headwater stream. The dimensions of the habitat type were measured and area (cm^2) recorded. Surface substrate composition was estimated by placing a wire 15 × 15 cm grid with 5 cm mesh on the stream bottom centered on the sample point. At each mesh intersection (12), the substrate type (boulder/bedrock, cobble, gravel, sand, or silt/clay) covered by the intersection was recorded (Cazier, 1993). Vegetative overhang was recorded as the amount overhanging the mesh screen, in percent. The life history stage (larva, transformed) and sex if an adult (inspection of cloacal lips) were recorded for all *R. variegatus* captured.

Data Analysis.—In our analysis of the relationships of landscape variables to the presence and relative abundance of *Rhyacotriton variegatus*, elevation (m) and forest age (0–80 yr) were considered independent continuous variables. Aspect was measured as a continuous variable (0–360°), but grouped into eight 45° octants and treated as a categorical variable. All other variables were treated as categorical variables. We used a stepwise logistic regression analysis, SLR (BMDP Version 7.0; Dixon, 1992) with 10 iterations and P values of 0.10 and 0.15 to enter and exit the model, respectively, to determine which of the landscape variables best predicted the presence of *R. variegatus*. We then used Chi-square tests (NCSS, Version 6.0; Hintze, 1995) on the variables 'aspect' and 'geology' to see which category was related to presence of *R. variegatus*. We divided the study area into a northern and southern region for further analysis because of a decreasing gradient of rainfall from north to south (Diller and Wallace, 1994). The Mann-Whitney U test (NCSS, Version 6.0) was used to compare the forest age of stands in the drainage to streams with and without *R. variegatus*.

For the stream reach data, we considered stream order and cover type as categorical variables; all others were treated as continuous variables. We used SLR (BMDP Version 7.0; Dixon, 1992) with 20 iterations and *P* values of 0.10 and 0.15 to enter and exit the model, respectively, to determine which of the reach habitat variables best predicted the presence of *R. variegatus* in the sample reaches. A Chi-square test (NCSS, Version 6.0) was then used on 'aspect' (divided into eight equal octants) to see if there was a relationship to presence of *R. variegatus.* Pooled t-tests were used to compare the continuous variables (slope, canopy closure, and the average amounts of each substrate type) at sites with or without *R. variegatus.*

In our study of microhabitat use, habitat type (high and low gradient riffle, cascade, and pool) was considered a categorical variable; all others were considered continuous variables. We used a logistic regression analysis (NCSS, Version 6.0) with 20 iterations and *P* values of 0.10 and 0.15 to enter and exit the model, respectively, to determine which microhabitat variables were related to the presence of *Rhyacotriton variegatus.* A Chi-square test was then used to determine the relationships between the three categories of habitat type and the presence of *R. variegatus.* (Cascade and high gradient riffle were combined into high gradient habitat, because only five observations were made in cascades.) ANOVA (NCSS, Version 6.0) was used to determine the relationship between relative abundance of *R. variegatus* and the three slope categories. We used a Mann-Whitney U test (NCSS, Version 6.0) to determine if differences existed between average percent surface substrate composition of microsites with and without *R. variegatus.* A significance level of 0.05 was set for all analyses.

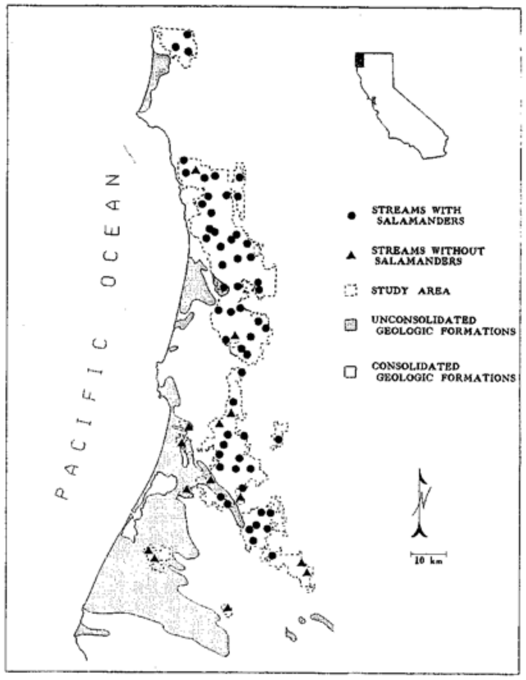

FIG. 1. Map of study area showing consolidated and unconsolidated formations, and streams with and without the southern torrent salamander (*Rhyacotriton variegatus*), north coastal California.

RESULTS

From 1990 through 1994, we found 1475 *Rhyacotriton variegatus* from 220 different streams across the study area, including 410 animals that were found incidentally at 107 sites while conducting other field work. The remaining 1065 individuals (including 72 found in a pilot survey in 1992 that were not included in further analysis) were located at 113 sites from stream surveys. A sample of these animals contained 415 transformed individuals and 498 larvae. The sex ratio of 252 adults was nearly 1:1 (121 females, 131 males).

Landscape Surveys.—We recorded 694 salamanders from 57 of the 71 streams (80.3%) randomly selected from across the study area. The SLR analysis with six independent landscape variables showed that only geology (improvement $\chi^2 = 16.53$, df = 1, $P < 0.001$) and forest age (improvement $\chi^2 = 4.01$, df = 1, $P = 0.045$) entered the model to predict the presence of *Rhyacotriton variegatus.* A greater percentage of streams flowing through the consolidated geologic materials contained *R. variegatus* than those flowing through the younger, unconsolidated materials ($\chi^2 = 21.37$, df = 1, $P < 0.001$). Only one of seven (14.3%) located in the unconsolidated geologic formation contained *R. variegatus* compared to 56 of 64 streams (87.5%) located in the consolidated geologic formations (Fig. 1). Forest age differed significantly among sites with and without *R. variegatus* (Mann-Whitney U test: $Z = 2.66$, $P < 0.007$). The average age of stands surrounding streams with and without *R. variegatus* was 38.6 years (SD = 30.35, N = 57) and 63.1 years (SD = 42.95, N = 14), respectively. A greater proportion of streams with a northerly aspect (34 of 36) had *R. variegatus* compared to those with a southerly aspect (10 of 18; $\chi^2 = 12.05$, df = 1, $P < 0.001$), and there was a greater proportion of streams with *R. variegatus* in the northern (37 of 39) compared to the southern portion of the study area (20 of 32; $\chi^2 = 11.64$, df = 1, $P < 0.001$). *Rhyacotriton variegatus* was found from 49 to 1219 m in elevation and relative abundance varied from 0.008–1.12 *R. variegatus*/linear m searched (overall average = 0.15/linear m).

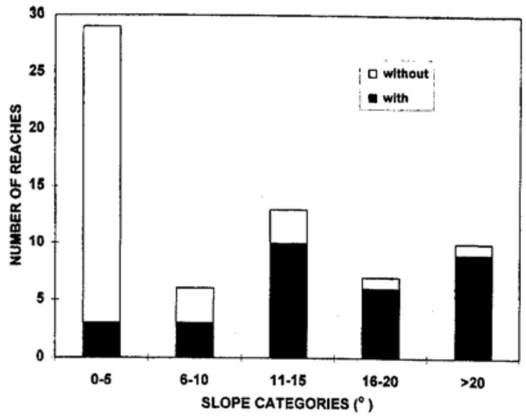

FIG. 2. Number of stream reaches with and without *Rhyacotriton variegatus* in five categories of stream slope (gradient), north coastal California.

TABLE 2. Comparison of the percent surface substrate composition of sites with and without *Rhyacotriton variegatus*. Significance based on Mann-Whitney U Tests.

Substrate composition	Sites with (N = 111)	Sites without (N = 216)	*P*
Boulder/bedrock	2.9	1.7	0.035
Cobble	11.0	19.4	<0.001
Gravel	54.3	46.3	0.004
Sand	10.2	16.2	NS
Silt-clay	21.6	16.4	<0.001

Stream Reach Surveys.—We recorded 109 *Rhyacotriton variegatus* from 31 of 64 stream reaches (48.4%) surveyed. A SLR with 16 continuous and two categorical variables determined that stream slope was the only significant variable to enter a model predicting the presence of *R. variegatus* (improvement $\chi^2 = 24.7$, df = 1, $P < 0.001$). The model with the single variable of slope provided a 82.5% correct classification of stream reaches. The strong relationship between stream slope and salamander presence is illustrated in Fig. 2. A Chi-square analysis of aspect (divided into 8 equal octants) was not significant. Pairwise comparisons indicated that reaches with *R. variegatus* had significantly greater slope, more small boulders, and less sand than those without *R. variegatus* (Table 1). All other comparisons were not significant. Canopy closure was greater and water temperature lower in reaches with *R. variegatus*, but these differences were not significant (Table 1). Estimates of salamander densities at sampling sites varied from 0.014 to 1.26 *R. variegatus*/m² from the 31 stream reaches of first and second order streams (overall average = 0.118/m²).

Microhabitat Surveys.—We collected 190 individuals from the 14 study streams known to have *Rhyacotriton variegatus*. Microhabitat sampling showed that abundance of *R. variegatus* was positively related to slope (ANOVA, F = 20.43, df = 2, $P < 0.001$). A logistic regression analysis without slope but with six continuous variables and one categorical variable showed that habitat type was the most important variable ($\chi^2 = 20.55$, df = 2, $P < 0.001$). Average percent overhang also was significant ($\chi^2 = 4.79$, df = 1, $P = 0.029$) with greater percent overhang at microsites with compared to those without salamanders. *Rhyacotriton variegatus* was found more often than expected in high gradient habitats (cascades and high gradient riffles), and less often than expected in low gradient riffles and pools ($\chi^2 = 53.64$, df = 2, $P < 0.001$). Of 147 microsites with *R. variegatus*, 89.8% were in high gradient habitats, 8.8% in low gradient riffles, and 1.4% in pools. In comparison, 53.3% of 212 mircosites without *R. variegatus* were in high gradient riffles, 36.8% in low gradient riffles, and 9.9% were in pools. The surface substrate in which *R. variegatus* was found was composed mostly of gravel (54%) and the overall ratio of substrate categories was significantly different from sites without *R. variegatus* ($\chi^2 = 141.29$, df = 4, $P < 0.001$; Table 2). Salamander densities at sampling sites varied from 0.09 to 5.0 *R. variegatus*/m² (overall average = 0.28/m²) in 14

TABLE 1. Continuous variables measured at stream reaches with and without *Rhyacotriton variegatus*, north coastal California, 1993. Significance based on pooled t-tests.

Habitat variable	Sites with (N = 31)		Sites without (N = 33)		*P*
	$\bar{x}$	SD	$\bar{x}$	SD	
Slope (°)	17.64	10.20	5.05	6.01	<0.001
Sand (%)	3.29	7.23	10.04	13.60	0.017
Small boulder (%)	11.74	13.65	5.79	8.23	0.037
Canopy cover (%)	85.94	28.78	72.13	37.33	NS
Water temp. (°)	12.51	1.40	12.87	1.27	NS

streams known to have populations of *R. variegatus*. Restricting the analysis to high gradient riffles of these same streams, densities varied from 0.18 to 5.5/m² (overall average = 0.83/m²).

Discussion

Rhyacotriton variegatus is widespread throughout most of the study area at the landscape level and was found in 80.3% of headwater streams surveyed. However, its presence was closely tied to the geological formation of the stream drainage. The small proportion of streams where this species was not found in the consolidated geologic region were typically in areas that had a high proportion of unconsolidated materials even though the site fell within a consolidated geologic type. When our search was confined to a randomly selected stream reach of fixed length (stream reach survey), *R. variegatus* was found only in 48.4% of the reaches. This illustrates that presence of *R. variegatus* was not effectively determined by a sampling methodology that was restricted to a relatively short (30–50 m) randomly selected sample reach.

Data are not available to make direct comparisons of our presence data of *Rhyacotriton variegatus* within headwater streams to other studies because different sampling procedures were employed. However, estimates of the proportion of streams with *R. variegatus* have varied from 28.5% in young forests to 86.4% in old growth areas (Carey, 1989; Corn and Bury, 1989; Welsh et al., unpubl. data).

We found an inverse relationship between presence of *Rhyacotriton variegatus* and forest age rather than a direct relationship as is often reported for *Rhyacotriton* spp. (Welsh and Lind, 1988; Carey, 1989; Welsh, 1990; Welsh et al., unpubl. data). However, this probably is a statisitical artifact produced by a secondary correlation with historical timber harvest patterns; we do not believe that *R. variegatus* favors landscapes dominated by young forests. Historically, coastal forests, where unconsolidated geologic formations were more likely to be encountered, were harvested first (late 1800s to early 1900s) and now have the oldest second growth forests. The more interior areas with steeper topography and shallower soils associated with consolidated geologic formations have been harvested within the last 30 yr. Therefore, we believe the strong association of *R. variegatus* with certain geologic formations and the history of harvesting in our study area produced a spurious association between forest age and presence of *R. variegatus*. The higher proportion of streams with this species in the northern portion of the study area relative to the southern region also is best explained by the fact that only one of the sample streams occurred in an unconsolidated geological formation in the northern portion of our study area. There also is a precipitation gradient with decreasing rainfall from north to south, which may create more favorable conditions for *R. variegatus* in the northern portion of the study area.

The strong association between the presence of *Rhyacotriton variegatus* and steep slopes suggests that this species prefers microhabitats with relatively loose gravel and cobble, open interstices, and minimal fine sediments. We believe that high gradient reaches are important because they are transport areas where finer sediments do not accumulate and gravel and cobble do not become embedded. Good and Wake (1992) also noted that *Rhyacotriton* is associated with areas of "considerable relief" and is generally absent from areas with low relief.

Rhyacotriton requires cold water (Nussbaum et al., 1983; Corn and Bury, 1989) and both aspect and canopy influence water temperature (Beschta et al., 1987; Bury and Corn, 1991). We believe that the positive association between the presence of *R. variegatus* and northerly aspects at the landscape level indicates that water temperature may be limiting to *R. variegatus* in some southerly exposures in our study area.

The lack of a correlation of aspect and canopy closure to presence of *Rhyacotriton variegatus* at the stream reach level would suggest that these variables should be measured over a larger area. We also believe these variables tend to have a lesser impact in our study area because of the influence of the coastal climate. Cool summer temperatures and coastal fog moderate the impacts of variation in aspect and canopy closure on water temperatures. The narrow range of water temperatures measured in all streams (10–16 C) would suggest that the climate of the area moderates impacts on water temperature.

At the level of the microhabitat survey, there was a positive relationship between higher stream gradients and abundance of *Rhyacotriton variegatus*. Corn and Bury (1989) found a similar relationship for streams flowing through forests logged between 14 to 40 yr prior to their study. They noted that this relationship might be suspect because only three streams of 20 contained *Rhyacotriton*, but the species was absent from all logged streams with gradients <11%. They found no relationship between abundance of *Rhyacotriton* and stream gradient in streams flowing through uncut forests.

We found *Rhyacotriton variegatus* significantly more often in high gradient habitats compared to other habitat types, which would be expected given the relationship between abundance and stream slope. This further suggests that *Rhyacotriton* prefers microhabitats where sand is not deposited and interstices remain open. However, *R. variegatus* apparently was selecting for specific microsites within the high gradient riffles where there was more gravel but also more of the finest of sediments. This same type of relationship was noted by Welsh et al., (unpubl. data). They hypothesized that the relationship may be due to the finest sediments being composed of organic material that is important to many aquatic invertebrates and thus may be linked to potential prey for the salamanders (Welsh et al., unpubl. data).

Our surveys were not designed to provide estimates of population densities. In addition, searches of headwater streams often were incomplete, because of large amounts of debris left from past logging. However, our data on relative abundance and salamander densities at sampling sites do provide useful information about the patterns of abundance. Although most headwater streams had *R. variegatus*, their abundance was highly variable from stream to stream (0.014 to 5.0 animals/m^2), a pattern similar to that reported by Welsh and Lind (1992). In addition, the species was patchily distributed within streams. Usually, the best habitat and most *R. variegatus* were located near the upper most portion of the wetted channel, although there was likely some bias in this observation because it was easier to locate animals where there was only minimal flow.

Estimates of salamander densities reported from other studies ranged from 0.01 to 6.7 *Rhyacotriton* spp./m^2 (Bury, 1988; Corn and Bury, 1989; Welsh and Lind, 1992). However, the highest densities reported from single isolated localities are 14–22 individuals/m^2 in a seep (Welsh and Lind, 1992) and 27.6–41.2 individuals/m^2 in a small Oregon headwater stream (Nussbaum and Tait, 1977). Direct comparisons of estimates of salamander densities from this or previous studies are not appropriate because of differences in study designs, and because none of the studies were designed to estimate population densities.

Comparisons between undisturbed and disturbed streams were not possible because virtually all of our study area has been harvested at least once. Consequently, it is difficult to assess the extent to which past timber harvest impacted populations of *Rhyacotriton variegatus*. We believe that in most streams in our study area, habitat probably existed further downstream in lower gradient reaches prior to timber harvest and was reduced or eliminated due to the accumulation of sediments. High gradient reaches were probably less impacted by timber harvest. We do not know how isolated springs and seeps may have been impacted because our surveys were restricted to continuous stream channels. However, incidental observations indicate that some of the highest densities of *R. variegatus* occur in these habitat types within our study area. We conclude that previous unregulated timber harvest practices caused a reduction in the number of individuals in most headwater streams in consolidated geologic areas, but probably did not often cause the total extinction of populations in a stream because virtually all streams in our study area have some high gradient reaches. Our data also suggest that *R. variegatus* is not tied to old growth per se; however, the specific microhabitats required by this species are more likely to exist in undisturbed areas.

Continued survival of this species in our study area cannot directly be assessed. However, several factors suggest that habitat for the species will be maintained and possibly improved. The mean age of forests surrounding streams with *Rhyacotriton variegatus* was 39 yr. Therefore, most stands immediately adjacent to streams with *R. variegatus* will continue to grow for decades. Current timber harvest regulations in California mandate protection for all streams with *R. variegatus* or their habitat. Whereas little or no protection was provided to headwater streams in the past, protection of streams now includes equipment exclusion zones and tree retention standards ranging from 15–30 m on each side of the stream. With these protection zones, better road construction, and improved logging practices (i.e., cable logging), current and future impacts of timber harvest will be significantly less relative to those of the unregulated past.

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Distribution and Habitat of *Ascaphus truei* in Streams on Managed, Young Growth Forests in North Coastal California

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ABSTRACT.—We studied the distribution and habitat of larval *Ascaphus truei* in first and second order streams of managed forests in north coastal California from 1993–1996. Of 72 streams randomly selected to relate landscape variables to the presence of *A. truei*, 54 (75%) contained larvae. Geologic formation was the only landscape variable that predicted the presence of *A. truei* in a stepwise logistic regression (SLR) model. A second survey was conducted to determine which habitat variables of stream reaches were related to the presence of *A. truei*. Larvae were found in 18 (37%) of 49 stream reaches with flows >1500 cm^3/sec, and percent fines (negative association with frog presence), stream gradient (positive association), and water temperature (negative association) were the only habitat variables of 21 measured that entered a SLR model to predict the presence of *A. truei*. Only stream gradient differed significantly between reaches with and without tailed frogs; canopy cover, temperature, and forest age were not significantly different. A final survey to further define the microhabitat used by larval *A. truei* showed that larvae were found more often than expected in high gradient riffles and less often than expected in pools and runs. Occurrence of larvae was positively associated with cobble, boulder, and gravel substrates with lower embeddedness, and negatively associated with fine substrates. We discuss the comparative habitat requirements and sensitivities to land management activities of the two amphibian headwater stream inhabitants, *A. truei* and *Rhyacotriton variegatus*, in our study area.

Ascaphus truei, the tailed frog, is unique among North American anurans because it is highly specialized for life in cold, clear, mountain streams (Nussbaum et al., 1983). The larval stage lasts from two to five years (Metter, 1964; Brown, 1990), and tadpoles have an enlarged oral disc modified into an adhesive, sucker-like structure enabling individuals to adhere to rocks in swift current. Tadpoles feed almost exclusively on diatoms which are scraped off rocks (Metter, 1964). Transformed individuals can be found under objects in streams or near the stream margins in daytime. At night, under appropriate conditions of temperature and humidity, they are found on top of objects along the stream and up to 20–30 m from the stream feeding on insects and other invertebrates (Nussbaum et al., 1983). The species is found from southern British Columbia south to northwestern California from the Cascade Mountains west to the coast (Metter, 1968). It also occurs inland as disjunct populations in the Blue Mountains of southeastern Washington and northeastern Oregon, and in the northern Rocky Mountains of northern Idaho and western Montana (Nussbaum et al., 1983).

Ascaphus truei is one of the stream amphibians reported to be at risk in the Douglas-fir (*Pseudotsuga menziesii*) zone and it has been suggested that local extinctions of this species will occur after clearcutting these forests (Bury and Corn, 1988a; Corn and Bury, 1989). These authors have speculated that recolonization may take decades because *A. truei* has limited dispersal abilities and adults tend to breed in their natal stream. They also stated there is a need to assess the effects of logging in streamside and upland forests on headwater and small stream amphibians, such as *Ascaphus* (Bury and Corn, 1988b). No studies have been conducted on the habitat requirements of this species in the redwood (*Sequoia sempervirens*) zone of northwestern California, where a mild coastal climate has been shown to modify its life history patterns (Wallace and Diller, 1998) and may also modify its distribution patterns and habitat requirements.

In 1993, we began an extensive sampling program across the study area to determine the distribution and habitat associations of *Ascaphus truei* at three hierarchical levels of survey. Our study focused on larval *A. truei* because we believe the larval stage, which is restricted to streams, is the most sensitive to the impacts of timber harvest. The objectives of this study were to determine the distribution and relative abundance of this species in relation to major landscape variables, to correlate the presence/absence of the species with stream reach variables, and to determine the specific microhabitat components associated with sites utilized by *A. truei*.

Materials and Methods

Study Area.—Our study area encompassed 1500 km^2 of private timber lands located west of the crest of the Coast Range in western Del Norte, Humboldt and Trinity counties, northwestern California. Most of the property is within 32 km of the coast, but extends up to 85 km inland in places. The study area is located mostly within the north coast redwood zone (Mayer, 1988) where fog is common. Near the coast, mean summer and winter air temperatures are about 18 C and 5 C, respectively, but extremes of 38 C in summer and −1 C in winter are not uncommon 48 km from the coast. Precipitation varies from 102 to 254 cm annually, with 90% falling from October through April (Elford, 1974).

Coast redwood and Douglas-fir are the codominant conifers over most of the study area, with Douglas-fir becoming more prevalent at higher, drier locations. Hardwoods, such as tanoak (*Lithocarpus densiflorus*), red alder (*Alnus rubra*), Pacific madrone (*Arbutus menziesii*), and California bay (*Umbellularia californica*) also are major stand components. Common species along the watercourses surveyed include red alder, big leaf maple (*Acer macrophyllum*), and willows (*Salix* spp.).

Three major types of logging have occurred in the north coast area in the past; clearcutting entire drainages, selective logging, and—since the late 1960s—even-aged management with small clearcuts and prompt artificial regeneration. As a result of this logging history, the study area mostly consists of 0–80 yr old second and third growth forests with a stand age distribution of: 0–9 yr, 13%; 10–20 yr, 16%; 21–60 yr, 60%; and 61+ yr, 11%. Before 1973, streams were not protected in areas being harvested. Since 1973, state law has required leaving variable-width forest buffers along streams supporting fish or other aquatic life.

Landscape Surveys.—In 1993, we began surveys of amphibians on the study area by using a stratified random sampling design to select up to four sections per township from U.S. Geological Survey maps. The number of sample sections per township was reduced if the study area was not located in the entire township. Sampling was designed to insure selection of one section per 1/4 township (9 sections). Each section chosen had to include at least one half study area and have road access.

We selected 72 sections for a presence/absence survey of *A. truei*. We sampled the first second-order stream encountered along the major road through the section that had at least 1000 m of channel with flowing water. Tailed frogs were surveyed by searching for larvae attached to rocks on the stream bottom. A glass-bottomed viewing box was used to search for larvae across the entire streambed. Each stream was searched for 1000 m or until presence was documented. Once the first *Ascaphus* was found, an additional 20 m was searched to establish relative abundance for that particular stream. Search effort for all streams was concentrated in the best available habitat. Life history category (larvae, juvenile, adult) and sex of adults (presence of tail in males) were recorded for all *Ascaphus* collected. Forest age of the stand adjacent to each stream, stream aspect, and elevation were taken from a geographic information system data base and aerial photographs (1:12,000 scale). Cover types were grouped into redwood, Douglas-fir, redwood/Douglas fir mix, and hardwoods. The geological formation in which the watershed occurred was taken from U.S. Geological Survey topographic maps overlain by State of California, Department of Forestry geology maps and photographic interpretations (O. Huber, pers. comm.). Thirteen geological formations were identified but grouped into two categories, consolidated and unconsolidated, based on formation age and particle type formed following decomposition. The consolidated geologic group was composed of older formations that form boulders, cobbles, and gravel during decomposition into fine sediments, whereas the unconsolidated category included younger marine deposits that decompose directly into silt and sand.

Stream Reach Surveys.—To determine which stream reach habitat variables predict the presence of *A. truei*, we used the same sampling protocol to select an additional 13 first-order and 41 second-order streams with flows greater than

1500 cm^3/sec within the contiguous portion of the study area. Fixed stream reaches were located 10 m above the roadway or culvert. We placed cross-stream transects (Platts et al., 1983) at 5 m intervals starting 2.5 m above the lower end of the reach and established 10 transects in each sample stream unless physical features (waterfalls, log jams, stream going underground) prevented this. In one stream, only 25 m (five transects) were sampled, but in all other streams at least 30 m (six transects) were sampled. We searched streams with a viewing box and turned the substrate in search of animals.

Habitat variables measured (in cm) at each transect included amount of living vegetative overhang (total linear length up to chest height), small organic debris (sod, total linear length of leaves, twigs, and sticks <10.2 cm in diameter on the substrate), and logs (total linear length of dead woody material >10.2 cm in diameter occurring over or in the stream but not on the stream bottom). Substrate was classified by measuring along each transect the amount of mud/silt, sand, gravel, cobble, small boulder, large boulder, and large organic debris in cm (Platts et al., 1983). Stream gradient at each transect was measured by placing the center of a 1 m rod on the transect line parallel to the water flow and at the stream surface and recording the gradient, in degrees, from a clinometer. Percent canopy closure was estimated by taking readings with a densiometer in each cardinal direction in the center of the stream at the first, fifth, and tenth transects.

We estimated stream flow and measured temperature, pH, and conductivity during August and September to reduce the effects of seasonal variation. Stream flow (cm^3/sec) was estimated by measuring stream depth (cm) at 1/4, 1/2, and 3/4 intervals across the stream and dividing by four to get mean depth (Platts et al., 1983), measuring stream width (cm) at this point, and timing the surface speed of a small floating object for three trials. Temperature was taken to the nearest 0.1 C with a Schultheis quick recording thermometer. Conductivity (ms) and pH were estimated by an Oakton water test kit. Cover type, forest age of the stand surrounding the stream, and stream aspect were determined from maps.

Microhabitat Surveys.—We conducted a final study to better quantify the microhabitat associations of *A. truei*. For this study, 17 streams were subsampled from the 54 streams of the landscape survey known to have *A. truei*, using a stratified, random design. We first conducted a stream layout by walking the stream and identifying reaches in each of three gradient classes, 0–5%, 6–10%, and >10%. A reach was recorded if it was at least 20–30 m long, allowing for the placement of two or more sampling belts within that gradient class. We continued upstream until two reaches in each gradient class were identified, or 300 m, whichever was less.

Sampling belts were started 10 m upstream from the road, or beyond the influence of the road, whichever distance was greatest. We randomly placed the first sample belt 0–5 m upstream from the start of the sample reach. Sampling belts were 1.5 m long and assigned to a habitat type (pool, run, low-gradient riffle, or high-gradient riffle). Additional belts were systematically placed at 10 m intervals with a maximum of 15 belts per gradient class. If one gradient class exceeded 150 m (more than 15 belts), we increased the distance between belts to systematically sample over the entire length of the gradient class. If placement of the belt occurred on an unsearchable portion of the stream or between two habitat units, we adjusted the placement of the belt upstream to include a single habitat unit.

Before quantifying microhabitat, the surface of the substrate was visually searched for *Ascaphus* using a viewing box. Five cross-stream transects were then placed within each belt by laying a measuring rod perpendicular to the stream channel at 3 dm intervals beginning and ending 1.5 dm from the lower and upper limits of the belt. We recorded the substrate particle (fines, sand, gravel, small cobble, large cobble, small boulder, large boulder; Platts et al., 1983; and sod or lod) at each 2 dm point. Average water depth (at the midpoint of each sample belt), vegetative overhang, and gradient of the belt were measured as noted above (stream reach survey). Canopy closure was estimated (as above) at the mid-point of each belt and the upstream distance to the nearest log or log jam was measured (directly if 10 m or less, estimated if from 10–30 m, and not recorded if >30 m). Embeddedness of cobbles was visually estimated and assigned to one of four categories (0–25, 26–50, 51–75, and 76–100%) for each sampling belt. Each belt was searched for *Ascaphus* by working upstream and removing all loose objects from the channel while holding an aquarium net downstream of the object. After all loose objects were removed from the channel, the entire belt was searched again with the viewing box. If *Ascaphus* was found, we recorded the following life history data: larva or transformed; snout to tail length and limb measurements of larvae; and snout to vent length and sex of adults or transformed *Ascaphus*.

Data Analysis.—In our analysis of the relationships of landscape variables to the presence of *A. truei*, elevation (m) and forest age (0–117 yr) were considered independent continuous variables. Aspect was measured as a continuous variable (0–360°) but grouped into eight 45° octants and treated as a categorical variable. Geologic formation and cover type were treated as categorical variables. We used a stepwise logistic regression analysis, SLR (NCSS, Version 6.0; Hintze, 1995), with 20 iterations and a P value of 0.20 to enter the model to determine which of the five landscape variables best predicted the presence of *A. truei*. We then used Chi-square analysis (NCSS, Version 6.0) to test for association with the presence of *A. truei* to the variables 'aspect' (divided into four quadrants) and cover type. We divided the study area into a northern and southern region for further analysis because of a decreasing rainfall gradient from north to south (Diller and Wallace, 1994). The Mann-Whitney U test (NCSS, Version 6.0) was used to compare the forest age of stands adjacent to streams with and without *A. truei.*

For the stream reach survey, five of the original 54 streams sampled were omitted from the analysis because of missing data. We considered stream order and cover type as categorical variables; all others were treated as continuous variables. We used SLR (NCSS, Version 6.0) with 20 iterations and a P value of 0.10 to enter the model to determine which of the stream reach variables best predicted the presence of *A. truei* in the sample reaches. Mann-Whitney U tests were used to compare the continuous variables slope, canopy closure, temperature, sod, forest age, and the average amounts of each substrate type of reaches with and without *A. truei*.

In our study of microhabitat use, habitat type (pool, run, low gradient riffle, and high gradient riffle) was considered a categorical variable. For each sample belt, average substrate composition, stream width (dm) and depth (cm), stream gradient (%), distance to the nearest log (m), vegetative overhang (%), and canopy closure (%) were calculated and considered independent continuous microhabitat variables. We used SLR (NCSS, Version 6.0) with 20 iterations, and a P of 0.20 to enter the model to determine which of the microhabitat variables best predicted the presence of *A. truei* in the sample belts. Because substrate particle size is associated with different habitat types (Rosgen, 1996), we ran a second SLR without habitat type as one of the independent variables. We did not use abundance of larval *A. truei* as a dependent variable because tadpoles in some streams were metamorphosing during the survey period and the larval population was declining throughout the survey. Mann-Whitney U tests were used to compare microhabitat variables of sample belts with and without *A. truei*. A Chi-square analysis then was used to compare habitat types and the presence of *A. truei*. Alpha for all analyses was 0.05.

RESULTS

From 1993 through 1996, we recorded 725 *Ascaphus truei* from the study area; 693 were larvae and 32 were transformed juveniles, subadults, or adults. The statistical analyses reported here are based on the 693 larvae.

Landscape Surveys.—We found 443 *A. truei* in 54 (75%) of 72 streams randomly selected from the study area. The SLR analysis with five independent landscape variables showed that only geologic formation (improvement χ^2 = 12.11, df = 1, $P < 0.001$) and forest age (negative association, improvement $\chi^2 = 7.68$, df = 1, $P < 0.01$) entered the model to predict the presence of *A. truei*. The model correctly classified 86% of the streams sampled. A greater percentage of streams flowing through consolidated geologic materials (54 of 67, 81%) contained *A. truei* than those flowing through the younger, unconsolidated materials (zero of five; Fig. 1). There was a significant difference in forest age of stands surrounding sites with and without *A. truei* (Mann-Whitney U test: Z = 1.95, df = 1, $P = 0.051$), with mean stand age greater at sites without (median = 39.5 yrs, range = 109, N = 18) compared to sites with *A. truei* (median = 32, range = 84, N = 54). There was no significant difference in the proportion of streams with a northerly aspect having *A. truei* compared to those with a southerly aspect (χ^2 = 5.47, df = 3, $P = 0.140$). However, there was a significantly greater proportion of streams with *A. truei* in the northern (29 of 30, 97%) compared to the southern area (25 of 42, 60%; χ^2 = 12.88, df = 1, $P < 0.001$). Only two cover types, redwood and Douglas-fir, were recorded in drainages of the sample streams and there was no significant difference ($P = 0.682$) in cover type between streams with and without *A. truei*. Relative abundance of *A. truei* varied greatly among streams. In nine streams, only one to three animals were found within 200–1500 m of stream that was covered in search of suitable habitat (90–450 m actually surveyed), while in four other streams, 24–56 animals were found in 30 to 50 m surveyed. *Ascaphus truei* was found from 24 to 1038 m in elevation. Of the 443 animals captured during the landscape survey, only five were transformed individuals (one juvenile, four adults).

Stream Reach Survey.—We recorded 63 larval *A. truei* from 18 (37%) of 49 stream reaches. The SLR with 18 continuous and three categorical variables determined that percent fines (negative association), stream gradient (positive association), and water temperature (negative association) were the only variables to enter a model predicting the presence of larval *A. truei* (improvement χ^2 = 3.82, df = 1, $P = 0.051$; χ^2

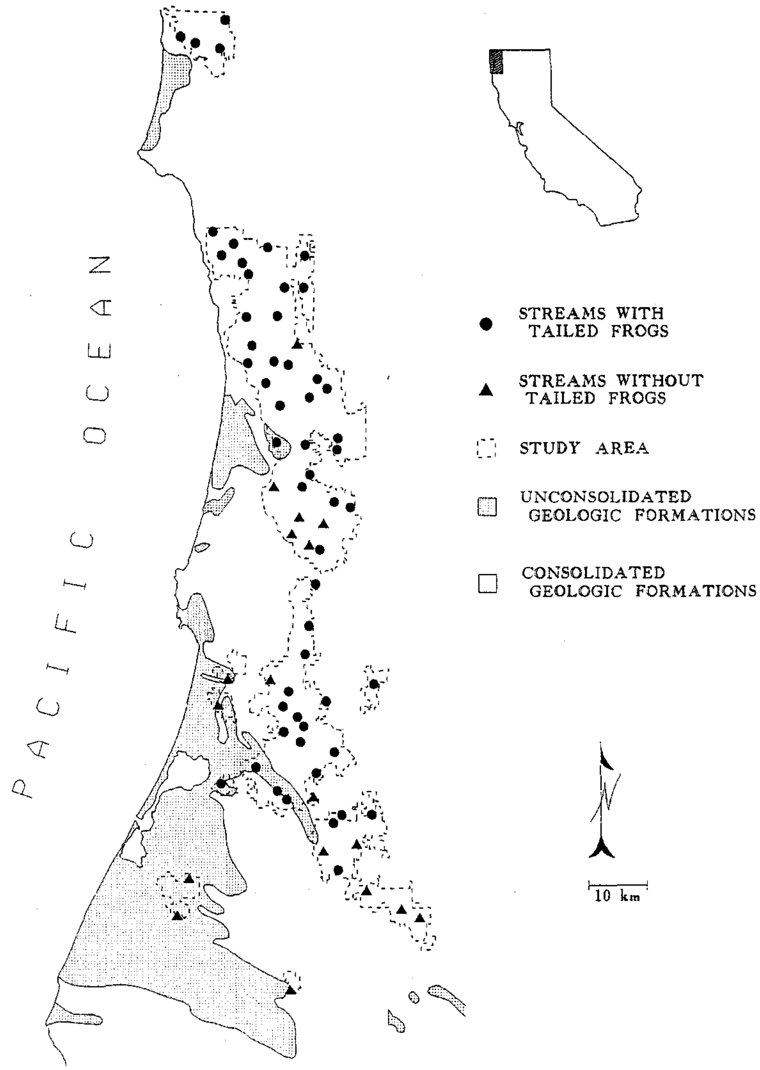

FIG. 1. Map of study area showing consolidated and unconsolidated geological formations, and streams with and without *Ascaphus truei*, north coastal California. Data obtained from landscape survey.

= 3.79, df = 1, P = 0.051; and χ^2 = 2.99, df = 1, P = 0.084, respectively, but none of these variables were statistically significant, assuming a strict interpretation of the alpha level). The model correctly classified 78% of the stream reaches sampled. Only gradient differed significantly between reaches with and without *A. truei* (Mann-Whitney U test: Z = 2.45, df = 1, P = 0.014) with tadpoles more likely to be found in higher gradient reaches. Canopy cover, temperature, forest age, and aspect did not differ significantly between reaches (Table 1). There were no significant differences in percent substrate composition between reaches with and without *A. truei*. In streams with *Ascaphus* larvae, relative abundance in sample reaches varied from 0.02–0.24 larvae/linear m (overall mean = 0.11).

Microhabitat Survey.—We recorded 192 larval *A. truei* from 17 streams surveyed to determine microhabitat associations of this species. A total of 349 1.5 m-belts was sampled, of which 82 (23%) had *A. truei*. A SLR analysis with one categorical and 15 continuous variables showed that the high gradient riffle habitat type was the first variable to enter the model (positive association, improvement χ^2 = 43.80, df = 1, P < 0.001). The next three variables entering the model with a significant improvement χ^2 were percent small cobble (positive association, χ^2 = 25.06, df = 1, P < 0.001), low gradient riffle

TABLE 1. Comparison of selected habitat variables between stream reaches with (N = 18) and without (N = 31) *Ascaphus truei*. Significance based on Mann-Whitney U Tests. * nonsignificant results.

Variable	Sites with *Ascaphus truei*			Sites without *Ascaphus truei*			P
	Median	Range	$\bar{x}$ (SD)	Median	Range	$\bar{x}$ (SD)	
Stream gradient (%)	7.1	18.1	9.1 (6.00)	3.6	28.2	5.9 (6.29)	0.014
Canopy cover (%)	100.0	99.3	81.3 (30.67)	100.0	93.5	87.6 (25.95)	0.385*
Water temp (C)	12.0	8.0	12.2 (1.71)	12.5	6.3	12.8 (1.33)	0.124*
Forest age (yrs)	21.5	46.0	23.0 (11.88)	22.0	81.0	25.8 (21.14)	0.884*
Substrate fines (%)	0.0	14.2	2.3 (4.04)	1.5	49.2	8.1 (13.81)	0.286*

(positive association, $\chi^2 = 11.90$, df = 1, $P < 0.001$), and percent fines (negative association, $\chi^2 = 6.21$, df = 1, $P = 0.013$). The first model with just one independent variable (high gradient riffle) provided 77% correct classification, while the final model with all four variables only increased the correct classification to 81%. A second SLR analysis, omitting habitat type as an independent variable, found that percent fines was the first variable entering the model (negative association, improvement $\chi^2 = 41.95$, df = 1, $P < 0.001$), followed by small cobble (positive association), water depth (negative association), and large boulder (positive association) (improvement $\chi^2 = 21.39$, df = 1, $P < 0.001$; $\chi^2 = 7.40$, df = 1, $P < 0.001$; and $\chi^2 = 4.48$, df = 1, $P = 0.034$, respectively). The model provided 78% correct classification. *Ascaphus truei* was found more often than expected in high gradient riffles and less often than expected in pools and runs ($\chi^2 = 52.37$, df = 3, $P < 0.001$). Of 90 belts with *A. truei*, 81.1% were in high gradient riffles, 15.6% in low gradient riffles, and 3.3% in pools and runs. Sample belts with *A. truei* contained cobble with significantly lower embeddedness, higher stream gradient, and less mean depth (Table 2) than belts without the species. Belts with *A. truei* also had significantly less fines, more gravel, and more cobble. Average densities of larvae in the sampling belts varied from 0.04–0.73 individuals/m^2 (overall average = 0.24/m^2) among the 17 streams sampled. However, if only high gradient riffles were considered, where most larvae were found, the average density varied from 0.20–7.25 larvae/m^2 (overall average = 1.23/m^2).

DISCUSSION

Ascaphus truei was widespread at the landscape level and was found in 75% of the streams sampled. However, its presence was closely tied to the geological formation of the stream drainage. No tailed frogs were found in five streams identified as being in an unconsolidated geologic region of the study area. Several of the remaining 13 streams without *A. truei* appeared during sampling to have a high proportion of unconsolidated geologic material influencing the stream sediments, even though they were identified from maps at the landscape scale as being in consolidated geologic regions. Therefore, we believe that a site specific quantification of the geology of streams sampled would further strengthen our conclusion that geologic formation of the stream basin was an important factor in predicting the occurrence of tailed frogs, due to the influence that it has on the composition of the stream substrate.

It is difficult to directly compare the proportion of streams sampled with *A. truei* in this study relative to other studies, because different sampling procedures were used. However, 75%

TABLE 2. Comparison of selected microhabitat variables between belts sampled with (N = 82) and without (N = 267) *Ascaphus truei*. Significance based on Mann-Whitney U tests. Embeddedness (N = 72 with and N = 151 without) based on a rating system where 1 = 0–25%, 2 = 26–50%, 3 = 51–75%, and 4 = 76–100% embedded.

Variable	Sites with *Ascaphus truei*			Sites without *Ascaphus truei*			P
	Median	Range	$\bar{x}$ (SD)	Median	Range	$\bar{x}$ (SD)	
Embeddedness score	2.0	2.0	1.99 (0.54)	3.0	3.0	2.85 (0.82)	<0.001
Fines (%)	5.1	43.3	7.11 (8.10)	15.0	94.6	21.88 (22.63)	<0.001
Gravel (%)	21.8	63.9	21.95 (11.92)	16.7	87.5	19.77 (15.91)	0.029
Small cobble (%)	17.0	56.1	18.91 (10.49)	8.7	43.3	10.04 (8.43)	<0.001
Large cobble (%)	19.1	46.7	20.46 (10.61)	13.6	100.0	15.29 (11.96)	<0.001
Gradient (%)	8.0	54.0	11.40 (9.70)	3.0	60.0	5.72 (8.44)	<0.001
Depth (cm)	5.1	12.0	5.32 (2.27)	6.8	45.3	8.46 (5.95)	<0.001

of the streams with *A. truei* in our study is intermediate to other studies where estimates varied from 35% in young forests to 96% in old growth areas (Corn and Bury, 1989; Welsh, 1990; Bull and Carter, 1996).

We found an inverse relationship between presence of *Ascaphus* and forest age rather than a direct relationship as is often reported for this species (Carey, 1989; Corn and Bury, 1989; Welsh, 1990). However, this probably is a correlation that resulted from past timber harvest patterns; we do not believe *Ascaphus* favors landscapes dominated by young forests. Historically, coastal forests, where unconsolidated geologic formations were more likely to be encountered, were harvested first (late 1800s to early 1900s) and now have the oldest second growth forests. The more interior sites with steeper topography and shallow, rocky soils associated with consolidated geologic formations have been harvested within the last 30 yr. Therefore, we believe that geologic formation has such a profound influence on stream substrate condition that it negates the potential impact of stand age on the occurrence of *Ascaphus* in our study area. The higher proportion of streams with this species in the northern portion of the study area relative to the southern region also is best explained by geology. All of the streams sampled that were in the unconsolidated geologic formation occurred in the southern portion of the study area. There also is a precipitation gradient with decreasing rainfall from north to south, which may create more favorable conditions for *A. truei* in the northern portion of the study area.

At the level of the stream reach, gradient was the only variable that was significantly different in reaches with versus without *A. truei*, and the same variable, with a positive association, entered the SLR model to predict the occurrence of this species. Percent fines, with a negative association, also entered the SLR model to predict the occurrence of *A. truei* within stream reaches. This is likely due in part to the association between stream gradient and substrate, where higher gradient reaches are typically transport areas that do not accumulate fine sediments (Rosgen, 1996). We suspect that the lack of any other significant results with substrate variables was due, in part, to the stream reach being too large of a scale for attempting to quantify variables that correlate best with stream habitat units. As noted below, we did observe significant differences in substrate variables among habitat units with and without frogs at the microhabitat scale.

Ascaphus requires cold water to complete larval development (Brattstrom, 1963; de Vlanning and Bury, 1970; Brown, 1975), and increased water temperature is thought to be one of the short-term impacts from timber harvest that may negatively affect populations of *A. truei* (Bury and Corn, 1988a). Our data provides no direct evidence that water temperature influenced the occurrence of *Ascaphus* in this study area. Water temperature, with a negative association, did enter the SLR model to predict the occurrence of the species, but the variable was not significant at the traditional alpha level of 0.05. In addition, the difference in mean temperatures of stream reaches with and without *A. truei* was small and not significant (12.2 versus 12.8 C, respectively). The minimal impact of temperature on the occurrence of *A. truei* in our study area probably was best explained by the ameliorating influence of the cool coastal climate of this region, which reduces the magnitude of the increase in water temperatures that could occur following timber harvest. We make this suggestion because the range of water temperatures recorded during the stream reach survey only varied from 7.5 to 15.7 C. Furthermore, there was no significant correlation between water temperature and aspect or canopy closure, even though both of these factors are known to influence water temperature (Beschta et al., 1987; Bury and Corn, 1991).

The association of *A. truei* with different substrate types was best seen at the level of the microhabitat survey. There was a consistent pattern of larval *A. truei* being associated with higher gradient riffles and substrate types such as small cobble and large boulder, while being less likely found in pools and runs, and habitat units with greater embeddedness and fine sediment. These findings are similar to those of Corn and Bury (1989) and Bury et al. (1991), who noted that *A. truei* preferred rocky substrates with cobble-sized rocks and was most commonly found in riffles. Hawkins et al. (1988) also found that higher density of larvae was associated with higher water velocities, lower embeddedness, and cobble-sized substrate (10–30 cm). There are a variety of possible reasons why larval *A. truei* might be associated with high gradient reaches, which have higher water velocities (e.g., increased oxygen and reduced predation). However, we believe that the strong association with high gradient riffles was at least partly due to larvae seeking out the habitat type that was less likely to have substrates embedded with fine sediment. This conclusion was reached because we observed that larvae could be found in low gradient riffles or runs when the substrate was not embedded. Unfortunately, this phenomenon did not occur with sufficient regularity to allow quantification of the relationship. The influence of water velocity on habitat selection in larval *A. truei* is largely unknown, and could not be readily elucidated without an experimental design in a controlled environment.

Of the three hierarchical levels of study, the microhabitat study came the closest to providing a density estimate for *A. truei*. We found that mean abundance of larvae in sample belts varied among streams from 0.04 to 0.73 larvae/m^2 (overall mean = 0.24/m^2). In addition to variation among streams, *A. truei* was often patchily distributed within streams, usually dependent on the distribution of appropriate habitat and substrate type. The upstream limit of tadpole distribution within streams was typically restricted to flows greater than 1500 cm^3/sec, but incidental observations indicate that subadult and adult frogs often can be found in small headwater portions of streams. The abundance of *A. truei* reported in our study suggests a lower density of larvae compared to uncut and logged streams in the Coast Range of western Oregon (Corn and Bury, 1989). They found a mean abundance of 0.76 *A. truei*/m^2 (23 uncut streams) and 0.37/m^2 (20 logged streams), and Hawkins et al. (1988) estimated mean densities of 0.58 to 4.40 larvae/m^2 in three different classes of watersheds in the Mt. St. Helens region of Washington. However, direct comparisons are not possible since in the first case these authors reported they reconnoitered the stream and then selected a "typical" section to sample a 10 m reach, and in the second case, two larval cohorts occurred in two of the three streams sampled. In the current study, most of our streams contained only one larval cohort (Wallace and Diller, 1998).

Rhyacotriton variegatus, the southern torrent salamander, is a stream species whose distribution overlaps that of *Ascaphus truei* in upper portions of streams. Both species are generally thought to be sensitive to the impacts of land management activities that either increase sediment delivery to the stream or increase water temperature (Bury and Corn, 1988a; Corn and Bury, 1989). Overall, the distribution of *A. truei* mimicked that of *R. variegatus* in our study area, both being associated with consolidated geologic formations and were found in a similar proportion of streams surveyed (Diller and Wallace, 1996). We have recorded *R. variegatus* at a greater number of sites within the study area compared to *A. truei* (304 versus 126, respectively; L. V. D., unpubl. data). However, sites with *A. truei* were generally larger in size relative to sites with *R. variegatus* (10s of m of stream length for *R. variegatus* versus 100s of m for *A. truei*).

At the level of the stream reach, both species showed a positive association with stream gradient, but the association was much stronger for *R. variegatus* compared to *A. truei* (mean gradient 31.8% and 9.1% for reaches with *R. variegatus* and *A. truei*, respectively; Diller and Wallace, 1996). Data from the microhabitat surveys further support the conclusion that both species are less likely to be found in areas with higher levels of fine sediments, although *A. truei* larvae are generally associated with larger substrate compared to *R. variegatus* (cobble versus gravel).

Both species are sensitive to the same types of impacts (increased sediment inputs that result in a higher proportion of fine sediments and embeddedness of the stream substrate, and to a lesser extent, increases in water temperature). However, the results of our studies provide no direct evidence for which species may be the most sensitive to these changes in the physical environment of the stream. In spite of this, we believe that it is possible to predict that based on their occurrence within a watershed, *R. variegatus*, being in the uppermost headwater areas, is more sensitive to direct impacts of land management activities, while *A. truei* is more likely to be influenced by indirect cumulative effects of these activities.

Comparisons between undisturbed and disturbed streams were not possible in our study area because virtually all areas have been disturbed at least once. Consequently, it is difficult to assess the extent to which past timber harvest impacted populations of *Ascaphus truei*. We believe that in most streams in our study area at least some habitat was eliminated due to the accumulation of sediments, with high gradient reaches being less impacted by land management activities. Our data also suggest that *A. truei* is not tied to old growth habitats per se; however, the specific microhabitats required by this species are more likely to exist in undisturbed areas.

Continued survival of this species in our study area cannot be directly assessed, but we believe that stream and riparian habitat conditions should be improving for *Ascaphus truei*. Whereas most streams in our study area with *A. truei* were logged at least once with little or no protection in the past, protection of these streams now includes equipment exclusion zones and tree retention zones from 15–30 m on each side of the stream. With these protection zones, better road construction, and improved logging practices (i.e., cable logging), current and future impacts of timber harvest will be significantly less relative to those of the unregulated past.

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C11.2 MONITORING OF SOUTHERN TORRENT SALAMANDER POPULATIONS

C11.2.1 Introduction

Torrent salamanders are generally found in springs, seeps and the most extreme headwater reaches of streams (Nussbaum et al. 1983; Stebbins 1985). They are a small salamander that appears to spend most of its time within the interstices of the stream's substrate, which make them difficult to locate and capture without disturbing their habitat. The larvae have gills and are restricted to flowing water while adults also appear to spend most of their time in the water, but are capable of movements out of the water. They are thought to have limited dispersal abilities and small home ranges so that recolonization of extirpated sites may take decades (Nussbaum and Tait 1977; Welsh and Lind 1992; Nijhuis and Kaplan 1998). Given the highly disjunct nature of their habitat, individuals at a given site (sub-population) are likely to be isolated from other adjacent sub-populations. The degree of isolation of these sub-populations probably varies depending on the distance and habitat that separates them so that torrent salamanders could be best described as existing as a meta-population.

Although there is some evidence for cumulative effects of sediment input in certain sites, torrent salamanders are primarily vulnerable to potential direct impacts from timber harvest (Diller and Wallace 1996). Direct impacts could include activities such as excessive canopy removal at the site leading to elevated water temperature, operating heavy equipment in the site, or destabilizing soil leading to excessive sediment deposits at the site. Past observations have indicated that these direct impacts can lead to extinction of the sub-population at the site. Due to the survey difficulties noted above, an attempt to get a statistically rigorous estimate of the number of individuals at monitored sites would be impractical. In spite of this, an index of the number of individuals at each site and record the life history stage of each individual captured will be determined. However, given the unreliability of the index of sub-population size, the persistence of individual sub-populations will be used as the primary response variable for the torrent salamander monitoring.

Concerns could be raised that there are too few sub-populations in the meta-population of torrent salamanders to expect to see significant changes over time, or that any loss in sub-populations would threaten the long-term persistence of torrent salamanders within the Plan Area. However, 598 torrent salamander sites (sub-populations) already have been located across Green Diamond's ownership in the HPAs, and it is estimated that no more than 25-30% of the total potential habitat has been surveyed. In addition, without a formal monitoring protocol, the apparent extinction and re-colonization of several torrent salamander sites have been documented. This would indicate that the meta-population concept does appear to apply to torrent salamanders in this region.

C11.2.2 Objectives

The primary monitoring approach for southern torrent salamanders will employ a paired sub-basin design. Changes in the persistence of sub-populations will be compared in randomly selected sites in watersheds with (treatment) and without (control) timber harvest. In some cases, control sub-basins will not be available in which case changes in sub-populations will be compared to the amount of timber harvest. In either case, the objective will be to determine if timber harvest activities have a measurable impact on the persistence of sub-populations. Therefore, the objective for torrent salamander monitoring will be to determine if their is a difference in the persistence rate for treatment and control sub-populations, and to document any apparent changes in the habitat conditions or index of sub-population size at each site. The monitoring reaches within each sub-basin will be sampled at least one year prior to operations that could influence the treatment sites and every year thereafter. New sub-basins will be added across the ownership until there are 12-15 paired sites well distributed across the Plan Area. Depending on the schedule of harvesting in the treatment sub-basins, it will likely be necessary to monitor a site for more than 10 years to determine if a treatment effect has occurred. (Refer to Appendix D for full details of the field protocol.)

A secondary monitoring objective will be to document long-term changes in torrent salamander populations across Green Diamond's ownership. Previous studies done within the Plan Area estimated that 80% of all surveyed streams (almost 90% excluding geologically unsuitable areas) had torrent salamander populations (Diller and Wallace 1996). Given that this occurrence rate is near the highest reported for the species even in pristine conditions (Carey 1989; Corn and Bury 1989; Welsh et al. 1992), an additional objective is to sustain the occupancy of torrent salamander populations in streams across the ownership at a minimum of 80% through time. To determine if this objective is being met, the landscape-level survey previously completed (Diller and Wallace 1996) will be repeated at 10-year intervals.

C11.2.3 Thresholds/Triggers

The extinction of a sub-population of torrent salamanders is a stochastic event that will not be likely to occur on a regular basis. As such it will not provide a responsive trigger to incremental changes in habitat conditions for torrent salamanders. However, any extinction of a sub-population will trigger a first phase (yellow light) evaluation to determine if the extinction was likely to be related to management activities. The apparent decline in the index of sub-population size in treatment sites compared to control sites would also trigger a first phase evaluation, but Green Diamond does not believe these data could be used to determine a reliable estimate of a population trend. Any significant increase in the extinction of treatment sub-populations relative to control streams would initiate a second stage review, but it is likely that this could be documented only after many years of monitoring.

The yellow light thresholds will be:

- any extinction of a sub-population, or
- an apparent decline in the average index of sub-population size in treatment sites compared to control sites.

The red light thresholds will be:

- a statistically significant increase in the extinction of treatment sub-populations relative to control streams, or
- a significant increase in the net rate of extinctions over the landscapes.

The change in the occurrence of torrent salamander populations across the ownership would not be suitable to use as a trigger to initiate management review due to the extended time-lag between successive data points. However, the occurrence of torrent salamanders in streams across the Plan Area would serve as corroborative evidence to support the findings of the meta-population monitoring, and a significant decrease in the occurrence rate would initiate a review of the probable cause of the decline.

C11.2.4 Temporal Scale

Based on previous monitoring of torrent salamander sites, the extinction of a site will likely be due to a catastrophic event (natural or anthropogenic). This will be detected during the first survey season following the event. Therefore, yellow light conditions will trigger an evaluation in a single year. As noted above, the torrent salamander monitoring is not well suited for a red light threshold, because the temporal scale would likely be too long for effective use in adaptive management.

C11.2.5 Spatial Scale

The zone of monitoring influence for a specific site will be determined on a case-by-case basis. Given that torrent salamanders are most likely to be impacted by direct site impacts, assessment of yellow conditions will include a field inspection of the affected site to determine likely causes. Results from all sites will be examined to determine if extirpations or declines are localized, area-wide, or associated with specific management activities, geologies, climatic variations, or other variables. Potential adaptive management changes could occur within a HPA, across the Plan Area, or in all areas with similar geology, for example, depending on the nature of the monitoring results.

C11.2.6 Feedback to Management

As noted above, the extinction of a sub-population of torrent salamanders due to management activities will most likely be caused by the direct impacts of timber harvest. Green Diamond believes that most of these impacts can be avoided by the proper identification of the site as a Class II watercourse. Ongoing training of the forestry staff will be designed to insure that improper watercourse classification does not occur. However, if it does occur, additional corrective measures such as only utilizing trained biologists to determine watercourse classification on small headwater streams will be employed. Extinctions or apparent declines in numbers that occur for more subtle reasons will be evaluated using habitat data collected at each site such as monitoring water temperature, canopy closure and substrate composition. If the apparent cause is management related, the appropriate adjustments will be made to mitigate future impacts.

C11.2.7 Results to Date

Eight paired sub-basins have already been selected for monitoring southern torrent salamanders including one sub-basin (Poverty Creek) that will serve as a control for two treatment sub-basins (Jiggs and Pollock Creeks). Five were initiated in 1998, two in 1999 and one additional paired sub-basin was selected in 2000 (Table C11-1).

Table C11-1. Summary of southern torrent salamander monitoring sites, 1998-2000.[1]

			Salamanders		
Paired Monitoring Sub-basin	**Site**	**Type**	**1998**	**1999**	**2000**
Blackdog Creek	BD 5400 A	C	6	4	4
Blackdog Creek	BD 5400 B	C	9	27	12
Blackdog Creek	BD 5300 A	T	8	3	5
Blackdog Creek	BD 5300 B	T	18	2	1
Lower NF Mad	Poverty A	C	13	27	18
Lower NF Mad	Poverty B	C	63	87	79
Lower NF Mad	Jiggs A	T	7	6	7
Lower NF Mad	Jiggs B	T	6	5	5
Lower NF Mad	Pollock A	T	9	3	1
Lower NF Mad	Pollock B	T	4	5	11
Upper NF Mad	Canyon A	C	20	21	20
Upper NF Mad	Canyon B	C	8	3	18
Upper NF Mad	Mule A	T	9	9	11
Upper NF Mad	Mule B	T	6	7	2
Panther Creek	O-5 A	C/h	4	6	5
Panther Creek	O-5 B	C/h	8	23	23
Panther Creek	O-6 A	T	8	6	3
Panther Creek	O-6 B	T	3	1	2
Rowdy Creek	R-1700 A	C/h		7	7
Rowdy Creek	R-1700 B	C/h		5	13
Rowdy Creek	R-1000 A	T		13	10
Rowdy Creek	R-1000 B	T		7	3
NF Maple Creek	B (F-10)	C/h		3	3
NF Maple Creek	C (F11.5-1)	C/h		2	2
NF Maple Creek	D (F11.5)	T		5	3
NF Maple Creek	A (F-13)	T		4	6
Surpur Creek	B700A	C			9
Surpur Creek	A400A	C			9
Surpur Creek	B1042B	T			4
Surpur Creek	A400B	T			24
Totals			209	291	320

Note
1 "C" indicates a control site with no timber harvest, C/h represents a control site that will have some limited timber harvesting and "T" indicates treatment sites that will have extensive timber harvesting.

C11.2.8 Discussion

This study has only been going on for three years and there has been no timber harvesting immediately adjacent to any of the torrent salamander monitoring sites. Unlike the tailed frog monitoring protocol (see Appendix D), the torrent salamander protocol is based on the persistence of sites as the primary response variable and not on estimates of abundance of individuals in monitoring reaches. However, the protocol does specify consistent collecting effort over the same sample reach each year so that comparisons of relative abundance of individuals at each site can be made. In spite of the less precise estimate of abundance relative to tailed frogs, there was little annual variation in the number of torrent salamanders collected at monitoring reaches. The mean number of individuals captured per year from 1998-2000 for the 18 sites that were monitored over the entire three years was 11.6, 13.6, and 12.6, respectively. If this pattern persists, it could lend support for using relative abundance as the primary response variable, which would provide much greater sensitivity to the treatment effects for this monitoring approach. Recently, Green Diamond experimented with marking individual salamanders with a fluorescent elastomer and the initial results have been promising. If this technique proves to be reliable, it will be used to obtain mark-recapture estimates of salamander abundance which will allow tracking of changes in abundance over time.

C11.2.9 Conclusion

This study is in its preliminary stages and it is too early to determine if there were any effects of timber harvest on the persistence of the sites by torrent salamanders. However, most sites seemed to have relatively constant numbers among years and there was no evidence of any local extinction.

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C11.3 MONITORING OF TAILED FROG POPULATIONS

C11.3.1 Introduction

Tailed frog habitat has been characterized as perennial, cold, fast flowing mountain streams with dense vegetation cover (Bury 1968; Nussbaum et al. 1983). To support larval tailed frogs, streams must have suitable gravel and cobble for attachment sites and diatoms for food (Bury and Corn 1988). Streams supporting tailed frogs have been found primarily in mature (Bury and Corn 1988; Welsh 1990) and old growth coniferous forests (Bury 1983; Welsh 1990). Bury and Corn (1988) reported that the frogs seem to be absent from clearcut areas and managed young forests (Welsh 1990). Although these authors did not establish a cause and effect relationship, it is hypothesized that tailed frog populations could be effected by both direct and indirect impacts of timber management. Direct impacts could include activities such as excessive canopy removal at the site leading to elevated water temperature, or destabilizing soil leading to direct sediment inputs at the site. However, tailed frogs may be vulnerable to cumulative impacts from the upper reaches of watersheds that result in elevated water temperatures or excessive sediment loads. In this regard they are similar to the salmonid species except that such cumulative impacts could effect tailed frog populations before the impacts were manifest in the lower fish-bearing reaches of the watershed.

The primary focus of the tailed frog monitoring will be on the larval population. While the adults can move between the stream and adjacent riparian vegetation, the larvae respire with gills and are tied to the stream environment. They require a minimum of one year to reach metamorphosis (Wallace and Diller 1998), which necessitates over-wintering in the streams. They feed on diatoms while clinging to the substrate with sucker-like mouth parts (Metter 1964) and have limited swimming ability. This makes them potentially vulnerable to excessive bed movement of the stream during high flows, which previously have been documented to drastically reduce the larval cohort. As a result of their life history requirements, the larvae provide the most immediate and direct response to changes in stream. In addition, larval tailed frogs can be captured with ease while causing minimal disturbance to the site. Ongoing studies have allowed us to develop a protocol that has been shown to be highly effective in estimating larval populations. Adults can also be captured with minimal disturbance to the site, but in contrast to the larvae, their population size can not be readily estimated. As a result of all the factors discussed above, the primary response variable for the tailed frog monitoring will be the size of the larval population.

C11.3.2 Objectives

The primary monitoring approach will employ a paired sub-basin design. Changes in larval populations of tailed frogs will be compared in randomly selected streams in watersheds with (treatment) and without (control) timber harvest. In some cases, control sub-basins will not be available in which case changes in larval populations will be compared to the amount of timber harvest. In either case, the objective will be to determine if timber harvest activities have a measurable impact on larval populations. The monitoring reaches within each sub-basin will be sampled at least one year prior to operations that could influence the treatment sites and every year thereafter. New sub-basins will be added across the ownership until there are 12-15 paired sites well distributed across the Plan Area. Depending on the schedule of harvesting in the treatment sub-basins, it will likely be necessary to monitor a site for more than 10 years to determine if a treatment effect has occurred. (Refer to Appendix D for full details of the field protocol.)

A secondary monitoring objective will be to document long-term changes in tailed frog populations across Green Diamond's ownership. Previous studies done within the Plan Area determined that 75% of all surveyed streams (80% excluding geologically unsuitable areas) had tailed frog populations (Diller and Wallace 1999). Given that this occurrence rate is not much lower than the highest reported for the species even in pristine conditions (Corn and Bury 1989; Welsh 1990; Bull and Carter 1996), a secondary objective is to sustain the occupancy of tailed frog populations in streams across the ownership at a minimum of 75% through time. To determine if this objective is being met, the landscape study previously completed (Diller and Wallace 1999) will be repeated at 10-year intervals.

C11.3.3 Thresholds/Triggers

The change in larval tailed frog populations can be used as a trigger to initiate both first and second stage review of management activities. Any significant decrease in the larval populations of treatment streams relative to control streams would initiate a first stage (yellow light) review. A significant decline in treatment streams relative to control streams over a three year period would initiate a second stage (red light) review.

The yellow light thresholds will be:

- any statistically significant decrease in the larval populations of treatment streams relative to control streams, <u>or</u>

- a statistically significant downward trend in both treatment and control streams.

The red light thresholds are:

- a statistically significant decline in larval populations in treatment streams relative to control streams in >50% of the monitored sub-basins in a single year;
- a statistically significant decline in treatment vs. control sites continuing over a three year period within a single sub-basin or;
- a statistically significant downward trend in both treatment and control streams that continues for three years or more.

The change in the occurrence of tailed frog populations across the ownership would not be suitable to use as a trigger to initiate management review due to the extended time-lag between successive data points. However, the occurrence of tailed frogs in streams across the ownership would serve as corroborative evidence to support the findings of the larval population monitoring, and a significant decrease in the occurrence rate would initiate a review of the probable cause of the decline.

C11.3.4 Temporal Scale

If a significant change occurs in the larval populations of treatment streams relative to controls, it will most likely occur during winter high flow events. This change would then be detected during the summer survey season immediately following the winter event. Therefore, the yellow light threshold for adaptive management could be initiated in a single year. The red light threshold would require three years to be initiated.

C11.3.5 Spatial Scale

The spatial scale over which results from an individual monitoring site should apply, (the zone of monitoring influence), will be analyzed on a case-by-case basis. The inherent variability associated with monitoring of a biological indicator necessitates this approach. If a yellow or red light condition is detected, results from all sites across the Plan Area will be examined carefully to determine if the observed population decline(s) appear to be associated with management activity, if they are localized or area wide, and if they appear to be correlated with other factors such as underlying geology or annual climate variation. Field inspection of the problem site(s) will also attempt to identify potential causes of the decline. Because populations in both treatment and control streams could decline for reasons beyond control that may not be related to habitat (e.g. stochastic disease outbreaks), it is essential to examine the results from all monitoring sites to look for patterns in the observed decline. The spatial scale of any resulting adaptive management changes will depend on the particular results. Potential management changes could occur within a HPA, across the Plan Area, or in all areas with similar geology, for example, depending on the nature of the monitoring results.

C11.3.6 Feedback to Management

A decline in tailed frog populations could be caused by a number of factors including elevated water temperatures, change in the algal community due to an increase in insolation or increase in sediment inputs. However, previous research and monitoring of tailed frogs indicated that they were most likely to be impacted by increases in sediment inputs. Given that water temperature, canopy closure, and substrate composition along with the larval populations will be monitored, Green Diamond believes that the likely cause of a future decline will be determined. If for example some future decline is attributed to sediment inputs, the source of the sediment can be determined, and if it is management related, the appropriate adjustments will be made.

C11.3.7 Results to Date

Eight paired sub-basins have already been selected for monitoring tailed frogs including one sub-basin (Poverty Creek) that will serve as a control for two treatment sub-basins (Jiggs and Pollock Creeks). Five were initiated in 1997, one in 1998, two more in 1999 and one additional paired sub-basin was selected in 2000 (Table C11-2).

Table C11- 2. Summary of tailed frog monitoring sites, 1997-2000.[1]

Paired Monitoring Sub-basin	Site	Type	Tailed Frog Larvae			
			1997	1998	1999	2000
Blackdog Creek	BD 5400	C	86	140	183	30
Blackdog Creek	BD 5300	T	25	76	290	99
Upper NF Mad	Canyon	C	88	103	370	98
Upper NF Mad	Mule	T	79	41	83	78
Lower NF Mad	Jiggs	T	127	136	389	106
Lower NF Mad	Pollock	T	148	272	242	159
Lower NF Mad	Poverty	C		53	90	50
Panther Creek	O5	C/h		107	182	36
Panther Creek	O6	T		122	311	58
Rowdy Creek	R1700	C/h			39	40
Rowdy Creek	R1000	T			153	75
NF Maple Creek	F-8	C/h			121	44
NF Maple Creek	F-line	T			65	30
Surpur Creek	West Fork	C/h				190
Surpur Creek	South Fork	T				27
Totals			553	1050	2518	1120

<u>Note</u>
1 "C" indicates a control site with no timber harvest, C/h represents a control site that will have some limited timber harvesting and "T" indicates treatment sites that will have extensive timber harvesting.

C11.3.8 Discussion

Only one treatment monitoring reach (Jiggs in 1998) has had any significant harvesting to date. In spite of this, the results to date indicate that there is considerable annual variation within monitoring stream reaches for both control and treatment streams. It also appears that the different sites were somewhat in synchrony such that there were generally good and bad years for tailed frog reproduction. For example, the mean number of tailed frog larvae captured per year from 1997-2000 for the 6 sites that were monitored over the entire four years was 92.2, 129.7, 259.5 and 95, respectively. There were almost three times as many larvae produced in 1999 compared to both 1997 and 2000. This may be the result of differential annual reproductive effort by the adult population or differences in larval survival among years. Currently, little is known about the adult population in terms of its size or life history characteristics so that it is difficult speculate as to the cause of these annual fluctuations. In spite of the annual fluctuations in the larval populations, the BACI experimental design that was incorporated in this monitoring program will still allow for the detection of treatment effects since the analysis will be based on a treatment by time interaction. However, these fluctuations will increase the variance in the analysis and therefore decrease the statistically power. As a result, Green Diamond intends to implement additional studies of the adult population to determine if the effects of annual variation can be removed from the analysis through the inclusion of one or more additional covariates. Green Diamond currently is experimenting with capturing and marking the adult frogs to determine the feasibility of estimating the size of the adult population. If this proves successful, it would be possible to estimate annual fecundity rates, and subsequently over winter survival rates of the larvae. Having several response variables to monitor would greatly increase the chances of isolating the life history stage that is most sensitive to management activities.

C11.3.9 Conclusion

This study is in its preliminary stages and there has been very little harvesting in any of the treatment sub-basins to date. Therefore, it would be premature to attempt to analyze the data to determine if there were any effects of timber harvest on larval tailed frog populations. However, the data do suggest that there was substantial annual variation in both control and treatment sites, which if not explained through future studies of the adult population, may reduce the statistical power of this monitoring approach.

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Appendix D. Effectiveness Monitoring Protocols

CONTENTS

Figures

Tables

INTRODUCTION

Currently, numerous studies are being implemented by state, federal and private industry biologists to monitor and assess stream conditions with respect to timber harvest and potential impacts to salmonid habitat. State and federal agencies in the Pacific Northwest use a variety of monitoring and assessment protocols. This inconsistency fuels the debate over the value and utility of various methods. This is due in part to the poor understanding of the inherent, regional variability of instream parameters associated with the unique and dynamic characteristics of geology, climate, vegetation and past management histories (Hughes et al. 1986). Study design is another limitation, along with the human and monetary resources to implement long-term studies (Hicks et al. 1991). For example, the temporal lag between hillslope processes (either natural or human induced) and measurable responses in the stream channel is highly variable and may be on the order of decades or longer.

Recently published monitoring and assessment protocols include:

- EPA's "Monitoring Guidelines to Evaluate Effects of Forest Activities on Streams in the Pacific Northwest and Alaska" (MacDonald et al. 1991);
- Ambient Monitoring Program Manual" (Schuett-Hames et al. 1994);
- "Methods for Stream Habitat Surveys" (Moore et al. 1993);
- "California Salmonid Stream Habitat Restoration Manual" (Flosi and Reynolds 1994).

This small sample from the many manuals and protocols available indicates the difficulty in quantifying conditions which reflect the dynamic variables of watershed processes across broad geographic ranges. There is also a wide range in the magnitude and intensity of monitoring proposed, indicating either different sets of objectives or a lack of consensus on how much information is needed to monitor watershed processes and channel conditions.

Green Diamond has carefully considered all of the above approaches in developing an appropriate methodology for its monitoring projects and this AHCP/CCAA (Plan). Green Diamond also is an active participant in the Fish, Forest, and Farms Communities (FFFC) Technical Committee, which provides an ongoing forum where monitoring and assessment protocols are being cooperatively developed and refined by industry, agency and academic biologists. The FFFC Technical Committee has compiled a set of field protocols to standardize data collection for assessing and monitoring salmonid habitat and populations in California. Green Diamond currently utilizes these adopted protocols in their assessment programs.

The Effectiveness Monitoring projects will measure the success of the conservation program in achieving the Plan's biological goals and objectives. Effectiveness monitoring will track trends in the quality and quantity of habitat for the Covered Species as well as the distribution and relative abundance of the Covered Species, and provide information to better understand the relationships between specific aquatic habitat elements and the long-term persistence of the Covered Species. The Effectiveness Monitoring projects are divided into four categories, Rapid Response Monitoring, Response Monitoring, Long-term Trend Monitoring/Research, and Experimental Watersheds Program. The first three categories are based on the minimum time frame over which feedback for adaptive management is likely to occur. The Experimental Watersheds Program provides a unique spatial scale for individual projects and for the development of new and refined monitoring approaches.

Each Effectiveness Monitoring project is based on current monitoring technology and methodologies and on current understanding of the limiting habitat conditions required by the Covered Species, i.e. LWD, sediment, and water temperature. It is reasonable to expect that monitoring techniques and related technology will change significantly through the fifty-year life of this Plan, and that our understanding of riparian function will also change. Therefore, it is essential to build flexibility into the monitoring program to respond to these changes. Some monitoring programs may be retired or replaced by more efficient and/or accurate techniques to address the same issues, and entirely new monitoring programs may be implemented to address currently unforeseen issues. Any changes to the monitoring program considered will be evaluated to insure that they do not reduce the ability of the program to achieve its objectives: to evaluate the effectiveness of the conservation measures and provide feedback for adaptive management. Periodic reviews, at least every ten years or following changed circumstances, of the monitoring and adaptive management program will provide the assessment needed to justify any changes. All changes to the monitoring program will be subject to the concurrence of the Services.

D.1 RAPID RESPONSE MONITORING

D.1.1 Introduction

Rapid Response Monitoring activities include:

- Summer water temperature monitoring
  -- Property-wide water temperature monitoring
  -- Class II (BACI) water temperature monitoring

- Spawning substrate permeability monitoring

- Road-related sediment delivery (turbidity) monitoring

- Headwater monitoring
  -- Tailed frog monitoring
  -- Southern torrent salamander monitoring

The Rapid Response Monitoring projects will provide the early warning signals necessary to ensure that the biological goals and objectives of the Plan will be met. Rapid Response Monitoring projects have the potential to provide feedback to adaptive management on a time scale of months up to two years. Each project has measurable thresholds which, when exceeded, initiate a series of steps for identifying appropriate management responses. To provide the ability to respond rapidly to early signs of potential problems while providing assurances that negative monitoring results will be adequately addressed, a two stage “yellow light, red light” process will be employed.

D.1.2 Property-wide Water Temperature Monitoring

D.1.2.1 Background and Objectives

Stream water temperature monitoring on Green Diamond Timberlands began in 1994 and is ongoing today. At the end of the year 2000, 400 summer water temperature profiles have been recorded at 150 locations within 108 Class I watercourses distributed throughout the Plan Area. An additional 210 summer temperature profiles have been recorded in 87 sites in approximately 70 headwater (Class II) watercourses within the Plan Area. As part of Rapid Response Monitoring, water temperature will be monitored on an annual basis within both Class I and II watercourses throughout the Plan Area.

The following objectives have been developed for water temperature monitoring:

- Document the highest
    - a) 7DMAVG (highest 7-day moving average of all recorded temperatures),
    - b) 7DMMX (highest 7-day moving average of the maximum daily temperatures),
    - c) seasonal temperature fluctuations for each site for both Class I and Class II watercourses.
- Identify stream reaches with temperatures that have the potential to exceed the monitoring thresholds relative to the drainage area above the monitoring site for both Class I and Class II watercourses.

D.1.2.2 Class I and II Watercourse Monitoring Methods

D.1.2.2.1 Calibration and Recorder Replacement

The annual calibration of all thermographs is necessary to remain assured that all recorders (loggers) are operating within the manufacturer’s specifications and that batteries are in good condition. The calibration process is not an attempt at documenting precision beyond that of the manufacturer’s specifications or an attempt at establishing correction factors to be applied the data after retrieval. The manufacturer’s specification for the current models, Onset’s Hobo® or TidbiT®, is ± 0.2 ºC at -5ºC to +37ºC. Any recorder that fails the first calibration will be repaired and recalibrated. If a second calibration failure occurs, the thermograph will be retired and replaced. Technological advances and replacement intervals for temperature loggers will ensure that recorders used for the monitoring program will not be more than five years old. The TidbiT®’s battery is not replaceable and the unit is only expected to last about five years before being replaced. The unit records data with very little draw on the battery but the download process, through a Light Emitting Diode (L.E.D.), is very demanding on the battery. Therefore, it is recommended that units only be calibrated once a year prior to deployment and that deployments run as long as reasonable to avoid frequent downloading. Green Diamond still maintains a few HOBO® models that are used primarily for high profile sites, where the recorder may be stolen, or in streams that have already been documented as being well within suggested temperature thresholds.

At the beginning of each field-monitoring season every logger, that is not currently deployed, is subject to calibration in an ice bath using the following procedure.

1. Set and start each of the loggers at a recording interval of 10 seconds and an appropriate delayed start time.

2. Obtain an ice chest or large garbage can capable of holding all of the recorders to be calibrated at once.

3. Fill the container half way with crushed ice.

4. Place the recorders in a single layer on a plastic tray or screen and place on top of the ice.

5. Finish filling the container with ice and then fill ¾ of the container with cold water. If available, place the container in a walk-in cooler or at the minimum insolate it with blankets and place in a shaded area.

6. A small water pump (i.e., a fishtank pump) should be set at the bottom of the container to circulate the water and prevent any measurable thermal gradation developing in the container.

7. Using an ASTM certified lab thermometer, verify the water temperature at periodic intervals. The water will be at or slightly less than 0^{o}C, depending on the purity of the water.

8. Continue to monitor for two to three hours allowing time for acclimation of both the recorders and the water.

9. Remove and download all of the recorders.

The available software for processing the thermograph data (Boxcar Pro 4.0©) does not allow for direct comparison of the data sets, therefore while downloading the thermographs each file should be exported as a text file (.txt extension) in an Excel© compatible format. Each text file will contain two columns: the date/time code in an Excel© format and the corresponding temperature data. Select a common one-half hour period in which the water temperature was stabilized at or near 0^{O}C. For each individual thermograph calculate the average temperature recorded during the calibration run period and the standard deviation around those temperatures. Thermographs with identical average temperatures and deviations are matched up and used in paired watershed studies (see BACI Protocol for Class II monitoring below). Those recorders that operate within the manufacturer's specifications are assigned to Class I and II monitoring sites and those that do not pass are recalibrated or retired and replaced.

A thermograph-tracking document will be maintained that documents each recorder's historical placement, calibration and maintenance history, deployment problems, and retirement date. When a logger is deployed the following will be documented in the tracking file: the stream, sampling interval, launch date and recovery. A record of all logger serial numbers, purchase dates, battery replacement dates, and battery life will be kept in a master temperature monitoring equipment file as part of the documentation.

D.1.2.2.2 *Stream Selection*

The streams and/or stream segments selected for water temperature monitoring will represent a variety of monitoring goals. Any particular monitoring site may serve multiple goals.

Green Diamond annually monitors:

- Individual streams with exceptionally diverse species composition or significant populations of torrent salamanders, tailed frogs or coho salmon.
- Individual streams that have been documented as having water temperatures potentially problematic for salmonids and amphibians.
- Stream segments (within those streams that have been documented as having elevated temperatures), to document the extent of the elevated temperatures.

Green Diamond will also periodically monitor:

- Streams and stream segments that have been documented as having no temperature problems. These streams are selectively monitored on a two-year schedule. This will provide a long-term database that allows for trend analysis.
- Streams and stream segments for which there are no temperature profiles in existence.

At a minimum, all 3rd-4th order Class I sub-basins (typically 3000-5000 acres) with >2500 feet of fish-bearing channel and >10% of the sub-basin harvested (average >1%/year using even age silviculture) over any rolling 10-year interval will have at least one monitoring site low in the sub-basin where summer water temperatures will be monitored on an annual basis. The monitoring may be discontinued after five years, if the highest 7DMAVG (7-day moving average of all recorded temperatures) for the stream falls below the trend line (least squares regression line) of 7DMAVG versus drainage area (see Summer Water Temperature Monitoring, Section 6.3.5) for all sub-basins in that particular HPA, and there is <5% additional harvest during that time interval. If at some future time the rate of harvest exceeds an average of 1%/year over a rolling 10-year interval, the monitoring will be re-initiated. In addition to the minimum described above, 10-15 streams from across the Plan Area that do not meet any of the criteria described above and were previously found to be below the temperature thresholds will be monitored on a three to five-year rotating basis to document general trends in water temperature throughout the Plan Area.

There are some previously established monitoring sites on Class I watercourses that have watershed areas greater than 10,000 acres. These monitoring sites will no longer be used since the scope of inference for the threshold equations is less than 10,000 acres. A new site will be established further upstream so the watershed size criteria will be met for the water temperature monitoring.

Water temperature monitoring of Class II watercourses will be distributed across the Plan Area as part of the Headwaters Amphibian Monitoring, Class II BACI Water Temperature Monitoring (see below) and other amphibian studies. In addition, if the highest 7DMAVG associated with a given 3rd-4th order Class I is at or above the yellow-light threshold level (see Summer Water Temperature Monitoring, Section 6.3.5), then a temperature profile for the mainstem and all the major Class II tributaries in the sub-basin will be determined at the warmest time of the year. Temperature loggers will be deployed in 2-3 of the warmest Class II watercourses to determine if they are within the threshold limits. Wherever possible, Class II watercourses in these sub-basins will be targeted for BACI water temperature monitoring sites.

D.1.2.2.3 *Temperature Monitoring Site Selection*

Within the stream or stream segment selected, the specific site for monitoring will be in the lowest portion of the stream on Green Diamond Property. Care will be taken to avoid tributary confluence's that may bias the temperature data. The temperature recorder will be either anchored to a length of steel rebar driven into the channel bed or secured to a cement block with cord or cable. In order to avoid any effects of thermal stratification within a Class I watercourse habitat unit, recorders shall be placed either in a deep well-mixed riffle or at the head of a pool in 1 - 2 feet of water. For Class II watercourses thermal stratification is generally not considered an issue, rather the goal would be to place the recorder in water deep enough that the unit will not be de-watered during summer low flow conditions. The intent is to monitor representative temperatures for the stream and avoid monitoring specific thermal refugia. In all cases each recorder shall be launched and deployed at a recording interval of no greater than 1.2 hours. This interval provides 20 recordings per 24-hour period. Recent upgrades in the memory capacity of the TidbiT® make it feasible to record at much shorter intervals but the increase in data volume does not add to the data quality. In addition the increased sampling interval requires more memory and thus longer to transfer the resulting data file. The file transfer operation is the most demanding on the logger's battery and can significantly reduce the life span of the recorder

Green Diamond's summer stream temperature monitoring activities are focused on documenting seasonal peak temperatures that can occur anytime from early June to late September. To document seasonal peaks in water temperature the recorders are deployed early in the year and left unattended until October or November. In a majority of the streams monitored, summer low flow conditions result in a dramatic lowering of the water surface elevation of what was a shallow pool or riffle during the spring. Therefore, care shall be taken in placing the recorder so that it does not become exposed to the air or to unrepresentative water conditions while deployed. Generally, the temperature recorder is placed in the stream with cobbles placed around it to help anchor and shield the recorder from direct solar radiation.

D.1.2.2.4 *Collection of Site Specific Variables*

Several variables will be collected and will contribute to a better understanding of the temperature data collected by the thermograph. These site-specific variables will be collected either while deploying the thermograph or upon its retrieval.

- Channel type using CDFG protocols (this will include bankfull width and depth measurements)

- Canopy Closure using CDFG protocols
- Water depth and discharge during placement and retrieval

These additional variables will be generated from GIS analysis and/or aerial photos:

- Site elevation
- Stream aspect
- Watershed area upstream of the thermograph
- Stand age

D.1.2.2.5 *Data Analysis*

The temperature monitoring data collected is intended to document the summer water temperature maxima. Several metrics shall be calculated from the data set in addition to the absolute maximum temperature. These metrics further describe the water temperature conditions during the summer period and the diurnal fluctuations immediately following the warm summer temperature conditions. The Seven-Day Moving Average (7DMAVG) is the seven-day period with the highest average temperature. The Seven-Day Mean of the Maximums (7DMMX) is the highest seven-day moving mean of the maximum daily temperatures. The absolute Maximum temperature (Max) may or may not occur during the 7DMAVG or the 7DMMX. The minimum temperature (Min) following the absolute maximum (Min. after Max.) is the minimum temperature on the day following the occurrence of the Max. This is intended to describe the diurnal range on the hottest day of the year. The raw temperature data is imported into Microsoft Excel to calculate every seven day moving average and every seven day moving mean of the maximum temperatures. The highest seven day moving average temperature (7DMAVG) and the seven day moving mean of the maximum temperature (7DMMX) is selected and the associated middle dates (Mid Date 7DMAVG and Mid Date 7DMMX) from both seven day period. The absolute maximum (MAX) is then selected along with the Min. after Max and the date of the maximum. This data is then entered into a spreadsheet along with the period of record, year, site name and number. A master list of all thermograph data processed is compiled and updated annually. Subsets of this data are submitted with Timber Harvest Plans to document water temperature conditions within the assessment area of that plan as shown in Table D-1 below. All new temperature summaries are analyzed in reference to the red and yellow light thresholds.

Table D-1. Example of temperature monitoring data set: summer water temperature monitoring summary for Little River HPA.

Stream Name	Class	Year	7DMAVG (°C)	Mid Date 7DMAVG	7DMMX (°C	Mid Date 7DMMX	Max (°C)	Max Date	Min after Max (°C)	Area (acres)
Little River, Upper SF	1	1994	14.5	8/19	15.9	8/16	16.2	8/3	14.0	3619.0
Little River, Upper SF	1	1995	14.7	8/3	16.5	8/3	17.0	7/31	13.7	3619.0
Little River, Upper SF	1	1998	15.0	8/14	16.5	7/20	16.8	7/18	13.7	3619.0
Little River, Upper SF	1	1999	14.8	8/27	15.2	8/27	15.6	8/29	14.5	3619.0
Little River, Upper SF	1	2000	15.3	7/31	16.5	7/31	16.8	8/1	14.6	3619.0
Little River, Lower SF	1	1994	14.6	7/24	16.3	8/5	16.9	8/3	14.5	3452.0
Little River, Lower SF	1	1995	15.2	7/30	16.7	8/3	17.2	8/1	14.0	3452.0
Little River, Lower SF	1	1998	15.9	7/23	17.4	7/23	18.1	7/26	15.2	3452.0
Little River, Lower SF	1	1999	15.6	8/27	16.5	8/23	17.2	8/22	14.5	3452.0
Little River, Lower SF	1	2000	16.1	7/31	18.0	7/31	18.5	8/1	15.2	3452.0
Little River (mid)	1	1994	15.2	7/30	16.4	7/29	16.9	7/31	14.4	13176.3
Little River (mid)	1	1996	16.0	7/28	17.5	7/28	17.9	7/29	14.8	13176.3
Little River (mid)	1	1999	15.5	8/27	16.2	8/27	16.6	8/29	15.3	13176.3
Little River (mid)	1	2000	15.8	7/31	17.0	7/31	17.4	8/1	15.0	13176.3
Little River (upper)	1	1994	13.4	8/21	14.2	8/21	14.5	8/19	13.3	8755.0
Little River (upper)	1	1995	14.0	8/3	15.2	8/3	15.8	7/31	13.3	8755.0
Little River (upper)	1	1996	14.1	7/28	15.3	7/27	15.8	7/30	12.6	8755.0
Little River (upper)	1	1999	14.1	8/27	14.7	8/27	15.3	8/29	13.1	8755.0
Little River (upper)	1	2000	14.3	9/18	15.1	9/18	16.1	9/19	13.9	8755.0

D.1.3 Class II BACI Water Temperature Monitoring

D.1.3.1 Background and Objectives

In summer 1996, Green Diamond initiated water temperature monitoring in nonfish bearing (Class II) watercourses to assess potential impacts of harvesting and adequacy of the riparian buffers. The goal of this effort was to examine changes in stream temperature after timber harvest by comparing maximum temperature differentials across fixed lengths of stream. These temperature differentials were measured on pairs of similar streams, one member of which ran through a harvest unit, the other of which was undisturbed. Measurements were initiated in both streams of a pair prior to harvesting timber surrounding one member of the pair. Monitoring of the stream pair will continue until the stream pair returns to pretreatment conditions. These data represent a BACI (Green 1979; Stewart-Oaten et al. 1986; Skalski and Robson 1992) observational study. While observational studies cannot infer cause and effect relationships, BACI studies represent the best available setup for detecting changes after disturbance. In 1999, three additional watersheds were added to the Paired Watershed (BACI) experimental design. Future paired watersheds may be added as needed to meet the Plan's Class II water temperature monitoring needs. New Class II BACI water temperature sites will be established across the Plan Area as opportunities exist. (New BACI sites cannot be initiated unless there is going to be harvesting in the area to create the treatment reach.) The goal is to have a minimum of 12-15 paired sites that are well distributed across the Plan Area to represent different physiographic regions. If there is little variance among sites in the response of water temperature to the treatment effect, this minimum number will be adequate to reach a definitive conclusion on the impact of harvesting on Class II water temperature. However, if there is substantial variation in the treatment response, it will be necessary to add additional sites. The actual maximum number is a statistical question that cannot be answered until the data are collected and analyzed.

D.1.3.2 Methods for Class II BACI Studies

D.1.3.2.1 <u>Calibration and Recording Interval</u>

Temperature recording devices were/will be calibrated prior to deployment. For calibration, all thermographs will be calibrated as described above in the Class I summer water temperature monitoring program. Only instruments with identical readings after three hours in the calibration ice bath will be used for the BACI experiments. All thermographs will be programmed to record temperature (ºC) every 1.2 hours or 20 times every 24 hours.

D.1.3.2.2 <u>Site Selection and Deployment</u>

Streams in areas where timber harvest is planned were, or will be, identified and paired with separate streams in close proximity that has similar size, streamflow, aspect, elevation, stand type and age and streambed geology. The stream of each pair running through a harvested area is designated as the “treatment” stream. The other stream of each pair was/will be designated as the “control” stream because no timber harvest is planned around these streams. At least one year prior to timber harvest, paired temperature-recording devices (HOBO’s® or TitBiTs®) will be placed in the treatment stream at the upstream and downstream edges of the harvest unit. At the same time, another pair of temperature recording devices was/will be placed in the control stream at locations which are the same (stream) distance apart as the recording locations in the treatment stream.

The upstream and downstream placement of temperature recording devices allow measurement of temperature differential across the treatment area and an assessment of the extent to which water temperature changed as it flowed through the treatment area. Interest is primarily in the amount of warming water experiences as it flows through the treatment area. Ground water inputs, climate, and microclimatic factors can all effect water temperature and consequently the paired stream design was adopted.

For all watershed BACI sites paired thermographs will be deployed to all streams in middle and late spring each year and collected after 15 September each year.

D.1.3.2.3 <u>Watershed and Stream Selection</u>

In the original monitoring program, data were recorded on five pairs of streams with each pair referred to as a site. As stated above three additional sites were added in 1999. Each stream pair (site) will be given a unique site name. The original five study sites were labeled Mitsui, D2010, D1120, 6001, and 5410. Mitsui was located in the headwaters of the Little River. D2010 was located in the Winchuck drainage. D1120 was located in the headwater tributaries of Dominie Creek. 6001 was located off the main stem of the Mad River. Site, 5410, was a pair of tributaries to Dry Creek. Timber harvest at Mitsui and D2010 took place in winter 1996/1997. Timber harvest at 6001 and 5410 took place in winter 1997/1998. As of winter 1999/2000, timber harvest had not occurred at D1120.

The sites added in 1999 are Windy Point, M1, and M155. Windy Point and the M1 are in tributaries to Maple Creek and the M155 is in a pair of tributaries to the Lower South Fork Little River. The Maple Creek units where harvested in winter 1999/2000 and the Lower South Fork unit has not been harvested yet.

D.1.3.2.4 Collection of Site Specific Variables

Several variables will be collected and will contribute to a better understanding of the temperature data collected by the thermograph. These site-specific variables will be collected either while deploying the thermograph or upon its retrieval.

- Canopy Closure
- Stream flow
- Water depth during placement and retrieval

These additional variables will be generated from GIS analysis and/or aerial photos:

- Watershed area upstream of the thermograph
- Site elevation
- Stream aspect
- Stand age

D.1.3.3 Data Analysis

For analysis, attention will be restricted to the time during the warmest water temperatures, which are generally late August to early September in coastal northern California.

Upstream and downstream temperatures collected on a single stream will be matched according to the time of day they were recorded and the difference between downstream and upstream temperature (downstream - upstream) will be calculated every 1.2 hours. The maximum downstream-upstream temperature differential will be computed each day. The time of day at which the maximum temperature differential was recorded will likely vary between days and streams.

The statistical analysis used to assess harvest impacts was/will be a modified BACI analysis. BACI analyses assess the lack of parallelness in response profiles through time. This lack of parallelness was/is measured by the treatment by time (year) interaction from an ANOVA with time as one factor and treatment as the other. The BACI analysis allows the level of responses to be different between control and treated sites both before and after treatment, but requires the after treatment difference in control and treated responses to be the same as the before treatment difference in control and treated responses. If the after treatment difference in responses is different from the before treatment difference in responses, the BACI analysis will conclude that there was significant change in treatment areas after application. Inference as to the cause of treatment differences is as a result of professional judgment based on a preponderance of evidence.

Differences between sites in the direction and magnitude of temperature differences after harvest can become apparent upon plotting of the data. In the face of these differences, each site was/will be analyzed separately and no statistical inference to other sites is possible. Discussion of other sites should be considered professional judgment and not directly based on inference from the data.

Details of the BACI estimation process can be found in McDonald (2000) (Attachment A to Appendix C3. The modification of standard BACI methods used here involves adjusting error estimates to account for estimated auto-correlations in the inter-day time series inherent in the data.

D.1.3.4 Literature Cited

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D.1.4 Spawning Substrate Permeability Monitoring

D.1.4.1 Background

Spawning gravel permeability will be monitored in selected Class I watercourses throughout the Plan Area to determine if conditions are currently suitable for the covered fish species and to track trends in permeability. Sedimentation can reduce the survival to emergence of the covered embryos by reducing subsurface flow (Reiser and White 1988). Permeability monitoring is a way to measure subsurface flow, and permeability has been correlated with survival to emergence of salmonids. Field measurements in streams across the Plan Area will be combined with the available literature and field data from additional streams, including pristine portions of the Prairie Creek watershed, to determine appropriate threshold and biological objective values. Approximately five years of initial trend monitoring is expected to be necessary for this process.

D.1.4.2 Threshold Development

Approximately five years of initial trend monitoring is expected to be necessary to determine appropriate permeability threshold values. At the end of the trend-monitoring interval a review and evaluation of the monitoring results will be conducted to set thresholds with agency collaboration. In addition, at other times agreed upon with the consensus of the Services, periodic reviews will be conducted to evaluate progress in determining substrate permeability thresholds. Concurrently with the initial trend monitoring efforts a literature re-evaluation and assessment will be conducted to assist in establishing threshold values for the protection of life-stages of anadromous salmonid sensitive to the effects of reductions in substrate flow and oxygen concentrations.

D.1.4.3 Monitoring Methods

D.1.4.3.1 Introduction and Permeability Theory

The condition of salmonid spawning habitat can be a factor limiting the success of salmon and steelhead populations. Assessing the quantity and quality of salmonid spawning habitat requires both field methods and analytical methods to first quantify the productive capacity of spawning gravels, then compare conditions in watersheds with different land use histories and remnant salmonid populations, and finally assess temporal changes in these factors. Suitable methods for spawning gravel assessment should allow quantitative prediction of egg survival to emergence (i.e., incubation success) with known accuracy and at reasonable expense to allow widespread application.

To date, the best methods available that partially meet these criteria are from Tappel and Bjornn (1983), who related survival-to-emergence of chinook salmon (*Oncorhynchus tshawytscha*) and steelhead (*Oncorhynchus mykiss*) eggs to two indices of the particle size distribution: the cumulative percentage of substrate finer than 9.5 mm and 0.85 mm. Their laboratory experiments provided regression equations allowing prediction of survival-to emergence with the cumulative percentage of particle size fraction as input variables. The Tappel and Bjornn (1983) method has proven extremely useful in the past decades primarily because it links a measurable physical condition of the watershed, i.e., the amount of fine sediment in spawning gravels, to a biological effect, the percentage of salmonid eggs that survive to emerge as fry (survival-to-emergence), in a cause and effect relationship. A significant weakness of this method is the enormous effort required to collect enough sediment samples to accurately assess variability in the cumulative percentage of fine sediment that regularly occurs at the reach--, tributary--, or watershed--scale. For example, if sediment samples collected for a particular site have a narrow range of particle size distributions (i.e., low variance), then predictions of survival-to-emergence can be useful. If variance is high, however, as is often found in impacted watersheds, the regression equations predict a broad range of survival, and the utility of the method is compromised.

Permeability may provide a better method of assessing the condition of spawning gravels for several reasons. First, salmonid egg incubation depends on the supply of oxygen delivered to incubating eggs, and removal of waste from the egg pocket. The rate of oxygen delivery and waste removal is determined in part by the permeability of the gravels surrounding the egg pocket. Permeability is thus a more direct measure of factors affecting egg incubation and survival. Second, as discussed below, permeability data are more easily obtained than particle size distribution; thus characterizing the range of variability with suitable accuracy requires less cost and effort than methods based on substrate composition analysis. Finally, permeability is independent of discharge, stage height, season, etc, and can therefore be measured accurately at any time.

The measure of permeability of spawning gravel has a relatively short history. Terhune (1958) recognized that to estimate the probability of survival (to emergence) of salmonid eggs, two quantities must be known: "the concentration of dissolved oxygen in the groundwater, and the apparent velocity of the water through the gravel in the immediate vicinity of the redd [egg pocket]." Apparent velocity is the rate of seepage, expressed as a volume of liquid per unit time passing a cross sectional area containing both solids and interstices. Apparent velocity of water flowing through gravel interstices depends, in turn, on two factors: the hydraulic head and gravel permeability (Pollard 1955). Hydraulic head in a spawning riffle is determined by the hydraulic gradient, which is the slope of the water surface ($S=\Delta h/L$). Because hydraulic head changes with discharge (via change in slope), apparent velocity also changes with discharge. Apparent velocity (V) is also difficult to measure. Pollard also showed that, for laminar flows occurring at the velocities usually encountered in spawning gravels, D'Arcy's coefficient of permeability, K, as defined by $K=V/S$, is independent of apparent velocity, V. Permeability depends only on the composition and degree of packing of the gravel, and viscosity of the water (viscosity is related to water temperature). In the equation $K=V/S$, slope is dimensionless, so permeability will have the same dimensions as apparent velocity (usually cm/hr). Terhune (1958) therefore suggested permeability as a surrogate measure to apparent velocity as an empirical measure of the quality of salmonid spawning gravels:

"The permeability of the gravel, the ease with which water can pass through it, may be used as a figure of merit for the gravel—the higher the permeability the greater the supply of oxygenated water that can reach the salmon eggs for a given river gradient." (Terhune 1958).

Determining the permeability of spawning gravels by mechanical analysis is not practical because it is impossible to evaluate the degree of packing of the streambed substrates *in situ* (Pollard 1955). The standpipe was thus developed as a way to measure permeability in the field (Pollard 1955). Several iterations and modifications to standpipe techniques resulted in the “Mark VI Groundwater Standpipe” (Terhune 1958). Terhune recalibrated the standpipe by constructing a permeameter (14-ft long flume) and performing multiple trials to relate the rate of water inflow into the standpipe to the permeability, as measured by the permeameter. Barnard and McBain (1994) performed additional calibration with their own permeameter and standpipe. The permeability calibration curve is shown in Figure D1-1. Techniques for measuring permeability will be discussed below, following an additional word about permeability theory.

As mentioned, past research has relied primarily on measuring the volume of fine sediment in gravels to assess the quality of spawning gravels. Intrusion of fine sediment into gravel reduces the intra-gravel flow of water by reducing permeability, which results in reduced rates of oxygen delivery to incubating embryos and removal of metabolic waste from the egg pocket. The volume of fine sediment in spawning substrates is thus an indirect measure of gravel conditions that affect survival to emergence, whereas permeability directly measures conditions affecting embryonic survival. Chapman’s (1988) review of the effects of fine sediments on the survival to alevin emergence noted that survival relates positively to both temperature and apparent velocity, and that survival also relates positively, and significantly, to permeability: for McCuddin (1977) data, r^2=0.83; for Koski (1966) data, r^2=0.33. Data from McCuddin (1977) and Tagart (1976) were plotted together (Figure D1-2), and show a significant correlation between permeability and survival-to-emergence. While plotting these data together shows a strong relationship exists between permeability and survival-to-emergence, this regression should be considered preliminary and used with caution, as the data are from studies involving two different salmonid species using different data collection methods. Additional studies are warranted to confirm/strengthen this important link. Despite this information, few researchers or resource managers have employed permeability techniques to assess salmonid spawning gravel quality.

Until recently, permeability measurement relied on Terhune’s (1958) methods, which employed a hand pump (a bicycle or bilge pump) to extract water from a 4.5 cm stainless-steel standpipe into a 2.0 L graduated cylinder. The quantity of water withdrawn into the cylinder and the corresponding time interval were used to calculate the “inflow rate” of water into the standpipe from the surrounding substrate. A correction factor was necessary to account for the 2.54 cm pressure head at the top of the standpipe, and the operator was required to pump vigorously and consistently for up to several minutes in low permeability conditions. Young (1988) demonstrated significant imprecision in this technique. He found significant differences in permeability samples withdrawn by different individuals (sampling bias), resulting in substantial variability in permeability estimates. Young also pointed out that previous research relied on only one replicate per sample to estimate permeability, when variation in permeability may be expected at a particular sample location.

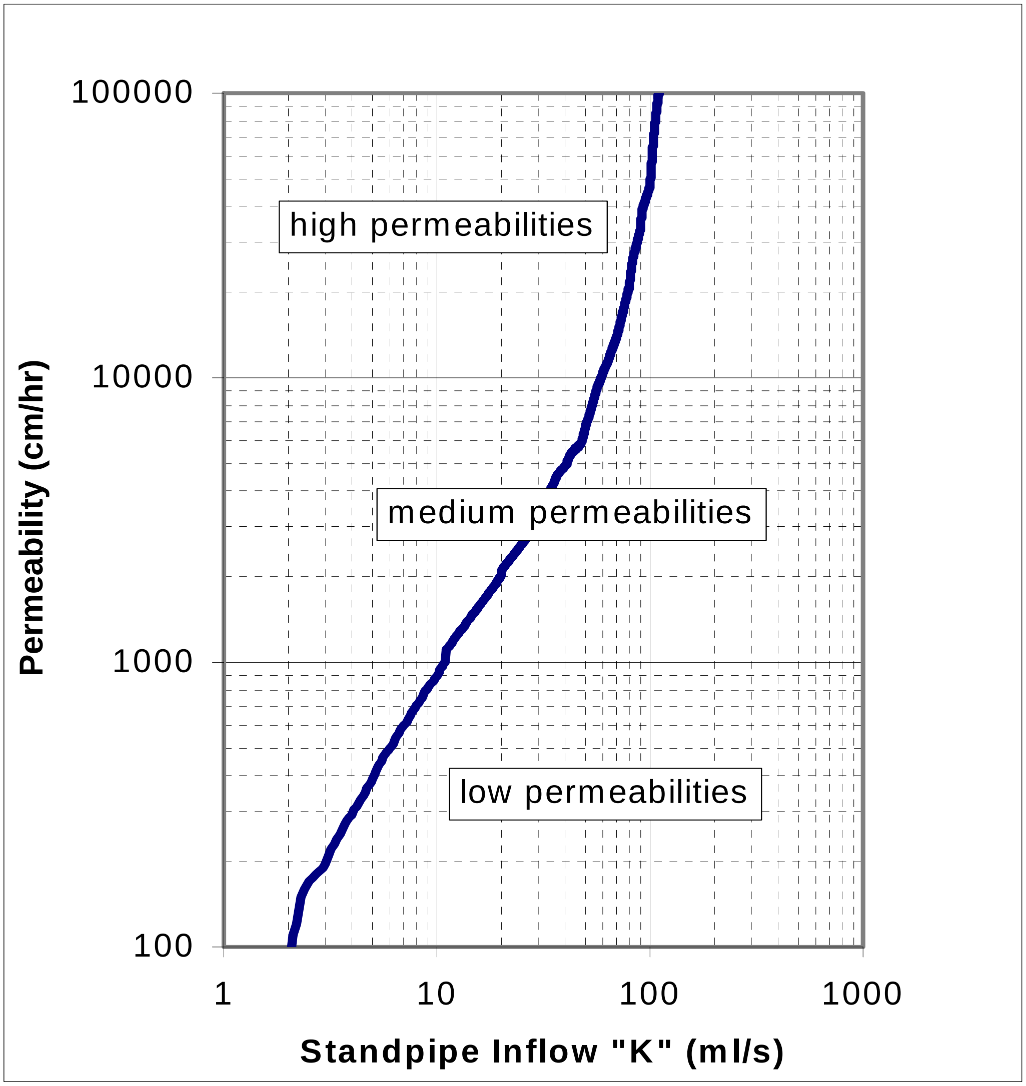

Figure D-1. **Relationship between inflow rate (ml/s) and permeability (cm/hr) used to convert field inflow measurements into permeability. Note that permeability ranges across three orders of magnitude, from 0 cm/hr to 100,000 cm/hr.**

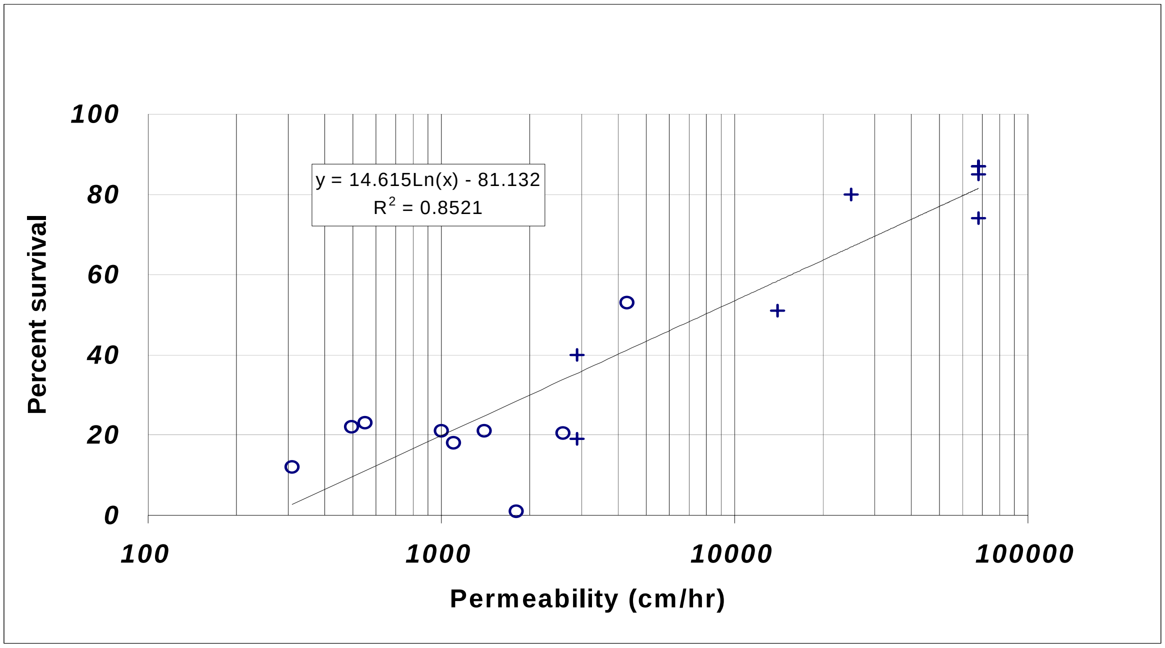

Figure D-2. Data from Tagart (1976) and McCuddin (1977) showing a highly significant relationship between survival of chinook (McCuddin data, "+") and coho salmon (Tagart data, "o"), and permeability of the incubation substrate [Figure provided by Stillwater Sciences].

D.1.4.3.2 Equipment, Operation, and Maintenance

To improve the accuracy of permeability measurements and eliminate potential user bias, several researchers have begun using a hand-made electric pump device to draw water from the standpipe into a volume chamber calibrated to measure the volume of water inflow per unit time (i.e., inflow rate). The device is mounted on a backpack frame for convenient use in the field. This new device allows consistent, replicate sampling in a short time (approximately 20-100 seconds per replicate), from which a mean permeability and variance can be computed for a single sample location. Collection of 5-10 replicates for each sample, and several samples (at least three) within a single spawning area or sediment facies can be completed in approximately one hour. Use of this new device in several independent studies and monitoring programs (Mendocino Redwood Company, B. Klatte 1998 HSU Master's Thesis, McBain and Trush 2000, Mesick 2000, Lower Tuolumne River Spawning Gravel Assessment, in progress,) has shown consistent and reproducible permeability measurements.

A detailed description of the equipment is provided here. The electric pump, a Thomas Inc. diaphragm vacuum pump (model 107CDC20), is mounted inside a box and connected to a 12-volt deep-cycle battery (e.g. Interstate® Battery model PC1270/ 7.0 AH). A toggle switch is connected in the circuit and mounted through the side of the toolbox so the switch can be turned on/off externally. A ⅜ inch diameter plastic hose connects from the pump to a plastic overflow bottle, then through the box to the vacuum chamber cylinder. The vacuum chamber is constructed of 3½ inch diameter clear PVC pipe (from Ryan Herco, Inc.), measures 20 inches in length, and has 3½ inch x 1½ inch PVC bushings on each end. The top bushing has a threaded plug, which allows easy access into the chamber to rinse out silt and sand that accumulates. The bottom bushing has a ¾ inch threaded nipple and a ¾ inch brass ball valve, which allows water to be drained off after each replicate measurement. A piece of ⅜ inch clear, rigid plastic tubing is installed to the vacuum chamber side to facilitate reading the meniscus (stage height) as accurately as possible. A ruler calibrated in millimeters/centimeters is attached next to the rigid tubing, and is used to read the stage height of water inside the vacuum chamber. A second piece of ⅜ inch plastic tubing, 6 ft long, is connected to the vacuum chamber and leads to a 5 ft piece of ⅜ inch copper tubing (stainless steel also works). This rigid copper tubing is inserted down into the standpipe and contacts the water surface. When the switch is turned on, the pump draws water into the vacuum chamber via the copper and plastic tubing.

The standpipe is constructed of one inch interior diameter Schedule-40 stainless steel pipe, approximately 4½ ft in length, open at the top and with a driving tip welded into the bottom (Figure 5, drawing of standpipe equipment). The heavy-duty stainless steel is used because it is durable and will not corrode. This one inch standpipe is smaller in diameter than the original Terhune (1955) model (1¼ inch), which slightly reduces disturbance to the gravel. A 3 inch band of perforations is located several inches from the bottom, and includes forty-eight ⅛ inch diameter holes drilled through the pipe to allow water to flow into the standpipe. Vertical grooves are cut into the pipe connecting the holes, to prevent small substrate materials from plugging the holes. To drive the standpipe into the streambed, a sledgehammer and a driving head of solid stainless steel or lead, machined to fit into the top of the standpipe, is used. The driving head protects the rim of the standpipe from becoming damaged by the hammer. Place duct tape 10-12 inches up from the middle of the band of perforations to indicate the depth to which the pipe should be driven into the substrate.

D.1.4.3.3 <u>General Field Methods</u>

A permeability measurement is made by pumping water from the standpipe into the vacuum chamber, and measuring the change in stage height in the vacuum chamber (mm) per unit time (sec). The only field data required for a permeability measurement are therefore the start and ending stage, time, and water temperature. Record detailed information about the site location, extent of spawning habitat and evidence of spawning usage, photograph, etc. A sketch map of the site is also useful. The stage height change (mm) is later converted to volume (ml) using a Microsoft Excel© spreadsheet, then the inflow rate (ml/sec) is converted to permeability (cm/hr) using the calibration curve developed by Terhune (1958) and refined by Barnard and McBain (1994). The permeability is also adjusted for water viscosity in the spreadsheet by a conversion factor using water temperature.

To initiate a measurement at a selected site, the standpipe is first driven into the substrate to the appropriate depth, using the driving head and sledgehammer. Once the pipe is in place, the driving head is removed and the copper tube (connected to the pump) is inserted into the standpipe. To locate the exact stage height of the groundwater inside the standpipe, perform a "slurp test", in which the pump is turned on and the copper tube is slowly lowered until the copper tip contacts the water and makes a slurping-straw sound. Then insert a one-inch spacer on the rim of the standpipe, and clamp needle-nose vise-grips on the copper tube precisely above the spacer. In this way, when the spacer is removed and the vise-grips rest on top of the standpipe rim, the tip of the copper tube gets lowered exactly one inch deep below the water surface elevation inside the standpipe. Pumping this one-inch fraction of water out of the pipe creates a pressure head outside the pipe, thus causing water to flow continuously into the standpipe through the perforations. The rate of inflow into the pipe is determined by the permeability of the surrounding gravels. The original calibration of inflow rate to permeability by Terhune (1955) employed the one-inch pressure head, and is essential to proper permeability measurements.

When the slurp test and vice-grips procedure is complete, the first permeability replicate sample can be taken. The pump is turned on to fill the volume chamber with water from outside the standpipe to the level of the bottom of the ruler. The copper tube can then be replaced into the standpipe to begin pumping water from the standpipe. Allow a few seconds to draw out the first one-inch volume of water to create the pressure head and stabilize the rate of water pumping, then record the initial stage and start the timer. Generally, allow at least 20 seconds and/or a change in stage of approximately 10 cm for each replicate. After the end stage height is noted and the timer stopped, turn off the pump and record the data. Drain the water back to the level of the bottom of the ruler, and begin the next replicate measurement. In the field, if the stage change is the same for each replicate measurement, then the time (seconds) is a surrogate for the actual permeability, and replicates can be compared to each other. Green Diamond has observed a general trend of increasing permeability during the first several replicates, noted by the decrease in time required to fill the same volume of the chamber. For example, the time to fill 10 cm in stage change might require 24.2s, 23.1s, 21.5s, 22.0s, and 20.8s. A general rule is to collect at least 5 measurements, and continue beyond 5 reps until the last rep is not the highest permeability. In the example above, if the 6th rep is 21.4s, then 6 reps would satisfy the general rule and therefore sampling could stop. When enough replicates are collected, the sample is complete, and the operator can move to the next sample location.

D.1.4.3.4 *<u>Data Entry and Analysis</u>*

Collect at least three samples at a given spawning site, so that a variance and confidence intervals can be computed. The included Microsoft Excel© spreadsheet will compute the inflow rate and convert it to permeability with the necessary adjustment for viscosity based on water temperature. The spreadsheet requires only the input values of the initial stage reading, end stage reading, time, and temperature. Up to 10 replicate measurements can be entered for each sample, and the spreadsheet will generate the mean permeability and several statistics that describe the variability of the sample. Once data for several samples have been entered, the spreadsheet will compute the mean, variance, and confidence intervals for the entire site. Ideally in the future, with a solid relationship established between permeability and egg survival-to-emergence, the spreadsheet could be designed to estimate or predict a range of survival values for eggs incubated in those particular gravels. Note that the conversions of raw data to true permeability, and statistical calculations make use of the "look-up tables" in the Excel© Workbook, and cannot be changed or removed from the file. Create a separate worksheet for each stream sampled by copying and pasting the template sheet and renaming the sheet with the stream name. Each new worksheet will continue to reference the look-up tables. Maintain the template file blank. Once several different streams are entered, copy the entire column Q and "Paste-Link" or "Paste-Values" (Excel operations) into a new sheet as a summary sheet. This allows comparisons between different streams.

D.1.4.3.5 Sampling Design

The primary objectives of permeability sampling are to:

- quantify the condition of salmonid spawning substrates in a manner that will allow prediction of egg survival-to-emergence or incubation success;
- document the variability in baseline or initial conditions of a particular river or stream reach with suitable precision to allow comparison to other reaches/stream, and to detect changes in conditions in subsequent years' monitoring; variability may occur within a chosen spawning site, from site to site within a stream, and/or from stream to stream;

To meet these objectives, the monitoring data should assess the mean or average condition of a particular study reach, and the variance in the mean. These variables can then be used to determine the confidence interval around the mean, and to compare two or more streams to determine if the means are statistically similar or different (generally with a t-test or ANOVA). In other words, the mean and confidence interval must be defined narrowly enough that a statistical comparison will detect a significant difference, if a difference exists. The confidence interval is dependent on the sample size and the variance (or standard deviation), according to the formulae:

$$\text{SE (Standard Error)} = s\sqrt{n};$$

and

$$\text{CI}_\alpha \text{ (Confidence Interval)} = y \pm t_\alpha * \text{SE}$$

where "s" is the standard deviation, "n" is the sample size, "y" is the sample mean, and "t" is the student t distribution at α significance level.

The standard approach to estimate the sample size necessary to ensure a level of variance that will allow meaningful statistical comparisons is to perform a power analysis. A power analysis uses a preliminary estimate of the expected variance (s^2) to determine the sample size necessary to achieve a specified level of variance. In other words, use an estimate of variance to estimate the sample size necessary to achieve the variance desired. The estimate of variance can be collected in a pilot-level assessment of a particular stream, or from the range of variability obtained in other studies. In addition to the estimated variance, three additional terms must be specified: "α", the significance level, "β", the power (or Type II error), and "δ", the minimum detectable difference. The minimum sample size can be computed from the following equation:

$$\text{Sample Size (n)} = (s^2/\delta^2) * (t_{\alpha(2),df} + t_{\beta(1),df})^2 \qquad \text{(Zar 1974)}$$

Sample size estimates based on this equation should be rounded up to the next highest integer. A conventional combination of significance and power is 95% significance (α=0.05$_{(2\text{-tail})}$) and 80% power (α=0.20$_{(1\text{-tail})}$). The standard deviation term (s^2) is the standard deviation of residuals from an ANOVA test, with log-transformed data. The minimum detectable difference, "δ", is a decision made depending on the study objectives (i.e., a subjective decision). The δ can be interpreted, for example, as the percent difference in permeability that the research expects to detect, with the sample size then determined by the above formula. If two tributaries are sampled with the objective of determining a significant difference of at least 10% (with 95% confidence and 80% power) between the means of permeability, then δ=0.10. With these initial objectives, the proposed study may not then detect a 9% or less difference in the mean permeability.

From ANOVA tests with permeability data from the Garcia River (McBain and Trush, 2000; with assistance from Stillwater Sciences), an estimated standard deviation of 0.7, was applied in the above equation to determine the sample size necessary to detect a difference between tributaries of: (a) a factor of 10, or the difference between 1,000 cm/hr and 10,000 cm/hr, and (b) a factor of 2, or the difference between 1,000 cm/hr and 2,000 cm/hr. These estimates yielded a sample size of 2 and 17 samples per tributary, respectively, to detect the corresponding level of difference between different tributaries:

Minimum Detectable Difference	Sample Size (n) Based on Z Values
Factor of 10	2
Factor of 2	17

A sample size of at least 20 samples per tributary, distributed among several different pool-tail or spawning sites within a reach is recommended. This initial level of sampling should allow an adequate number of samples to define the variability within a study reach with good precision. Additional samples may improve the precision in the data. Sample sites should be selected and distributed randomly throughout the spawning habitat or particle facies (i.e., a pool-tail) identified for sampling. Once the variability has been assessed within each study site, subsequent sampling may require fewer samples to define the desired range of variability. Each sample should consist of numerous replicate measurements, as discussed above. Selection and collection of at least 20 samples within a stream study reach should be possible in a single field day's work for a crew of two technicians.

Substrate permeability will be initially employed in the long-term channel monitoring reaches and the four streams in Little River where summer and winter populations are estimated. Additional Class I watercourses within each HPA will be monitored so there will be an adequate zone of monitoring influence once thresholds values are established.

D.1.4.4 Literature Cited

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D.1.5 Road-related Sediment Delivery (Turbidity) Monitoring

D.1.5.1 Introduction

Increases in suspended sediment and turbidity are potential impacts associated with various land management activities. Primary sources associated with timber harvest practices are road erosion runoff, hillslope erosion and inchannel inputs (inner gorge slides and displacement of stored sediment). The road erosion runoff can be considered to be all management related while hillslope erosion and inchannel inputs have a natural or background component as well as management influenced inputs. This monitoring is intended to isolate and quantify suspended sediment inputs from the surface and inboard ditch of roads (not mass wasting events associated with roads and culverts). Road upgrading measures and winter use limitations are expected to reduce road erosion and resulting turbidity (suspended sediment). To monitor the effectiveness of the reduction of road erosion, turbidity and suspended sediment monitoring will be conducted.

Turbidity monitoring will be focused on the four watersheds that make up the Experimental Watershed Program: the Little River, South Fork Winchuck River, Ryan Creek, and Ah Pah Creek). Within each of the 4 experimental watersheds, turbidity will be measured immediately above and below selected road crossings in 1st and 2nd order streams that have consistent flows during winter. The difference in observed turbidity between the monitoring locations is assumed to due to surface runoff (erosion) from the road. The road surface erosion monitoring will also compare this change in turbidity on individual road segments before and after road upgrading, and between roads which have been upgraded and those which have not. Continuous turbidity monitoring stations will also be employed within specific streams in the four experimental watersheds. Continuous turbidity monitoring stations will be monitoring all changes in the experimental watersheds (i.e. all effects). These data can be used for comparing all changes within each of the experimental watersheds.

Appropriate threshold values for turbidity monitoring cannot be determined at this time. Approximately five years of initial trend monitoring are expected to be necessary to set the appropriate biological objectives and threshold values. At the end of 5 years a review and evaluation of trend monitoring results will be conducted and threshold values determined.

D.1.5.2 Monitoring Methods

Two samples will be taken in the watercourse: one upstream of the crossing above the influence of any inboard ditch contribution and one just downstream of the watercourse crossing. Successive samples (flow and grab) must be taken at the same location each time. The difference between the upstream sample and the downstream sample is the contribution of the road surface and connected inboard ditches. Suspended sediment measured at watercourse crossings along road segments is the response (dependent) variable that will be used in the analysis. However, the amount of sediment that enters at a watercourse crossing will also depend on the following independent variables: rainfall intensity, length (or area) of road contributing to a watercourse, amount and type of road use, age/construction of the road, and status of the road (upgraded or not upgraded). Rainfall will be measured with collecting gages at the sample road segments and a primary event recording rain gage located at an appropriate location (e.g. Pollnow Peak in the Little River HPA) will measure rainfall intensity. The length of inboard ditch (or road surface area) contributing sediment will be measured at each sample site. Road segments will be selected based on road status (upgraded vs. not upgraded) with a wide range of anticipated use. Sites on roads that have not been upgraded will be used to establish a "baseline condition" from which a treatment effect could be determined when the road is upgraded. Road upgrading will involve rocking or re-rocking road surfaces intended for winter use and hydrologically disconnected inboard ditches from watercourses. Road upgrading involves other treatments as well, however, the measures described above will likely have the greatest effect at controlling road-related surface erosion.

D.1.5.2.1 *Site Selection*

Various road segments representing categories of road use and road condition will be selected. The categories will be low and moderate-use versus high-use roads, and road segments scheduled for upgrade in an upcoming year versus roads that have already been upgraded. For each of these road sections, a random starting point will be selected from the first 5 crossings with every fifth crossing systematically selected for sampling beyond that point. This sampling intensity may change depending on how many crossings are available for sampling in a given area. The selection of sites will be reviewed by the agencies.

D.1.5.2.2 *Field Measurements*

Inexpensive plastic rain gauges will be dispersed throughout the monitoring area. A record log will be associated with each gage to track daily, site specific rainfall quantity. The date, time and quantity will be recorded during every sampling event. A separate rain gage will be maintained at a crew member's residence in order to track relative rainfall and possibly provide a trigger to initiate sampling at higher intensity rainfall events. For example; if the target is a 1" storm event, current rainfall at the off-site location can be tracked until the threshold is approached, at which time field sampling can begin.

Flow will be measured at each site during each sampling event. Standard flow measurements for low flow streams will be employed. Flow will be measured as the product of cross-sectional area and water velocity over a known length (usually 1-2 m) of channel with relatively uniform depth and width. Stream depth will be estimated by measuring at 1/4, 1/2, and 3/4 intervals across the stream and dividing by four (Platts et al. 1983). Water velocity will be estimated by timing the surface speed of a small floating object for three trials over the pre-determined length of stream. If site selection allows for flow measurements at culvert outlets then that method may be preferable. A calibrated bucket can be placed at the outlet and the amount of time it takes to fill to a certain level will provide a flow measure. The total length of the contributing inboard ditch to the watercourse will also be measured.

The grab samples will be collected in 0.5 L plastic bottles from a well mixed area of the stream that will remain consistent for all sampling events. The bottle should be filled as much as reasonably possible, especially if the sample is relatively clear, to insure that a measurable amount of sediment will be available following the filtration process. This grab sample will be taken back to the lab for processing. Sample processing and analysis will follow the Redwood Sciences Laboratory protocols.

D.1.5.3 Literature Cited

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D.1.6 Headwaters Monitoring

D.1.6.1 Introduction

Most of the research and protocols developed for monitoring forest aquatic systems in the Pacific Northwest have focused on anadromous fish populations and their habitat conditions within third order or larger streams. Using the fish populations as indicators of watershed health is problematic, as factors outside the freshwater system have a major impact on population levels. As a result, much of our monitoring program is focused on the habitat conditions within the fish-bearing reaches of streams. However, it is possible that habitat conditions will be shown to improve throughout the life of the plan, but fish populations will continue to decline. It is critical to the monitoring program to provide a definitive biological link to freshwater habitat conditions.

The headwaters monitoring project will provide this biological link by focusing on the populations of the two obligate headwater species (tailed frog and southern torrent salamander) that are the most sensitive to the potential impacts of timber harvest. These species are unique relative to anadromous fish species in lower stream reaches in that they have relatively limited vagility and typically live out their entire lives in or immediately adjacent to a relatively short reach of stream. Therefore, the population levels of obligate headwater species are influenced by the conditions that exist within or immediately adjacent to the stream course. Although there are many demonstrated risks associated with the use of biological indicator species, the population levels of the headwater amphibian species covered in this plan should provide a good biological indicator of the general effectiveness of the plan in achieving the biological goals of maintaining cold water temperatures and reducing excessive sediment inputs into streams.

In addition to the need to provide a biological indicator, the focus of the headwaters monitoring will be on populations because there are no well defined protocols that can be directly applied to monitor the habitat conditions within headwater streams. Research in smaller headwater streams has typically focused on the populations and habitat associations of the species that live in these streams. In comparison to numerous studies designed to monitor the impact of watershed processes on stream morphology in fish bearing streams, little has been done to monitor the impact of those same processes on headwater streams. It is known that headwater streams typically have higher gradients and more confined channels than lower stream reaches, and as a result are primarily sediment transport reaches. There are no readily implemented techniques to monitor how sediment movement through these systems impacts the quality of the habitat in the stream. Although Green Diamond will monitor some elements of habitat conditions in headwater streams, the headwaters monitoring program will be primarily focused on populations of the two obligate headwater species covered under this AHCP/CCAA, the tailed frog and southern torrent salamander. Populations of tailed frogs and southern torrent salamanders should provide the best indicator of overall habitat conditions in headwater streams.

D.1.6.2 Tailed Frog Monitoring

Tailed frog habitat has been characterized as perennial, cold, fast flowing mountain streams with dense vegetation cover (Bury 1968; Nussbaum et al. 1983). To support larval tailed frogs, streams must have suitable gravel and cobble for attachment sites and diatoms for food (Bury and Corn 1988). Streams supporting tailed frogs have been found primarily in mature (Bury and Corn 1988; Welsh 1990) and old growth coniferous forests (Bury 1983; Welsh 1990). Bury and Corn (1988) reported that the frogs seem to be absent from clearcut areas and managed young forests (Welsh 1990). Although these authors did not establish a cause and effect relationship, it can be hypothesized that tailed frog populations could be affected by both direct (on site) and indirect (off site) impacts of timber management. Direct impacts could include activities such as excessive canopy removal at the site leading to elevated water temperature or changes in the algal community of the stream, or direct physical disturbance by operating heavy equipment within the site. However, tailed frogs may be vulnerable to indirect impacts that occur off site from the upper reaches of watersheds that result in elevated water temperatures or excessive sediment loads. In this regard they are similar to the salmonid species except that such indirect impacts could affect tailed frog populations before cumulative impacts can be manifested in the lower fish-bearing reaches of the watershed.

The primary focus of the tailed frog monitoring will be on the larval population. While the adults can move between the stream and adjacent riparian vegetation, the larvae respire with gills and are tied to the stream environment. They require a minimum of one year to reach metamorphosis (Wallace and Diller 1998), which necessitates over-wintering in the streams. They feed on diatoms while clinging to the substrate with sucker-like mouth parts (Metter 1964) and have limited swimming ability. This makes them potentially vulnerable to excessive bed movement of the stream during high flows, which have previously been documented to drastically reduce the larval cohort (Green Diamond unpublished data). As a result of their life history requirements, the larvae provide the most immediate and direct response to changes in stream. In addition, larval tailed frogs can be captured with ease while causing minimal disturbance to the site. Ongoing studies have allowed us to develop a protocol that has been shown to be highly effective in estimating larval populations. Adults can also be captured with minimal disturbance to the site, but in contrast to the larvae, their population size cannot be readily estimated. As a result of all the factors discussed above, the primary response variable for the tailed frog monitoring will be the size of the larval population.

D.1.6.2.1 <u>Study Design</u>

The primary monitoring approach will employ a paired sub-basin design. The goal will be to compare changes in larval populations of tailed frogs in randomly selected streams in sub-basins with (treatment) and without (control) timber harvest. Paired sub-basin will be based primarily on geographic proximity, because this increases the likelihood of similar weather patterns, elevation and geologic formations. However, geology can show dramatic local differences, which would preclude utilizing some potentially paired sub-basins that are in close proximity and otherwise quite similar. Finding a large number of streams in paired sub-basins from which to randomly choose will be difficult. Therefore, sampled streams will sometimes be selected based on being the only available stream for pairing within an adjacent sub-basin. When possible, streams in sub-basins scheduled to be harvested (treatment streams) will be paired with streams in sub-basins scheduled for little or no harvest (control streams). However, finding a control stream to match with every harvest stream will not be critical to the statistical validity of the overall project. In some cases, control sub-basins with no timber harvest will not be available in which case changes in larval populations will be compared to the amount of timber harvest in the sub-basin. The advantage of pairing is that statistical power may be increased if the variable (timber harvest) which forms the basis for pairing affects the response variable of interest (larval population).

All of the streams within the study area have been impacted from past land management activities. Many of these streams were heavily impacted from unregulated timber harvesting and other land management activities, which presumably adversely affected tailed frogs. Since the inception of the California forest practice rules in the mid 1970s, protection of headwater streams has increased and it is our assumption, corroborated by review of past aerial photographs, that stream conditions have generally improved for tailed frogs in recent years. Therefore, Green Diamond also assumes that populations of tailed frogs that currently exist in streams either managed to survive the heavy impacts of the past or recolonized the stream some time after the initial impacts occurred. In either case, Green Diamond also assumes that lacking any new impacts, current populations of tailed frogs will continue to persist into the future. Therefore, the assumption is that tailed frog populations in control streams will persist at or above their current levels, for the life of the Plan and that statistically significant changes in tailed frog populations in treatment streams relative to control streams will be due to the treatment effect. Assumptions of persistence of the control populations will be further tested through future graduate studies of the adult populations to estimate demographic parameters. Specifically, Green Diamond will use mark-recapture methods to estimate the size of the adult population, mean fecundity and age-specific survival.

Within each treatment and control stream, one tailed frog reach within the primary breeding zone for tailed frogs will be selected for sampling. The sampling reach in treatment streams needs to be located below the treatment area such that the stream reach has the potential to be influenced by all direct and indirect impacts of the treatment. Control reaches should be located in a similar position in the sub-basin relative to the paired treatment reach. Logistical constraints will be used to limit the potential placement of the monitoring reach, but the specific starting point will be randomly chosen from some reasonable access point. The monitoring reach within each sub-basin will be sampled at least one year prior to operations that could influence the treatment sites and every year thereafter. New sub-basins will be added across the ownership as the opportunities exist. (New sites cannot be created unless there is going to be harvesting in the area to create the treatment reach.) The goal is to have a minimum of 12-15 paired sites that are well distributed across the Plan Area to represent different physiographic regions. If there is little variance in the treatment effect among sites, this minimum number will be statistically adequate to reach a definitive conclusion on the impact of harvesting on tailed frogs. However, if there is substantial variation in the treatment response, it will be necessary to add additional sites. The actual maximum number is a statistical question that cannot be answered until the data are collected and analyzed. The duration of the monitoring will be dependent on the inherent within and among stream variation in tailed frog abundance along with annual variation among and within streams (i.e. statistical power of the monitoring). The amount of harvesting in the treatment sub-basins may also influence the duration of the monitoring, since one cannot conclude that no treatment effect has occurred until the maximum treatment level has been achieved. All of these unknown variables make it impossible to set a minimum duration for this monitoring, but it will likely take at least 5 years before the minimum number of sites have been established, and it is anticipated that each monitoring site will be monitored for at least 10 years.

D.1.6.2.1.1 *Monitoring Protocol*

Chronology

Sampling will begin in the late spring or early summer when flows are sufficiently low to allow working efficiently in the stream. The animal sampling must be completed by late July to avoid sampling after larvae have begun metamorphosing and leaving the stream. Substrate sampling can be most efficiently done in late summer or early fall during minimal flows and after the larvae have metamorphosed. Flow measurements will be done in August to get a standardized low flow estimate among all streams. Stream temperature profiles will be obtained from mid-summer to early fall (July – October).

Physical Stream Characteristics

Water temperature data recorders will be placed near the lower end of each monitoring reaches from mid-summer (July) until early fall (October) to determine temperature profiles during the warmest time of the year. In addition, potential differences in mainstream water temperatures due to side tributaries will be measured and recorded with a hand-held thermometer.

Discharge (flow) will be measured as the product of cross-sectional area and stream velocity over a known length (usually 1-2 m) of stream with relatively uniform depth and width. Stream depth will be estimated by measuring at ¼, ½, and ¾ intervals across the stream and dividing by four (Platts et al. 1983). Water velocity will be estimated by timing the surface speed of a small floating object for three trials over the pre-determined length of stream. This will be measured on all streams in late August during minimal flows to reduce the effects of seasonal variation.

Changes in substrate composition will be estimated by assessing material deposited at the tail-out of pools. From the beginning of the sample reach, walk upstream sampling each low gradient riffle (0-5%) at the tail-out of the pool. Riffles will be excluded from sampling if bedrock, LWD or some other stream feature prevents the substrate from being deposited by normal hydrologic scouring and depositional processes of stream flow. In addition, riffles should not be sampled if they have been recently disturbed by the stream layout or sampling procedures.

At each site chosen for substrate sampling, measure and record the gradient using the "measuring stick" and clinometer, and record canopy closure using a concave densiometer (4 cardinal directions measured at the center of the stream). Place a 45x45cm grid with 5cm mesh in the center of the stream at the pools tail-out. Record each substrate type, based on particle size, at each intersection on the grid. The particle size is determined by measuring the secondary axis of the substrate. This results in a total of 100 readings for each sample. Continue upstream repeating this process until the end of the monitoring reach has been reached. Table D-2 provides the particle size and types classification.

Table D-2. Particle size and type.

Size	Particle Type
<0.06mm	Fine (F)
0.06-2mm	Sand (S)
2mm-6cm	Gravel (G)
6-13cm	Small Cobble (SC)
13-26cm	Large Cobble (LC)
26-51cm	Small Boulder (SB)
>51cm	Large Boulder (LB)

Stream Reach Layout (Selection of Habitat Units)

1. The sample reach of the treatment stream is located below the treatment area so that the stream reach has the potential to be influenced by all direct and indirect impacts of the treatment.

2. A similar stream reach needs to be designated in the same watershed position in an adjacent watershed to serve as the control stream. The logistics of getting to the designated stream reaches will normally dictate the general of the monitoring reach, but the specific starting point of a stream reach will be randomly chosen.

3. Habitat units will be delineated by hiking up the designated stream reach with a hip chain and recording fast and slow-water stream habitat units that are at least 1.5m in length (fast-water = riffles and cascades; slow-water = pools and runs).

4. Selection of sites where sampling belts will be placed is as follows: all fast-water habitat units in a stream reach will be identified and measured for length. All fast-water habitat will be in theory placed end-to-end as if it was all contained in one long habitat unit. A random start, labeled m, between 1 and 3 will be chosen, the m-th belt from the beginning of the linear assemblage of fast water habitat will be sampled, the (m+3)-th belt from the beginning of the linear assemblage will be sampled, the (m+6)-th belt from the beginning of the linear assemblage will be sampled, and so on. In the end, every third belt after the m-th will be sampled.

5. Each fast water unit is considered to be 3m in length. Therefore, every ninth meter of fast water will be sampled as a 1.5 to 3m belt. Sample every tenth slow-water unit of at least 1.5m in length from a random starting point.

6. For long slow water units, such as more than 6m, the large unit can be broken up into smaller units of approximately 3m each in order to maintain consistent sampling intensity. If the designated unit is unsearchable due to water depth, organic debris or excessive gradient, go on to the next available unit.

7. Continue up the stream until 30 sample belts are identified. Record and sum the total number length of each of the habitat types within the sample reach.

8. The starting point of the monitoring reach will be permanently monumented, but the first belt to be sampled will be randomly selected each year that the reach is re-sampled.

Animal Sampling

1. If the selected slow-water unit is between 1.5-3.0m in length, delineate the entire unit for sampling. If the unit is greater than 3.0m, randomly select the starting point based on 1m increments and sample a 3.0m belt.

2. If a fast-water belt crosses a habitat unit boundary and more than 1.5m of a belt is in one of the habitat units, then sample that length for the belt. If less than 1.5m of a belt is in one unit while the rest is in another, then sample the next belt or, if possible, move that belt back within that unit to sample 1.5 to 3m.

3. Prior to any disturbance of the unit to be sampled, place a blocking net at the downstream end of the unit. Measure the gradient of the unit, depth of the water at the mid-point of the unit (measure at ¼, ½, and ¾ intervals across the channel and divide by 4 to get average depth), width at the beginning, mid-point and end of the unit and length of the unit (1.5-3m).

4. Remove all the substrate that can be moved by hand within the sample unit and collect any animals that may be incidentally seen during this process.

5. Do the first visual search of the unit using a viewing bucket and remove all tailed frogs seen. Place all of these animals in an appropriate container and repeat the visual search three additional times.

6. Check the blocking net after each pass and place any animals seen with the other animals collected during the search. If the number of frogs obtained in a removal pass is larger than the number of frogs obtained in a previous pass, perform an additional pass for a total of five.

7. Record the sex (adults only) and age class for each tailed frog captured. Following the final search, remove the blocking net, put the substrate back into the stream and release the animals back into the stream.

D.1.6.2.2 Tentative Analysis

The tailed frog monitoring protocol will yield the following data: 1) an estimate of the total number of tailed-frogs for each monitoring reach (using removal/depletion techniques), and 2) various physical measurements associated with each monitored reach (water temperature, flow, canopy cover and etc.). In addition, the distance and amount of disturbance (treatment sites) will be recorded.

D.1.6.2.3 BACI Analysis

For this analysis, the estimated number of tailed frogs in each stream will be analyzed using standard before-after-control-impact (BACI) analyses (Skalski and Robson 1992; McDonald et al. 2000). These analyses will make use of the paired nature of monitored streams and will adjust for nuisance variables that were used to form the pairs.

Following the philosophy of BACI analyses, the lack of parallelism in time trajectories of tailed-frog abundance on the control and treatment sites will be estimated from raw data. If the time trajectories of responses on the control sites are not parallel to those on the treatment sites, the treatment will be deemed associated with changes in the response. Parallelism will be estimated using the interaction effect in a univariate repeated measures analysis of variance (Little et al. 1996), where the pair and treatment-control factors are applied to "main plots" and the before-after disturbance factor forms the repeated measure. Using the systematic nature of sampled segment, a single tailed-frog abundance estimate will be computed for each monitored stream. Assuming p treatment-control stream pairs and t years of monitoring, the anticipated BACI analysis of variance table appears in Table D-3. Following McDonald et al. (2000), interest lies in components of the timeXtreatment interaction factor because they quantify the lack of parallelism in time trajectories. Significance of the interaction components will be assessed using standard likelihood ratio tests and will adjust for estimated overdispersion. Other environmental variables, such as flow and canopy cover, will be considered for inclusion in the repeated measures analysis of variance model to adjust estimated parallelism for these types of nuisance variables.

Table D-3. Anticipated analysis of variance table for the BACI analysis of tailed-frog monitoring data[1]

Source	Degrees of Freedom
Pair	p-1
Treatment	1
Pair X Treatment	p-1
Time	t-1
Time X Treatment	t-1
Residual	2(t-1)(p-1)
Total	2pt-1

Note
1 The BACI analysis is a repeated measures analysis and follows the philosophy of McDonald et al. (2000) where interest lies in the Time X Treatment factor.

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D.1.6.3 Southern Torrent Salamander Monitoring

D.1.6.3.1 Introduction

Torrent salamanders are generally found in springs, seeps and the uppermost headwater reaches of streams (Nussbaum et al. 1983; Stebbins 1985). They are a small salamander that appears to spend most of its time within the interstices of the stream's substrate, which make them difficult to locate and capture without disturbing their habitat. The larvae have gills and are restricted to flowing water while adults also appear to spend most of their time in the water, but are capable of movements out of the water. They are thought to have limited dispersal abilities and small home ranges so that recolonization of extirpated sites may take decades (Nussbaum and Tait 1977; Welsh and Lind 1992; Nijhuis and Kaplan 1998). Given the highly disjunct nature of their habitat, individuals at a given site (sub-population) are likely to be isolated from other adjacent sub-populations. The degree of isolation of these sub-populations probably varies depending on the distance and habitat that separates them so that torrent salamanders could be best described as existing as a meta-population.

Although there is some evidence for cumulative effects of sediment input in certain sites, torrent salamanders are primarily vulnerable to potential direct impacts from timber harvest (Diller and Wallace 1996). Direct impacts could include activities such as excessive canopy removal at the site leading to elevated water temperature, operating heavy equipment in the site, or destabilizing soil leading to excessive sediment deposits at the site. Past observations have indicated that these direct impacts can lead to extinction of the sub-population at the site. Due to the survey difficulties noted above, an attempt to get a statistically rigorous estimate of the number of individuals at monitored sites would be impractical. In spite of this, the project will provide an index of the number of individuals at each site and a record of the life history stage of each individual captured. However, given the unreliability of the index of sub-population size, the persistence of individual sub-populations will be used as the primary response variable for the torrent salamander monitoring.

Concerns could be raised that there are too few sub-populations in the meta-population of torrent salamanders to expect to see significant changes over time, or that any lose in sub-populations would threaten the long-term persistence of torrent salamanders within the Plan Area. However, Green Diamond has already located 598 torrent salamander sites (sub-populations) across the Plan Area and estimates that no more than 25-30% of the total potential habitat has been surveyed. In addition, without a formal monitoring protocol, Green Diamond has already documented both the apparent extinction and re-colonization of several torrent salamander sites. This would indicate that the meta-population concept does appear to apply to torrent salamanders in this region.

D.1.6.3.2 Study Design

The primary monitoring approach for southern torrent salamanders will employ the same paired sub-basin design that was described above for tailed frogs. Monitoring for tailed frogs and torrent salamanders will be geographically linked whenever possible by selecting monitoring sites for torrent salamanders in the same sub-basins where a tailed frog monitoring reach has already been selected. Therefore all the same criteria used to select sub-basins for monitoring described above will also apply to torrent salamander monitoring. However, instead of using larval populations as the primary response variable, Green Diamond will compare changes in the persistence of sub-populations of torrent salamanders in treatment and control sub-basins. In addition, within each sub-basin (treatment and control), two torrent salamander sites in the uppermost reaches of first order tributaries will be randomly sampled.

As noted above, all of the streams within the study area have been impacted from past land management activities. Many of these streams were heavily impacted from unregulated timber harvesting and other land management activities, which presumably adversely affected torrent salamanders. Since the inception of the California forest practice rules in the mid 1970s, protection of headwater streams has increased and it is our assumption, corroborated by review of past aerial photographs, that stream conditions have generally improved for torrent salamanders in recent years. Therefore, Green Diamond also assumes that populations of torrent salamanders that currently exist in streams either managed to survive the heavy impacts of the past or recolonized the stream some time after the initial impacts occurred. In either case, Green Diamond also assumes that lacking any new impacts, current populations of torrent salamanders will continue to persist into the future. Therefore, it is assumed that torrent salamander populations in control streams will persist at or above their current levels, and that statistically significant changes in torrent salamander persistence in treatment streams relative to control streams will be due to the treatment effect.

The sampling reaches in treatment sub-basins need to be located such that they will be located within a future treatment area (harvest unit). Control reaches should be located in a similar position in the sub-basin relative to the paired treatment reaches. Logistical constraints will be used to narrow the potential placement of a monitoring reach, but the specific starting point will be randomly chosen from some reasonable access point. The monitoring reaches within each sub-basin will be sampled at least one year prior to operations that could influence the treatment sites and every year thereafter. New sub-basins will be added across the ownership as the opportunities exist. (New sites cannot be created unless there is going to be harvesting in the area to create the treatment reach.) The goal is to have a minimum of 12-15 paired sites that are well distributed across the Plan Area to represent different physiographic regions. If there is little variance in the treatment effect among sites, this minimum number will be statistically adequate to reach a definitive conclusion on the impact of harvesting on torrent salamanders. However, if there is substantial variation in the treatment response, it will be necessary to add additional sites. The actual maximum number is a statistical question that cannot be answered until the data are collected and analyzed. The duration of the monitoring will be dependent on the inherent within and among stream variation in persistence of torrent salamander sites along with variation in abundance of salamanders (i.e. statistical power of the monitoring). The amount of harvesting in the treatment sub-basins may also influence the duration of the monitoring, since one cannot conclude that no treatment effect has occurred until the maximum treatment level has been achieved. All of these unknown variables make it impossible to set a minimum duration for this monitoring, but it will likely take at least 5 years before the minimum number of sites have been established, and it is anticipated that each monitoring site will be monitored for at least 10 years.

D.1.6.3.3 *Monitoring Protocol*

Chronology

Sampling should be done in the fall after enough rain to insure that the riparian habitat is cool and moist, but while stream flows are still low. The larger torrent salamander streams with higher flows should be surveyed first with other streams surveyed in order of decreasing flow. The goal is to insure that adult salamanders are active at the surface at a time when stream flows are low enough to make searching for larvae efficient. Stream temperature profiles will be obtained from mid-summer to early fall (July – October).

Physical Stream Characteristics

Water temperature data recorders will be placed near the lower end of each monitoring reach from mid-summer (July) until early fall (October) to determine temperature profiles during the warmest time of the year.

The total length and the amount of searchable habitat within the sample reach will be determined using a hip-chain or tape measure. The total amount of searchable habitat should be at least 10m with a maximum of 30m.

Measurements of active channel width will be made where obvious scouring can be seen somewhere near the beginning, middle, and end of the reach. Canopy closure will be measured with a spherical densiometer on the four cardinal directions at the same points (beginning, middle, and end) along the reach.

Discharge (flow) will be measured as the product of cross-sectional area and stream velocity over a known length (usually 1-2 m) of stream with relatively uniform depth and width. Stream depth will be estimated by measuring at ¼, ½, and ¾ intervals across the stream and dividing by four (Platts et al., 1983). Water velocity will be estimated by timing the surface speed of a small floating object for three trials over the pre-determined length of stream. This will be measured on all streams in late August during minimal flows to reduce the effects of seasonal variation.

Gradient of the reach will be measured using a clinometer. The gradient measurement can be broken up into more than one measurement depending on the length of the reach. Cascades are not included as part of the gradient measurement.

Where possible, changes in substrate composition will be estimated by assessing material deposited at the tail-out of pools in the same manner that was described for tailed frogs above. However, many torrent salamander reaches are sufficiently short and with such high gradient that no low gradient riffles are available at the tail-out of pools. As a result, it will be necessary to record a qualitative description of the habitat conditions associated with the reach. Include substrate composition, signs of recent sediment inputs, bank erosion or scour, inner gorge slides, and overall assessment of the quality of habitat for torrent salamanders. If it is feasible, the site will be photographed to document the general habitat conditions.

Animal Sampling

The entire length of the reach will be searched carefully for animals by moving upstream from the bottom of the reach turning the substrate by hand or with a rake when necessary. When flows are sufficiently high to allow animals to escape downstream without being detected, the search will be conducted while holding an aquarium net downstream of the area searched. Also search along the margins of the stream by turning rocks and moveable woody debris.

For each salamander found record: distance from the beginning of the reach to where the salamander was found, age class and sex of adults and habitat type where the salamander was found (e.g. low, medium, or high gradient riffle, pool, rock or log cascade and etc.).

D.1.6.3.4 Tentative Analysis

The torrent salamander monitoring protocol will yield the following data: 1) presence or absence of torrent salamanders at each monitoring site, 2) an index of salamander abundance (i.e., the raw count) associated with each monitored stream reach, and 3) various physical measurements associated with each monitored reach (water temperature, flow, canopy cover and etc.). In addition, the distance and amount of disturbance (treatment sites) will be recorded. Throughout the period of monitoring, the field sampling protocol applied at each monitored reach will not change. Among other things, this insures that the search effort expended at each monitored reach will be constant and that raw counts of torrent salamanders will be comparable through time. Two monitoring reaches are planned for each stream in each set of paired streams for a total of four monitoring reaches per steam pair.

Two analyses are proposed below. While both analyses will be useful for assessing impacts of timber harvest on torrent salamanders, it is unknown which analysis will be most statistically powerful prior to data collection.

Analysis 1: Regression

For this analysis, the paired nature of stream segments is ignored and the probability of torrent salamander extinction at a site is related to the distance and amount of disturbance (treatment). Let $y_i = 0$ if torrent salamanders were found on the i-th monitored reach before disturbance and *k* years after disturbance. Let $y_i = 1$ if salamanders were found prior to disturbance of the i-th monitored reach, but were not found during the field visit(s) that occurred *k* years after disturbance. Monitored reaches that did not contain salamanders prior to disturbance will be ignored in this analysis. Let x_i be either the distance from the i-th monitored reach to the nearest disturbance (treatment), or an index of the amount of disturbance incurred by the i-th monitored site.

A logistic regression equation (Hosmer and Lemeshow 2000), relating the expected value of y_i to x_i, will be estimated to assess the potential effects of disturbance on probability of extinction. The logistic regression equation will be of the form, logit (pi) = b0 + b1xi where pi = E[yi] is the probability of extinction at the i-th site, logit(pi) = log(

pi/(1-pi)), and b0 and b1 are parameters to be estimated. If β_1 is significantly different from 0, the probability of extinction will be declared significantly related to disturbance as quantified by x_i. The significance of β_1 will be assessed using standard likelihood ratio test and will adjust for any estimated overdispersion (Hosmer and Lemeshow 2000).

It is anticipated that the above logistic regression equation will be estimated for *k*=1, 2, 10 years post-disturbance. Other physical variables, such as flow and canopy cover, will be investigated in the logistic regression model to potentially adjust β_1 for these types of nuisance variables.

Analysis 2: BACI Analysis

For this analysis, the raw count of torrent salamanders, or count per unit effort (CPUE), will be analyzed using standard before-after-control-impact (BACI) analyses (Skalski and Robson 1992; McDonald et al. 2000). These analyses will make use of the paired nature of monitored streams and will adjust for nuisance variables that were used to form the pairs.

Following the philosophy of BACI analyses, the lack of parallelism in time trajectories of salamander count or CPUE on the control and treatment sites will be estimated from raw data. If the time trajectories of responses on the control sites are not parallel to those on the treatment sites, the treatment will be deemed associated with changes in the response. Parallelism will be estimated using the interaction effect in a univariate repeated measures analysis of variance (Little et al. 1996), where the pair and treatment-control factors are applied to "main plots" and the before-after disturbance factor forms the repeated measure. Using the systematic nature of sampled segment, a single torrent salamander abundance index will be computed for each monitored stream. Assuming p treatment-control stream pairs and t years of monitoring, the anticipated BACI analysis of variance table appears in Table D-4.

Table D-4. Anticipated analysis of variance table for the BACI analysis of southern torrent salamander monitoring data.[1]

Source	Degrees of Freedom
Pair	p-1
Treatment	1
Pair X Treatment	p-1
Time	t-1
Time X Treatment	t-1
Residual	2(t-1)(p-1)
Total	2pt-1

Note
1 The BACI analysis is a repeated measures analysis and follows the philosophy of McDonald et al. (2000) where interest lies in the Time X Treatment factor.

Following McDonald et al. (2000), interest lies in components of the timeXtreatment interaction factor because they quantify the lack of parallelism in time trajectories. Significance of the interaction components will be assessed using standard likelihood ratio tests and will adjust for estimated overdispersion. Other environmental variables, such as flow and canopy cover, will be considered for inclusion in the repeated measures analysis of variance model to adjust estimated parallelism for these types of nuisance variables.

D.1.6.3.5 *Literature Cited*

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D.2 RESPONSE MONITORING

D.2.1 Introduction

Response Monitoring activities include:

- Class I channel monitoring
- Class III sediment monitoring

The Response Monitoring projects, like the Rapid Response projects described above, monitor the effectiveness of the conservation measures in achieving specific biological goals and objectives of the Plan. These monitoring projects are distinguished from the Rapid Response projects by the greater lag time required for feedback to the adaptive management process. The Response Monitoring projects are focused on the effects of cumulative sediment inputs on stream channels. Natural variation in stream channel dimensions, combined with the potential time lag between sediment inputs and changes in the response variables of these projects, make it difficult to determine appropriate thresholds for adaptive management at this time. When yellow and/or red light thresholds are determined, they are expected to require more than three years of results to be triggered in most cases.

D.2.2 Class I Channel Monitoring

D.2.2.1 Background and Objectives

The monitoring objectives of the Class I channel monitoring project are to track long term trends in the sediment budget of Class I watercourses as evidenced by changes in channel dimensions. The long term channel monitoring project is one of four monitoring projects designed to measure the effectiveness of the conservation measures in reducing management related sediment inputs to area streams. This technique is generally best suited for establishing long term trends due to the potential lag times between sediment inputs and the measured response in the monitoring reach. Nine monitoring reaches are currently established in eight streams across the Plan Area. Two additional reaches are established with a reduced protocol (thalweg profile only), because the sites do not meet the criteria necessary for doing the full protocol. An additional monitoring reach on Ah Pah Creek within the Klamath Basin will be established in the near future. These twelve reaches will be measured at least every other year for the duration of the Plan. The channel dimensions measured in each reach include cross-sectional and thalweg profiles, substrate size distributions (pebble counts), and bankfull and active channel widths.

D.2.2.2 Methods

Once a watershed has been selected for long-term cumulative effects monitoring, a sample reach, or reaches, should be located with respect to the channel's overall longitudinal profile. Generally, field inspection is necessary to properly identify desired low gradient (less than 1.5-% slope) stream reaches as potential sample sites; poor resolution of the longitudinal profile constructed from USGS topographic maps often obscures the true longitudinal profile of low gradient reaches.

The entire profile can be subdivided initially into transport, transitional, and aggradational/alluvial reaches. However, cutoff criteria (channel gradient changes) between these subdivisions are not always clear. Major tributary junctions and/or abrupt changes in channel type (e.g., a canyon segment within the low gradient, alluvial reach) may justify finer subdivision. A delta affected by flood backwaters of a larger channel may require an additional reach assignment, or not be selected for sampling.

Rather than sampling randomly or systematically throughout a channel's longitudinal profile, channel reaches most responsive to long-term cumulative effects should be selected. Low gradient channels (less than 1.5%) with alluvial, erodible banks probably are the most sensitive to changes in the watershed sediment budget.

A sample reach should be a minimum of three meander wavelengths long, but in many streams the entire low gradient, alluvial segment of the longitudinal profile probably should be included as one sample reach. A meander wavelength is approximately 7 to 9 bankfull widths long, therefore a monitored channel reach should be approximately 25 bankfull widths long. If the low gradient, alluvial-banked reach of the selected stream is extensive the selection of an appropriate monitoring reach should follow one of several methods. First, divide the entire lower reach into sample reaches of proper length (25 bankfull widths) and randomly select one for long-term monitoring. Second, monitor the uppermost section of the depositional reach. Third, select a monitoring reach for all parameters using one of the first two methods, but collect data for several of the parameters (such as thalweg profile, and pebble counts) for the entire depositional reach. Site selection may change as trend is analyzed from initial surveys to capture the section of the channel most responsive to change.

The following will be collected, analyzed, and reported:

- Determine drainage area at head of sample reach;
- Plot longitudinal profile from USGS topographic maps;
- Distinguish transport, transitional, aggradational segments on the longitudinal profile;
- Estimate average annual rainfall;
- Estimate average annual runoff;
- Estimate annual maximum flood duration curve;
- Inventory available aerial photographs, especially historical photos;
- Acquire personal photographs, verbal accounts; and
- Check CDFG for documented stream surveys.

The acquisition of historical information and photographs of the long-term monitoring watershed, especially historic photos of the actual monitoring reach are vital to evaluating the present channel condition with respect to recovery.

D.2.2.2.1 *Plan Mapping*

The plan map is a template for all additional measurements and can be produced either by hand or by various computer software packages. The following steps provide the data for plan map development and locations for all additional measurements within the monitoring reach.

Center Tape

After selection of the monitoring reach, place the beginning of the first 300 foot tape at a randomly selected starting point within the first ten feet of the beginning the monitoring reach. Because some measurements off the center tape occur at set 10-foot intervals, a random starting point creates a random, systematic sampling design. Methods to select random numbers include a random number chart, random # function on a calculator, or a roll of a pair of dice.

Two 300 foot centerline tapes are set up the channel, end to end, roughly between the bankfull channel edges, in straight segments. The ends of each tape are held in place with rebar stakes driven into the streambed.

Drive four-foot long rebar stakes into the streambed at any turning points along the tapes. Fix the position of the tapes by clamping them to the rebar turning points. Short pieces of hose (4-6 inches) are slipped over each rebar stake to protect the center tape from abrading on the rebar.

Record the length (nearest 1/10 of a foot) and azimuth (0^o to 360^o) of each leg or segment along the center tapes.

All measurements include a point location described in reference to the center tape. A point location includes a station number (feet upstream from the beginning of the tape) and the shortest distance between bankfulls for channel dimension measurements; or a perpendicular distance from the center tape for thalweg measurements. Measurements to the right of the center tape are recorded as a positive number. Measurements to the left of the center tape are recorded as a negative number. For example, (STA. 57.2', -14.3') is a point 57.2 feet upstream and 14.3 feet to the left of the center tape. Negative and positive numbers are used to reference spatial locations off of the center tape so that the plan map can be generated by the Microstation software program.

Temporary Benchmarks (TBMs)

The rebar stakes used for turning points are also used for TBMs. Assign an arbitrary elevation of 100' to the TBM furthest downstream.

Using a surveyor's level, survey elevations of all TMBs in reference to the arbitrary 100' TBM. "Close the loop" frequently (at least daily) to catch any surveying or note recording mistakes in a timely fashion.

In your field notebook record the point location of each TBM. A simple sketch of TBM locations can also be helpful for future reference.

D.2.2.2.2 Channel Dimensions

Working upstream, record the active channel width (Qa), and the bankfull channel width (Qbf) at the shortest distance between bankfulls on 10 foot intervals off of the center tape. For plan map purposes, take an azimuth at every channel dimension measurement. When the precise location of Qa or Qb is uncertain, use a surveyor's level to shoot the elevation of the nearest appropriate unambiguous channel dimension break. The elevation of the known channel break should be matched (allowing for the necessary change in stream gradient) to find the undecided point location.

Active channel (Qa) definition and indicators: Qa width is the wetted width during base winter flow. Some indicators of Qa along exposed cutbanks are fine exposed alder and willow roots and the lower extent of lichen and moss. "Bathtubs rings" and young (less than two years old) alder and willow growth are indicators along point bars. Combine these indicators with slight breaks in bank slope and slight changes in substrate particle size to locate the Qa margin.

Bankfull channel (Qbf) definition and indicators: Qbf is the wetted width during an elevated flow with a recurrence interval of approximately 1.5 to two years. This is the elevation at which bedload movement initiates and when elevated flows begin to spill onto flood terraces and dissipate scouring forces. Some indicators of Qbf include the edge of perennial vegetation such as mature alders and occasionally conifers. On the outside of meander bends look for the tops of exposed point bars. Combine these indicators with significant breaks in bank slope and changes in substrate particle size to locate Qbf.

D.2.2.2.3 Thalweg Profile

Working upstream, survey the thalweg depth (elevation) and location in reference to the center tape. The thalweg is the deepest point of the flowing channel, excluding any detached or "dead end" scours and/or side channels. These features are important and could be surveyed later and added to the plan map, however do not include these deep points in the thalweg profile or analyses of thalweg residuals.

At 10 foot intervals, measure the perpendicular distance (left or right) from the center tape and shoot the thalweg elevation.

Use the nearest TMB to determine the elevation of the surveyor's level. Record each thalweg elevation to the nearest 1/100 of a foot.

Record the point location of each elevation measurement using the numerical referencing described in Section D.2.2.2.1.

Always record the thalweg elevation at the maximum depth and the crest of the tailout of all pools so that pool depth variation (thalweg elevation residuals) can be calculated.

D.2.2.2.4 *<u>Cross Sections</u>*

Cross sections are elevation surveys conducted perpendicular to the channel. Two types of cross sections are surveyed; either across a thalweg cross-over or across a point bar. Each is designed to detect changes in channel morphology. On either side of the creek, (well above bankfull) drive a 4 foot piece of rebar to permanently mark the end points of the cross section. Attach a tape from the left bank to the right bank pins. Using the Total Station, survey every significant change in elevation along the cross section tape. Features such as bankfull (Qbf), active channel (Qa) and thalweg location can be noted during the survey and confirmed later during the cross section analysis.

Additional information that is collected at the thalweg cross-over cross sections is substrate roughness. In general terms, collect information regarding the substrate and vegetation composition along the cross section. This information can be applied in standard open channel flow equations such as Manning's equation to evaluate the bankfull elevation and bankfull area.

D.2.2.2.5 *<u>Pebble Counts</u>*

Straight channel reaches, exceeding three to four bankfull widths long, are the best sites for pebble counts. These sites usually coincide with thalweg crossovers (where the thalweg switches from one bank to another). These areas are generally uniform in cross-sectional dimensions and are resistant to adjustments in channel width. With relatively less change in water surface slope over a wide range of discharges, deposition is less likely to include secondary deposits overlaying primary deposits (each having its own particle size distribution). This substantially reduces surface particle size variation.

Each pebble site location includes the area from the top to the bottom of the riffle and the width of the Qa channel. Most riffles are diagonal to the flowing channel - be sure to sample only the riffle area. This may necessitate truncating either or both ends of the riffle in order to sample a roughly rectangular area. Measure the area of, and then exclude any LWD or localized sand deposits, which are larger than 10% of the sample area.

Divide the rough length of the sample area by ten to determine the location of ten approximately equally-spaced transects across the riffle. Randomly select a starting point for the first transect (between zero and the determined spacing) then systematically locate the remaining nine transects to be sampled. For example, if the riffle was 100 meters long - first divided the riffle into ten, 10 meter sections. Then using a random number chart you pick a number between 0 and 10 - lets say "4". The first transect to conduct a pebble count would be at 4 meters, the second transect at 14 meters, the third at 24 meters, etc.

Along each transect, randomly select fifteen pebbles at approximately equal intervals and measure the secondary axis to the nearest millimeter. To randomly select a rock, walk along the transect without looking at your feet or the channel bottom. After the appropriate number of steps required to achieve equal spacing along the transect, stop and place your finger at the tip of your right foot and touch to the ground. The first pebble you touch is the one you pick up and measure to the nearest millimeter. Repeat this procedure until 150 pebbles have been sampled across the entire riffle. The secondary axis is the diameter that would allow the pebble to pass through a sieve.

In the field notes record the surveyor and transect number, along with the appropriate measurements. Sketch each pebble count sample site in the field notebook including area measurements of the site and any LWD and/or local sand deposits.

D.2.2.2.6 *Repeat*

Continue to measure the channel dimensions, thalweg profile, and conduct pebble counts along each 600-foot reach of center tape until the end of the monitoring reach.

D.2.2.2.7 *Measurement Error Calibration*

Due to the subjective nature of several of the proposed monitoring variables it is necessary to quantify the measurement error between crews. Once the monitoring reach is selected, determine the number of 300-foot tapes required for the survey. Then, randomly select two numbers between zero and total number of tapes (again - use a random number chart, random # function on a calculator, or roll a pair of dice). These two reaches will then be surveyed twice for channel dimensions and the thalweg profile following the appropriate protocols. Because this exercise is testing for measurement error, it is necessary for each crew to use the same randomly selected starting points. Two crews, of two members each, will independently survey each channel reach. The difference between the two surveys is considered the measurement error.

D.2.2.2.8 *Permanent Bench Marks*

Two permanent benchmarks will be installed, one at each end of the monitoring reach. These benchmarks should be located well above the bankfull channel margin near an established, easily recognized feature (bridge, old-growth stumps, and boulder). Include a sketch of the bench mark location in the field notebook. Permanent benchmarks are constructed using one to two bags of redimix concrete and a carriage bolt. Dig a hole about one foot in diameter and two feet deep. Fill the hole with redimix and mix in water to from concrete. Sink the carriage bolt upright into the middle of the concrete pad, leaving about 1" of the bolt exposed. Survey the elevation of the permanent benchmarks using the nearest TBM as a reference. The datum and associated elevations should reference mean sea level, otherwise (or until surveyed) an arbitrary elevation can be assigned to the downstream benchmark. Recent state legislative activity may soon require licensed land surveyor approval. Record locations of both permanent benchmarks in reference to the center tape. If a GPS unit is available, enter the positions of both benchmarks.

D.2.2.2.9 *Video and Still Photography*

Videotaping the entire monitoring reach in an upstream direction will be conducted. This will capture important features within the reach including: location of permanent bench marks, location of cross sections, instream structures, side channel habitat, terraces, and riparian composition. Accurate descriptions of all these features will be made verbally while filming and include the date that the filming occurred. Still photos of the important features described above will also be obtained. While photographing, notes documenting what frame corresponds to which feature will be made so that the developed slide can be labeled with an accurate title.

D.2.2.2.10 Sampling Frequency

The variables will be re-measured every other year. The re-measurement will be conducted to capture changes in channel features resulting from relatively small, yet important, channel-forming flows, such as:

- A coarsening of riffle-bed surfaces by mobilizing fines previously deposited by a major storm event.

Re-measurement will include:

- Taking pebble counts along uniform straight reaches;
- Estimating the peak discharge of the previous winter's high flow to include an update for the flood frequency curve used to determine the occurrence of a 5-year or greater storm event;
- Measuring thalweg profile and calculating thalweg depth residuals; and
- Measuring Qa and Qbf channel widths; and
- In addition, all established monitoring locations in fish-bearing watercourses will be re-mapped as well as re-measured the summer following a storm event with at least a five-year recurrence interval.

D.2.2.3 Data Analysis

Data analyses are performed using the methods of McDonald (1998). This analysis focuses on assessing changes in bank width, thalweg elevation, and shifts in substrate (pebble) size distributions. A section of each creek will be monitored using the methods and at the frequency outlined above. Monitored sections are chosen from the highest (closest to headwaters) depositional reach in each creek. Depositional reaches are chosen because if changes in sediment load or other stream morphology parameters occur anywhere in the watershed, such changes are likely to be reflected in the first depositional reach downstream. During each channel monitoring interval thalweg elevation (defined as the height of the deepest part of the channel), bank full width, active channel width, and substrate (pebble) sizes will be recorded on the monitoring reaches. Thalweg elevation will be analyzed for change in mean elevation. Thalweg elevation residuals (variability in pool depths) will be analyzed for changes in variance. Bank full and active channel widths will be analyzed for changes in average width. Substrate sizes will be analyzed for changes in distribution.

Thalweg elevation will be analyzed for change in mean elevation and thalweg depths will be analyzed for change in variance. These analyses both use statistical models appropriate for correlated data. The basic data are pairs of points, (d_i, y_i), where y_i is thalweg elevation and d_i is the distance from the upper terminus of the reach to the point where y_i is measured. Because thalweg elevations are measured relatively close together (approximately every 10 feet) the measurements (i.e., the y_i) are potentially spatially correlated and do not represent independent observations. Therefore, the analysis accounts for this lack of independence by adjusting model coefficients and significance levels using a one dimensional spatial regression model (Cressie 1991;

Venables and Ripley 1994). The spatial regression model estimates a one dimensional correlation function among residuals then adjusts estimates and p-values via generalized least squares regression techniques. The spatial regression techniques and the adjustment for auto-correlation are described in more detail in Appendix A of McDonald (1998).

For the analysis of thalweg elevation, a regression model relating elevation of the thalweg to a cubic polynomial in distance is estimated. Included in this model is a year factor so that the interaction between year and the cubic polynomial in distance can also be estimated. In equation form and provided the reach is monitored for three or more years, the regression relationship is:

$$\begin{aligned} E[\, y_i \,] = {} & \beta_0 + \beta_1 x_{1,i} + \beta_2 x_{2,i} \\ & + \beta_3 d_i + \beta_4 d_i^2 + \beta_5 d_i^3 \\ & + \beta_6 d_i x_{1,i} + \beta_7 d_i^2 x_{1,i} + \beta_8 d_i^3 x_{1,i} \\ & + \beta_9 d_i x_{2,i} + \beta_{10} d_i^2 x_{2,i} + \beta_{11} d_i^3 x_{2,i} \end{aligned}$$

where y_i is the thalweg elevation measured at a distance of d_i meters from the top of the reach, $x1_{,i}$ is an indicator variable for year 1 (i.e., 1 if observation $_i$ was taken in year 1, 0 otherwise), and $x2_{,i}$ is an indicator variable for year 2 (i.e., 1 if observation $_i$ was taken in year 2, 0 otherwise). These models effectively fit separate cubic polynomials in d_i each year.

The analysis for change in thalweg residual variance is a statistical test designed to detect increased (or decreased) variance in residuals which is indicative of increased (or decreased) pool depths and complexity of the reach habitat. Thalweg residuals are defined as the residuals of thalweg elevation in the above regression model; $ry_i = yy_i - y\hat{}y_i$, where yy_i is observed elevation at distance d_i in year y and $y\hat{}y_i$ is the predicted elevation at distance d_i in year y. The test for change in thalweg residual variance is carried out using a modified version of Levene's test (Neter et al 1991). Absolute deviations of the residuals from their median are calculated as $d_{yi} = |r_{yi} - m_y|$, where d_{yi} is the absolute deviation associated with the i-th observation in the y-th year and m_y is the median of residuals in the y-th year. Levene's test entailed carrying out a one-way analysis of variance on the d_{yi}, with year defining the groups. Because the r_{yi} are potentially (spatially) correlated, the d_{yi} are also potentially correlated and the one-way analysis of variance is adjusted using the spatial regression techniques outlined in Appendix A of McDonald (1998). Variance of the original residuals is deemed significantly different across years if the (spatially adjusted) one-way analysis of variance rejected the hypothesis of equal average deviations. The distribution of thalweg residuals can be also plotted as a visual interpretation aid.

Both bank full and active channel widths are analyzed for changes across years. To conduct this analysis, a systematic sample of widths is computed from available data after field sampling is completed each year. Such a systematic sample of widths is necessary because the field sampling protocol dictated that each bank of the creek is measured separately. Consequently, width measurements are not taken completely across the creek, but rather from each bank to a center tape. Furthermore, measurements from one bank to the center tape are not necessarily in the same place as measurements to the opposite bank. Therefore width cannot be computed directly from the raw data and consequently a systematic sample of widths is computed and analyzed by the following methods. The systematic sample of widths is computed by first connecting left and right bank width measurements with straight lines to form an approximate stream channel. A random starting point along the center tape is then chosen and widths (across the whole channel) are computed at regular intervals along the center tape. The number of systematic points in the sample is equal to the smaller of the two sample sizes taken on each bank. For example, if 50 measurements were taken on the left bank and 75 measurements were taken on the right bank, 50 systematic measurements of width were taken to analyze. An example of the systematic sample of widths computed at Cañon Creek in 1996 is presented as Figure D-3 below.

The above described systematic sample of widths will be computed each year for each creek. Average width is analyzed using one-way analysis of variance (anova) techniques analogous to the modified Levene's test (Neter et. al. 1991) described for analysis of thalweg residual. A one-way analysis of variance (two sample t-test if only two years) is computed, with year as the grouping factor, to test for changes in mean stream width. Because measurements in the field are taken relatively close together and because spacing of the systematic sample of widths are relatively tight, computed widths are potentially correlated and consequently the analysis of variance can be modified to adjust for spatial correlations using the techniques outlined in Appendix A of McDonald (1998). This analysis of variance was parallel to the modified Levene's test described for analysis of thalweg residual variance.

Substrate size, or pebble size, is measured at approximately 10 sites within each monitored reach. Each site is approximately 50 feet by 50 feet in size and consisted of riffle bed areas within the stream. At each site, field personnel measure the secondary axis of rocks (pebbles) which are collected by selecting one near the toe of their right foot as transects were walked around the site. Collection and measurement continues until 150 rocks are measured. All measurements are reported in millimeters and the smallest measurement is one millimeter.

The distribution of pebble size is plotted and analyzed for changes across years assuming independence of the measurements. Due to the large distances (relative to average pebble size) at which rocks are measured and the fact that several independent systematic samples are taken at each site, spatial correlations among observations are highly unlikely and consequently no adjustments for such correlation are made.

The hypothesis of no change in distribution is tested using two sample Wilcoxon rank sum tests (Wilcoxon 1945; Hollander and Wolf 1973) or three sample Kruskal-Wallis tests (Lehmann 1975; Hollander and Wolf 1973) depending on the number of years data are collected from a stream. Substrate size measurements from all sites within a year are combined for testing because site to site differences in substrate size are not of interest and, if such differences existed, would tend to inflate the distributions variance and provide a conservative analysis. Treating the systematic measurements as if they were purely random (i.e., by assuming independence) also inflates the distributions variance and further contributes to a conservative analysis.

Three quantiles from each substrate distribution are estimated. The 16-th, 50-th, and 84-th quantiles are estimated from each distribution to facilitate comparison with sediment movement models previously developed (USEPA 2000). The 16-th quantile is defined as that point in the distribution which was greater than 16% of the observations and less than 84% of the observations. By symmetry, the 84-th quantile is defined as that point in the distribution which was greater than 84% of the observations and less than 16% of the observations. The 50-th quantile is defined similarly and corresponded to the median. The standard error of each quantile is estimated using standard bootstrap methods (Manly 1997).

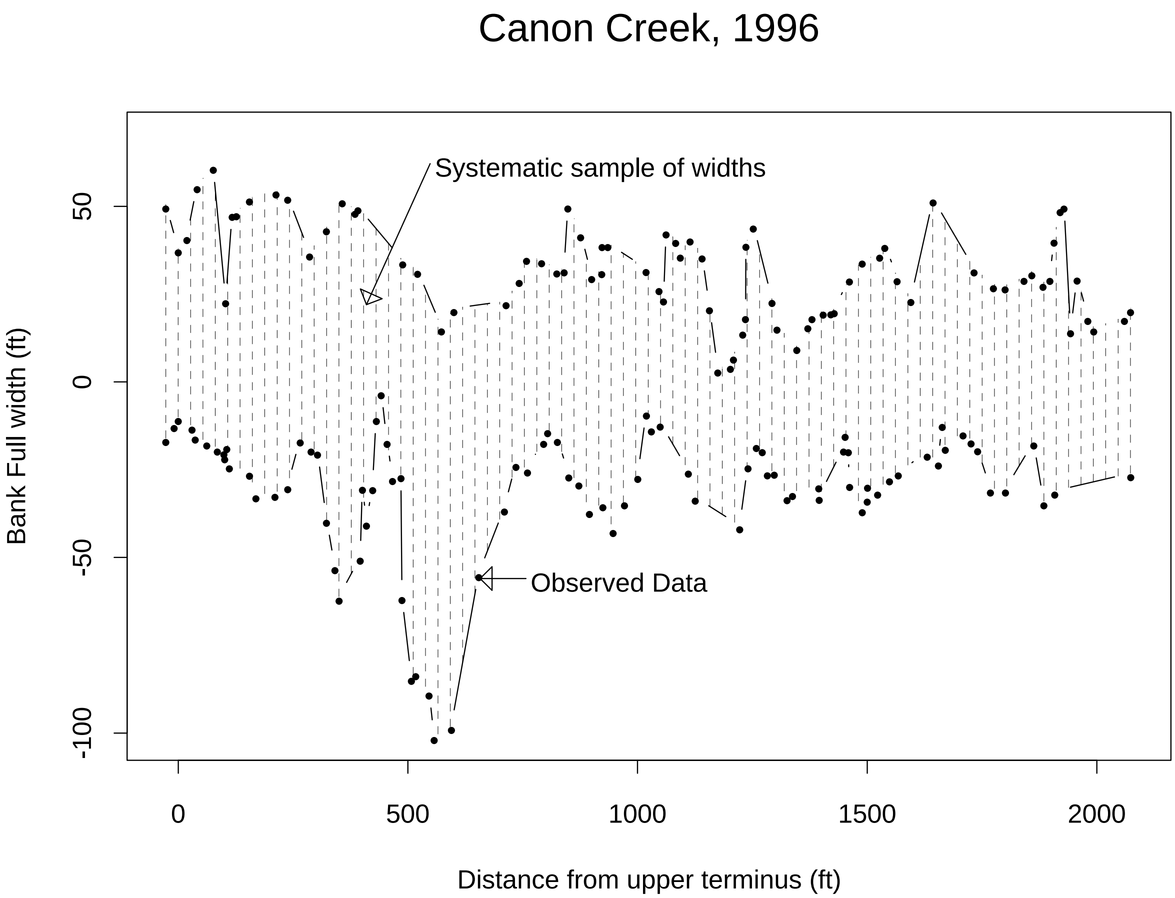

Figure D-3. Diagram of the systematic sample of widths taken for the investigation of width. This example shows bank full width at Cañon Creek in 1996. The zero in vertical dimension represents the center tape while negative numbers represent the left bank and positive numbers represent the right. Dots are observed bank full measurements with linear interpolation between each. Dashed lines show the systematic sample of widths.

D.2.2.4 Literature Cited

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D.2.3 Class III Sediment Monitoring

D.2.3.1 Background and Objectives

Concerns have been raised that complete removal of trees from Class IIIs will result in destabilizing these headwater areas resulting in an upslope extension of the channel and increased risk of shallow rapid landslides. The mechanisms that could trigger these potential effects may not be fully mitigated by the existing forest practice regulations: loss of root strength in the soil column that could increase mass wasting, and increased incident precipitation and storm runoff that could increase mass wasting and fluvial erosion processes in Class III watercourses. There is some evidence suggesting the latter from Caspar Creek (Lewis 1998). The net effect is that there could be significant increases in sediment production from watercourses even though Class I and II watercourses may have ample buffer retention. Because the majority of a channel network is made up of the first order channels, the overall impact of destabilized Class IIIs may be quite large even though increased sediment delivery in any given Class III is small. There is also the concern that if a debris torrent is triggered from one of these Class III areas, there will be no opportunity for delivering LWD into the channel below if no trees are retained in the uppermost reaches of these watercourses. The role of LWD in erosion and sedimentation processes in Class III channels is also potentially significant. LWD provides sediment storage sites, controls channel grade by preventing channel bed erosion, and deflects and concentrates stream flow thereby both protecting banks from erosion and magnifying fluvial bank erosion processes.

There are few empirical data available to assess the magnitude of these potential problems in northern California forestlands. Based on the protocols used in the retrospective study the results from across Green Diamond's ownership between 1992 and 1998 of 100 Class III watercourses indicated that changes in Class III channels following timber harvest were subtle and indistinguishable from natural channel changes over time (see Appendix C3: *Assessment of Sediment Delivery from Class III Water Courses: A Retrospective Study*). There was no evidence of substantial changes in channel morphology (e.g. increased width, depth or "head cutting"), few slides or bank erosion and no evidence for debris torrents. However, inferences related to more subtle changes in Class III watercourses following timber harvest were not possible given the retrospective study design. A more detailed examination of the channel, pre-harvest, with subsequent multiple surveys post-harvest would be required to detect these subtle changes. As a result, Green Diamond initiated a prospective study of sediment delivery from Class III watercourses scheduled for harvest utilizing a BACI (before, after, control, impact) experimental design. The objectives are to monitor Class III watercourses to quantify the amount of sediment delivered from treatment channels following timber harvest relative to control channels. Quantification of sediment delivery will be estimated utilizing four basic approaches: 1) documentation of changes in channel morphology (e.g. channel width, depth, bank scour, head cutting along with landslides, debris flows and areas of bank scour); 2) monitoring of turbidity (suspended sediments) during storm events; 3) sediment traps placed on the stream bank at selected high potential sediment delivery sites, and 4) silt fences placed at the lower extent of watercourse below the harvest unit. Each of these techniques will quantify sediment delivery in different ways, and tend to be measuring a different component of the total sediment budget in Class III watercourses, but collecting the different protocols should provide a comprehensive evaluation of sediment delivery from these streams. This monitoring program will only be employed in the four basins that make up the Experimental Watersheds Program.

Appropriate biological objectives and threshold values for Class III sediment delivery cannot be determined at this time. Approximately five years of initial trend monitoring are expected to be necessary to set the appropriate biological objectives and threshold values. At the end of 5 years a review and evaluation of trend monitoring results will be conducted. In addition, at other times agreed upon with the consensus of the Services, periodic reviews will be conducted to evaluate progress in determining turbidity thresholds.

D.2.3.2 Channel Morphology (In-channel Survey)

This protocol is designed to estimate sediment delivery from Class III channels by quantifying changes in channel morphology. Even using a BACI experimental design, Green Diamond does not expect to be able to quantify subtle changes that might result from small of amounts of fine sediment inputs. However, this technique should provide good estimates of more significant sediment inputs and it will also allow one to assess the mechanism of the sediment delivery.

Before going into the field, delineate the Class III channel on the proposed THP map to determine the drainage area. A minimum survey length will be 200 feet. In the field, assess the watercourse beginning at the lowest point on the channel within the THP unit.

This point may be at the culvert inlet on a road crossing or at a Class II/Class III break. Take channel measurements systematically up the channel at 10-foot intervals based on a random start within the first 10-foot interval. At each 10-foot sampling interval, if an active channel is evident, measure its width, maximum depth, and determine if there is evidence of recent scour (sediment erosion by fluvial processes). Also measure the linear length of exposed bank within 15 feet of the channel on both banks. If the exposed bank is part of an earth flow or slide, measure the entire limits of the exposed ground. Game trails and animal burrows are not included in measurements of exposed banks, but their occurrences should be noted. In order to facilitate subsequent re-surveys of the channel following timber harvest, install benchmarks along the channel at 25-50 foot intervals. Scribe the in-channel distance and benchmark number onto the tag.

At every 50-foot interval, measure the bank angle perpendicular to the channel on the left and right banks. At every 100-foot interval, measure the mean understory vegetation height and percent overstory canopy closure using a densiometer. Measure the channel gradient with a clinometer at the beginning of the layout and at all major breaks in slope throughout the remaining channel length. Measure the diameter and length all large woody debris (LWD) greater than 6-inch diameter wherever it occurs throughout the channel. (There is no minimum LWD length requirement.) Record if the LWD is hardwood or conifer. The LWD classification is intended to give an indication of its expected longevity within the channel. If the classification cannot be determined, default to the hardwood classification under the assumption that the piece of wood is rapidly decaying. Also note if the LWD is acting as a control point. A control point is any in-channel feature that retains sediment and/or prevents headcutting with a minimum of a 6-inch drop. Record the location and type of all other control points (roots, boulders, bedrock, etc.). Include the dimensions of the control point, vertical drop, scour below the control point and note the predominant channel substrate. Measure the area of all significant channel scour holes (hole in the channel > one foot in depth where, when there is flow, the flow would go subsurface) or other major in channel areas of scour. Benchmark all major control points (>1 foot drop), scour holes or other major in channel areas of scour. The benchmark needs to be designated in such a way that it will allow for an accurate assessment of changes in both the area and depth of these features. In addition, benchmark and construct a cross-section for any areas of significant entrenchment (>l foot depth). Cross-sections are constructed by first setting two fixed points that establish a line perpendicular to the desired site. The fixed points (benchmarks) must remain in place without any movement throughout the entire monitoring period so aluminum tags are typically attached by nails to large stumps or stable large woody debris with nails. A line is affixed between the two points and leveled using a line level. Depth measurements are taken at intervals along the fixed line using a stadia rod. Accompanying the depth measurement is a distance measurement taken from one of the designated fixed point (primary benchmark) to the various depth measurement points on the fixed line. If only one fixed point can be established due to a lack of suitable stable structures, then a different method is used. A primary fixed point is placed on one stable object (e.g. stump or LWD) “distal” to the cross-section. A secondary fixed point is then placed “medially” on the same stable object such that the line passing through the two points forms the desired cross-section. The line is leveled as above and depth measurements are taken at fixed intervals as described above.

Photo document the channel both upstream and downstream at the beginning, middle and top of the channel. In addition, photo document at major gradient breaks in the channel that precludes visibility, major control points, channel scour holes, significant mass wasting, or other major features that affect the channel. Note the presence and flow of water, changes in predominant vegetation and the occurrence of any aquatic vertebrates.

Continue the in-channel survey until the Class III channel ends at a headwall or spring, or at the harvest unit boundary, if the channel is a “run-through”. Survey the associated road system within the sub-basin and sketch the drainage area onto a topographic map. Record any stream piracy or diversions associated with the road system and include it in the drainage area. On the topographic map, record road failures, inner gorge slides or other larger scale sediment delivery features within the sub-basin.

D.2.3.3 Sediment Traps

This technique is designed to estimate delivery of sediment from stream banks by direct overland movement. The traps are set to capture a portion of the sediment being delivered directly to the channel through bank erosion (raveling or colluvial inputs). The technique does not allow for an estimate of the total sediment being delivered to the stream by this process, because it not possible to adequately estimate the total “input zone” for each trap. Rather, this technique is designed to estimate changes in the delivery rates between treatment and control streams, before and after harvest.

To maximize the potential to gather samples of sufficient size to allow for quantification and statistical analysis, the watercourse is first assessed prior to the placement of any of the sediment traps. All of the sites with highest potential to deliver sediment (with the exception that active slides need to be avoided) are flagged, and beginning with a random start, the sediment traps are distributed systematically at sites such that the entire length of the watercourse is sampled. During placement of the sediment trap, consider micro-topographic features to maximize collection of bank material that is mobilized, and to allow for an assessment of the micro-drainage area for each trap.

Set up the trap above the high water level but as close to the channel as possible. Sediment collected in the trap is assumed “delivered” to the Class III watercourse. This assumption may be violated if the trap is placed too far from the edge of the channel, because there is a possibility that the collected sediment actually would not have reached the channel. At the selected trap site remove the small organic debris so a tight seal is achieved between the ground surface and the edge of the trap. Next, push the leading edge of the traps into the hill slope. Position the slope of the trap so that it is sufficiently steep to insure that sediment will be carried into the collection bucket. Measure and record the slope of each sediment trap to insure that they are all placed at a similar slope. Drive rebar through the retaining rings on the trap and into the ground to stabilize the trap. Place a collection bucket at the outfall of the trap to collect the sediment generated by surface erosion. Place a plywood cover over the trap and collection bucket, to avoid collecting rainfall. At each sample site measure and record the following information: micro-drainage area above the trap, bank slope, distance of exposed soil above trap and canopy closure.

Check the sediment traps after every storm event that exceeds 1 inch of rainfall. Discard the first sample following the initial trap setup to “clear” material that was mobilized by the installation. Pour the collected sample through a number 230 testing sieve and transfer the sediment into a sample bag. In the field, measure the total volume of water that was collected. Bring the sample bags back to the lab for analysis.

Record precipitation from the rain gage that was placed in the vicinity of the monitoring site. Sediment bags are dried and weighed prior to taking samples. After sampling, the bags are dried and reweighed in the laboratory. To obtain sediment weight, subtract the empty bag weight from the total weight of the bag with the sediment.

D.2.3.4 Turbidity and Suspended Sediment Sampling

Class III channels only flow in response to storm events, and by definition, are capable of transporting sediment to receiving Class I and II watercourses, but do not support "aquatic life". Sediment sampling will take place during storm events, since turbidity and suspended sediment are highly dependent on discharge and the vast majority of sediment transport occurs during high flows. The turbidity and suspended sediment sampling element of the Class III monitoring program was designed to determine the validity or accuracy of the sediment traps in quantifying the sediment contributions from timber harvesting activities. In addition, it will measure suspended sediment and turbidity generated from in-channel scour and remobilization of stored sediments. The latter sediment contributions should correspond to changes detected from the in-channel survey.

Grab samples will be taken at the downstream end of the Class III channel (but above the silt fences) within the BACI unit. Automated samplers and depth-integrated samplers will not be used, since these watercourses are generally very shallow and only flow during storms. Water samples are taken from a well-mixed area of the watercourse using a 0.5 L plastic bottle and stream discharge is measured at the same location. It also will be noted if the inboard ditches are contributing flow. The grab samples are analyzed for turbidity and suspended sediment in the laboratory.

A storm event that is expected to deliver 1 inch of rain will trigger the crews to collect turbidity and suspended sediment samples. Repeat measures are taken during the rising limb, peak, and falling limb of the hydrograph. Following the sampling period, record the rainfall amounts from the rain gages located in the vicinity of the BACI unit.

Turbidity is measured making sure the sample is well mixed. Filter papers (Whatman glass microfibre filters) are labeled, dried and weighed. Volume of sample is measured before sample is poured through the vacuum filtration system. Filter papers are removed, dried and reweighed. Sediment weight is post filtration weight minus original filter paper weight.

D.2.3.5 Silt Fences

The final portion of the Class III sediment monitoring includes estimating fine sediment production using silt fence check-dams. The relatively small size and low ephemeral discharge of Class III channels makes it possible to attempt to construct relatively low cost, low maintenance sedimentation basins using silt fence material. The principle behind this approach is that the silt fence check-dams act as a velocity break to the flow in the channel, which allows suspended sediments greater than some particle size to be deposited above the fence. The actual particle size that is deposited depends on the size of the silt fence and the degree to which it impounds the flow. These data will be used primarily to correlate with the turbidity (suspended sediment) sampling to determine consistency between the two methodologies. If the turbidity sampling correlates well with the silt fence results, it may be possible to eliminate the more labor-intensive turbidity sampling.

The proposed design will include three successive sedimentation basins created by silt fences in close proximity to accommodate potential overflow as the silt fence pores become clogged with sediment (Britton et al. 2001). Successive basins will provide for additional capture of sediment in the flow. In addition, if the upstream basins fail, downstream basins will be in position to capture flow and sediment.

The design of sedimentation basins will include steel rebar and sections of chain link fence to support the silt fence. Silt fence will be fastened to the bed and banks to prevent flow under and around the structure. It is expected that there will be some leakage around the edges, adding another design purpose in setting up three successive basins. The data to be collected seasonally is the dry weight of sediment accumulated in the sedimentation basins. Colloidal material will not be collected, but most silt, sand and gravel should be captured. Data on soil particle size distribution will be collected to estimate the efficiency of the sedimentation basins.

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D.3 LONG-TERM TREND MONITORING/RESEARCH

D.3.1 Introduction

Long-term trend monitoring includes:

- Road-related mass wasting monitoring,
- Steep streamside slope delineation study,
- Steep streamside slope assessment,
- Mass wasting assessment,
- Long term habitat assessments,
- Large woody debris (LWD) monitoring,
- Summer juvenile salmonid population estimates, and
- Out-migrant trapping.

The long term trend monitoring projects are those monitoring projects for which no thresholds for adaptive management are set. For some projects, this reflects the multitude of factors which affect the response variables, in others, the long time scales required to distinguish the ‘noise’ from the underlying relationships. Research projects designed to reveal relationships between habitat conditions and long-term persistence of the covered species are also included in this section. Each of these projects has the potential to provide feedback for adaptive management, but in some circumstances, decades may be required before that can occur.

D.3.2 Road-related Mass Wasting Monitoring

D.3.2.1 Background and Objectives

Roads can lead to increases in the frequency and severity of all types of mass soil movement. Increased sediment inputs to streams can in turn negatively impact all six of the Covered Species. The road upgrading and decommissioning process described in Section 6.3.3 is expected to significantly reduce the frequency and/or severity of road related mass wasting sediment inputs. As such, it is an integral component of the suite of conservation measures designed to achieve the biological goal of reducing management-related sediment inputs to Plan Area streams.

The road-related mass wasting monitoring project will monitor the effectiveness of the road upgrading and decommissioning measures in reducing the frequency and severity of road related mass wasting inputs. This will involve before and after monitoring of particular road segments, comparisons within basins or sub-basins of treated and non-treated roads, and Plan Area wide comparisons of treated and non-treated roads. If no significant effect (i.e. reduced frequency and severity of road-related mass wasting inputs) can be attributed to the road upgrading and decommissioning measures, the monitoring results will be used to adjust and revise the road upgrading and decommissioning measures to improve their effectiveness.

D.3.2.2 Site Selection

The road-related mass wasting monitoring project will be employed in the four basins that make up the Experimental Watershed Program. Various road segments representing different categories of road use and road condition will be selected for monitoring. The categories will be seasonal versus rocked, low (or moderate) versus high-use, upgraded versus scheduled for upgrades (not yet upgraded) and decommissioned versus scheduled for decommissioning (not yet decommissioned. The goal will be to have a minimum of 12-15 crossings in each road category selected for monitoring. Within a given experimental watershed, watercourse crossings or road related landslide features to be sampled will be selected from all of the combinations of road use and condition categories using a stratified random sampling approach. Within a given selected road segment, the individual crossings or road related landslide feature to be monitored will be selected using a systematic sample with a random start. For example, assume that a 20% sample achieves the desired sample size for a given road use and condition category. Then all of the sites that have or will be upgraded along a selected road segment will be identified. A random starting point will be selected from the first 5 sites with every fifth site systematically selected for sampling beyond that point.

D.3.2.3 Field Measurements

Road related mass wasting sediment inputs to streams are episodic in nature and typically triggered by intense rainfall events. As such the sample sites will be resurveyed the summer following a flow event with a 5 year return interval. The volume of sediment delivery that occurred from each sample site will be determined. The time scale required to collect enough data and accurately assess the effectiveness of road upgrading and decommissioning may be on the order of decades.

D.3.3 Steep Streamside Slope Delineation Study

The goal of the Steep Streamside Slope (SSS) Delineation Study is to determine the minimum slope gradient and maximum slope distance of SSSs for each HPA. The initial default minimum slope gradients and maximum slope distances for the HPA Groups will be adjusted for each HPA based on the results of this study.

The quantitative criteria for determining SSS minimum slope gradients and maximum slope distances will be the same as described Section 6.3.2.3. The minimum slope gradient will be based on an 80% cumulative sediment delivery volume from streamside slopes in the all HPAs. The maximum slope distance will be based on a 80% cumulative sediment delivery volume from streamside slopes in the Blue Creek and Coastal Klamath HPAs, and 60% cumulative sediment delivery volume from streamside slopes in all other HPAs.

Initially, the procedure will be based on the assumptions described in Section 6.3.2.3 and it will utilize similar methods as were employed in the three pilot watershed areas to determine the initial default SSS slope gradients and distances. This will include conducting an office-based Steep Streamside Slope and landslide inventory using aerial photographs and published geologic maps, designing a statistically valid field-based data collection program based on the SSS and landslide inventory, field verifying the office-based SSS and landslide inventory, collecting geologic data (e.g. landslide-related information or lithologic data during on-site review), data analysis, reporting results and implementation of adaptive SSS slope gradients and distances.

In order to collect data that will allow statistical inferences to be made that will apply to the entire HPA, it will be necessary to sample study sites across the HPA using a probability based sampling design that is spatially distributed. The specific sampling design has not been determined yet, because the sampling frame or acceptable levels of variance in the estimates has not been set. Once this has been done, there are a variety of possible sampling schemes that will achieve the objective of obtaining a statistically valid sample from which to draw inferences to the entire HPA, and the specific sampling scheme selected will be based on minimizing variance and while maximizing efficiency of data collection. Data collection will emphasize landslide type, landslide crown distance to watercourse, natural pre-existing slope gradient, geologic and geomorphic setting, and land-use or management history. Causal mechanisms for individual landslides may also be assessed.

The SSS Delineation Study for each HPA will be completed with priority given to completing the HPAs that are anticipated to have substantial timber harvesting operations in the near future.

The SSS Delineation Studies for all 11 HPAs will be completed within 7-years following the effective date of the Permits. The modified slope and distance criteria for each HPA may be applied starting on the 30th day after a letter of notice with a summary map that summarizes the data and describes the findings of the data analysis for each HPA is sent to The Services. Subsequent updates to the SSS Delineation Study for each of the HPAs will be conducted depending on climatic cycles and landscape response.

The adaptive management account will not be credited or debited based on the results of the first SSS Delineation Study for each HPA following Plan approval. Instead, the baseline for credits or debits to the adaptive management account will be reset according to these results. The subsequent modifications to SSS maximum slope distances and minimum slope gradients will be handled through the adaptive management account.

D.3.4 Steep Streamside Slope Assessment

The goal of the SSS Assessment is to determine the effectiveness of SSS prescriptions and to recommend appropriate changes to the SSS conservation measures, if any such change is necessary, that will more closely achieve the effectiveness goal of the SSS conservation measures. The SSS conservation measures are designed to be at least 70% effective at preventing management -related sediment delivery from landslides compared to that from appropriate historical clear-cut reference areas. A maximum of a 30% relative increase in landslide-related sediment delivery compared to merchantable-sized, advanced second-growth uncut SSS areas may be used as another comparative standard to determine the effectiveness of the conservation measures. The objectives of the SSS Assessment are to collect data relevant to landslides in SSSs and to determine the effectiveness of the SSS conservation measures by comparative analysis of cumulative sediment delivery volumes and associated data. The procedure will utilize similar methods as were employed in the three pilot watershed areas to determine the initial default SSS slope gradients and distances. For each HPA, this will include conducting an office-based Steep Streamside Slope inventory and a landslide inventory using aerial photographs and field surveys, designing a statistically valid field-based data collection program (as described for the SSS Delineation Study), field verifying the office-based SSS and landslide inventory, collecting field data, data analysis, reporting and implementation of adaptive SSS slope gradients and distances.

A California Registered Geologist (R.G.) will oversee data collection. Data collection is expected to focus on landslide location and type, geologic composition and setting, distance of landslide crown from watercourse, pre-existing natural slope gradient, landslide dimensions, volume of sediment delivery, land-use or management history, and causal mechanisms. Other data parameters may also be collected based on the professional discretion of the supervising R.G. All data will be stored in a database and appropriately represented on maps in order to facilitate data analysis.

Data analysis to determine the effectiveness of the prescriptions will be performed by a scientific review panel, which will consist of independent experts on the subject at hand. The panel will have three members, one appointed by the Services, one appointed by Green Diamond, and a third selected by the first two panel members. The analysis will be performed after the 15th winter following the effective date of the Permits.

The role of the scientific review panel will be to provide technical analysis of the data and to attempt to reach conclusions on the effectiveness of the SMZ prescriptions relative to the goal of the SMZ conservation measures. The criteria for determining appropriate modifications to the SMZ conservation measures, if any modification is necessary, will be based on the comparison of the cumulative sediment delivery volumes from harvested SSS, unharvested SSS, and historically clearcut SSS. Modifications to the initial default prescriptions can range from clear-cut to no harvest and may vary from HPA to HPA and possibly within individual HPAs. Modifications will not be made to the default SSS prescriptions unless the analysis is conclusive in the opinion of a majority of the scientific review panel.

If the results are not conclusive, the monitoring protocol will be evaluated to ensure that appropriate methodologies are being applied and the monitoring will be extended for another 5 years. Any adjustments to the conservation measures will be in keeping with the Adaptive Management Reserve Account and changed circumstances. For comparative purposes, harvested SSS may be subdivided into those areas harvested using the default prescription and those areas harvested using alternative prescriptions developed through onsite geologic review. Historical clearcuts may be used as a comparative standard to determine the effectiveness of the conservation measures. Unharvested or advanced second growth stands may be used to represent background landslide-related sediment delivery rates as a comparative standard to determine the effectiveness of the conservation measures. Both harvested and unharvested SSSs may also be subdivided for comparison according to geologic conditions, forest stand type, management zone (RSMZ and SMZ) land-use, and other sub-groupings as may be appropriate, in order to ascertain the most meaningful results in each HPA or subunit thereof. If modifications are made to the initial default SSS prescriptions, the Services will be notified prior to the implementation of the modified prescriptions.

D.3.5 Mass Wasting Assessment

Green Diamond will conduct a property-wide Mass Wasting Assessment (MWA) within 20 years. The Goal of the MWA is to examine relationships between mass wasting processes and timber management practices. The objectives of the Mass Wasting Assessment are to collect a thorough data set that represents a wide range of mass wasting processes and management practices, to analyze the data, and to present the results in a report or in several reports. The results of the MWA will not be subject to the adaptive management mechanisms provided by the plan.

A preliminary MWA will be completed within 7 years of the effective date of the Permits. The preliminary MWA will primarily include a landslide inventory and some statistical reporting with limited comments and discussion. The landslide inventory and analysis will generally follow the procedures outlined in the Washington State Department of Natural Resources (WDNR) methodology for mass wasting analysis, with some modifications. Modifications to the WDNR method may be implemented based on data or at the professional discretion of the supervising geologist.

The final MWA will be complete in 20 years of the effective date of the Permits. The final MWA will include updating the preliminary data and it will attempt to identify patterns or trends in mass wasting processes as they relate to management practices. The final MWA will be presented in a report or in several reports.

Green Diamond and the Services will jointly review the final MWA results to determine if the MWA Assessment should continue. If The Services and Green Diamond cannot reach agreement on the finality of the MWA, a scientific panel shall be convened to determine if continued slope stability monitoring is necessary. If the scientific panel is required, the panel shall be convened in the same manner and generally follow the same procedure as the panel for the SSS Assessment.

D.3.6 Long-term Habitat Assessments

D.3.6.1 Background and Objectives

Channel and habitat typing assessments were previously conducted by Green Diamond personnel during 1994 and 1995 following CDFG methods (Flosi and Reynolds 1994; and Hopelain 1994). Sixteen streams within the Plan Area were assessed identifying 75 reaches by channel type, for a total of nearly 104 miles of stream channel. Additional channel and habitat typing assessments on Plan Area streams have also been conducted by the Yurok Tribal Fisheries Program (YTFP), the California Conservation Corp (CCC), the Louisiana Pacific Corp., and CDFG. Those parties have conducted assessments on 42 streams, covering 140 reaches for a total of 131.0 miles of channel being assessed. All streams assessed were selected based on their biological significance as producers of salmonids, and the size of Green Diamond's ownership in the watershed's anadromous reaches.

Future channel and habitat assessments will be conducted to provide information about the health of these streams, especially with respect to salmonid habitat. Channel and habitat variables including the following will be collected:

- Percent canopy cover
- Percent LWD as structural shelter
- Habitat types as a percent of length
- Dominant substrate composition
- Pool embeddedness
- Pool depths
- Shelter rating in pools

The trends observed through this long term, comprehensive assessment will be valuable for comparison with the results of the other more specific monitoring projects. The habitat assessment monitoring project will ensure that the individual biological objectives (i.e. permeability, channel dimensions, water temperature monitoring projects), are accurately depicting overall aquatic habitat health and function.

The channel and habitat assessment process will be repeated on the original 58 surveyed streams every 10 years for the life of the Plan. As the first assessments were completed in 1994 and 1995, the next assessment will be conducted in 2004 and 2005. Detection of significant trends will probably require at least a third assessment beginning in 2014 and 2015. The channel and habitat typing reaches are distributed throughout Green Diamond's entire ownership except for properties in Trinity County. Each assessment will identify the channel types and habitat features in the particular stream assessed. The objective of the Habitat Assessment Monitoring Project is to document long term trends in habitat quality and quantity across the ownership.

D.3.6.2 Methods

To evaluate salmonid stream habitat value and quality, channel and habitat assessments for streams that are known to have historically contained coho salmon will be conducted. Brief inspections (spot checks) for fish presence will also be conducted at the same time at the lower, middle, and upper reaches of the streams assessed. These assessments will be utilized to assess channel conditions in anadromous reaches of Plan Area streams. This protocol is based on the CDFG Habitat Inventory Methodology as described in the California Salmonid Stream Habitat Restoration Manual, (Flosi et al. 1998) and the FFFC Channel and Habitat Typing protocols (FFFC 1997). The assessment of anadromous streams will consist of the following primary components: 1) channel classification, 2) habitat typing, and 3) riparian vegetation assessment

D.3.6.3 Channel Classification

The channel classification data will be utilized to describe specific stream reaches by channel type and sequence within a watershed. This will help predict a stream's behavior from its appearance (e.g. predicting a channel's response to upstream sediment inputs). The method will assist in stratifying streams by channel types for conducting subsequent habitat typing surveys.

Streams will be classified using the system developed by Rosgen (1994) and will use the following eight morphological characteristics to describe the stream channels:

- Channel width
- Depth
- Velocity
- Discharge
- Channel slope
- Roughness of channel materials
- Sediment load
- Sediment size

The stream channel delineation criterion includes general description, width/depth ratio, water surface slope/gradient, dominate particle size, entrenchment, and sinuosity. Descriptions and definitions of these classification criteria are found in the CDFG Salmonid Stream Habitat Restoration Manual (Flosi et al. 1998). Field data will be entered on standardized worksheets using the instructions and methods for completing the stream channel type worksheets provided in (Flosi et al. 1998). The results of this classification will result in the categorization of the target stream reaches into 1 of 34 single thread channel stream types or 1 of 7 multiple thread channel types (Rosgen 1994).

D.3.6.4 Habitat Typing

The stream-level habitat typing data yields the most detailed information of the assessment methods. Habitat typing of a watershed's anadromous reaches provides information that physically describes the anadromous habitat within the wetted channel. Habitat typing reveals factors that may limit production of salmonid smolts. These assessments also facilitate planning, prioritizing, and implementing fisheries restoration projects. Finally, habitat typing evaluates habitat responses to restoration efforts.

Habitat typing will be conducted on the entire target stream from mouth to the upper extent of anadromy using CDFG methods as specified in Flosi et al. (1998). These methods are a variation of a system originally developed by Bisson et al. (1982) and modified by others. Level II habitat typing will be conducted to describe the specific pool, flatwater, and riffle habitats within each target stream. Each habitat unit type is determined based on riffle or pool type and location. The following variables are measured for each habitat unit:

- length, width, depth of pools and riffles
- shelter rating based on shelter complexity
- substrate composition including percent exposed
- percent canopy cover
- percent coniferous and deciduous trees
- pool tail
- bank attributes

The level II habitat typing will describe each habitat unit and categorize into the following habitat types:

- Riffle:
    - Low-gradient riffle
    - High-gradient riffle
    - Cascade
    - Bedrock sheet

- Flatwater:
    - Pocket water
    - Run
    - Step run
    - Glide
    - Edgewater

- Pool:
    - Plunge pool
    - Mid-channel pool
    - Dammed pool
    - Step pool
    - Channel confluence pool
    - Trench pool
    - Lateral scour pool
    - Root wad enhanced
    - Boulder formed
    - Bedrock formed
    - Log enhanced
    - Corner pool
    - Secondary channel pool
    - Backwater pool-boulder formed

Root wad formed
Log formed

Habitat inventory data will be collected and recorded onto standardized data sheets following the instructions provided by the CDFG Salmonid Stream Restoration Manual (Flosi et al. 1998). Data will be entered into a data management system (Access) for subsequent analysis using the CDFG developed program HABITAT©.

D.3.6.5 Riparian Vegetation Assessment

A riparian vegetation assessment will be conducted for each target stream. This consists of a large organic debris (LOD) survey. This survey will be conducted in 200 foot sections to cover a minimum of 20% of each channel type in each target stream. Variables measured will include:

- all LOD within 50 ft. of each bank tallied
- percent bank slope
- dominant vegetation/LOD percent and type
- large debris accumulation (noting those that retain gravel upstream)

D.3.6.6 Field Survey

Surveys are conducted by two person teams and are begun at the downstream end of the stream reach. The surveys continue by walking upstream and measuring the variables throughout the length of the entire survey reach. All data are collected on standardized data forms while in the field. For each habitat unit, its length is measured and recorded. When conducting the habitat typing inventory all variables are measured and recorded for each first-time encounter of each habitat type in a channel type, starting with the units above the hydraulic influence of its receiving stream. All variables for all randomly selected habitat units are measured and recorded. These include depths, widths, and embeddedness in all pool habitats.

D.3.6.7 Literature Cited

Flosi, G. and F.L. Reynolds. 1994. California salmonid stream habitat restoration manual. Second Edition. IFD, CDFG, Sacramento, CA.

Flosi, G., S. Downie, J. Hopelain, M. Bird, R. Coey, and B. Collins. 1998. California salmonid stream habitat restoration manual. Third Edition. IFD, CDFG, Sacramento, CA.

Hopelain, J. 1995. California salmonid stream habitat restoration manual. IFD, CDFG, Sacramento, CA.

D.3.7 LWD Monitoring

D.3.7.1 Objectives and Background

The objectives of the LWD monitoring are to document long term trends in the abundance and size class of inchannel and potential LWD under this Plan.

The development of potential LWD in riparian areas throughout the Plan Area is relatively predictable. Green Diamond has projected future stand composition in riparian zones through the life of the plan. In contrast, the recruitment of potential LWD into the stream (inchannel LWD) is a highly stochastic process that occurs over long time scales. For this reason, the LWD assessment project does not lend itself to be used as measurable thresholds for adaptive management. The conservation measures as a whole are expected to increase potential LWD, and may increase inchannel LWD, over the life of the Plan, and this monitoring project will document whether this expectation is met.

LWD inventories have been conducted previously on fifteen streams distributed throughout the Plan Area. Information regarding the distribution of LWD was also obtained in the channel and habitat typing assessment process, but the importance of LWD to biological and physical processes in the stream channel justified the need for a more thorough assessment of this critical habitat component. The LWD inventory covers two distinct zones:

- LWD within the bankfull discharge area of the stream channel; and

- LWD and live trees within the "recruitment zone," defined as the area encompassing the floodplain and 50 feet of the hillslope beyond the bankfull channel margin.

The objectives of the LWD inventory include:

- Accurately documenting the current abundance, distribution, and characteristics of instream LWD.

- Providing a repeatable methodology for monitoring long-term changes in the abundance, distribution, and characteristics of instream LWD.

- Accurately identifying the source of instream LWD (naturally recruited or restoration structure) and the species composition of instream LWD (hardwood or conifer).

The LWD inventory will be conducted using the CDFG methods (Flosi et al. 1998). This methodology was designed with the objective of quickly identifying stream reaches lacking in LWD for prioritizing restoration projects. After analyzing previously collected data on Green Diamond properties, it became clear that the following modifications to the in-channel CDFG methodologies were necessary to meet Green Diamond’s objectives:

- A 100% inventory of LWD instead of a 20% sub-sample;

- A more precise breakdown of LWD size classes;
- Identification of LWD as either deciduous, conifer, or redwood; and
- Designation of LWD as naturally recruited or as stream enhancement structures.

D.3.7.2 Methods

Personnel conducting the LWD inventories will be familiar with channel typing methods of Rosgen (1996) and the equipment needed to conduct LWD inventories. Training and daily sight calibration will be conducted as needed to assist in categorization and recording of field data. Equipment required for LWD inventories includes:

- Clinometer
- Hip chain
- 50' diameter tape
- Waders
- Clipboard and data forms

Inventory teams consisting of 2 people will walk upstream within the stream channel recording LWD information as they proceed upstream. One team member inventories the defined right bank and the stream channel while the other member inventories the left bank. LWD inventories will be conducted after stream habitat typing surveys have been conducted and channel habitat types and lengths have been determined. LWD inventories will be conducted throughout the entire length of the anadromous reach of all streams inventoried.

Standardized LWD data forms will be used to capture inventory data. Inventory data collected are that described by Flosi et al. (1998). The LWD data will include tallies of diameter and length categories, and the condition (e.g. live, dead, perched), and wood type (e.g. conifer, deciduous, redwood).

D.3.7.3 Literature Cited

Flosi, G., S. Downie, J. Hopelain, M. Bird, R. Coey, and B. Collins. 1998. California salmonid stream habitat restoration manual. Third Edition. IFD, CDFG, Sacramento, CA.

Rosgen, D.L. 1996. Applied River Morphology. Printed Media Companies, Minneapolis, Minnesota.

D.3.8 Summer Juvenile Salmonid Population Estimates

D.3.8.1 Background and Objectives

The objectives of the summer population estimates are to estimate summer populations of young-of-the-year coho and age 1+ and older steelhead and cutthroat trout, and to track trends in these populations over time. This protocol has been modified from previous methodologies to provide more consistency between individual crews and from year to year. The definition distinguishing deep or shallow pools has been modified so that determination is made solely on depth. A pool less than 1.3 meters is considered a shallow pool regardless of cover. This provides better consistency between crews, allowing comparisons of population estimates between different streams, crews, and property owners.

The sampling and process variance associated with the population estimates and the uncertainty related to the possible causes of observed long-term trends preclude the use of summer population estimates as measurable thresholds for adaptive management purposes. While changes (positive or negative) in summer population estimates will clearly be a source of interest, it remains unclear what, if any, changes can be related to management. The summer population data, in combination with other monitoring efforts, may provide valuable information about the relationships between coho populations in different streams throughout the Plan Area, and the climactic and/or habitat conditions which affect summer population size. In addition, trends in summer population estimates will be valuable in determining the recovery status of the coho populations within the Plan Area.

The protocol for estimating summer populations of young-of-the-year coho salmon and yearling or older steelhead was developed by Dr. W. Scott Overton (Oregon State University, retired) and Dr. David G. Hankin (Humboldt State University) and is that of the Fish, Farm and Forest Communities (FFFC). The methodology is an extension of earlier sampling designs developed, in part, by Dr. Hankin (Dolloff et al. 1993) that utilized a combination of direct observation counts and electrofishing. This protocol relies less on electrofishing to calibrate dive counts, instead employing multiple-pass dives for calibration. Electrofishing is still utilized to calibrate a proportion of the dive units (in habitat units with 20 or more fish of each species counted on the initial dive pass).

D.3.8.2 Methods

The methods were modified from Hankin and Reeves (1988) single stream fish population estimate. The summer population estimation method allows for increased use of diver counts for estimating the abundance of juvenile salmonids in streams. This approach reduces the need for electrofishing and related possible mortality of special status species (e.g. coho salmon).

The first phase of the sampling design:

- classifies habitat units into riffles, runs, pools, and deep pools,
- measures dimensions of each unit,
- randomly selects a fraction of units in each habitat class for Phase 1 sampling (employing the adaptive sequential independent sampling [ASIS] method [Hankin, in press]).

Phase 1 sampling consists of diving each selected unit to obtain an initial count of fish within that unit. Riffle segments are electrofished as diving cannot be conducted in riffles. A subset of the sampled units is then randomly selected for calibration using the

ASIS method. The mode of calibration (2nd phase sampling) is determined by the following procedure. If the initial dive counts of the target species is less than 20 individuals then calibration is conducted by a bounded count methodology (Robson and Whitlock, 1964) using 3 additional independent diver counts. If the initial dive count of the target species exceeds 20 fish, then calibration is made by four-pass removal electrofishing method. Calibration within deep-pool stratums is made only by diver counts, as electrofishing is inefficient in this stratum. In riffles selected for calibration, a 2 to 3 pass-removal electrofishing method is the mode of calibration.

If the method of bounded counts is the mode of calibration, the 3 additional dive counts are made immediately following the Phase 1 dive counts. If the Phase 2 sampling is conducted by the 4 pass-removal electrofishing method the electrofishing is conducted within no more than 2 days following Phase 1 sampling. The methods employed for sample selection and estimation, the ASIS methodology, and Phase 2 calibration methods are those of Hankin (in press). Additional discussion of the applicability and assumptions of the population estimation methodology employed by Green Diamond are found in (Hankin, in press).

D.3.8.3 Fish Survey: Phase I

The initial fish counts are obtained by snorkeling each of the flagged shallow pools, deep pools, and runs while progressing upstream to each successive unit. The diver(s) will enter the unit to be surveyed from the lower end without disturbing the fish and progress through the length of the unit counting the fish as they go. A "clicker" will be carried to record fish numbers for abundant species. Fish counts will include 0+ coho, 1+ coho, "trout" and other species in the survey. The presence of 0+ trout may be noted, but not counted. The length of time that it takes to complete the snorkel count is recorded in case the unit is selected for calibration. Following completion of a snorkel count of the fish in a designated unit, the appropriate Phase II ASIS number is drawn to determine if the surveyed unit will be selected for calibration. It is critical that the diver(s) doing the initial pass in the unit do not know if the unit is to be calibrated prior to doing the dive.

D.3.8.3.1 Phase I Snorkel Survey

The snorkelers will record the following data:

- Unit Number-The unit number assigned by the habitat crew. This number is found on the flags that bound the habitat unit.

- Diver-Initials of the diver for that unit. If divers on the same team share the same initials, follow the initials with a number and indicate in comments which diver uses the numbers.

- Species Code-Code indicating species:

CODE	SPECIES
CO	Coho
CH	Chinook
SH	Steelhead
CT	Cutthroat
UT	Unknown Trout
Oi	Other species #I

(Other species is for use if surveyor is interested in species not on the list. The surveyor assigns number I and notes in comments the species names with the corresponding number.)

- Age class-age class of the group counted for that row of data entry (0+, 1+)

D.3.8.3.2 *Dive 1*

- Count-the number of fish counted in the dive of that species within the age class for the row.

- Duration-the duration of the dive. Note the start and stop times in the diver notebook

- Vis-visibility for the initial dive. If visibility becomes worse in later repeat dives, make detailed notes in a comment page, noting the habitat unit. If visibility for repeat dives becomes too clouded, the calibration must be electrofishing, or, if electrofishing is not possible, the unit reclassified as other. Visibility codes:

Code	Visibility
E	**Excellent** - no problems seeing anything in unit
G	**Good** - Approximate minimum of 10 feet of visibility. (Visibility from bank to bank with minimal movement, or, for two divers, from midline to bank.)
P	**Poor** - Visibility not good enough for reliable counts
Z	**Fails** - Visibility near zero, counting impossible

D.3.8.3.3 *ASIS*

- Phase II Number- ASIS strip number including the yes or no. Each ASIS selection strip provided by FSP will have:

- the selection probability for that strip,

- the sequence number for the strip, and

- a ‘YES’ or ‘NO’. Record both the sequence number and the response. If the third run entry has a ‘YES’, then record as 3YES.

- E fish-record a ‘Y’ or ‘N’. If the unit was selected by phase II ASIS as a calibration unit, and if the unit is a Run or Shallow Pool with a 0+ coho count greater than 20, then record a ‘Y’ and the unit is flagged as an electrofishing unit. Otherwise, record a ‘N’ for no phase II electrofishing.

D.3.8.4 Fish Survey: Phase II (Calibration)

If the ASIS number indicates that the unit needs to be calibrated, a decision will be made based on the number of 0+ coho seen. If the count for coho 0+ exceeds 20, then the unit is flagged for later 4-pass electrofishing calibration. If the fish count is less than or equal to 20 fish and there is not excessive complexity in the pool that would preclude seeing all the fish without risk of double counting, then the calibration will be done by the bounded count method. This involves three additional passes through the same unit following a brief (5 minutes) wait with approximately equal effort in each pass. The wait between dives must be long enough to insure that the water has cleared and the fish have had time to settle down. If other species of interest exceeded 20-fish threshold while 0+ coho did not, the unit may be flagged for electrofishing of the other species that exceeded the threshold. However, three additional dives are required.

The sampling of riffles, which is only done by electrofishing and the calibration of phase I units by electrofishing should be done within two days of the initial snorkel surveys. Block nets must be placed at the top and bottom of the units to be electrofished, and three depletion passes are made through the unit. The effort (time spent electrofishing) on each pass should be approximately equal.

D.3.8.4.1 Phase II Snorkel Survey

Phase II snorkel dive data are recorded onto the same data sheets as the Phase I snorkel dive. The Phase I data was recorded in the Dive 1 column while the Phase II data will go into the appropriate Dive 2, Dive 3, or Dive 4 column for the appropriate dive pass. These dives will be immediately following the dive 1 pass and determining the Phase II status.

D.3.8.4.2 Phase II Electrofishing Survey

Each electrofishing data sheet is for one habitat unit. Each phase II electrofishing unit will be subjected to four depletion passes.

- On the top table record the following:

    a) E-fish time-the start and stop time for the electrofishing

    b) Duration-the duration, in seconds, of time the electrofishing unit was on for each pass

    c) Processing time-only if desired for those taking lengths and weights

    d) Water temp (°C)-the water temperature at the beginning of the pass

    e) Conductivity-measured conductivity, if means available

- On the bottom table record the following:

  a) Pass number

  b) Species Code-the following code indicating species:

CODE	SPECIES
CO	Coho
CH	Chinook
SH	Steelhead
CT	Cutthroat
UT	Unknown Trout
Oi	Other species #i

(Other species code is for use if surveyor is interested in species not on the list. The surveyor assigns number i, and notes in comments the species names with the corresponding number.)

  c) Age Class-age class of the group counted for that row of data entry (0+, 1+)

  d) Fish Count-the number of fish captured for that species and age class on the given pass number. Note, if taking scale samples, weights, and/or lengths, treat fish count as fish id number and one row will be one fish.

  e) Mortality Count-the number of mortalities for that species and age class on the given pass number. Note, when taking scale samples, weights, and/or lengths record a zero if the individual fish was alive or 1 if that fish was dead.

  f) Length-fork length (mm) of an individual fish. If fork length not appropriate for the particular species, make a note in comments about which length measurement was taken.

  g) Weight-weight of an individual fish in grams.

  h) Scales-denote with a ‘Y’ if scale samples were taken.

  i) Comments-note any difficulties encountered that may affect the reliability of the results.

D.3.8.5 Literature Cited

Dolloff, C.A., D.G. Hankin, G.H. Reeves. 1993. Basinwide estimation of habitat and fish populations in streams. USDA Forest Service, General Technical Report SE-83.

Hankin, D.G, 1999. Unpublished MS, a modification of the "Hankin and Reeves" (1988) survey designs, as summarized in detail by Dolloff et al. (1993).

Hankin, D.G. and G.H. Reeves. 1988. Estimating total fish abundance and total habitat area in small streams based on visual estimation methods. Canadian Journal of Fisheries and Aquatic Sciences 45: 834-844.

Robson, D.S. and J.H. Whitlock. 1964. Estimation of truncation point. Biometrika 51: 33-39.

D.3.9 Out-migrant Trapping

D.3.9.1 Background and Objectives

The out-migrant trapping monitoring project is designed to monitor the abundance, size, and timing of emigrating salmonid smolts. Furthermore it is conducted to look for long term trends in any or all of these variables. The results of the out-migrant trapping are used in conjunction with the summer population monitoring to estimate overwinter survival in the Little River HPA. Eventually this information will be further analyzed to correlate specific habitat conditions with overwinter survival of coho salmon.

The objectives of monitoring out-migrant salmonid smolts are threefold: to estimate overwinter survival of juvenile coho by comparing out-migrant abundance to the summer population estimates; monitor the abundance, size, and timing of out-migrating smolts; and look for long term trends in any or all of these variables. Juvenile smolt out-migration is monitored to:

- Determine the diversity of salmonid species.
- Identify physical and age-specific characteristics of each species.
- Determine species specific out-migration timing.
- Establish baseline data to ascertain the viability and abundance of salmonids.
- Monitor long-term trends in smolting populations.

D.3.9.2 Methods

This monitoring method uses a combination of a weir, pipe, and live-box to capture juvenile salmonids (Figure D-4). Smolting populations of coho salmon (*Oncorhynchus kisutch*), chinook salmon (*O. tshawytscha*), steelhead trout *(O. mykiss iridens*) and coastal cutthroat trout (*O. clarki clarki*) are the species targeted. The data are used to estimate the relative population sizes of those species. The equipment and methods utilized allow for variation in fish sizes being trapped, maximize the number of out migrants entering the trap, and minimize the stress and mortality of fish.

D.3.9.2.1 Establishing the Survey Area

Selecting an area to sample juvenile salmonids can be a difficult task, and the location must be based on several criteria. There are four very important factors to consider during a field visit to select the trap site. Items considered are: access, stream flow, stream gradient, and substrate composition.

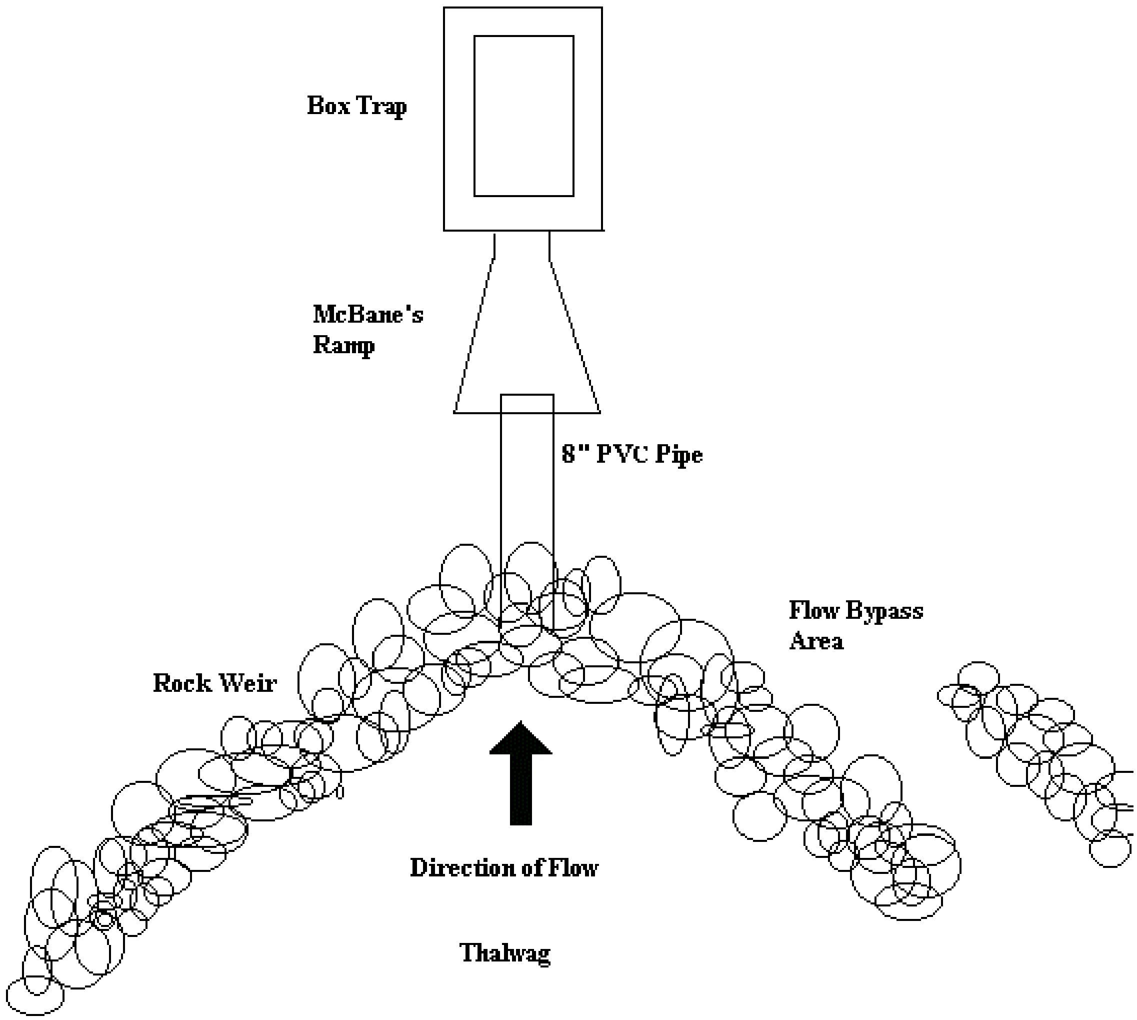

Figure D-4. **Out Migrant Fish Trapping System. The weir is constructed out of boulder and cobble from the streambed. Four to five sections of 20 foot, 8 inch diameter PVC connect the weir to the McBane's ramp downstream. The McBane's ramp diffuses the water velocity before entering the box trap. Diagram not shown to scale. Note: A wooden pallet weir can take the place of the rock weir.**

Access is extremely important, since juvenile out-migration trapping is a time consuming methodology that requires daily visits, 7 days a week, potentially 150 days of the year (February to July). Having a nearby road or well-established trail is very important, for both delivery of equipment and daily trap maintenance. Flows ranging from 150 to 200 cubic feet per second are the maximum volume of water that has been dammed with this technique. Flows of greater volume may require the use of a screw trap or alternative method (if flow regularly exceeds 150 cfs). Weir strength limits water volume being trapped and may be exceeded during early February and March peak flows, resulting in a loss of trap efficiency.

Stream gradient is important to create the vacuum needed to draw fish towards the mouth of the pipe (positioned at the v-notch in the weir). A minimum of one to three percent drop in stream gradient is sufficient and can be best located in a pool to riffle transition area. By placing the weir on the tail-out of a pool or run, the pipe can be placed in the riffle, capturing the drop in stream gradient that will create the suction necessary to attract fish towards the mouth of the pipe. Substrate composition determines what material will be used in weir construction. Depending on whether there is a sandy bottom with loose gravels or a large cobble/boulder dominant reach, the weir can be fashioned from fence posts and pallets or boulders taking advantage of on site material. This is important, because if planning to use fence posts as anchoring points, it is impossible to drive them through a large cobble/boulder dominated substrate.

D.3.9.2.2 Duration of Surveys

Juvenile out-migration trapping can run from early February to late July, and may start earlier and run later depending on the out-migration timing of the species being studied. Out-migration timing can vary widely in parts of northern California, and can be triggered by environmental conditions, competition, or egg deposition time. To ascertain site specific out-migration timing for each species during the first year of trapping, allowance for up to 150 days of continuous trapping (February through July) should be made.

Smolt abundance, size, and timing will be monitored annually. The time required to correlate these results with habitat information and summer population estimates is truly unknown, but will probably require a minimum of ten years due to the high variability observed in both summer population estimates and smolt abundance.

D.3.9.2.3 Equipment List

The following equipment is needed for each trap site established:

- One to two wood or plastic box(es) (for retaining fish)
- McBane ramp (dissipate water velocity)
- 6’ Steel fence posts (optional)
- Wooden Pallets (optional)
- 60 to 100 feet of 4’ T Galvanized Hardware Screen (optional)

- Galvanized bailing wire (optional)
- 5-6 20'L x 8"D PVC pipes
- Car jacks (scissors variety)
- Nylon rope

The list of equipment that will be needed to maintain and check each trap daily is listed below:

- Three black 5-gallon buckets
- Large meshed fishing net
- Large dip net
- Small dip net
- Ventilated holding cage
- Measuring board
- Viewing chamber
- Data Sheets
- Scissors
- Clear plastic cup
- Alka-Seltzer® (or other anesthetic)
- Long handled scrub brush

D.3.9.3 Weir Construction and Trap Installation

Weir construction is the most time consuming yet important part of trap installation. Once the substrate composition of the streambed is determined, the appropriate material for weir construction will be selected. If large cobble and boulder dominate the substrate of the surrounding stream reach, constructing the weir from streambed material (boulders and large cobble) maybe more appropriate. If the streambed material is small cobble, gravel, or sand, fence posts will be used in weir construction, in combination with either wooden pallets or wire mesh to retain the water.

D.3.9.3.1

Weir Construction

First, locate the thalweg of the run or pool. This will be where the mouth of the pipe is placed and the weir converges to form a V-shape. Clear a location for a 20' section of PVC pipe by removing streambed obstructions (i.e., large rocks and boulders). Take a section of PVC (20'L x 8"D) pipe and place it in the thalweg of the downstream riffle running parallel with stream flow. Submerge the PVC pipe in the unobstructed area of the pool thalweg and quickly place large boulders or several fence posts along the pipe to secure it in place. This location will serve as a convergence point to start constructing both sides of the weir. The rock wall or fence post wall should be shaped as a V, with the mouth of the PVC pipe being at the V-notched end of the weir. Before weir construction is complete, create a small, shallow channel at the edge of the weir, which will serve as a bypass area for escaping steelhead adults. Make sure the bypass area is just shallow enough to pass fish moving upstream. Too much flow may draw out-migrants to this section of the weir. Construction materials and procedures for building the weir depends on the weir type required:

- If building a rock weir, form a large base to the wall like a pyramid (4'-5' thick). Around the mouth of the pipe should be the strongest, thickest portion of the weir. This is the location that needs the most protection, because if this section blows out, it will be very difficult to reconstruct under strong flows. Begin construction by building out from the converging points towards the stream bank, keeping the same thickness of wall. Start adding height to the wall until the majority of flow is trapped and funneled towards the mouth of the pipe. This style of weir is very effective in swift flowing, higher gradient (3-4%) streams.

- If building a fence post/pallet weir, use wood pallets as a marker for fence post placement. Construct the weir by moving from the convergence of the weir out towards the stream banks forming a classic or slightly altered V– shape (depending on thalweg location). Using a fence post driver, place one post deep into the substrate of the streambed. Place one end of the wooden pallet over the fence post and sink it until you reach the streambed. Take another fence post and secure the pallet in place making sure the angle and direction form the V-shape necessary to corral the fish. Continue this procedure until the stream banks are reached and the majority of stream flow is dammed behind the weir. Scrape streambed gravel and cobble around the foundation to cover gaps at the foot of the weir. Be sure to cut wooden pallets in half sections and fill in gaps between the wood with redwood slats or other material to effectively block flow. Half pallets can be stacked to form the weir and will make for a very effective system to manage flow during peak events. This style of weir capable of damming large flows in swift flowing streams, and may be the most efficient method for long-term trapping (if substrate allows).

- If building a fence post/screen weir, placement of the fence posts with use of the fence post driver can be done before the screen is attached to the posts. Place the fence posts by moving from the convergence of the weir out towards the stream banks forming a classic V– shape (depending on thalweg location). The fence posts can be 2 – 4 feet apart, spaced closer together if trapping late in the winter during higher flows. Using bailing wire, attach the screen to the fence posts. Scrape streambed gravel and cobble around the foundation of the screen to cover gaps at the foot of the weir. The majority of flow will be filtered through the wire mesh creating very little incentive for fish to move towards the mouth of the pipe. Sometimes, young fish can become impinged on the surface of the screen during higher flows. To avoid this, angle the V of the weir as much as possible, to avoid perpendicular angles to the direction of stream flow. This system works well for low gradient streams that will not experience large flow events, and may be best suited for late season trapping in flows 1 – 50 cfs (if algae blooms are common, strongly consider the use of pallets or boulders).

D.3.9.3.2 Pipe Installation

The section of pipe that has already been laid parallel with stream flow and sits at the notch of the weir is connected together with the remaining sections of pipe. This string of pipe is run down the full length of the riffle to take advantage of the change in stream gradient (three to six sections of PVC pipe may be needed to run the full length of the riffle). This will help to create the suction at the mouth of the pipe and draw fish in. An attempt is made to empty the pipe into the next habitat unit, preferably a run or pool with slack water, near the stream bank. Place large boulders or use fence posts to keep the pipe stationary during large flows.

D.3.9.3.3 Ramp Installation

If using a rock walled weir, the system is working well when the majority of trapped water is being funneled through the pipe, with the lot of head pressure coming out the far end of the pipe. This will not occur with a fence post/screen weir. In order to reduce the potential for fish mortality, do not place the downstream end of the pipe directly into the box. Alternatively, using a scissors jack, raise the downstream end of the pipe off the ground, and place the mouth of the McBane ramp under the pipe. The McBane ramp is a graduated ramp made out of perforated sheet metal, which will dissipate large volumes of water. Adjust the flow over the McBane ramp by moving it forward or backward. The majority of flow will dissipate through the McBane ramp leaving a smooth sheet of water to carry fish into the box trap. Be sure there is just enough water to gently glide the fish into the live box. There should be 10 – 12 inches of clearance between the surface of the water and the mouth of the McBane ramp, and five to six inches of clearance between the surface water and the portion of the ramp entering the box. The ramp at this point should be at a downward angle when entering the box trap, leaving enough room under the ramp for increased water levels. If there is not enough water flowing over the McBane ramp to create a constant flow, place plastic on the ramp to cover the holes and achieve more flow. This may be necessary when using a fence post/screen weir.

D.3.9.3.4 Box Trap Installation

Attach the box trap to the McBane ramp. Slide the McBane ramp into the pre-formed board created to support the ramps' exit point. The box trap can be submerged or remain out of the water depending on the box trap type being used. Preferably, the box trap should be submerged five to eight inches in the water, which will leave room for increased water levels. Placing rocks or other material along the base of the trap should slow flow against the box trap screen. Installation of a second box trap is optional, but highly recommended to reduce in-trap predation from sculpin, cutthroat and/or steelhead. If a second box trap is used attach the second box trap behind the first box with a connector. The second box trap will hold the young-of-the-year fish. Slide two different gauge screens into the series of box traps, one at the rear of the first box trap and the other at the mouth of the second box trap. This will serve as a barrier to separate large salmonids from smaller salmonids, which will naturally segregate themselves into the two boxes.

D.3.9.3.5 Fine Tuning the Trap Systems

During the first few weeks of trapping, trap and weir maintenance are required daily, especially if higher flows are present. Use this period to fine tune both the weir and the trap to increase trap efficiency and eliminate potential mortality associated with higher flows.

D.3.9.4 Daily Monitoring Procedure

There are three basic steps to the daily monitoring procedure; remove fish from box, identify and measure, and release. All fish entering the box and the number observed are recorded including incidental catches of non-target species such as lamprey, suckers, and sculpins. To initiate the monitoring procedure, the following steps are completed:

D.3.9.4.1 Organization

Data sheets are prepared to measure the day's catch. The organization of the data sheets is important, and species are arranged systematically. Before opening the traps, boards are slid into the screening area and screens are removed. This will stop young-of-the-year from moving back and forth between boxes. The first box trap is opened and any steelhead trout down-runners are removed first. These adult fish are measured and any hatchery marks are noted. After removal of adult steelhead, preparations to measure, mark (if necessary) and release the day's catch are made.

D.3.9.4.2 Preparation of Holding Containers

A 5-gallon bucket is prepared by filling it with three to four inches of water. One Alka Seltzer® tablet is dissolved into the water in the bucket. This bucket will be used to anesthetize the first batch of fish (CO_2 and MS-222 may be substituted for Alka-Seltzer®. If Alka-Seltzer® is being used, the tablet must be fully dissolved. A second 5-gallon bucket is filled 2/3 full with water and placed next to the trap. This is used as a recovery bucket for processed fish. A sheet-metal live box or other holding cage is placed in three inches of flowing water next to the trap to serve as a temporary holding cage for clipped fish.

D.3.9.4.3 *Capture Fish*

The day's catch is then sampled by sweeping the large dip net through the trap. A group of twenty to twenty-five fish are selected and place into a bucket for identification, measurement and for potential clip (marking). Until fish-handling proficiency is perfected during the first few weeks of trapping, fewer fish will be selected. Later in the season when water temperatures increase, additional handling stress may occur to fish and therefore will be checked at one time. Fish are placed into the bucket to be anesthetized. Smolting steelhead trout, cutthroat trout, and coho salmon are the only species that will be used to test trap efficiency.

D.3.9.4.4 *Check Fish*

After the selected fish are fully anesthetized they are identified and measured. All measured parr, pre-smolt and adult fish are placed into the release bucket, and data for: species, length and age class are recorded on the data sheet. If a smolt is being measured, an appropriate caudal clip is made to mark fish and the fish is placed into the holding trap. Recaptured smolts are noted at the bottom of the data sheet but are not included in the day's total count. The procedure for marking smolts is found below. Checking fish should take no longer than ten minutes per group, and less time if water temperatures increase or fish are anesthetized quickly. Water in the anesthetizing bucket and recovery buckets are changed every time a new group of fish is selected for data collection.

D.3.9.4.5 *Marking Smolts*

Coho salmon smolts, steelhead trout smolts and cutthroat trout smolts will be marked with fin clips. A total of four different clips will be used throughout the trapping season. Clips will be used for a period of seven-days. The easiest clips to see are caudal fin clips. A horizontal upper caudal, vertical upper caudal, vertical lower caudal and horizontal lower caudal clip will be used for each seven-day period, in any sequence seen fit. After the first 28 days, the same sequence of clips is repeated. Having at least a 28-day period before repeating a sequence of clips is absolutely necessary. Up to 16 smolts of each species will be marked. It may be necessary to increase this number if the number of recaptured fish remains extremely low.

D.3.9.4.6 *Release Recovered Fish*

After the first bucket of fish are measured and recorded, they are checked to see if the unmarked fish in the recovery bucket are ready to be released. If these fish have recovered from the anesthetic, these fish are released to a portion of slack water near the trap site. Procedures are repeated until the first box trap is empty.

D.3.9.4.7 *Check Young-of-the-Year*

The majority of young-of-the-year chinook salmon, coho salmon and trout will have already separated themselves into the second box trap and there will be no need to anesthetize these young-of-the-year fish. A 5-gallon bucket is filled with water and the young-of-the-year fish are removed from the second box using additional caution on hot days. Twenty measurements from a sample of each species are obtained and recorded.

Once the twenty measurements are recorded, tallies of the remaining young-of-the-year are made until all fish are observed.

D.3.9.4.8 *Release Marked Fish*

Fish clipped for trap efficiency tests will be fully recovered from the anesthetic and placed into a 5-gallon bucket for transport above the weir. Clipped fish are then placed into the pool a few yards above the weir (but not at the mouth of the pipe). Trap efficiency is designed to test how well the weir is working not predation or any other factor. If these fish are recaptured, they are not used again for efficiency testing. In some cases the fish may be held until dusk before releasing. This accomplishes several things; testing to see if there is some handling mortality; release of the fish to coincide with peak diurnal movements; and allows fish to fully recover from handling prior to re-approaching the weir. This is an optional procedure step in this protocol.

D.3.9.4.9 *Record Mortality*

Very little trap-related mortality will be generated with this method. Under federal and state salmonid trapping permits, some mortalities are retained for genetic studies. Appropriate handling and preservation techniques will be implemented if permit requires that mortalities be archived.

D.3.9.5 Daily Trap Maintenance

All accumulations of debris will be cleaned and removed daily from the McBane ramp, interior of both boxes, behind the weir and screens that segregate the fish. The majority of mortality is caused by debris accumulations on the ramp or inside the live boxes. The weirs will be checked for leaks or debris accumulations that may have piled up.

D.3.9.5.1 *Calculating Trap Efficiency*

A “mark-recapture” method is used to estimate trap efficiency. Accurate population estimates depend on this portion of the protocol.

A 28-day period will be used to test trap efficiency, utilizing coho salmon smolts, cutthroat trout smolts, and steelhead trout smolts, as described above. Trap efficiency will be calculated by using only species that are actively leaving the drainage on their seaward migration (“smolts”). These tests will be run to determine what percentage of the population is missed by inefficiencies in the weir. Marks (fin clips) will be changed every 7-days to account for variations in environmental attributes. Trap efficiency will be calculated using a software package (DARR: Darroch Analysis with Rank-Reduction) that analyzes stratified mark-recapture data (Bjorkstedt 2000).

D.3.9.5.2 *Population Estimation*

Population estimates will be made for smolt year classes of coho salmon, steelhead and coastal cutthroat trout. Population size will not be estimated for chinook salmon due to their size and abundance during out-migration. Chinook are too small when first entering the traps to mark with a caudal fin clip. Population estimates are not made for young-of-the-year, parr, or pre-smolts of the same species because these life stages are only redistributing themselves within the watershed, and not actively emigrating to the ocean.

The out-migrant smolt population estimates will be calculated using a software package (DARR: Darroch Analysis with Rank-Reduction) for analysis of stratified mark-recapture data (Bjorkstedt 2000).

D.3.9.6 Literature Cited

Bjorkstedt, E.P. 2000. DARR (Darroch Analysis with Rank-Reduction): A method for analysis of stratified mark-recapture data from small populations, with application to estimating abundance of smolts from outmigrant trap data. U.S. Department of Commerce, NOAA, NMFS, SWFSC, Admin. Rep., Santa Cruz, SC-00-02. 261 Kb, 28 p.

D.4 EXPERIMENTAL WATERSHEDS PROGRAM

While the majority of the Plan's monitoring projects will be conducted throughout the Plan Area, four experimental watersheds judged to be representative of the different geologic and physiographic provinces across the Plan Area have been designated for additional monitoring and research on the interactions between forestry management and riparian and aquatic ecosystems. Those watersheds are the Little River HPA, the South Fork Winchuck River in the Smith River HPA, Ryan Creek in the Humboldt Bay HPA, and Ah Pah Creek in the Coastal Klamath HPA (see Figure 6-9 in Section 6.3).

In general, the program will entail:

- Effectiveness monitoring projects and programs that due to their complexity and expense of implementation can only be applied in limited regions (these include turbidity monitoring, Class III sediment monitoring, and road-related catastrophic sediment input monitoring;

- BACI studies of harvest and non-harvest areas, allowing for more effective evaluation of conservation measures and increased understanding of the effects of forest management on the habitats and populations of the Covered Species.

- BACI studies of conservation and management measures, allowing for a refinement of measures and an assessment of the relative benefits of different measures under the Plan; and

- Development and implementation of new or refined monitoring and research protocols.

In addition, Green Diamond may expand Out-migrant Trapping in the Little River HPA to one or more of the other experimental watersheds.

In the program, management will be implemented as a large scale experiment where possible, allowing for more effective evaluation of conservation measures and increased understanding of the effects of forest management on the habitats and populations of the Covered Species. Where possible, harvest with a variety of different conservation measures will be the "treatments" in a BACI experimental design, with an adjacent unharvested area as the control. Specific effectiveness monitoring projects will compare the treatment and control before and after harvest to determine the effectiveness of the conservation measures.

The turbidity monitoring and catastrophic sediment input monitoring are designed in part to measure the effectiveness of the road management plan's upgrading and decommissioning measures in reducing road-related sediment inputs. For these road-related monitoring projects, the experimental design occurs as monitoring is implemented both spatially and temporally to allow comparisons of road-related sediment inputs before and after road upgrading and decommissioning.

Upgrading and decommissioning the roads as effectively and efficiently as possible is the first priority, therefore monitoring will essentially be conducted "around" the road work schedule. The prioritization process (see Section 6.3.3.) used to schedule the road work will provide the information needed to design an effective monitoring program without slowing the implementation of the road upgrading and decommissioning process. For example, the prioritization table may dictate that, within a specific sub-basin, one road work unit will be upgraded before another. Monitoring could begin in both units before any work is done, and continues while first one, and then the other work unit is upgraded. This experiment would not be conducted in a true BACI design, because Green Diamond will not leave any sub-basins as "controls" in the untreated condition. However, over time it will be possible to make a cumulative comparison of treated versus untreated roads and sub-basins to determine if the road management plan is effective in reducing road related catastrophic sediment inputs or road-related increases in turbidity.

Green Diamond and CDFG are already implementing an experimental management program in the Little River HPA to assess the relative benefits of two different mitigation measures to protect aquatic resources following timber harvest. A randomized BACI experiment will be conducted in blocks of three streams, wherein the two sets of mitigation measures are viewed as two different treatments with the third stream as a control. During the course of the experiment, both mitigation measures will be applied to an approximately equal number and linear distance of streams. The primary objectives of the study will be to:

- determine if there are any detectable changes in environmental and biological variables measured on watercourses following timber harvest, and if there are,
- which mitigation strategy is more effective in reducing negative impacts.

The response variables will be monitored pre and post harvest and will include water temperature, shallow landslide activity, Class III sediment delivery, and potential LWD. Air temperature, relative humidity, wind speed, turbidity, and stream amphibian populations will also be monitored in selected sites.

The development and implementation of new research and monitoring protocol will provide an opportunity for Green Diamond to refine existing conservation measure to make them more effective and efficient. This will include state-of-the-art existing study designs along with original research approaches that will require the input from academic, agency and private scientists.

No experiment which involves the application of conservation measures other than those prescribed in the Plan will occur without the concurrence of the Services.

Appendix E. Potential Effects of Timber Management on Covered Species and their Habitats

CONTENTS

Figures

Tables

E.1 INTRODUCTION

The effects of timber harvest on aquatic life depend on many factors and studies often produce contradictory results (Spence et. al. 1996). Factors that may influence responses include: aquatic species' diversity and adaptability, physical and vegetative conditions and harvest methods, biotic interactions and wide-ranging migratory behaviors can act to reduce impacts of habitat alterations, independent impacts that can accumulate, or interact collectively resulting in compensatory or synergistic responses, and large natural (catastrophic) events that create variable baseline conditions confusing other smaller scale variability.

Despite the difficulties of separating timber harvesting effects from natural disturbance regimes, there has been considerable research on the potential impacts of timber harvesting on aquatic species and their habitats. For example, Chamberlain et al. (1991) summarized four timber harvesting effects that may modify the hydrologic and geomorphic processes and channel formations that determine salmonid habitat:

- Possible increases in peak flows or occurrences of channel-forming flows from increased snow-melt or run-off, resulting in increased bed scour and bank erosion;

- Significant increases in sediment supplies from surface erosion, mass soil movement and bank erosion, leading to channel aggradation, loss of pool volume and degradation of spawning gravels;

- Destabilization of streambanks due to removal of riparian vegetation, physical breakdown, or channel aggradation, increasing sediment supplies and leading to losses of channel formations that constrict flows and promote a diversity of habitat types required by salmonids; and

- Loss or reduction of LWD by direct removal, debris torrents, or management practices that convert riparian corridors to younger stands of predominately hardwoods, contributing to reduced sediment storage sites, reduced pool numbers and volumes, and less rearing habitat for juvenile salmonids.

There has been less research on the potential impacts of timber harvesting on the covered amphibian species, but most of the potential impacts to salmonids and their habitat are believed to also impact the cool-water adapted stream amphibians. In general, timber harvesting activities have the potential to impact all of the Covered Species through altering one or all of the following processes: hydrologic cycle, sediment inputs and transport, large woody debris (LWD) recruitment and distribution, thermal regimes and nutrient inputs.

E.2 ALTERED HYDROLOGIC CYCLES

E.2.1 Potential Effects of Timber Harvesting Activities on Aquatic Habitats

The basic components of the hydrologic cycle are precipitation, infiltration, evaporation, transpiration, storage and runoff. In the Pacific Northwest where annual precipitation is highly seasonal the timing, quantity and quality of rain and snow fall has great influence on salmonid life histories. Thus the interactions of timber harvest activities on the hydrologic cycle are important. This section reviews how timber management activities may influence the hydrologic cycle and the possible impacts on salmonid populations.

Timber harvest temporarily reduces or eliminates leaves and stems. The surface area of this vegetation normally intercepts precipitation for short-term storage that is either evaporated or released as drip. The loss of forest vegetation also reduces the amount of water extracted from the soil by root systems via evapotranspiration and increases soil moisture and piezometric head. This was demonstrated by Keppeler and Brown (1998) after harvest of second growth redwood forest. Such increases in soil moisture can contribute to increased risk of mass wasting (Sidle et al. 1985, Fig. 10; Schmidt et al. in press). This is discussed further in Section 5.3.2.2. The effect of any reduction in evapotranspiration is typically short lived (3-5 years), as rapid regrowth of vegetation may consume more water than pre-timber harvest amounts (Harr 1977). This is likely to be true in redwood forests as well, in part owing to the stump-sprouting habit of redwood.

The primary effects of timber harvest on surface water hydrology pertain to (Spence et al. 1996):

- peak flows,
- low (base) flows,
- water yield, and
- run-off timing.

Paired watershed experiments to measure changes in flow following timber harvest have been conducted north of the project area (Oregon) and south of the project area (Mendocino County, California). In relatively small watersheds (about 150 to 1200 ac), peak flow magnitude following harvest tends to increase, with the largest increases occurring in smaller runoff events (less than 1-yr) (Beschta et al. 2000; Ziemer 1998). For 1-yr recurrence interval events, peak flow magnitude increased 13-16%; these increases were 6-9% for 5-yr recurrence interval events (Beschta et al. 2000). At Caspar Creek in Mendocino County, increases in peak flow magnitude were about 10% for 2-yr storm recurrence interval events. The effect of timber harvest on peak flows generally diminishes with increasing watershed size and with increasing flow magnitude (Beschta et al. 2000; Ziemer 1998). Effects for larger watersheds are difficult to asses because they are influenced by many additional factors, including regulatory controls on the proportion of the landscape that can be harvested at any given time (e.g., clearcut adjacency and rotation age restrictions adopted by the Board of Forestry) and the extreme variability introduced when attempting to study large basins that experience relatively infrequent major hydrologic events.

The extent of harvest-related changes in hydrology within a watershed may be affected by whether the system is rain or snow dominated. Keppeler and Ziemer (1990, as cited by Spence et al. 1996) found increased summer flows in a Northern California stream following timber harvest but this diminished after five years. In many cases, for rain-dominated systems in the Coast Range, increases in peak flows (particularly in the fall) following timber harvest, are documented (Spence et al. 1996). The principal increases in peak flows following timber harvest in rain-dominated systems are likely as a result of reduced interception and evapotranspiration rates resulting from the loss of vegetation and the more rapid routing of water to stream channels because of soil compaction and roads (Spence et al. 1996; Ziemer 1998). In contrast, generally in snow-dominated systems in the Northwest, peak flows have been shown to change little following timber harvest. In transient-snow systems studies have been somewhat inconclusive as to the effects of timber harvest on peak flows. However, Harr (1986 as cited by Spence et al. 1996) found that in transient-snow systems where harvest had resulted in increased peak flows, the removal of vegetation increased the delivery of water to the soil from the snow-pack during rain-on-snow events. Other research has shown that increased snow melt rates and delivery of water to the soil occurs during rain-on-snow events accompanied by relatively high temperatures and wind speeds (Coffin and Harr 1992, as cited by Spence et al. 1996). The commercial timberlands within the 11 HPAs are entirely rain-dominated. Therefore, the effects of snow-dominated and rain-on-snow hydrology are not an issue for this Plan.

When logging activities compact or disturb surface soils the infiltration capacity is reduced, possibly increasing surface runoff, peak stream flows and sediment inputs. The soil structure of forested hillslopes regulates the downslope movement of water through the soils and into watersheds. On forested hillslopes the infiltration capacity of the soils usually exceeds rainfall or snowmelt intensities so that all water is absorbed by these soils and transported to stream channels through subsurface pathways (Dryness 1969; Harr 1977). Timber harvest activities that compact or disturb the soil can reduce the infiltration capacity of soils and alter the process of subsurface water movement.

Water and sediment from roads can enter stream channels by many mechanisms (Furniss et. al. 2000):

- Inboard ditches that deliver road drainage to stream channels at truck road stream crossings,

- Inboard ditches that deliver flow to culverts, road drainage dips or water bars with sufficient discharge to create a gully or generate a sediment plume that extends to a stream channel,

- Improperly spaced or located road drainage structures that discharge sufficient water to create a gully or generate a sediment plume that extends to a stream channel, and

- Roads located close enough to a stream that fill slope erosion or fill failures result in sediment discharge in to stream channel.

Some studies have shown that forest roads increase peak flows and sediment inputs to small watersheds when as little as 2.5%-3.9% of the watershed is composed of road surfaces (Harr et al. 1975; Cederholm et al. 1980; King and Tennyson 1984). Studies reporting increases in water yield from logged watersheds indicated that these increases were most evident in the start of the fall/winter wet season when rain quickly filled soil pore spaces in the logged areas and then ran off as surface flow. Differences were less apparent later in the rainy season as soil under mature canopies also became saturated, and runoff from harvested and un-harvested areas became similar (Hibbert 1967; Harr et al. 1979). Other studies have also shown that road construction and some timber harvest activities may lead to increased flows in the first (fall/early winter) small rain events but have no significant effect on larger flow events (Wright et al. 1990; Johnson and Beschta 1980).

Many paired watershed studies have found increases in summer base flow and total water yield (Bosch and Hewlett 1982), particularly in humid coniferous forest types. Studies north of the Plan Area in southwest Oregon (Harr et al. 1979) and south of the Plan Area at Caspar Creek in Mendocino County (Keppeler 1998) found increases in both total water yield and seasonal base flows.

Coastal watersheds of northern California receive a majority of their precipitation as rain. However, some watersheds in the Plan Area have upper sections within the transition zone between rain and snow. Along these hillslopes the forest canopy intercepts snowfall, redistributes the snow, shades the snowpack and acts as a windbreak. In these transient areas the snow is generally wet and sticks to the forest canopy longer than colder, drier snow. In transitional areas snow usually reaches the ground in clumps under trees or as snow melt so that snow pack in forested areas tends to vary in distribution and depth compared to logged hillslopes (Berris and Harr 1987).

Snow melt from hillslopes in coastal watersheds is usually the result of warmer rainfall or latent heat in air moisture rather than from solar radiation. Snow packs in transitional areas may accumulate and melt several times during the wet season. When the forest canopy has been removed more of the snow pack is directly exposed to rainfall, warm air and direct sunlight. Harr (1986) reported there was more heat available to melt snow in a clear-cut stand than in an old-growth Douglas-fir stand during a rain storm with a two year recurrence interval. Plot studies in paired watersheds (logged and unlogged) have reported increases in peak streamflow after rain on snow events in the logged areas (Harr and McCorison 1979; Christner and Harr 1982).

E.2.2 Potential Effects on the Covered Species

The effects of temporary changes in watershed yield, peak flow magnitude and timing, and summer base flows on salmonids and key salmonid habitat characteristics are difficult to assess. The life-cycles of salmonids species have adapted to temporal variations in flow conditions by timing the phases of their life cycles to take advantage of seasonal discharges characteristics (Sullivan et. al. 1987). Increased runoff in the early part of the rainy season may, in some cases, benefit salmonids by reducing water temperatures, improving water quality, and providing more flow for immigrating adult spawners. However, a harvest-related increase in peak flows may increase the number of times that channel substrates are mobilized by storm events and potentially damage developing eggs and alevins in redds (Hicks et al. 1991 as cited by Spence et al. 1996). Damage to developing eggs and alevins in redds would constitute take. Channel forming flows may occur more frequently as a result of an increase in peaks flows and thus habitats for spawning, rearing and foraging may be affected, either adversely or beneficially. Increased peak flows may also affect the survival of over-wintering juvenile salmonids by displacing them out of preferred habitats. Displacement of juveniles could cause take if the displacement impairs individual sheltering needs to the extent of killing or injuring individuals. These flow increases could have marginal beneficial effects by increasing available aquatic habitat. Short-term increases in summer baseflows may improve survival of juveniles (Hicks et al. 1991 as cited by Spence et al. 1996) and increase the amount of aquatic habitat. However, these effects are proportional to harvested area and diminish with regrowth of forest vegetation, so the effects are greatest for small watersheds.

The specific effects of altered hydrology on the amphibian Covered Species and their habitat are not known currently and are equally difficult to assess. Green Diamond is not aware of any studies that have addressed this potential effect on species such as the torrent salamander or tailed frog. The speculation is that, in general, these headwater species would be less likely to be affected relative to salmonid species that spawn and rear lower in the watershed. Tailed frog habitat overlaps with the upper reaches of salmonid habitat, and it is possible that increases in peak flow during winter may have a negative impact on larval tailed frogs. This could occur through entrainment of the substrate, which may displace or directly harm the larvae. Further in extreme circumstances, such increases in peak flow could cause take, which may result in local declines in tailed frog populations. However, this would not likely result in long-term changes in the habitat for the species, and therefore it would not likely to result in major changes in populations of the species. Increases in summer low flows due to harvesting activities may be beneficial to larval tailed frog populations, especially during drought years, so it is not possible to know if the overall impact of altered hydrology on tailed frog populations is positive or negative.

Southern torrent salamanders live in seeps and springs and the uppermost reaches of watercourses. The speculation is that increases in peak flow would be unlikely to have any negative impact on this species. Limited field observations of torrent salamanders during high flows suggest that they simply move to the margins of the channel and would not be impacted by entrainment of the substrate. Since torrent salamanders live in aquatic sites with minimal flows, it seems likely that increases in summer low flows would be beneficial for this species. However, they live in association with Pacific giant salamanders that have the potential to prey on or compete with torrent salamanders. Torrent salamanders specialize in utilizing sites with the most minimal flows, so biotic interactions may change with increases in summer low flows. All of these considerations are highly speculative, and Green Diamond does not believe it is possible to predict whether or not altered hydrology would have an impact, positive or negative, on southern torrent salamanders.

Increased runoff and peak flows and decreased infiltration capacity of soils due to timber management and road construction are also correlated with increased sediment inputs to watercourses (Harr et al. 1975; Cederholm et al. 1980; King and Tennyson 1984). The negative effects of increased sediment inputs on the Covered Species and their habitats are described below in Section E.3.

To summarize, the extent to which watershed hydrology is altered by timber harvesting activities and, similarly, the extent to which such altered hydrology may negatively impact the Covered Species, is a function of the amount and timing of those activities in a sub-basin or watershed. Given the cumulative relationship among those activities and this type of environmental effect, it is difficult to assess the potential for these activities to cause altered hydrology itself, and it is also difficult, in turn, to evaluate the potential for altered hydrology to cause take of the Covered Species. For example, as noted above, management-altered hydrology has the potential to harm both the early stages of development (eggs and alevins) as well as over-wintering juvenile salmonids. On the other hand, the effects of altered hydrology may be beneficial for adults returning to spawn in the fall and summer juvenile populations. Therefore, depending on which potentially limiting factors are actually limiting for salmonid production in a given sub-basin, some levels of altered hydrology may be beneficial. However, if other factors are limiting, altered hydrology may cause take and lead to local declines in populations of salmonids. For instance, if summer water temperatures are limiting, increases in summer base flows could be beneficial. In contrast, increases in winter peak flows could cause take and lead to local declines if spawning or over-wintering survival rates were limiting. In conclusion, the potential impacts of altered hydrology are highly complex, and although it has the potential to cause take that could lead to local declines in populations of the Covered Species, the actual impact of various levels of altered hydrology remain unknown. In any event, as a means of avoiding or minimizing and mitigating any negative impacts that could result from altered hydrology, the Plan provides measures to minimize the potential for harvest operations to cause altered hydrology.

E.3 ALTERED SEDIMENT INPUTS AND TRANSPORT

Timber harvest and the construction and use of the associated road system have the potential to increase sediment inputs. Increased sediment inputs from such activities can reduce the quality of aquatic habitats for all six Covered Species through reduced depth of deep water habitats (primarily pools), increased embeddedness of gravel and cobble substrates, and the effects of chronic turbidity on the Covered Species.

Hillslope erosion, sediment delivery to streams, and sediment transport and sorting within streams are natural dynamic processes that are responsible for creating aquatic habitat for the Covered Species. Steep, geologically young, coastal mountains are especially prone to high natural rates of erosion and the Covered Species have evolved in this environment. However, excessive inputs of sediment from a combination of anthropogenic and natural sources can overload a stream’s ability to store and transport sediment, reducing the quality and quantity of aquatic habitat for the Covered Species.

E.3.1 Northern California Sediment Yields and Sources

The variations in bedrock geology, tectonics, and associated geomorphic characteristics in northern California result in different erosion and sedimentation conditions in different stream reaches (the geology and geomorphology of the Plan Area and 11 HPAs are described in the Geologic and Geomorphic Setting Section of the EIS). Sediment production (erosion) may be highly variable depending on the presence or absence of Franciscan mélange and other geologic formations that contain abundant deep landslides and earthflows and locally extensive shearing and faulting in sedimentary rocks. Lisle (1990) cited previous studies discussing factors affecting annual sediment yield in the Eel River, where geologic conditions are most similar to those found on Green Diamond lands south of Redwood Creek. Of note are observations that sediment yield increases with annual rainfall and with drainage area, unlike many other regions. The increase in sediment yield within the drainage area is attributed to abundant deep-seated landslides adjacent to large mainstem river channels, which greatly increase sediment inputs per unit watershed area to the channel network relative to more stable terrain in smaller watersheds. The following data, as well as other data in Table E-1, illustrate this point.

Brown and Ritter (1971) reported mean annual suspended sediment yield for the Eel River to be 1,720 t/km^2/yr (about 4,900 t/mi^2/yr) for a drainage area of 9,400 km^2 (3,600 mi^2). Kelsey (1980) estimated the sediment yield of the upper Van Duzen River to be 2,500 t/km^2/yr (about 7,100 t/mi^2/yr) for a drainage area of 1,500 km^2 (580 mi^2). Both of these watersheds have abundant deep-seated landslides. In comparison, a stream draining an earthflow in mélange terrain was estimated to produce 24,000 t/km^2/yr (about 68,000 t/mi^2/yr) for an area of 3.4 km^2 (1.3 mi^2), or about 10 times more sediment yield per unit area than the basin as a whole. In Redwood Creek (Table E-1), active earthflows yielded about 5 times more sediment than the basin average.

It should be noted that although earthflows are a form of deep-seated landslide, earthflows are less common than rockslides that have slower episodic rates of movement, and that this comparison overstates the rate of sediment production by deep-seated landslides as a whole. Although earthflows may be more persistent sediment sources, rock slides may deliver more sediment in short time periods. For example a debris flow associated with the Floodgate Slide in Mendocino County on the Navarro River (Sowma-Bawcom 1996) delivered at least 200,000 metric tons of sediment from a landslide area of 0.04 km^2 (0.16 mi^2). Hence, stream reaches affected by active deep seated landslides may be more likely to exhibit transport-limited conditions, and are likely to have high suspended sediment loads. Furthermore, data indicating increasing sediment yield with increasing drainage area are also consistent with the hypothesis that sediment deposits in stream channels are a significant sediment source during periods of peak runoff when stream channels are fully occupied by flow and surficial bed armor layers are disrupted.

In contrast to regions where active earthflows and rockslides contribute massive amounts of sediment to streams, Lisle (1990) observed that more competent sandstone units of the Franciscan Formation deliver less sediment. In these areas, hillslope geomorphology is characterized by V-shaped valleys with steep hillslopes where debris slides are the primary mass wasting process. This description is similar to that given for "coherent sandstone" in Redwood National Park (Cashman et al. 1995), with the exception that in Redwood National Park, these characteristics occur on inner gorge slopes. In addition, abundant coarse sediment is generated in erosion events, most of which is of a size that can be transported during annual flood events (Lisle 1990). In these areas, under forested conditions, sediment yields are approximately 300 t/km2/yr (about 860 t/mi^2/yr Lisle 1990). Kelsey (1982) suggests typical rates in the upper Van Duzen River headwaters to range from 80 to 540 t/km^2/yr (230 to 1,500 t/mi^2/yr). Short-term measurements for largely unlogged Franciscan sandstone in Redwood National Park range from about 30 to 110 t/km^2/yr (90 to 310 t/mi^2/yr). Long-term measurements at Caspar Creek, long-term estimates for Freshwater Creek, and short-term measurements for Freshwater and Jacoby Creeks (Table 5-1), all of which include significant logging effects except perhaps Jacoby Creek, have sediment yields in a range characteristic of "competent" Franciscan sandstone, despite the prevalence of weaker geologic materials (Wildcat Group) in Freshwater Creek. Hence, where active deep-seated landslides do not contribute a major component of sediment inputs, sediment yields are approximately an order of magnitude (a factor of 10) lower.

Table E-1. Characteristic Northern California Coast Range sediment yield.

Watershed (Source)	Drainage Area-mi2 (km2)	Sediment Yield-t/mi2/yr (t/km2/yr)
Large Rivers		
Eel River (Brown & Ritter 1971)	3,600 (9,400)	4,900 (1,700)
Van Duzen River 1941-1975 (Kelsey 1980)	580 (1,500)	7,100 (2,500)
Redwood Creek	280 (725)	6,300-7,700 (2,200-2,700)
Earthflows and Rockslides		
Active Earthflows, Van Duzen River (Kelsey 1978)	1.3 (3.4)	71,000 (25,000)
Active Earthflows, Redwood Creek, (2 sites, annual average 1978-1982) (Nolan and Janda 1995a)	0.01-0.05 (0.023-0.13)	32,800 (11,500)
Rock Slide, Navarro River (minimum estimate from volume estimate for associated debris flow, Floodgate Slide, Sowma-Bawcom 1996)	0.016 (0.04)	13,000,000 (4,600,000)
Small Rivers, Few Active Deep Landslides		
Freshwater Creek Sediment Budget (1988-1997, Suspended load estimate from sediment input budget, Pacific Lumber Co. Watershed Analysis (WPN 2001))	13 (34)	340-430 (120-150)
Freshwater Creek Gauge Data (Suspended load yield, Redwood Sciences Laboratory, WY 2000)	13 (34)	380 (130)
Jacoby Creek (Suspended load yield extrapolation from data, Lehre and Carver 1985)	14 (36)	440 (155)
Redwood Creek (Suspended load yield extrapolation from data for 1973-74, Nolan and Janda 1995b)	0.6-4.0 (1.6-10.3)	90-310 (30-110)
North Fork Caspar Creek (Suspended load yield, 1990-1995, post-logging period, Lewis 1998)	1.8 (4.7)	130 (47)
North Fork Caspar Creek (Suspended load yield, 1963-1995, Cafferata and Spittler 1998)	1.8 (4.7)	380 (130)

These data suggest that aquatic ecosystems and organisms have evolved with relatively high levels of erosion and sedimentation, and that in watersheds where deep seated landslides and earthflows characteristic of Franciscan mélange are common and active, high levels of erosion and sedimentation are to be expected, regardless of management influences. Furthermore, the data suggest that smaller watersheds where large, active landslide complexes are found, such as those found on the Eel River and Redwood Creek, are less likely to have extremely high erosion rates. Thus, anadromous fish may have historically had access to watersheds with lower erosion and sedimentation rates, despite extreme erosion rates in some locales. These generalizations are limited by local geologic and management history at the watershed scale, however, it is fair to say that regardless of the efficacy of any future efforts to prevent excessive erosion from management, there will be episodes of high erosion and sedimentation rates at various spatial and temporal scales in streams draining the northern California forest landscape.

Comparison of erosion rates attributed to forest management to background erosion rates provides valuable perspective on the significance of high natural erosion rates and management impacts. Recent investigations of northern California erosion rates at the watershed scale have been conducted by a variety of contractors for the U.S. Environmental Protection Agency (EPA) and the North Coast Regional Water Quality Control Board (NCRWQCB). These studies are accessible via the internet at http://www.epa.gov/region09/water/tmdl/index.html. Several of these studies were analyzed to develop a common quantitative format allowing for the results to be compared and to assess whether any general conclusions may be drawn with respect to harvest effects and road effects on erosion rates in the Plan Area. The most general form of the results of this review is presented in Table E-2 below.

Table E-2. Summary results of recent regional erosion source studies in northern California.

Watershed	Background[1] (% of total)	Management Sources	
		Mass Wasting[2] (% of total)	Surface Erosion, Road Erosion, Other Sources[3] (% of total)
Sproul (S.Fk.Eel)	24	19	57
Tom Long (S.Fk.Eel)	71	5	24
Hollow Tree (S.Fk.Eel)	43	24	33
Noyo River	58	13	28
Upper S. Fk. Trinity	66	11	23
Lower S. Fk. Trinity	68	21	10
Hayfork Cr. (S. Trinity)	49	1	50
Freshwater Cr.4	40	16	44
Mean	**52**	**14**	**34**
Range of Values	**24-71**	**1-24**	**10-57**

Notes
1 Includes streamside landslides thought to be of natural origin and all deep seated landslides.
2. Includes road and harvest related slides; harvest related slides are typically assumed to be triggered by harvest if they are observed in harvested area, regardless of actual triggering mechanism.
3 Road surface erosion (sheet and rill erosion of road tread and cut slopes) is the dominant surface erosion process assessed; additional road erosion is from gullies and other road-drainage related erosion. Other sources (e.g. bank erosion) are relatively small.
4 Pacific Lumber Co. Watershed Analysis (WPN 2001)); all others are TMDL studies by USEPA or NCRWQCB.

These data on erosion sources represent conditions over roughly the past 30 years, with the implementation of the California Forest Practice Act beginning early in the period for which the data are summarized. The studies shown were selected in part because they are generally comparable with respect to the erosion processes for which rate estimates were developed and to techniques used to develop erosion rate estimates. The summary presented in Table E-2 should be interpreted with some caution owing to remaining differences in the methods employed and differing scales of different studies. The mean values reported in Table E-2 are in agreement with a similar, prior investigation based on intensive erosion surveys of Redwood National Park (Hagans and Weaver 1987), suggesting that the data in Table E-2 are reasonably well-supported and representative of regional conditions. On the basis of these data, management-related erosion at the watershed scale typically induces increases in erosion of about 100%, ranging from about 30% to over 300%. The data indicate that management erosion sources other than mass wasting, primarily road-related erosion, are believed to be at least as large or larger than management-related mass wasting (Lower S. Fk. Trinity in Table E-2 is the lone exception).

E.3.2 Erosion Sources and Processes

E.3.2.1 Surface Erosion

A common source of sediment input to watersheds is surface erosion. Surface erosion can be major contributor of sediment in areas where soils are composed of granite or highly fractured marine sedimentary rocks (Furniss et al. 1991). Surface erosion is a two-part process in which particles are first detached and then transported downslope. The two hydrologic processes that transport surface erosion are channelized erosion by constricted flows (rilling and gullying) and sheet erosion in which soil movement is non-channelized (rolling and sliding) (Swanston 1991).

Increases in channelized and non-channelized erosion occur when the infiltration capacities of soils are reduced by management activities, large storm events or fires. Chamberlain et al. (1991) reported that the potential for surface erosion is directly related to the amount of bare soil exposed to rainfall and runoff. A study in Redwood National Park using erosion pins (Marron et al. 1995) found that erosion following logging on soils derived from sandstone was not significant to the watershed sediment budget, but that logging on soils derived from schist may be significant. Higher erosion rates tended to occur where rill erosion was more common, which was associated with tractor-harvest, and to a lesser extent, cable yarding, on schist soils. The study examined soil detachment and local ground surface lowering, but did not assess delivery of eroded sediment to streams. Hagans and Weaver (1987) analyzed the data used by Marron et al. (1995), as well as data on percent bare soil following harvest and data on sediment delivery to streams from surface erosion processes on logged areas, including skid trails, for the lower Redwood Creek basin for the period c. 1954-1980, and concluded that only 4% of erosion was caused by sheet and rill erosion. Rice and Datzman (1981) conducted detailed surveys in northern California of 102 harvested plots averaging about 11 acres in size over a range of geologic and slope conditions. In aggregate, they found that two-thirds of the observed erosion was associated with roads, landings or skid trails. Surface erosion in the form of rills and gulleys not associated with roads, landings or skid trails (i.e. harvested areas) accounted for about five percent of total erosion.

Surface erosion by rainsplash and sheetwash processes from roads (including cut slopes), watercourse crossings, landings, skid trails and ditches may all contribute to substantial increases in surface erosion and increased delivery of sediments into stream channels (Reid and Dunne 1984; Luce and Black 1999; Duan 2001). Road erosion estimates in Table E-2 include substantial quantities of sediment from rainsplash and sheetwash processes delivered to streams.

E.3.2.2 Mass Wasting

In steep mountainous terrain, mass soil movement is a major type of hillslope erosion and sediment source in watersheds (Sidle et al. 1985; Swanston 1991). The frequency and magnitude of mass soil movements is governed by hillslope gradient, level of soil saturation, composition of dominant soil and rock types, degree of weathering, type and level of management activities, and occurrence of climatic or geologic events.

Mass soil movements are usually episodic events and tend to contribute significant quantities of sediment and organic debris to stream channels over time intervals ranging from minutes to decades (Swanston 1991). The resultant sediment and organic debris may have a profound effect on a stream channel including large increases in coarse and fine sediments, shifts of existing bed-load, and increases in woody debris that can lead to partial or complete stream blockages.

Forest management practices can affect slope stability by changing vegetative cover, hillslope shape, and water flow above and below the ground surface. Different forest management operations have distinct effects on the factors that control slope stability. For two of the major components of forest management operations—road construction (and to a lesser extent skid trail construction) and harvesting trees—the potential consequences with respect to shallow landslide processes and slope stability are relatively well known. These are described briefly below, with more detailed discussion following.

Road and skid trail construction may:

1. create cut slopes and fill slopes too steep to be stable,

2. result in deposition of sidecast material (spoils) that overburdens and/or oversteepens slopes, and

3. divert and/or concentrate both surface and subsurface runoff.

Harvesting trees:

1. reduces effective soil cohesion by disrupting networks of interlocking roots from living trees, and

2. increases soil moisture by reducing interception of precipitation and evapotranspiration of soil water.

The actual influence of specific forest management activities on slope stability, however, depends on the design and construction of the road network, density of residual trees and under-story vegetation, rate and type of revegetation, topography, material strengths, patterns of surface and subsurface flow, and patterns of water inflow (Sidle et al. 1985; Yoshinori and Osamu 1984). Landslide rates associated with roads are generally much greater than landslide rates associated with timber harvest alone (Sidle et al. 1985).

Changes in canopy interception and evapotranspiration following timber harvest tend to increase soil moisture. This is significant because greater soil moisture reduces the amount of precipitation from a given storm event required to cause soil moisture levels to reach a critical level. This relatively simple qualitative statement regarding soil moisture does not account for complex spatial and temporal effects of vegetation change on hillslope hydrology that could affect slope stability. The potential hydrologic effects are less understood in comparison to the foregoing effects, and therefore have greater attendant uncertainty with respect to effects of forest management.

Timber harvest activities (falling and yarding) not directly associated with roads can increase direct sediment input to streams through surface erosion and mass wasting. Timber harvest may increase the amount of bare soil exposed to rainfall and runoff, leading to increased surface erosion. The occurrence of mass wasting may also increase after timber harvesting, depending in part on the type and intensity of harvest methods (Rood 1984; Swanson et al. 1987). Sidle et al. (1985) reviewed mass wasting surveys concluded during the 1970's and found that mass wasting rates (landslide volume per unit area per unit time) increased from 0 to 40 fold, with the median increase being 3.7 times the rate for undisturbed forest. The substantially lower proportion of increase in erosion from harvest-related landslides relative to data in Table E-2 may be attributable to at least three factors. First, Sidle's review represented historical harvest practices prior to 1980, whereas the reviews in Table E-2 begins in the 1970's. Second, Sidle's review does not distinguish between eroded sediment and delivered sediment, and probably represents erosion rates rather than sediment delivery rates. Consequently, sediment eroded and subsequently deposited on hillslopes is presumably included in Sidle's ratios. Third, the background erosion rates in northern California (e.g. Table E-1) are generally higher than in the areas cited in Sidle's review; hence the proportional increase related to harvest would be lower in the Plan Area (consistent with Table E-2).

Separating the effects of timber harvest activities from the associated yarding, construction, maintenance and use of skid roads and the forest road system may be difficult. Further, the results vary between watersheds. Most studies indicate that the sediment inputs from timber harvesting alone are less than those of the associated road network (e.g. Table E-2, also see Sidle et. al. 1985, Raines and Kelsey 1991, Best et al. 1995).

The Oregon Department of Forestry study of landsliding associated with the high intensity, low frequency storms and flood events in February and November 1996 (Robison et al. 1999) revealed that in areas with slopes > 60%, average sediment delivery was about 2.5 times higher for 0-9 year age class of timber compared 100-year plus age class. In contrast, for the 10-30 year old age class, half of the study areas had lower erosion rates compared to 100-year plus age class. The results reflect the short-term impact of a very large storm and therefore likely overestimate the long-term impact of harvesting.

Federal and state regulatory programs have recently required development of TMDL calculations for designated watersheds in northern California. The primary tool utilized to date for development of TMDLs has been quantitative sediment source assessments (sediment budgets for erosion sources). Although the data collected from the TMDL studies is not sufficient to quantitatively evaluate the impact of harvesting, the data does suggest that harvest-related slides, on average, contribute less sediment than background ("natural") and road-related erosion sources.

In connection with regulatory action by the State of California North Coast Regional Water Quality Control Board against Pacific Lumber Company (PALCO), sediment source studies were conducted by Pacific Watershed Associates for Bear River (PWA 1998b), Jordan Creek (PWA 1999b), North Fork Elk River (PWA 1998a), and Freshwater Creek (PWA 1999a). The latter study included a landslide inventory that was expanded in a subsequent Watershed Analysis. The results from the Bear, Jordan and North Elk can be interpreted to reveal a 2.3 to 11 times increase landslide rates associated with harvesting when the effects of recent harvest and high intensity, low recurrence rainstorms and floods in 1995 and 1997 are considered over a period of about 25 years. However, these interpretations on the impacts of harvesting must be viewed with caution, since the majority of those landslides were associated with very large storms that occurred over a very short time period.

The PALCO Freshwater Creek Watershed analysis is the only recently published analysis, which specifically looked at landslide rates in clearcut, partial cut, and forested areas in northern California. Landslide rates (#/acre/yr) in clearcut areas were on average 2.3 times higher compared to second growth unthinned areas. The impact in headwall swales was higher, about 5 times higher for clearcuts compared to second growth. No difference was apparent between thinned second growth and uncut second growth.

Preliminary landslide data from Hunter Creek mass wasting assessment revealed that landslide delivery rates in clearcut units were between 1.0 and 1.7 times higher than uncut forested areas for the 1958 to 1972 air photo period. The majority of this impact was associated with the intense 1972 storms and long term impacts will likely be less. Cafferatta and Spittler (1998) found little difference between landslide rates in clear cut areas and mature forest in northern California.

These results suggest that landslide rates on harvested areas do not uniformly increase, and that there is considerable uncertainty regarding the circumstances under which reported increases have occurred. It is possible that increased risk avoidance in development of timber harvest plans under present-day professional and regulatory standards help to explain the results of the three studies noted above and those in Table E-2 relative to earlier studies (Sidle et al. 1985).

The changes in physical processes associated with timber harvesting (timber removal alone) are reduced root reinforcement of shallow soils by root-wood deterioration and, to a lesser extent, temporary increases in water input and soil moisture because of reduced evapotranspiration and reduced rainfall interception (or increased throughfall). Whether or not sediments related to timber harvest activities actually enters a watercourse is related to local topography and the proximity of the timber harvest to a watercourse.

E.3.2.2.1 Reduced Root Reinforcement

After forest removal, the gradual decay of small tree roots can predispose certain slopes to failure (Burroughs and Thomas 1977; O'Loughlin and Ziemer 1982; Wu and Swanston 1980; Ziemer 1981a; Ziemer 1981b; Ziemer 1981c; Ziemer and Swanston 1977). Root systems contribute to soil strength by providing effective cohesion (Sidle et al. 1985). Studies have shown that most of the original root reinforcement is lost 4 to 15 years following harvest in a Douglas-fir and pine forest. Redwood and hardwood stands, which dominate the commercial timberlands in the 11 HPAs, resprout after cutting; in these stands a significant loss in root strength is less likely to occur. Landslide susceptibility may also be a function of species composition and spatial variability of root reinforcement (Schmidt et al. in press).

The timing of landsliding, however, may not always be coincident with maximum root deterioration because of the relatively low frequency of occurrence of required storm thresholds (Cafferata and Spittler 1998). Recently harvested areas in the Elk River

(PWA 1998b) and Bear Creek (PWA 1998a) watersheds in Humboldt County experienced unusually high landslide rates in part because a series of low frequency, high intensity storms between 1994 and 1997. These landslide rates may reflect hydrologic influences as much or more than root strength losses. On the other hand, according to Montgomery et al (2000), storms with recurrence intervals less than four years are associated with many landslides in the Oregon Coast Range. In any case, the extent to which losses in root reinforcement of soil trigger landslides depends in part on the intensity of harvest and in part on the timing of subsequent rainstorms, particularly in the “window” of reduced root reinforcement up to about 15 years.

The effect of root strength is most apparent in shallow cohesionless soils on steep slopes (Chatwin et al. 1994; Sidle and Swanston 1982). Soil cohesion from root systems rarely extend to a depth of > 1.5 ft in coastal Oregon (Schmidt et al. in press). Most of the soil reinforcement by roots is therefore a function of the lateral spread of roots. The root strength in the upper portion of the soil column provides little, if any, additional stability to deep-seated landslides where failure planes often exceed 20 feet in depth (Sidle et al. 1985; Yoshinori and Osamu 1984) or in soils that have high cohesion. Landslide susceptibility may also be a function of species composition and spatial variability of root cohesion (i.e. spacing and distribution of root networks of conifers, hardwoods and shrubs; Schmidt et. al. in press).

Modeling studies of shallow landslides and the effects of different silvicultural systems on root strength suggest that partial cutting, thinning and shelterwood techniques result in substantial increases in root strength and substantial decreases in probability of slope failure (Sidle 1992; Krogstad 1995). In addition, understory vegetation often represents a substantial component of total root cohesion (Schmidt et al. in press), suggesting that efforts to suppress understory vegetation following timber harvest may reduce root cohesion and increase the potential for shallow landslides on susceptible slopes.

E.3.2.2.2 Decrease in Evapotranspiration and Rainfall Interception

Evapotranspiration can influence soil water recharge and subsurface flow and thus has the potential for affecting shallow and deep-seated slope stability. The removal of vegetation from a hillside may locally increase the level of ground saturation by reducing the amount of water intercepted and transpired by the canopy (Keppeler et al. 1994; Keppeler and Ziemer 1990; Swanson et al. 1987; Swanson 1981). Where slopes are marginally stable, the resulting increased soil moisture and higher pore pressures may increase both the rate and duration of slope movement. The effects of reduced transpiration and rainfall interception are diminished as vegetation becomes re-established.

Most shallow slides are triggered by peak groundwater levels during high-intensity rainfall events in the winter months when vegetative transpiration rates are already low. Once winter moisture conditions are attained, generally by early December, the difference in soil moisture between logged (clear-cut) and unlogged slopes is virtually indistinguishable (Gray 1977). On the other hand, reduced evapotranspiration may allow near-surface soils to become wetter sooner and stay wetter longer and therefore expose the slope to a potential triggering storm event for a longer period during the wet season.

Canopy interception during storms reduces water delivery to the soil by about 15 to 35% in coniferous forests (Dunne and Leopold 1978). To the extent that some landslides are triggered by relatively short duration bursts of high intensity precipitation (Wieczorek 1996), the loss of canopy would be expected to increase the potential for landslides in susceptible areas. Over seasonal time periods, median canopy interception for coniferous forests is about 22% (Dunne and Leopold 1978). This increment of additional water input to soil that could result from timber harvest could increase the frequency of critical soil moisture conditions when landslides are most likely to occur.

The actual effect of an individual timber harvest on porewater pressures, however, is site specific, dependent upon the characteristics of the underlying parent material (hydraulic conductivity, storativity, shear strength, etc), hillslope geometry, water input, and density of the residual stand. Little change in porewater pressures will be realized in materials with high hydraulic conductivity (i.e. drain rapidly) and/or high storativity (i.e. high porosity) compared to materials that have both low hydraulic conductivity and low storativity.

Upslope clearcut harvesting may potentially influence downslope failures by altering the water balance at the hillslope scale. This was suggested as a potential mechanism contributing to observed landsliding in Bear Creek (see PWA 1998a), a tributary to the Eel River in southern Humboldt County, California. The hypothesis holds that the scale dependent effect would be greatest on larger drainages and where the entire slope from ridge top to near valley bottom was harvested (Tom Spittler, pers. comm., 1998). Modeling studies revealed potentially significant effects of upslope harvest on the stability of historically-active, deep-seated landslides at a site in western Washington (Miller and Sias 1998). Additional research would be required to test this hypothesis and to quantify the attributes where such a process is most important.

E.3.2.3 Deep-Seated Landslides

Natural mechanisms that may trigger deep-seated landslides include intense rainfall, earthquake shaking, and erosion of landslide toes by streams. It is generally acknowledged that deep-seated landslides (earthflows and rockslides) may be destabilized by undercutting of the landslide toe (e.g. by streambank erosion or excavation of road cuts), by adding significant mass to the landslide body (e.g. disposing of spoils from grading or excavation projects), or by significantly altering the groundwater conditions in a landslide (e.g. diversion of road drainage into head scarps or lateral scarps) (TRB 1996, Ch. 16). Deep-seated landslides may also be affected by these hydrologic changes associated with reduced evapotranspiration reduced canopy interception during rainstorms (DMG 1997). Potential increases in groundwater associated with timber harvest in areas upslope of active deep-seated slides may also be important.

Reduced evapotranspiration may add substantially to the annual groundwater flux. Measurements of change in annual stream runoff provide an estimate of the magnitude of change in evapotranspiration following timber harvest. Data from two watershed experiments at Caspar Creek in the redwood region of coastal California using partial cut (65% volume removal) and clearcut harvest (50% of watershed) techniques both indicated an average annual increase in runoff of 15% (Keppeler 1998). The increase in groundwater implied by these experiments is a potential risk factor for increased activity of deep-seated landslides.

Miller and Sias (1998) modeled the effect of timber harvest on groundwater conditions and slope stability of a large, deep-seated landslide in glacial lacustrine sediments adjacent to a large river channel. They predicted that timber harvest in the groundwater recharge area of the landslide would produce very small decreases in the factor of safety, suggesting that harvest would contribute to landslide movement only if the landslide were at or near the threshold of stability. This suggests that active deep-seated landslides are most likely to be affected by harvest-induced changes in groundwater, while inactive and dormant slides are less likely to be affected.

Iverson (2000) developed a model of landslide triggering in response to rainfall. For large deep seated slides with low hydraulic conductivity and large contributing drainage area, landslide force balances driven by hydrologic factors change over periods of time on the order of months to years. Consequently, large deep seated landslides are expected to be sensitive to long-term cycles of precipitation. While it is implied that changes in evapotranspiration could also affect landslide force balances, this potential effect would be substantially reduced by limiting the extent or intensity of harvest to avoid sharp, persistent declines in evapotranspiration.

The relatively few regional empirical landslide studies have produced varying conclusions on the effect of timber harvesting on earthflow stability. Short-term increases in ground displacement following clear cutting have been documented on several active earthflows in the Coast Range and Cascades of Oregon (Pyles et al. 1987; Swanson et al. 1988; Swanson et al. 1987; Swanston 1981). In contrast, work by Pyles et al. (1987) on the Lookout Creek earthflow in central Oregon concluded that timber harvesting was unlikely to induce a large increase in movement, primarily because the slide was well drained.

In summary, previous studies suggest that forest management activities can potentially increase the occurrence or rate of movement of deep-seated landslides. Recognition of active landslides and avoidance of management practices that are known to increase risks of movement can reduce the overall risk of erosion associated with deep landslides. Site-specific conditions pertaining to individual slides will always be important in development of site-specific forest management plans, nevertheless, substantial uncertainty is likely to remain regarding predicted effects of management on slide activity. Deep landslides are relatively common, naturally occurring geologic features in northern California that will continue to generate substantial quantities of sediment delivered to streams, regardless of management influences.

E.3.2.4 Sediment Input from Roads

In the past 25 years studies and reports have shown that road construction for timber harvesting causes great increases in erosion rates within a watershed (Haupt 1959; Gibbons and Salo 1973; Beschta 1978; Cederholm et al. 1980; Reid and Dunne 1984; Swanson et al. 1987; Furniss et al. 1991). Roads affect watersheds by modifying natural drainage patterns and by accelerating erosion and sedimentation, thereby altering channel stability and morphology. If proper construction techniques and maintenance practices are not followed, sediment increases following road construction can be severe and long-lasting. Gibbons and Salo (1973) concluded that the sediment contribution per unit area from forest roads is usually greater than that contributed from all other timber harvesting activities combined. Cederholm et al. (1980) reported a significant positive correlation between the percentage of basin area in road surfaces and percentage of fine sediments (less than 0.85 mm) in spawning gravels.

Forest road systems and their associated watercourse crossings in steep coastal watersheds have the potential to be a major cause of mass soil movements (Best et. al. 1995; Sidle et al. 1985; many others). Road inventories conducted in the Pacific Northwest have reported that erosion from older roads may contribute 40 to 70 percent of the total sediment delivered to the system (Best et al. 1995; Durgin et al. 1988; McCashion and Rice 1983; Raines and Kelsey 1991; Rice and Lewis 1991; Swanson and Dryness 1975).

Raines and Kelsey (1991) developed a sediment budget for Grouse Creek, a tributary to the South Fork Trinity River. These authors concluded that within the Grouse Creek watershed, erosion rates from managed lands were 1 to 6 times higher than erosion rates in unmanaged lands, and erosion rates from roads were 20 to 140 times higher, depending on the time period studied. Road related erosion was the largest single source of sediment volume per unit area (Raines and Kelsey 1991). Sidle et al. (1985) reported that mass soil movements associated with forest roads were 30 to 346 times greater per unit area (median=125) compared to undisturbed forest, consistent with findings by Raines and Kelsey (1991) for Grouse Creek. Fluvial erosion of gullies related to road drainage problems (plugged culverts and resulting stream diversions) accounted for 16% of the sediment budget for Garrett Creek in Redwood National Park (Best et al. 1995). Cederholm et al. (1980) reported that in Washington’s Clearwater watershed 60% of road related sediment production was from associated hillslope failures and that road surfaces accounted for 18-26% of all sediment production. These and many other studies demonstrate that roads are typically the dominant element in management-related erosion in forested upland watersheds.

Increases in hillslope failures due to roads are affected by variables such as hillslope gradient, soil type, soil saturation, bedrock type and structure, management levels and road placement. However, the literature suggests that road placement is the single most important factor because it affects how much the other variables will contribute to slope failures (Anderson 1971; Larse 1971; Swanston 1971; Swanston and Swanson 1976; Weaver and Hagans 1994). Specific road-related landslide triggering mechanisms responsible for road-related mass wasting are described below.

Recently, techniques have been developed to improve the construction and maintenance of forest roads which minimize erosion and sedimentation and should be incorporated into new and existing road networks (Weaver and Hagans 1994). However, a road construction and maintenance crew that is skilled in these techniques and motivated to do quality work is vital to the success of a low impact forest road network.

E.3.2.4.1 Oversteepening

Midslope roads may require cut slopes which create a slope angle too steep for stability. These steep new slopes, coupled with the loss of root strength and increased water inputs (as discussed below) may be subject to surface erosion and landsliding. Modern road building techniques make these roads infrequent in new construction.

E.3.2.4.2 Fill Materials

Placement of thick unengineered fill onto steep and potentially unstable slopes can lead to slope failures by increasing slope weight and altering local groundwater conditions. In addition, inadequate or poorly designed road drainage can result in runoff diverted onto loose and potentially unstable fill material. Saturation of fill significantly increases potential for slope failure. Further, loss of soil strength from decomposition of organics incorporated within the fill may ultimately result in slope failures several years or decades after road construction.

E.3.2.4.3 Concentration of Road Drainage

The concentration of road runoff from inadequately or improperly spaced road drains and/or the augmentation of runoff resulting from rerouting flow in road ditches from one drainage basin to another can saturate the soil more quickly and more frequently, leading to increased likelihood of slope failure. Undersized culverts that become plugged with debris and overtopped during large rainfall events can lead to failure of the fill at the crossing or, if runoff is diverted down the road, failure of an adjacent slope.

Whether or not sediment from road-related surface erosion or mass wasting events actually enters a watercourse is related to local topography, the proximity of the road to a watercourse, and whether or not it the road is connected hydraulically to that watercourse.

E.3.2.4.4 Reduced Bank Stability

Timber harvest in riparian areas has the potential to reduce bank stability and reduce the capacity of the riparian zone to act as a filter strip for sediment transport from upslope sources. These potential losses of riparian function result from soil compaction and exposure via heavy equipment operation, and loss of vegetative root strength and structure due to the removal of harvested trees and damage to other riparian vegetation.

Reducing the capacity of the riparian zone to act as a filter strip essentially increases sediment input to watercourses, the effects of which are described above. Loss of bank stability may lead to channel widening, increased sediment input (from the eroding banks), and a decrease in habitat depth and complexity. Channel widening in turn reduces canopy cover, increasing stream temperatures, and reducing organic input to the stream.

E.3.3 Sediment Transport Processes

The following discussion addresses several aspects of erosion and sedimentation processes. First, sediment transport mechanics are described, followed by a discussion of general watershed scale erosion and sedimentation phenomena. The spatial and temporal relationship between specific erosion processes and sediment transport processes are then developed in greater detail, including considerations regarding timing of sedimentation and effects on aquatic habitat. Next, a discussion of natural factors that mitigate sedimentation effects is presented, followed by management considerations regarding strategies available to minimize sedimentation impacts to aquatic habitats.

This Section discusses the distinctive characteristics of three modes of sediment transport in stream channels: bedload, intermittent suspended load, and suspended load. Although each of these processes corresponds to a generally consistent size range of sediment, it should be noted that these processes occur over a physical continuum, and that there is substantial overlap between these modes of sediment transport. Depending on the intensity (i.e. velocity) of stream flow, the sediment transported in one mode may be transported in another mode. Many textbooks provide a description of sediment transport mechanics (e.g. Richards 1982, Raudkivi 1990, Yang 1996).

E.3.3.1 Bedload Sediment

The typical size of material transported primarily as bedload in upland streams is gravel (2 mm to 64 mm diameter) and cobble (64 mm to 256 mm diameter). Larger material (boulders) are also transported as bedload, however, sediment particles of this size move relatively slowly and are more likely to form nodes of stability in stream channels (i.e. boulder steps or transverse bars, Grant 1990).

Bedload is transported by sliding, rolling, or skipping along the streambed. Bedload particles are rarely found in the water column far above the bed. Bedload sediment is typically routed through mountain channel systems slowly, with average annual transport distances from tracer studies of about 300 ft, ranging from about 60 to 1500 ft (NCASI 1999, p. 289). The volume of bedload sediment deposits is typically large in comparison with the annual transport rate.

Bedload sediment is broken and abraded as it collides with other sediment clasts on the bed or in transport; this gradual process of breakage and declining size is known as attrition. The attrition process converts a portion of the bedload to suspended load as larger sediment clasts produce smaller sediment particles. The attrition rate is usually estimated as a function of transport distance in the channel network. The magnitude of attrition varies, but as much as half of bedload material may be converted to suspended sediment over transport distances of about 20 km (Collins and Dunne 1989). Where bedrock is extremely weak (e.g. Wildcat Group rocks near Humboldt Bay), however, the attrition rate may be much higher, and where bedrock is relatively strong, the attrition rate much lower. Intermittent suspended load (also called “saltation load” by Raudkivi (1990)) is typically comprised of fine gravel and coarse sand. It is transported partly in contact with streambed, and partly in suspension, depending on flow intensity and local channel morphology. These sediment sizes are often found in sorted deposits in the lee of channel obstructions or in pools, and are typically finer than typical median grain size on the surface of point bars and alternate bars. Intermittent suspended load is transported through channel systems more quickly, provided it is not deposited underneath coarse armor layers of bed and bar deposits. The typical annual velocity of intermittent suspended load is between that of bedload and suspended load, and is on the order of 1000’s of ft to miles.

E.3.3.2 Suspended Sediment

Sand, silt and clay sizes (< 2 mm diameter) comprise the suspended sediment load in most upland stream systems. The sand fraction (> 0.06 mm and < 2 mm) is often a major constituent of the intermittent suspended load and a substantial constituent of the bedload. In many low-gradient rivers, sand is the dominant component of the bedload.

Such conditions are found at the mouths of several coastal watersheds in northern California.

Suspended load is transported in suspension in the water column in relatively low-intensity flows. It typically is transported through the channel system rapidly; sediment velocity for suspended load is nearly equal to water velocity. If suspended sediment is present in or on the margins of channels it will be entrained rapidly with increasing stream discharge. This suspended sediment can be subsequently deposited in low-velocity areas downstream as stream discharge declines. Sediment of this type is rarely deposited in large quantities within the streambed in upland channel networks except in low-velocity environments such as unusually low gradient or hydraulically rough reaches, channel margins, side channels, and behind flow obstructions.

A finer component of the suspended load is sometimes referred to as “wash load” (Raudkivi 1990; Reid and Dunne 1996). Wash load is usually comprised of clay and fine silt, and is distinctive in that once entrained in the water column of a stream, it will not settle out. Hence, this size fraction is found in only very small quantities in the bed of upland streams.

Much of the suspended load is removed from the upland stream system very rapidly and is deposited in floodplains, estuaries and offshore marine environments. Suspended load accounts for about 70 to 90% or more of the total sediment load in northern California watersheds. This includes the wash load, the suspended load and, depending on measurement technique, some portion of the intermittent suspended load measured

Suspended load transport in many northern California streams (e.g. Caspar Creek, Lewis 1998) is correlated with turbidity (an optical characteristic of water quantifying its clarity or cloudiness). Hence, the supply of suspended load sediment size fractions is the chief control on stream turbidity, a measure of water quality used by the California Regional Water Quality Control Board in its Basin Plan for northern coastal California. The silt and clay fraction in the suspended load (this is typically equivalent to the wash load in most upland streams in northern California) strongly influences turbidity, hence control of sediment sources rich in silt and clay will provide the greatest reduction in turbidity.

E.3.3.3 Watershed-scale Sediment Transport Concepts

The relationship between sediment inputs to a channel network and sediment transport capacity of the channel network will have a strong influence on channel sedimentation status (e.g. Montgomery and Buffington 1993, Buffington and Montgomery 1999). For example, channel systems that are said to be “transport-limited” are expected to contrast with “supply-limited” systems. The influence of sediment supply on bedload transport processes has been the subject of much research, including recent field-based modeling work (Lisle et al. 2000) that suggests that a higher proportion of the streambed tends to be mobilized by competent flows in channels with a higher sediment supply. In addition, some studies suggest that high sand loads in a gravel bed stream may increase bed mobility (Iseya and Ikeda 1987). Increased bed mobility would increase bed scour potential.

E.3.3.3.1 Transport-limited Channels

Transport-limited channels are defined by high sediment supply such that supply is greater than sediment transport capacity. Under such conditions, sediment transport rates would be proportional to flow, that is, abundant transportable sediment is available and the primary limit on sediment transport is flow magnitude and duration. The channel bed in transport-limited channels is expected to be relatively fine, typically composed of finer gravel and sand with little armoring of the bed surface. Transport-limited channels may be found where there are abundant sediment inputs (e.g. recent concentrated inputs from landslides) or where channel slope declines rapidly (e.g. where a relatively steep confined channel reaches a broad valley with lower channel gradient).

E.3.3.3.2 Supply-limited Channels

Supply-limited channels are defined by high sediment transport capacity relative to sediment supply. Sediment transport rates are high when sediment is available for transport, but relative to the transport-limited condition, the relationship between stream flow and sediment transport is erratic. The channel bed is expected to be relatively coarse, with frequent armoring of bed deposits and frequent bedrock exposures. Although conditions are variable, depending on channel and valley morphology and watershed erosion history, many of the smaller, steeper upland streams important for anadromous fish would be expected to be supply-limited. This expectation is conditioned largely on the high degree of confinement, moderately high slopes, and moderate to intense storm runoff typical of such streams (i.e. factors suggestive of high sediment transport capacity).

Climatic variability is also an important temporal factor in that, during periods of low frequency of intense rainstorms (regional-decadal scale), sediment transport capacity could be significantly reduced. This could conceivably shift channel conditions toward transport-limited from supply-limited in systems or reaches where sediment supply and transport capacity are relatively balanced.

E.3.3.4 Spatial and Temporal Relationship Between Sediment Input Mechanisms and Sediment Transport Phenomena

Coarse sediment is approximately equivalent to bedload sediment excluding the sand that is transported in intermittent suspension and which is thought to be detrimental to spawning habitat. Although the precise size range corresponding to coarse sediment varies among observers and objectives, coarse sediment referred to in this Section is considered to be > 2 mm diameter, and includes gravel, cobbles and boulders.

Landslides are generally the major sources of coarse sediment. Shallow rapid landslides (debris slides and debris flows) generally include significant proportions of coarse sediment, depending on the proportion of gravel in the displaced soil and colluvium. Deep-seated landslides (translational/rotational slides) include coarse sediment from soil and colluvium overlying the slide, as well as coarse sediment derived from the underlying bedrock. Consequently, deep-seated slides have the potential to introduce large sediment clasts (boulders). Even earthflows, which have high proportions of fine sediment inputs (DMG 1997), may introduce very coarse rock that cannot be mobilized by the stream, thus inducing a steepening of the stream channel (Kelsey 1980).

Channel erosion in headwater streams, particularly in mélange (Best et al. 1995), bank erosion, and soil creep processes also introduce coarse sediment to streams in proportion to the concentration of gravel in the soil material. In combination with natural landslides, bank erosion and soil creep are a major source of natural or "background" sediment inputs. Fluvial erosion associated with gullies created by blocked culverts where roads cross streams, or where blocked culverts cause fill failures, may also introduce large quantities of sediment (Best et al. 1995), with the quantity of coarse sediment depending on the proportion of gravel in the soil material. Under past management practices, road construction may have introduced large quantities of sediment to streams as a result of uncontrolled sidecast disposal of soil, or a as a result of poor road construction techniques and poor maintenance. Although such practices are now prohibited, northern California rivers may yet be affected by the legacy of former practices.

The timing and frequency of coarse sediment inputs tend to be dominated by mass wasting processes. With the exception of channel erosion, bank erosion and soil creep, the erosion processes noted above typically generate sediment inputs that are relatively concentrated near the point of entry to the channel network. Landslide deposits in channels typically include abundant coarse and fine sediment and LWD. Deposits may fill existing channels and induce erosion along stream banks. The transport and downstream routing of such coarse sediment budgets have been investigated both in model and field studies of upland rivers (Benda and Dunne 1997a, 1997b; Lisle et al. 1997 and Lisle et al. in press (re: Floodgate slide)). While it is generally agreed that the local effect is greatest at the point of entry, consistent theoretical statements regarding the magnitude and timing of effects downstream and the governing processes are elusive. Benda and Dunne (1997a) hold that concentrated coarse sediment inputs to a channel network are routed downstream in a kinematic wave that persists downstream. Kelsey (1980) observed this phenomenon in the Van Duzen River. In contrast, Lisle et al. (in press) believe that diffusive processes control the routing of sediment, and that such pulses of input gradually disperse downstream. In either case, the greatest short-term effects with respect to coarse sediment are localized, with only gradual (over a period of years to decades) translocation of effects (typically increased depth of gravel deposits and changes in size distribution of bed material).

Landslide inputs of coarse sediment also tend to be concentrated in time in response to periods of unusual precipitation and streamflow. Conditions that are likely to trigger shallow landslides occur relatively infrequently in northern California (Cafferatta and Spittler 1998). The activity of deep-seated landslides tends to be related to longer-term periods of precipitation (Iverson 2000), or to periods of high streamflow that erode toe deposits and destabilize deep landslide blocks.

Debris flows and debris torrents may have broader impact on streams because high concentrations of sediment and woody debris may be carried several thousand feet or more from the initiation site to the distal end of the deposit. Large portions of the affected channel may be scoured to bedrock, while reaches affected by deposition may aggrade substantially. Moreover, in terrain prone to debris flow (e.g. debris slide amphitheater/slope (DMG 1997)), many potential initiation sites may be present in colluvial hollows, creating potential for more frequent debris flow impacts to downstream channels (Benda and Dunne 1997a). Field evidence indicates that episodes of major debris avalanching in headwater channels in northern California probably occurred at intervals of 300 to 2,000 years, and that smaller but significant episodes probably occur more frequently (Kelsey 1982).

E.3.4 Potential Effects on Covered Species

This Section reviews known potential effects of sediment on the Covered Species and the characteristics of their aquatic habitat. The summary of sediment transport and sedimentation processes provided above is applied to give perspective on the relationship between sediment sources and sedimentation effects on habitat. The discussion distinguishes between the effects and sources of “fine” and “coarse” sediment, and is oriented toward conditions found in northern California streams and watersheds.

Although this Section focuses on sediment effects on aquatic habitat, it must be recognized that sediment is not a singular environmental factor affecting habitat conditions. Stream temperature and habitat morphology, particularly in relation to the influence of LWD, are two other major controls on habitat conditions, and both of these have been or potentially may be affected by watershed management. Areas with generous riparian buffers provide a means to recruit LWD and reduce sediment inputs; when mass wasting events occur in such areas, both LWD and coarse sediment will be recruited to channels along with fine sediment. Coarse sediment (in modest amounts) and LWD can both contribute positively to aquatic habitat conditions in the long term (and often in the short term, particularly LWD). In contrast, chronic erosion from roads (road tread surface erosion, small scale mass wasting of road cut slopes, fluvial erosion of ditches and gullies formed by road drainage) contributes fine sediment to streams. Hence, to the extent that fine sediment negatively affects aquatic habitat, erosion from roads is expected to be relatively more likely to degrade aquatic habitat conditions than modest degrees of mass wasting inputs from riparian buffer zones.

E.3.4.1 Coarse Sediment

In the most extreme case, landslide deposits may bury a channel reach to depths sufficient to entomb any organisms present such as larval tailed frogs, southern torrent salamanders and salmonid eggs in redds in the stream bed. More common and widespread effects resulting from increases in bedload sediment supply may also result in channel aggradation and associated decreases in mean channel depth, decreases in pool depth and more mobile, less stable channels, reducing the quantity of rearing habitat for juvenile salmonids and potentially reducing emergence from redds (Bisson et al. 1992, Sullivan et al. 1987). If water temperatures are not increased, aggradation of the channel due to coarse sediment inputs potentially would have less of an impact on the amphibian Covered Species, because they select for riffle habitat and are generally not found in pools (Diller and Wallace 1996, 1999; Welsh and Lind 1996). Coarse sediment inputs of competent material with a small fraction of fines may actually be beneficial to southern torrent salamanders. Material of this type contains an extensive interstitial network through which the salamanders can move.

Effects of excess coarse sediment on pool habitat are believed to be potentially significant for the salmonid Covered Species. Pool abundance and depth has been positively correlated with salmon and trout abundance and density (Bisson et al. 1982; Murphy et al. 1986). Juvenile coho salmon as observed in Green Diamond’s summer population estimates are found almost exclusively within pool habitats in Plan Area streams (Appendix C7). Pool habitats provide summer rearing habitat, and may act as cool water temperature refugia in the summer (Steele and Stacy 1994). The input of coarse bed materials can result in both increased and decreased rearing capacity for juvenile salmonids (Hicks et al. 1991). Coarse sediment inputs have the potential to negatively impact the fish Covered Species through infilling of pool habitat and the localized burial of redds. Such habitat modification could constitute a take of salmonids if it interfered with the ability of those present to shelter or if it destroyed their eggs.

The relatively slow rate of transport of bedload sediment results in relatively persistent effects, depending on local transport rates and the magnitude of the effect. The slow movement of bedload sediment and the tendency for bedload inputs to be concentrated in space in association with landslides suggests that coarse sediment effects may frequently be localized, affecting stream reaches rather than entire watersheds. With the passage of time, assuming inputs of coarse sediment are reduced, negative effects of coarse sediment on salmonid habitat can be expected to dissipate (Sullivan et al. 1987).

E.3.4.2 Fine Sediment

There are two size fractions of fine sediment to consider, each with different effects on habitat. First, there is the intermittent suspended load comprised primarily of fine gravel and sand. This is distinguished from the suspended load/wash load fraction comprised of fine sand, silt and clay particles.

The erosion sources that supply fine sediment to streams include those identified for coarse sediment, however, they also include significant quantities generated by rainsplash, sheetwash, rill, and gully erosion processes occurring primarily on roads and skid trails.

The timing and frequency of fine sediment inputs are potentially distinct from timing and frequency of coarse sediment inputs. Both coarse and fine sediment inputs resulting from landslides tend to be concentrated in time and space. More dispersed and chronic inputs of fine sediment are likely, however, owing to widely dispersed sources and the high frequency of rainfall-runoff events capable of mobilizing fine sediment from sources areas, particularly roads. Most rainstorms are likely to provide sufficient energy to erode and deliver available sediment from road surfaces to streams. Hence, even in relatively dry years when mass wasting processes are insignificant, substantial road surface erosion would occur. Given the propensity for landslide events to be triggered during relatively intense rainstorms, mass wasting episodes tend to be concentrated in a few years over periods of decades at the watershed scale. During the intervening years of relatively low mass wasting, erosion of fine sediment from roads would likely be persistent, potentially magnifying its impact on aquatic habitat.

As described above, wash load and suspended load travel at velocities similar to average stream velocities. Consequently, suspended sediment effects are transient, but may be persistent if the erosion source is persistent. The intermittent suspended load travels at substantially lower velocities, but is nevertheless significantly faster than coarse bedload. Consequently, fine sediment entering the stream system is rapidly dispersed far downstream, and sand and fine gravel deposits on the bed surface can be routed through channel reaches relatively quickly (Lisle and Hilton 1999).

The effects of increased fine sediment input on the Covered Species vary with sediment particle size. Increased inputs of fine sediments are associated with increased embeddedness of spawning substrates and high turbidity levels (Chapman 1988). Increases in fine sediments deposition into stream gravels can lead to a reduction in spawning success, reduced food production, and loss of benthic cover for over-wintering juveniles (Hicks et. al. 1991; Wood and Armitage 1997). The larvae and adults of the southern torrent salamander and larval tailed frogs utilize the interstices within gravel and cobble substrate, and are not typically found in sandy or silty streams (Bury and Corn 1988; Diller and Wallace 1996, 1999). Salmon and trout spawn in gravel and cobble substrates, and sedimentation or burial of these substrates would likely result in reduced reproductive success for these species (Chapman 1988). Subsurface flow through redds is essential in providing dissolved oxygen to embryos and carrying away metabolic wastes. Sedimentation can reduce the survival to emergence of the covered embryos by reducing subsurface flow, and by creating a sediment 'cap' which prevents hatched fry from emerging (Reiser and White 1988). Accordingly, increased embeddedness caused by increased input from Covered Activities could result in take of salmonids by destroying eggs or fry. Laboratory studies have demonstrated that increases in fine sediment in redds reduces survival to emergence either by entombment or by reducing the supply of oxygenated water to the redd, but field experiments have found more variable effects depending on the experiment, region and other environmental factors (Everest et al. 1987).

As noted above, there are several potential habitat effects associated with the coarser fraction of fine sediment (i.e. sand and fine gravel, intermittent suspended load). These include infiltration of fine sediment into coarse sediment that degrades the quality of spawning habitat. Infiltration of sand and fine gravel in coarser gravel and cobble streambeds has been investigated in both laboratory (Carling 1984) and field (Lisle 1989) studies and show that infiltration rate is proportional to sediment transport rates in the stream. Hence, reductions of fine sediment inputs are expected to result in improved spawning conditions for all Covered Species. The coarser fraction of fine sediment has also been found to collect in pools in some stream systems, reducing the quantity and quality of summer rearing habitat and winter refugia in pools (Lisle and Hilton 1999). The extent of pool filling by fine sediment appears to be related to watershed sediment supply.

Additional effects of excessive sediment inputs of either size class on aquatic habitat include aggradation of stream channels and loss of bank stability, resulting in a wide, shallow channel with low canopy cover, higher water temperatures, and intermittent surface flows in low flow conditions (Swanston 1991). These secondary effects are typically seen in the depositional reaches of streams, making them likely to impact the salmonids but not the amphibian Covered Species.

The finer fraction of fine sediment, primarily silt and clay transported in suspension in water column (suspended load and wash load) is highly correlated with turbidity. High levels of suspended sediment have been found, primarily in laboratory experiments, to have a range of deleterious effects on salmonids. An increase in fine sediments can also lead to chronic levels of turbidity, which may damage the gills of salmonids, reduce their growth rate, impair the ability of fish to locate food, and negatively impact the macroinvertebrate production (Bozek and Young 1994; Sigler et. al. 1984; Newcombe and MacDonald 1991). Negative effects of suspended sediment on juvenile salmonids depend on sediment concentration and duration of exposure, and the interaction of these factors is not well understood (Newcombe and MacDonald 1991). In addition, the availability of localized refugia from high suspended sediment concentrations, such as side channels and backwater pools, may also affect both concentration and duration of exposure. Gregory (1993) indicated that suspended sediment may have some beneficial effects as well, such as providing cover from predators. Thus, fine sediment inputs from the Covered Activities could take salmonids by impairing their ability to breathe, grow and eat.

It is not known if there are any direct effects of increased suspended sediment or turbidity on the amphibian Covered Species. Green Diamond speculates that it has the potential to impact the aquatic dependent larval stages of these amphibians in the same manner as was noted above for the salmonids. In addition, suspended sediments could influence the growth of diatoms on the stream's substrate, which is the sole food for larval tailed frogs. Southern torrent salamanders are less likely to be impacted by suspended sediments, because they occur in seeps, springs and the uppermost reaches of streams that are generally not influenced by the downstream transport of fine sediments. However, Green Diamond believes that it is more likely that increases in suspended sediment (especially the larger particle sizes) would impact the amphibians indirectly by reducing interstices in the substrate and causing substrate embeddedness.

E.3.4.3 Potential Severity of Effects of Suspended Sediment on Salmonids

Newcombe and Jensen (1996) developed a model based on results of previous experiments for the effect of suspended sediment on salmonids in terms of concentration of suspended sediment and duration of exposure. They developed a concentration-exposure function that predicts the severity of the effect on adult and juvenile salmonids. To gain perspective on the effects of suspended sediment on salmonids in the study area, a series of paired suspended sediment and discharge observations representing hydrographs of peak runoff at eight USGS stream gauging stations in northern California with drainage areas ranging from about 1.5 to 30 square miles was evaluated with respect to predicted severity of effects according to Newcombe and Jensen's model.

Subsequent work by Newcombe and Jensen (1996) developed a model to predict quantitatively the effects of elevated suspended sediment on salmonids. This model was employed to evaluate available regional data and assess the potential magnitude of effects of suspended sediment on salmonids summarized below. The objective of this analysis was to develop quantitative perspective on the effects of suspended sediment on salmonids, particularly coho salmon, according to Newcombe and Jensen's (1996) model predicting severity of effects.

E.3.4.4 Methods-Hydrograph Analysis

Nine gauged sites were sampled from existing regional data that was readily available from internet data libraries. A series of paired suspended sediment and discharge observations representing hydrographs of peak runoff at nine USGS stream gauging stations in northern California with drainage areas ranging from about 1.4 to 30 square miles were evaluated (Table E-3). The data was collected in the 1970s, 1980s and early 1990s. Sites with at least 100 paired observations were chosen in order that a relatively large set of potential hydrographs were available for evaluation.

In order to compare discharges among different gage sites, discharge data were divided by the site's drainage area, thus producing unit discharge data (cfs/mi^2). Unit discharge data also provide perspective regarding approximate flow recurrence interval for the flows evaluated. Regional values of the 2 yr recurrence interval event range from about 40 to 80 cfs/mi^2 for streams with the range of drainage areas for these sites (Table E-3).

Table E-3. Summary of USGS suspended sediment gauging stations.*

Station #	Station Name	Area (mi^2)	# Obs	First	Last
11482110	Lacks C Nr Orick Ca	16.9	224	11/22/74	03/20/91
11482125	Panther C Nr Orick Ca	6.1	108	01/12/79	01/04/91
11482130	Coyote C Nr Orick Ca	7.8	100	12/11/78	05/15/89
11482225	Harry Wier C Nr Orick Ca	3.0	169	11/07/73	02/20/80
11482260	Miller C A Mouth Nr Orick Ca	1.4	134	11/07/73	01/22/81
11482450	Lost Man C Nr Orick	4.0	124	10/23/73	02/20/80
11482468	Little Lost Man C A Site No 2 Nr Orick Ca	3.5	192	03/29/74	05/11/89
11530020	Supply C A Hoopa Ca	15.9	123	11/03/81	01/01/84
11532620	Mill C Nr Crescent City	28.6	107	01/16/74	12/24/80

Note

* Of these nine stations, only the Little Lost Man Creek had little historic commercial forestry in its watershed.

After each sites' data were reviewed, data were selected for more intensive sampling periods during peak flows where at least 3 samples were taken in no more than four days; most of the selected data had several samples during a period of up to 4 days. For selected periods of flow, observations of suspended sediment concentration and unit discharge were plotted against time to generate sedigraphs. An example of one of these more intensively sampled events is shown in Figure E-1.

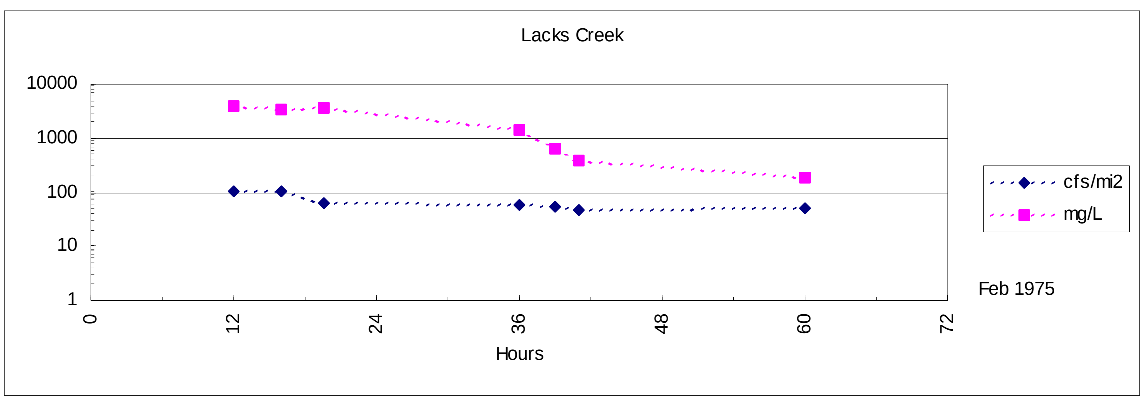

Figure E-1. Example of a more intensively sampled site in Lacks Creek during February 1975.

In some cases, these sedigraphs were then decomposed into shorter time periods based on interpretation of the hydrograph. The intent was to identify discrete storm hydrographs where suspended sediment concentrations appeared to reach sustained high levels, representing a 'worst case' scenario. Within these shorter time periods, the average suspended sediment concentration and unit discharge was calculated (Table E-4). The peak suspended sediment discharge and peak unit discharge also was noted. The time duration was determined by subtraction between the first and last observation.

E.3.4.5 Methods-Severity of Effects

Newcombe and Jensen (1996) reviewed 80 studies that documented fish responses to suspended sediment. Their analysis resulted in six empirical equations that quantify the biological response of fishes to duration of exposure and suspended sediment concentration. The authors quantified fish responses by creating the severity of ill effect (Table E-3).

The equation used to determine the SEV is:

$$Z=a +b(\log_e x) + c(\log_e y)$$

where z is the severity of ill effect, x is duration of exposure (in hours), y is concentration of suspended sediment (mg SS/L), a is the intercept and b and c are slope coefficients (Newcombe and Jensen 1996).

The natural log of the duration of exposure and the suspended sediment concentration were calculated and rounded to the nearest whole number. From there, the tables in Newcombe and Jensen (1996) were used, specifically Figure 1 (juveniles and adult salmonids), Figure 2 (adults salmonids only), and figure 3 (juvenile salmonids only). . Each of these figures presents SEV values for a given suspended sediment concentration and duration. Additionally, data from the appendix also were used to examine SEV values specifically for underyearling coho salmon (see Figure E-2). Similar curves were constructed for smolt and juvenile coho life stages, but the distribution of data was quite uneven and was difficult to interpret directly (unlike the data in Figure E-2).

E.3.4.6 Results

This analysis indicated that the worst case effects were “paralethal” (SEV=9). These conditions existed in Lacks Creek in February 1975 as well as in February 1979, and in Supply Creek near Hoopa in April 1982. All other scenarios indicate sublethal SEV values between 5 and 8.

Table E-3. Potential Severity of Effects (SEV) of suspended sediment on salmonids at nine gauged stations in Northern California.*

Gage Station	Hydrograph Dates	Mean Discharge (cfs/mi^2)	Peak Discharge (cfs/mi^2)	Mean SSC (mg/L)	Max. SSC (mg/L)	Hydrograph Duration (hours)	SEV Case 1	SEV Case 2	SEV Case 3	SEV Case 4
Lacks Creek Near Orick	2/12/1975 2/13/1975	80	102	3115	3900	24	9	9	9	8
	2/13/1975 2/14/1975	65	53	407	658	21	7	8	7	6-7
	2/10/1979 2/11/1979	8	11	127	159	24	7	7	6	4
	2/12/1979 2/13/1979	26	51	734	1140	34	9	9	9	6-7
Harry Wier Creek Near Orick	11/7/1973 11/8/1973	45	94	754	2070	11	7	8	7	7
	11/8/1973 11/9/1973	35	49	348	646	27	7	8	7	5-6
	3/1/1974 3/2/1974	17	21	148	377	19	7	7	6	5
Miller Creek Near Orick	11/7/1973 11/8/1973	46	76	1644	2730	13	8	8	8	8
	11/8/1973 11/9/1973	26	32	404	557	21	7	8	7	6-7
	2/12/1975 2/13/1975	29	32	1340	1530	11	7	8	7	7
	2/13/1975 2/14/1975	26	32	259	495	24	7	8	7	6-7
Lost Man Creek Near Orick	11/7/1973 11/8/1973	43	55	1089	1790	12	7	8	7	7
	11/8/1973 11/9/1973	40	53	224	388	27	7	7	6	5-6
Supply Creek Near Hoopa	4/11/1982 4/15/1982	30	49	337	822	98	9	9	9	5-6
Mill Creek Near Crescent City	3/17/1975 3/18/1975	65	84	286	406	16	7	8	7	6-7
	3/18/1975 3/19/1975	133	153	1119	1450	26	8	8	8	7
Panther Creek Near Orick	3/14/1980 3/15/1980	37	46	328	407	27	7	8	7	5-6
	1/15/1988 1/16/1988	25	29	200	258	23	7	7	6	6
Coyote Creek Near Orick	3/13/1980 3/14/1980	56	61	1783	1880	26	8	8	8	7
	3/15/1980 3/17/1980	24	29	228	296	50	7	7	7	5-6
Little Lost Man Creek Near Orick	3/18/1975 3/18/1975	185	192	2455	2830	1	7	8	6	6

Notes

* SEV values between 4 and 8 are considered "sublethal" and values equal to or greater than 9 are considered as "paralethal to lethal" (see Table E-3). SEV values for Cases 1 through 4 are shown; see below for definitions of cases.

SEV 1 - Adult and Juvenile Salmonids (Figure 1, Newcombe and Jensen 1996)

SEV 2 - Adult Salmonids (Figure 2, Newcombe and Jensen 1996)

SEV 3 - Juvenile Salmonids (Figure 3, Newcombe and Jensen 1996)

SEV 4 - Underyearling Coho (Appendix, Newcombe and Jensen 1996)

Table E-4. Scale of severity of ill effects associated with excess suspended sediment.

SEV	Description of effect
0	No behavioral effects
1	Alarm reaction
2	Abandonment of cover
3	Avoidance response
4	Short term reduction in feeding rates; short term reduction in feeding success
5	Minor physiological stress; increase in rate of coughing, increased respiration rate
6	Moderate physiological stress
7	Moderate habitat degradation; impaired homing
8	Indications of major physiological stress; long term reduction in feeding rate, long term reduction in feeding success; poor condition
9	Reduced growth rate; delayed hatching, reduced fish density
10	0-20% mortality; increased predation; moderate to severe habitat degradation
11	>20-40% mortality
12	>40-60% mortality
13	>60-80% mortality
14	>80-100% mortality

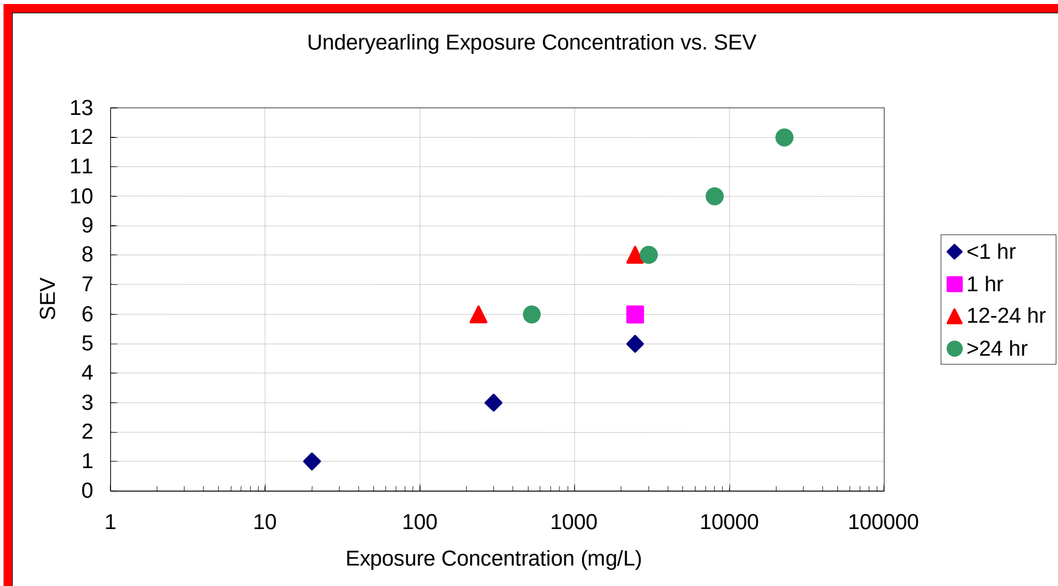

Figure E-2. Data from Newcombe and Jensen's (1996) appendix for "underyearling" coho salmon (SEV Case 4 in Table E-2).

E.3.4.7 Discussion

The suspended sediment data from all but one of the sites evaluated presumably reflect the effects of substantial pre-Forest Practices Act logging practices in watersheds that were managed largely for timber production. Consequently, these data represent relatively poor watershed conditions, probably substantially worse with respect to management-induced erosion than would be found under current conditions.

The results of this analysis show that these streams experience periods of elevated sediment concentrations that are predicted to induce physiological stress on salmonids. This suggests that such stressing conditions are likely to be present to some degree in many northern California streams. Given regional erosion rates (see section E.3.2 above), it may well be that these stressing conditions occur regardless of management influences.

Presumably, forest management would tend to increase the frequency and/or magnitude of these stressing events. The long-term effects of these stressing events on salmonids are not well known. It has been suggested that extended periods of higher turbidity (generally correlated with higher suspended sediment concentration in the region), could interfere with feeding success of juvenile salmonids, reducing the size of smolts, which in turn would presumably reduce survival rates in the oceanic life-stage. In this analysis, data for juvenile coho salmon (see Figure E-2) indicate a somewhat lower SEV score (SEV Case 4) than for salmonids in general. This suggests the possibility that coho salmon may be somewhat better adapted to cope with suspended sediment. Given the typical geologic conditions in the coastal watersheds where these fish evolved, this possibility appears plausible. No similar studies have been done for the amphibian Covered Species to quantify the impact of suspended sediments on any life stage.

Sediment inputs, both coarse and fine, are absolutely essential to maintain a healthy lotic system. However, excess sediment inputs can have diverse and highly negative impacts. As described in the discussions above, the potential impacts from increased sediment inputs vary depending on the primary particle size involved (i.e. coarse versus fine). The impacts are generally cumulative in nature, especially for the finer particle sizes that can stay suspended in the water column and potentially impact regions at great distances downstream of the sediment source. The life history stage of the Covered Species that are potentially impacted by various types of sediment inputs is also variable, but there is the potential for all life history stages to be negatively impacted in a manner resulting in take. Increased sediment inputs can produce a myriad of negative impacts on habitat, such as increased pool filling, embeddedness, increased temperature and turbidity can potentially result in direct mortality, and decreased survival rates of various life history stages of the Covered Species, particularly in early life stages. Such impacts, and more importantly, changes in population demographic parameters, may result in local population declines. Such declines could negatively affect the regional populations of the Covered Species.

E.3.5 Altered Thermal Regimes

E.3.5.1 Altered Riparian Microclimate

The riparian microclimate has potentially important indirect effects on the salmonid Covered Species and aquatic forms of the amphibian Covered Species through alteration of water temperature, which will be discussed in the following Section. However, the riparian microclimate also has potentially important direct effects on the adult forms of the amphibians. Reduction of riparian overstory canopy through timber harvesting could result in increased levels of incident solar radiation reaching the stream and riparian zone during the day and reduced thermal cover at night (Welch et al. 1998). It could also increase exposure to wind in the riparian areas due to an edge effect from an adjacent harvest unit with the overall net effect of increasing daily fluctuations in air temperature and relative humidity. Studies done in areas outside the coastal influence of the 11 HPAs indicate that microclimatic edge effects can be detected as much as 240 meters (787 feet) from the edge of a clearcut (Chen 1991). However, the greatest attenuation of edge effects on microclimatic changes occurs within the first 30 meters (98 feet) of the buffer (Ledwith 1996). Although the impact of altered riparian vegetation on the microclimate is ameliorated by the cool coastal climate in the region, reduction of riparian cover due to timber harvesting has the potential to cause greater daily and seasonal fluctuations in the microclimate of the riparian areas.

In addition, increased coarse sediment inputs from management activities, particularly when it occurs in the form of debris torrents, can result in widening of the channel and loss of streamside vegetation (Swanston 1991). Just as in overstory canopy loss, this has the potential to alter the riparian microclimate by increasing daily fluctuations in air temperature and relative humidity. It is unlikely that increases in air temperature with corresponding decreases in relative humidity during the day would directly impact the amphibians, because the adults are not surface active during the day. However, the corresponding drying effect of increased air temperature and decreased relative humidity could result in the loss of some daytime refugia habitat and nighttime foraging sites. It is also possible that the reduction of thermal cover at night may impact the ability of adults to forage at night.

E.3.5.2 Altered Water Temperature

Loss of riparian overstory canopy through timber harvesting and increased coarse sediment inputs from management activities could result in alteration of the riparian microclimate as described above. However, changes in the riparian microclimate will also result in corresponding changes in the daily water temperature regime. In addition, both reduction of overstory canopy and increased coarse sediment inputs can result in altered water temperature through direct mechanisms. Removal of the riparian canopy will result in elevated summer water temperatures, often in direct proportion to the increase in incident solar radiation that reaches the water surface (Chamberlain et al. 1991). For a given exposure from solar radiation, water temperature increases directly proportional to the surface area of the stream and inversely proportional to stream discharge (Sullivan et al. 1990). Exposed channels will also radiate heat more rapidly at night. In addition, increased sediment inputs that results in aggradation will result in a wider and shallower channel that gains and losses heat more rapidly. Therefore, reduction of riparian vegetation and aggradation of a channel act synergistically to cause greater daily and seasonal fluctuations in water temperatures.

While the increases in summer water temperatures may be large after removal of riparian vegetation, the changes in winter water temperatures are usually less dramatic. However, slight changes in temperature may have a large impact on salmonids when water temperatures tend to be low. Studies on a coastal watershed in British Columbia revealed that the number, size and migration timing of coho smolts were most affected by small increases in late-winter and early-spring water temperatures (Hartman et al. 1987). Generally, the removal of riparian vegetation resulted in increases of winter water temperatures in low elevation coastal watersheds due to increases of solar energy (Beschta et al. 1987). Conversely, in northern latitudes and at higher elevations decreases in winter water temperatures may occur due to the loss of insulation from riparian vegetation, leading to an increase in radiative cooling from the watershed.

Changes in water temperatures from the removal of riparian vegetation may benefit or negatively impact salmonid populations. Among the potential benefits is an increase in primary and secondary production that would increase the amount of available food. Studies have reported that after logging, increases in filamentous algae promoted the abundance of invertebrate grazers such as baetid mayflies, grazing caddisflies and midges that were more likely to contribute to the drift and be available as food for salmonids (Hawkins et al. 1982). Increased water temperatures during winter months are usually less dramatic than summer increases; however these slight increases may have a great effect on salmonids. Studies conducted on Carnation Creek in British Columbia revealed that slight increases in winter water temperatures resulted in accelerated development of coho embryos, thus an earlier emergence of juveniles (Hartman et al. 1987; Holtby 1988). The earlier emergence resulted in a longer growing season for the juvenile coho salmon, but also increased their risk to downstream displacement during late-winter storms. The increased growth of juvenile coho resulted in higher over-wintering survival rates of 1+ fish. However, in Carnation Creek the out-migration of coho smolts was highly correlated with spring temperatures, thus the slightly elevated temperatures resulted in an earlier out-migration (Hartman et al. 1987). These early migrants probably reached the estuarine environment when conditions were not favorable for smolt survival. Additional studies predicted that the marine survival of 1+ coho smolts declined from 14.3% to 10.7% and the marine survival of 2+ coho smolts declined from 15.6% to 10.7% (Bilton et al. 1982). Apparently earlier migration into unfavorable marine conditions negated any survival advantage of increased smolt growth due to increases in water temperature (Hartman et al. 1987).

Increased water temperatures can also have negative impacts on the salmonids (Beschta et al. 1987) as well as the amphibians. Potential impacts to salmonids from increased stream temperatures include (Hallock et al. 1970; Hughes and Davis 1986; Reeves et al. 1987; Spence et. al. 1996):

- reduction in growth efficiency,
- increased disease susceptibility,
- changes in age of smotification.
- loss of rearing habitat, and
- shifts in the competitive advantage of salmonids over non-salmonid species.

There is a potential secondary negative impact of increased water temperatures that is related to levels of dissolved oxygen in the water. During summer months, low flows and increased water temperatures accelerate respiration and reduce the solubility of oxygen. The reduction of available oxygen may reduce growth rates of individual fish and may limit the production capability of an entire watershed.

Although the specific mechanisms are not known, many of the same physiological or ecological factors associated with elevated water temperatures presumably exist for the amphibian species, which have temperature thresholds below those of the fish Covered Species.

Although elevated water temperatures can be a relatively localized phenomenon, this factor generally functions in a cumulative manner throughout a sub-basin or watershed. The impact of elevated water temperature also tends to be cumulative on a temporal scale, such that short-term increases are less likely to be harmful compared to more chronic increases in water temperature. The potential harm or death associated with this factor would primarily influence the juvenile salmonids and larval amphibians during summer and early fall. Take of Covered Species could occur as the result of temperature increases causing the impairment of essential functions and injury or mortality. The potential impacts of such taking include potential reductions in the local or regional populations of the Covered Species and could affect a possible need to list currently unlisted Covered Species under the ESA in the future.

E.3.6 Altered Nutrient Inputs

Unlike lentic systems and the mainstem of many rivers in which runoff from agricultural, suburban, industrial and other areas lead to eutrophication, the portion of lotic systems throughout the Pacific Northwest and Northern California in which salmonids spawn and rear are thought to be naturally oligotrophic due to low levels of nitrogen (Allan 1995; Triska et al. 1983). However, additions of nitrogen in these systems will only result in limited increases in primary productivity, because most of these streams, especially heavily shaded lower order channels, are also limited by light (Triska et al. 1983). While autochthonous inputs (derived from within the aquatic system through photosythesis) are important in higher order channels, much of the energy and nutrients in lower order channels (where many salmonids rear) comes from allochthonous inputs (derived from outside the aquatic system typically through detrital inputs). One of the most important sources of detrital inputs in streams throughout the Northwest comes from red alder, because it is readily available to the aquatic invertebrate community and its leaves are high in nitrogen (Murphy and Meehan 1991; pers. comm. K. Cummins, Humboldt State University). The fact that red alder fixes atmospheric nitrogen also has important implications for increasing the total available nitrogen in these potentially oligotrophic lotic systems. In contrast to red alder leaves that can be 50% decomposed in less than 2 months, Douglas-fir needles may take over 9 months to reach the same level of decay and have far less nitrogen. Woody debris, even twigs and small branches, has limited nutritional value to streams because it decays so slowly and is very low in nitrogen (Murphy and Meehan 1991). Another potentially important source of nutrients to streams comes from annual spawning runs of anadromous salmonids. Reduced ocean-derived nutrients to stream and riparian ecosystems due to declines in salmon returns in many regions have received considerable attention in recent years (AFS: Nutrient Conference 2001). This has lead to numerous studies looking at the potential benefits of artificially increasing the productivity ("jump-starting") of these systems through the addition of salmon carcasses or other sources of nutrients.

Reduction of riparian vegetation due to timber harvest is likely to increase productivity of streams in several ways. Increased incident solar radiation would likely increase periphyton production (unless it is limited by nitrogen), which may increase the abundance of invertebrates and fish due to an enhanced quality of detritus. The mechanism of this increase is tied to the algae, a higher quality food than leaf or needle litter, which increases the abundance of invertebrate collectors, which in turn, can increase the abundance of predators such as juvenile salmonids (Murphy and Meehan 1991). In addition, timber harvest in riparian areas may reduce the number of conifers and increase deciduous vegetation such as red alder. Therefore, with increased input of nutritionally rich leaf detritus compared to conifer needles, productivity of the stream may increase. Of course, the salmonid response would only be realized if the alteration of the riparian vegetation did not also lead to adversely high water temperatures. An increase in stream productivity may also not ultimately result in increased production of salmonids, because it will primarily benefit summer rearing populations when the "bottleneck" (i.e. limiting factor) for many salmonid streams is winter rearing habitat (Murphy and Meehan 1991).

Larval tailed frogs feed exclusively on diatoms that grow on the surface of the stream's substrate (Metter 1964). Growth of the diatoms is influenced by factors such as sunlight, water temperature and nutrients, but there have been no studies to determine if diatomaceous growth is ever limiting for larval tailed frogs. As a result, it is not possible to speculate on how altered nutrients may influence this life history stage of tailed frogs. The adult frogs presumably feed in the riparian zone, but there is little known of their foraging ecology and it would not be possible to speculate on how altered nutrients in the stream might influence the adults. Larval and adult southern torrent salamanders feed primarily on small aquatic invertebrates whose numbers would be influenced by detrital inputs. However, it is not known if food is very limiting for this species such that changes in aquatic invertebrates would influence survival or growth of individual salamanders.

The impacts of altered nutrient inputs would most likely be subtle and difficult to predict. The greatest potential impact would be to juvenile salmonid populations that need to reach some threshold in size before smoltification and out-migration can occur. Decreases in nutrient inputs would not likely result in direct harm, but they may reduce survival during the freshwater rearing period. In addition, ocean survival would likely be decreased if smolts out-migrate at smaller sizes.

E.4 LWD RECRUITMENT AND DISTRIBUTION

Historically, the mainstems of watersheds were utilized to transport logs downstream to processing mills. Thus, extensive clearing of debris jams occurred on most coastal watersheds (Sedell and Froggart 1984). Splash damming was another management technique to transport logs downstream that tended to dislodge established LWD from stream channels. These channel clearing activities directly removed salmonid habitat from watersheds and also reduced the probability of additional LWD retention within the channel.

Inchannel salvage logging and the clearing of LWD from streams in the Pacific Northwest began shortly after the 1964 Flood. Much of this activity was sponsored by the federal government as a measure to protect bridges and to reduce cases of property liability in court (Maser and Sedell 1994). Removal of LWD from stream channels also occurred during the 1970s and 1980s when state and federal agencies spent over six million dollars annually in efforts to remove debris jams and improve fish habitat (Maser and Sedell 1994). Many of the large debris jams were probably barriers to fish migration and required modification. However, these stream clearing programs often went too far and now fisheries managers have spent the past 15 years reintroducing LWD to streams along the Pacific Northwest. Currently, some fisheries biologists consider the placement of LWD restoration structures in streams as an interim, short-term measure until large conifers are reestablished in riparian zones to provide a source of LWD (House et al. 1989).

Decades of timber harvesting in the riparian zone has altered the species composition and age classes of trees along stream channels. The removal of valuable conifer species has led to the predominance of early successional species such as alders and willows. Short-rotation harvesting has decreased the numbers of large trees available as potential LWD. Woody debris from second-growth forests has a shorter residence time in stream channels than debris from uncut watersheds (Grette 1985). Managed riparian zones of predominately red alder may have a greater input rate of wood to the stream channel than conifers in an uncut riparian zone, but the reduced longevity of alder debris results in reduced cover and fewer pools than in uncut watersheds (Grette 1985).

In-channel LWD is recognized as a vital component of salmonid habitat, and to a lesser extent, but still important to the amphibian Covered Species. The physical processes associated with LWD include sediment sorting and storage, retention of organic debris, and modification of water quality (Bisson et al. 1987). The biological functions associated with LWD structures include important rearing habitats, protective cover from predators and elevated stream flow, retention of gravels for salmonid redds, and regulation of organic material for the instream community of aquatic invertebrates (Murphy et al. 1986; Bisson et al. 1987). Decreased supply of LWD can result in (Hicks et. al. 1991 as cited by Spence et al. 1996):

- reduction of cover,
- loss of pool habitats,
- loss of high velocity refugia,
- reduction of gravel storage, and
- loss of hydraulic complexity.

These changes in salmonid habitat quality can lead to increased predator vulnerability, reduction of winter survival, reduction in carrying capacity, lower spawning habitat availability, reduction in food productivity and loss of species diversity.

In headwater streams, LWD is also known to dissipate hydraulic energy, store and sort sediment, and create habitat complexity (O'Connor and Harr 1994). Creating and providing cover for pools, a primary function of LWD for salmonids, may be of limited benefit to the headwater amphibian Covered Species since torrent salamanders and larval tailed frogs prefer riffle habitats (Diller and Wallace 1996, 1999; Welsh and Lind 1996). The primary benefit of LWD to the amphibians is the creation of suitable riffle habitat through the storing and sorting of sediment. In addition, LWD that is perched a short distance above the streambed will often form a dam composed of coarse sediment and small woody debris through which water percolates. In streams that are otherwise too embedded with fine sediments to be used by torrent salamanders, this appears to form the only habitat that still supports the species (Diller, pers. comm.). There is circumstantial evidence that these same sites are utilized for egg laying by tailed frogs, but searching such sites is too destructive to adequately investigate the phenomenon (Diller, pers. comm.).

The decline of recruitment of potential LWD from riparian zones can be expected to reduce LWD recruitment to streams for decades following timber harvest of riparian areas. High in the watershed, the potential impacts would be primarily localized, but in larger streams lower in the watershed, LWD can be transported during higher flow events and the impacts may be cumulative. A decline in pool density, pool depth, instream cover, gravel retention, and sediment sorting are likely to result if LWD recruitment is reduced. These habitat changes may reduce the growth, survival, and total production of salmonids as well as the amphibian species (Steele and Stacy 1994; Murphy et al. 1986). Given that LWD is likely critical to provide habitat and cover for juvenile salmonids in both summer and winter, survival rates of these life history stages may be limited by the amount of LWD in some streams. Such potential impacts that reduce survival rates of key life history stages of the Covered Species may result in local population declines. Such declines could negatively affect the regional populations of the Covered Species.

E.5 CUMULATIVE WATERSHED EFFECTS

In general, cumulative watershed effects (CWEs) can be categorized as incremental changes that induce changes in watershed processes that alone are not overwhelming, yet if combined, the impacts on stream channels and habitat for aquatic species are detrimental. This is largely a theoretical concept without empirical data, because the identification of CWEs is difficult due to both the technical complexities of designing statistically valid field studies, and because few research efforts have been sustained for extended time periods. Recently, efforts have been made to examine the cumulative effects of timber harvesting on salmonid bearing watersheds. For example, the Carnation Creek watershed study in British Columbia was an 18 year project to examine CWEs due to timber harvesting (Hartman et al. 1987). Poulin (1984) created synoptic study designs that could examine CWEs by simultaneously studying numerous watersheds at various stages of timber management. Technological advances such as time series analysis of aerial photography, vegetative dating techniques, sediment analysis, and computer modeling systems also provide information about CWEs (Chamberlain et al. 1991). Likewise, extensive literature searches and reviews of historical media and agency files also have assisted in defining past management treatments and resultant effects on Pacific Northwest watersheds (Sedell and Froggart 1984). Historic documents also were used by Sedell et al. (1991) to investigate the transportation and storage of logs in watersheds and the impacts to channel formations and salmonid habitat.

Records of natural changes also are essential to assessing CWEs. Natural change usually occurs within limits that are “normal” for a particular watershed and the biological communities are usually adapted to those changes on either an individual or population level. However, naturally occurring events can inflict catastrophic change on watersheds. Wild fire, drought, floods, volcanic eruptions and earthquakes can all drastically alter physical and biological watershed processes. These events may force salmonid populations to utilize other habitats and undergo reductions in population numbers until the aquatic habitat recovers. The key to the recovery of biological communities is that catastrophic events occur sporadically and the events may also only impact a certain portion of a watershed. Management practices may or may not allow aquatic ecosystems the time to recover before additional impacts are imposed.

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Appendix F. Sediment Delivery Studies and Modeling Efforts

CONTENTS

INTRODUCTION

This appendix presents a description of the sediment delivery studies and the sediment modeling efforts conducted by Green Diamond Resource Company (Green Diamond). These projects were undertaken to estimate future long-term sediment delivery volumes to watercourses from roads and landslides within the Plan Area. The empirically-based model was designed to comparatively evaluate average long-term sediment delivery from roads and landslides under different management scenarios. The structure of the model enables Green Diamond to examine a wide range of management scenarios to identify the most efficient and effective prescriptions that will sufficiently reduce future management-related sediment delivery to meet the needs of aquatic resources of concern.

Model Data Base

Green Diamond conducted two extensive sediment delivery studies. One study involved the compilation of landslide inventories to evaluate landslide-related sediment delivery. Average long-term sediment delivery volumes from shallow and deep-seated landslides were evaluated for three pilot watersheds covering roughly 10% of the Plan Area: Salmon Creek, Little River, and Hunter Creek. Delivery rates were based on standard interpretations of aerial photographs with a limited amount of field verification. Sediment delivery from deep-seated landslides was also estimated in the Upper Mad River pilot watershed based on published data. The impact of harvesting on landslides and landslide-related sediment delivery was evaluated from the landslide inventory data collected in the pilot watersheds, from published reports, and complemented by professional judgment where data were lacking. A summary of the results of the landslide inventory and associated analysis is included as Appendix F1.

The second data collection effort was a field-based road inventory of 518 miles of road in five pilot watersheds to evaluate future sediment sources and sediment delivery related to the road network. The road-related sediment source inventories employed standard road inventory protocols developed by Pacific Watershed Associates, which have been used on forest and ranch lands throughout the north coast. The inventories were designed to quantify potential future sediment delivery from road-related landslides, watercourse crossing failures, and “other” sites (such as problems with ditch relief culverts and related gullies) associated with Green Diamond’s road network. As part of Green Diamond’s modeling effort described in this appendix, the road inventory data were summarized and applied to the Green Diamond ownership within the 11 HPAs to develop potential road-related sediment delivery estimates. These data were also instrumental in developing site-specific erosion prevention measures as well as general road-related erosion prevention measures that were incorporated into the Plan. A summary of the road inventory data is included as Appendix F2.

Green Diamond used the road-related sediment source data and landslide-related sediment data to parameterize a simple sediment delivery model for the Plan Area. This model was subjected to Monte Carlo simulation analyses to evaluate changes in forecast variables given ranges of uncertainty in the model’s parameters. The use of empirically-based sediment inventory methods and Monte Carlo simulation enabled Green Diamond to comparatively analyze average long-term sediment delivery under a variety of management scenarios and conservation measures. It was through this comparative analysis that Green Diamond developed the accelerated road-related erosion prevention strategy (see Section 6.3.3) and appropriate slope stability conservation measures (see Section 6.3.2) that are expected to meet the needs of the aquatic resources of concern. A description of the Plan Area model and the Monte Carlo simulation results are included as Appendix F3.

Limitations of the Model

The model quantified only those sediment sources and processes that were considered to be among the most prolific sediment contributors and that may be affected by management prescriptions. The conservation measures developed from the model focused on those prescriptions that were expected to have the greatest benefit to the covered species, provide the highest confidence of success, and are logistically and economically feasible. Conversely, prescriptions that were expected to result in only a marginal benefit, provide low confidence of success, and that are logistically or economically infeasible were avoided.

The model is best suited for comparative analysis of road and landslide related sediment delivery, and it is not intended to be a comprehensive sediment budget. Although the model does not address all possible forms of management-related sediment delivery, such as legacy skid trail erosion, in-unit hillslope erosion, and stream bank erosion, conservation measures and BMPs have been developed, following the advice of experts both within the government and within the private sector, to address those potential sediment inputs.

The model does not differentiate between fine- and coarse-grained sediment. While the effectiveness monitoring and the adaptive nature of the conservation measures will be based only on sediment delivery and potential sediment delivery volume, the conservation measures as a whole are expected to have a significant effect on fine-grained sediment contributions. This is particularly true of the road-related and harvest-related-ground-disturbance conservation measures described in the Plan.

Finally, the sediment model does not address cumulative watershed effects (CWEs). It is not site specific, and it does not integrate past, current, and reasonably foreseeable projects. Instead, the sediment model is spatially-averaged over the Green Diamond ownership within the 11 HPAs and time-averaged over the next 50 years. This does not reflect actual sediment delivery processes, which are prone to occur in more of an episodic nature and vary locally, depending mostly on climatic conditions. However, the significance of this limitation is reduced by the adaptive management mechanisms in the Plan that are expected to provide appropriate elasticity for the conservation measures within individual HPAs to meet the needs of the aquatic resources of concern.

Although Green Diamond's modeling approach may overestimate sediment delivery in some places and underestimate it in other places, it is thought to be reasonably accurate overall. Therefore, Green Diamond believes the model is adequate for evaluating the most efficient and effective prescriptions to limit management-related sediment delivery in order to meet the needs of the species of concern, keeping in mind that some of the initial prescriptions are subject to adaptive management.

Appendix F1. Assessment of Long-term Landslide Sediment Delivery under Existing and Proposed Plan Conditions

CONTENTS

Tables

F1.1 INTRODUCTION

The following chapter outlines the methodology, assumptions, limitations, and results of a modeling exercise designed to estimate approximate long-term landslide delivery rates from the road and skid trail network and from hillslopes to watercourses in several pilot watersheds within the Plan Area. The modeling is also intended to estimate long-term sediment delivery under various silviculture options.

The purpose of this exercise was to evaluate the potential impacts of forest practices on landslide-related sediment delivery and to assist in evaluating the most effective and efficient slope stability measures. Such evaluations are the focus of Appendix F3, which takes the models and results developed in this chapter and applies them to the Plan Area to develop property-wide sediment delivery estimates.

A general discussion of landslide types and processes is summarized in Appendix B. A general discussion of the potential impact management activities can have on these processes is summarized in Section 5.

Estimates of landslide delivery rates are based primarily on landslide data collected from the historical set of aerial photographs. Historical rates of landslide delivery from grading activities (i.e., roads, skid trails, landings, etc.) and from hillslopes were estimated separately. A simple model was developed to estimate management-related landslide delivery rates in harvest areas that are attributable to silvicultural treatment. Landslide rates for the pilot watersheds were applied to the remainder of the Plan Area based on professional experience.

A mechanistic modeling approach was considered. However, due to the inherent variability in many of the input parameters that can affect slope stability, the difficulty in obtaining the precise data required for any mechanistic model, temporal and spatial variability of the parameters, and limitations in the slope stability models, Green Diamond does not believe that accurate results could be obtained from such a model.

The information provided in this appendix is specific to sediment production and delivery from shallow and deep-seated landslides associated with roads and silvicultural treatment. Sediment production and delivery from other processes, such as surface erosion, channel bank erosion, or erosion of watercourse crossings are not addressed in this appendix, although the potential for such sediment causing effects is addressed elsewhere in the Plan.

F1.1.1 Approach

Total sediment delivery from landslides is the sum of natural landslide sediment and management induced landslide-related sediment. Management induced landslide related sediment includes sediment derived from cut slopes and fill slopes of roads (including skid trails and landings) and from harvest units (as influenced by silvicultural treatment). This relationship is illustrated by the following equation:

Equation 1: $\mathbf{SED_{tot} = SED_{background} + (SED_{road} + SED_{harvest})}$

Landslide delivery volumes were estimated based on empirical evidence that related management activities to increased erosion rates. These models are based largely on the results of preliminary mass wasting assessments (MWAs) conducted on several pilot watersheds within Green Diamond property. The impact of harvesting on sediment delivery was estimated from landslide inventory data collected throughout north coastal California and Oregon published scientific literature, and complemented by professional judgment where data were lacking.

Average long-term sediment delivery volumes from shallow and deep-seated landslides were estimated for both current management practices and those under the proposed Plan measures for three pilot watersheds: Salmon Creek, Little River, and Hunter Creek. Sediment delivery from deep-seated landslides was also estimated in the Upper Mad River pilot watershed.

F1.1.2 Limitations

It should be recognized that estimating landslide rates across all of Green Diamond ownership property with its diverse terrain and types of landsliding is a complicated process. Sediment delivery rates are temporal and spatially variable. The sediment delivery volumes presented here are long-term averages using empirically determined associations between sediment delivery and land management. The model is based on best available data.

Short-term sediment delivery rates may be higher or lower than the average presented here due to land-use and metrological events. Sediment delivery will be higher than average following major events and lower during relatively dry periods. Moreover, the post harvest impact immediately after harvesting is expected to be higher than average, diminishing as vegetation becomes reestablished. Sediment delivery is also not spatially characterized by the models presented herein. Local differences in geology, terrain, land use, and climate may result in locally different rates of sediment delivery to watercourses.

Ranges in model parameters have been provided in an attempt to evaluate ranges in sediment delivery due to uncertainties in estimates or measurements of the parameters. These ranges were useful in the Monte Carlo simulation exercise reported in Appendix F3.

The sediment delivery volumes presented here are intended as a means for evaluating the relative effects of different management scenarios on landslide sediment delivery to develop a physically based approach to prescription development. The results from this modeling effort are considered approximate and are not intended as detailed sediment budget of each watershed.

F1.2 MODEL DESCRIPTION

The following sections provide a detailed description of the data and analytical methods used to determine sediment delivery volumes for both shallow and deep-seated landslides. The impact of harvesting on shallow landslide processes was considered separately from the impact of harvesting on deep-seated landslides because of the difference in landslide processes and the availability and quality of existing data. Each of the following sections also includes a description of the limitations and assumptions used in the development of the model, and the limitations that should be understood during the application of the model output.

F1.2.1 Shallow Landslides

Shallow landslides are characterized by debris slides, debris flows, channel bank failures and small to large hillslope failures. These landslides are typically rainfall-activated, relatively fast-moving, shallow (less than 10 feet deep), and generally incorporate only the overlying surficial mantle of soil, colluvium, and weathered bedrock (see Appendix B).

F1.2.1.1 Methods

Average long-term sediment delivery from shallow landslides was calculated from preliminary landslide sediment delivery data collected in the MWAs of five pilot watersheds: Salmon Creek, Ryan Creek, Little River, Hunter Creek, and Tectah Creek. Sediment delivery from road-related landslides was estimated directly from the aerial photograph-based landslide inventory. Sediment delivery from hillslope landslides was estimated by applying a simple model that relates the relative impact of different harvest scenarios to landslide rates. The landslide inventories for Ryan Creek and Tectah Creek are incomplete at present; therefore, only the results from shallow, road-related failures in these areas were used as a supplement to the analysis.

F1.2.1.2 Total Sediment Delivery

Historical rates of sediment delivery from shallow landslide processes operating in each of the five pilot watersheds were estimated from an analysis of the historical set of aerial photographs (Table F1-1). Landslides were mapped from the historical set of aerial photographs and, with the exception of Ryan Creek and Tectah Creek, their location entered into the geographic information system (GIS) database for further analysis. The age of the slide was reported as the year of the photograph the slide was first observed. The input of landslide data from Ryan Creek and Tectah Creek into the GIS is pending.

Table F1-1. Landslide inventory photo record.

Pilot Watershed	Acreage	Photo Years
Salmon Creek[a]	7,889	1997, 1991, 1978, 1958, 1954
Ryan Creek	7,590	1997, 1990, 1984, 1978, 1966
Little River	28,755	1997, 1987, 1978, 1966, 1948
Hunter Creek	10,126	1997, 1984, 1972, 1958
Tectah Creek[b]	12,675	1997

Notes
a: 1958 photos used where 1954 photos were unavailable
b: Landslide inventory for earlier years incomplete at present

Pertinent data associated with each landslide were recorded into a database for further analysis. This included landslide type, estimated size (ft^2), estimated depth (ft), sediment delivery ratio (%), slope form (convergent, divergent, planar) and location (headwall swale, inner gorge, midslope), any association with graded areas (road, skid trail, landing, railroad tracks, etc.), and level of harvest (clearcut, partial cut, forested, grassland).

Limited field verification of mapped landslides was undertaken in all pilot MWA areas except Ryan Creek. Additional fieldwork in all watersheds is pending. Sediment delivery from each of the pilot watersheds is summarized in Tables F1-2 and F1-3.

In Tables F1-2 and F1-3, the road category is the sum of landslide sediment derived from all graded areas including roads, skid trails, landings, railroad tracks, etc. It is assumed that any landslide that initiates at, or adjacent to, a graded area is a result of that grading. The Non-Road category is the sum of all landslide-derived sediment that is not associated with grading. The % Historical Road category is the percentage of the total sediment for the period of the air photo record that is road-related (including all graded areas), whereas the % 1997 Road category is the percentage of 1997 sediment that is road-related. The % Historical Road can be higher or lower than the % 1997 Road depending on road construction history. The % 1997 Road is considered a better estimate of the current relative impact of roads on shallow landslide sediment delivery.

Table F1-2. Shallow landslide sediment delivery volumes.

Watershed	Acres	Years of Record	Landslide Delivery (cy)			% Historical Road[1]	% 1997 Road
			Total	Road[1]	Non-Road		
Salmon Creek	7,889	58	156,732	41,650	115,082	26%	17%
Ryan Creek	7,590	46	27,903	9,240	18,663	33%	56%
Little River	28,755	64	139,457	28,491	110,966	20%	40%
Hunter Creek	10,126	54	494,523	306,751	187,772	62%	39%
Tectah Creek	12,675	n/a	104,121	550	84,982	n/a	18%

1 Road includes all graded areas including roads, landings, skid trails, railroad tracks and other graded areas.

Table F1-3. Long-term shallow landslide delivery rates.

Watershed	Cy/ac/yr			$T/mi^2/yr$[b]		
	Total	Road[c]	Non-Road	Total	Road[c]	Non-Road
Salmon Creek	0.34	0.09	0.25	295	80	217
Ryan Creek	0.08	0.03	0.05	69	22	46
Little River	0.08	0.02	0.06	65	13	52
Hunter Creek	0.90	0.57	0.34	781	485	297
Techtah Creek[a]	--	--	--	--	--	--

Notes
a: Pre-1997 landslide data unavailable at present
b: Assumes a unit weight of soil of 100 pcf.
c: Road includes all graded areas including roads, landings, skid trails, railroad tracks and other graded areas.

F1.2.1.2.1 *Confidence of Landslide Volume Estimates*

The accuracy of identifying and characterizing landslides in aerial photographs is variable and depends, in part, on the size of the slide, thickness of the vegetative cover, and timing and quality of the photographs. Large landslides, or landslides mapped in recently harvested areas or through thin canopy, are identified with relatively high accuracy. However, small streamside failures, which are often numerous, are difficult to identify because of thick riparian canopy. Therefore, aerial photo analysis will only allow for a partial identification of the total number of landslides in the Plan Area. As a result, the number of slides inventoried for use in landslide delivery should be considered a minimum representation of the actual number of slides that are present in the area. To illustrate this point, the Oregon Department of Forestry's (ODF) evaluation of storm impacts and landslides for 1996 (Robison et al. 1999) revealed that air photo inventories may underestimate sediment production from landslides by as much as 50 percent. The error is greatest in mature forests with thick canopy and less apparent in recently harvested areas.

Field verification of air photo measurements was conducted in Hunter Creek and to a lesser extent in Salmon Creek, Little River, and Tectah Creek. Where field verification is complete, air photo estimates of sediment production are generally within 30 percent of field measurements. This relatively high level of accuracy may be partly explained by data indicating that small slides, potentially undetected in the aerial photograph record, do not deliver large volumes of sediment to streams and are not a large component of the total sediment budget. This leads to the conclusion that the majority of sediment is probably delivered by large slides that have a high likelihood of detection in the air photo record. It should be noted, however, that Green Diamond has accounted for uncertainty in landslide sediment delivery rates in its modeling efforts. Appendix F3 contains a description of four assumption variables that address such uncertainties: Delivery From Road-Related Landslides, Little River Sediment Multiplier, Hunter Creek Sediment Multiplier, and Salmon Creek Sediment Multiplier.

Table F1-4 summarizes the expected range of shallow landslide sediment delivery volumes relative to measured aerial photograph volumes. The range is based on limited field reconnaissance and verification of slides in Salmon Creek, Little River and Hunter Creek, and professional judgment. The range in landslide delivery volumes incorporates uncertainties in slide identification and volume estimates. The higher range in Salmon Creek and Little River compared to Hunter Creek is a result of the expected higher incidence of small stream bank failures that were apparent during field reconnaissance of the watershed but may not be apparent in the air photos.

Table F1-4. Assumed range in landslide delivery volumes relative to air photo estimates.

Watershed	***Lower Bound***	***Most Likely***	***Upper Bound***
Salmon Creek	80%	100%	150%
Ryan Creek	80%	100%	150%
Little River	80%	100%	150%
Hunter Creek	70%	100%	130%
Tectah Creek	--	--	--

F1.2.1.3 Road -Related Landslide Sediment

Landslide delivery volumes from road-related landslides were calculated directly from the air photo inventory. Failures were identified as road, landing, skid trail or "other" related landslides. "Other" related landslides included failures originating from railroad fill and building pads. It was assumed that any landslide on or adjacent to one of these road features occurred as a result of the construction of that feature. Cutbank failures were not inventoried unless they overtopped the road and delivered sediment directly to a watercourse.

The classification of failures related to grading activities is relatively straightforward in harvested areas or areas with thin canopy. Some small roads may have been classified as skid trails; likewise, some large skid trails may have been classified as roads. Identification of roads or skid trails in areas of thick canopy is speculative at times and therefore it is possible that some failures in these areas may have been misclassified. Landslide delivery volumes from roads are summarized in Tables F1-2 and F1-3.

F1.2.1.4 Harvest-Related Landslide Sediment

Harvesting can potentially impact landslide rates through reduced root reinforcement and changes in the hydrologic regime (See Section 5). Determining the contribution of sediment from harvest areas is a much more difficult endeavor than estimating sediment contribution from roads. Unlike roads, the simple existence of a slide within in a harvest unit is insufficient to make a causal link between that particular slide and the harvesting activity. This is because natural landslides may occur within harvest units therefore determining the casual mechanism of failure of any given in unit slide often requires in-depth field review. Although many studies have addressed the impact of roads on sediment production, there are few comparable studies in the region that have quantitatively evaluated the impact of harvesting (i.e., tree removal alone) on sediment production and delivery rates, and those studies that have been completed give widely varying results.

With respect to sediment delivery, the relative impact of timber harvesting on landsliding is probably best evaluated using an empirical approach that compares landslide delivery rates from harvested areas to forested ground. Unfortunately, few studies of this kind have been conducted in northern California.

The difficulty in evaluating the impact of harvesting is further compounded by the fact that different harvest methods are expected to have different implications for slope stability. For example, a selection harvest is not expected to have the same impact on slope stability as clearcutting. Similar problems exist with differences in terrain and geology. For example, the reduction of root strength in cohesionless soils is expected to have a greater impact on shallow landsliding than harvests in soils with relatively high cohesion. Further, it is possible that some harvests may have impacts on slope stability offsite. For example, it has been hypothesized that in some areas, extensive upslope harvesting may have an impact on downslope areas through alterations in the hillslope hydrology (see Section 5).

In this study, the harvest contribution of non-road-related, shallow, landslide-derived sediment was estimated using a relatively simple empirical model that applies a regional average ratio between harvest-related sediment (timber removal alone) and natural "background" sediment [herein referred to as *"harvest ratio"* (HR)] to the non-road-related component of shallow landslide sediment measured in each pilot watershed (see Equation 3).

The average clearcut HR was estimated from published and unpublished studies, including total maximum daily load (TMDL) studies, Pacific Lumber Company (PALCO) sediment source assessments, the ODF study, and from preliminary results from Green Diamond's Hunter Creek pilot MWA (these studies will be discussed in detail later in this appendix). HRs for other silvicultural prescriptions are not reported. Therefore, adjustments to the clearcut HR were required to account for differences in silvicultural prescriptions and expected differences in mass wasting rates as a result of inherent sensitivity of the hillside as delineated by the mass wasting prescription zones (MWPZs).

Green Diamond has assumed that sediment delivery from harvest areas can be reasonably estimated based on the following equation:

Equation 2: $\mathbf{SED_{harv} = SED_{nonroad} / (HR_{clearcut} * N_{partcut\ (y)} * N_{terrain})}$,

where SED_{harv} is the rate of sediment delivery resulting from timber removal alone, $SED_{nonroad}$ is the rate of non-road-related sediment delivery measured from the historical set of aerial photographs, $HR_{clearcut}$ is the clearcut harvest ratio, $N_{partcut\ (y)}$ is a factor to account for different silvicultural techniques (y) other than clearcutting, and $N_{terrain}$ is a factor to account for terrain differences.

The model assumes that the rate of harvesting has remained relatively constant over time. In addition, the model assumes a direct spatial link between harvesting and slope failure. In other words, the analysis assumes that vegetation retention has only a local effect on slope stability. Any offsite impact of harvesting (such as changes in downslope hillslope hydrology from upslope harvesting, or increased stream flow from upstream harvesting) is assumed to be negligible and was not modeled.

While Green Diamond recognizes that upslope harvesting may have an impact on downslope harvest areas, there is little data at present to model this process. Nonetheless, Green Diamond believes the model provides a reasonable and simple method to evaluate the relative impact of different silvicultural methods. As more data are collected and the understanding of the impact of harvesting increases, the model can be revised.

F1.2.1.5 Harvest Ratio

HR is defined as the ratio between the average long-term rate of sediment delivery (cy/acre/yr) derived from harvest blocks (includes harvest-derived sediment and background sediment) compared to uncut or advanced second growth forested ground (background sediment):

Equation 3: $\mathbf{HR(n) = (SED_{harvest}(n) + SED_{background})/SED_{background}}$,

where n is the type of silviculture applied, $\mathbf{SED_{background}}$ is the measured volume of sediment generated from undisturbed or advanced second growth forests, $\mathbf{(SED_{harvest}(n) + SED_{background})}$ is the measured volume of sediment generated from failures originating in harvest blocks, and $\mathbf{SED_{harvest}}$ is the volume of extra sediment above background that is generated as a result of harvesting. This value cannot be directly measured because it is generally not possible to distinguish between individual natural and harvest-caused landslides within harvest blocks.

The model assumes that the impact of harvesting is uniform and constant across the landscape. It is likely, however, that HRs are quite variable, depending on terrain, geology, hydrology and vegetation type. Moreover, the period during which a slope is most prone to shallow instability is a function of the magnitude of the hydrologic event and the decay time to a critical root cohesion value low enough to allow for landsliding, and the duration of time spent below the critical root strength (SWS 1999; Ziemer and Swanston 1977). With the amount of data available at present, however, it is not possible to tailor the HR to individual watersheds or sub-watersheds.

As a first approximation, a regional long-term average clearcut HR ($HR_{clearcut}$) was estimated based on published and unpublished reports. HRs for other silvicultural strategies are not presented in the literature. Therefore for the purpose of this model, the clearcut HR was then modified to account for other silvicultural prescriptions (e.g., 85 percent overstory retention, selection, hardwood retentions, etc.) based on what data was available, review of deterministic models and professional judgment.

F1.2.1.5.1 *<u>Clearcut Harvest Ratio</u>*

An average clearcut harvest ratio was estimated from a review of published and unpublished landslide inventories, including TMDL studies, the ODF study on the impacts of 1995 and 1996 storms (Robison et al. 1999), PALCO Sediment Source Investigations (PWA 1998a, 1998b, 1999a, 1999b), PALCO Freshwater Creek Watershed Analysis (PALCO 2001a), and Green Diamond's preliminary Mass Wasting Assessment for Hunter Creek. The results of these studies are summarized in Table F1-5. Results from the other pilot watersheds are pending.

Based on the foregoing, the historical average **long-term** increase in sediment delivery from clearcut areas ranges between 1.25 and 4.0 times background (most likely equal to 2.0). The results from Freshwater and Hunter Creek were weighted more heavily than the other studies because these were the most rigorous in evaluating the impact of clearcut harvesting, and because they are more representative of geologic and terrain conditions on Green Diamond lands. In addition, each of these cases includes periods of record in which extensive clearcut harvesting occurred a few years prior to intense triggering storms.

It is important to note the clearcut harvest ratio likely presents a 'worst' case scenario for a long term average given that the ratio is based on data originating from areas recently subjected to very intensive land use dominated by the effects of recent large storm events (i.e., Hunter Creek and Freshwater Creek). Recent work by Schmidt et al. (in press) on root cohesion and susceptibility to shallow landsliding found that 100-year-old industrial forests had lower root strength and inferred higher landslide rates in comparison to natural forests. However, these results should be viewed with caution since the lower root strength in the 100-year-old industrial forests is attributed to forestry practices a century ago that did not include replanting of conifer, therefore allowing the site to regenerate with hardwood. Conceptual modeling by Schmidt et al. (in press) suggests that if the site is replanted with conifer immediately following harvesting root cohesion values can return to pre-harvest levels within 16 years.

It is important to note that the HR used for modeling is intended to be a long-term average over the 50-year period of the harvest. Short-term impacts may be higher or lower depending on the occurrence of triggering hydrologic events and the rate of vegetation regrowth.

F1.2.1.5.2 *Partial Cut Harvest Ratios*

Because partial cutting retains understory vegetation and leaves a substantial live root mass, it has less impact on root strength and slope stability than clearcutting. Further, harvesting in redwood or hardwood forests, which maintain a viable root network and generally sprout vigorously after cutting, should have less impact on slope stability.

Few studies have been conducted that evaluate the impact of different residual stand densities on slope stability and shallow landslides. The ODF study of the effects of the 1995-96 storms revealed that comparatively few landslides originated in partially cut areas (Robison et al., 1999). Similarly, little change in landslide rates was documented in partial cuts in the *Draft Freshwater Creek Watershed Analysis* (PALCO 2001).

When relating landslide occurrence to changes in vegetation crown cover, studies in Idaho revealed that landslide frequency increases only slightly as overstory crown cover is reduced from 100 percent to 11 percent. However, a notable increase in landslides occurs when crown cover is reduced below 11 percent (Megahan et al. 1978). The Idaho study may not be applicable to the north coast area because of differences in geology and vegetation; nonetheless, it illustrates that in some areas, even a rudimentary root network can increase soil stability on a hillside. The relatively low impact that partial cuts have on landslide occurrence is also supported by the preliminary data from the Green Diamond MWA pilot watersheds.

Modeling studies of shallow landslides and the effects of different silvicultural systems on root strength suggest that partial cutting results in substantially greater residual root strength and a substantially lower probability of slope failure compared to a clearcut scenario (Krogstad 1995; Schmidt et al. in review; Sidle 1991, 1992; Ziemer 1981a, b). For example, Sidle (1992) reports “A 75 percent partial cut reduced the maximum probability of failure more than five times compared with clearcut simulation." Ziemer (1981a) suggests that under shelterwood removal silviculture, where 70 percent of the original stand is harvested followed by removal of the remaining trees 10 years later, root reinforcement dropped to about 70 percent of its uncut value at 2 to 3 years post harvest, then rose to about 10 percent above the uncut value after about 7 years after harvest as the residual trees quickly expand. About 15 years after the residual trees were harvested, root reinforcement again dropped to about 50 percent of the uncut value. Under a light selection harvest where 20 percent of the trees were cut every 10 years, root strength would decrease by about 3 percent 2 years after harvest, then increase to about 7 percent above the uncut strength as a result of rapid expansion of the roots of the remaining trees. It is important to recognize that the foregoing modeling results are for maximum short-term impact. Long-term impact over complete rotations (i.e., 50 years) would be substantially less.

Table F1-5. Summary of clearcut harvest ratios.

Study	Clearcut Harvest Ratio ($HR_{clearcut}$)
Early Oregon and Washington Studies (summarized in Sidle et al. 1985)	1.9 – 8.7[a]
Oregon Department of Forestry (ODF): 1996 Storm Impacts in Oregon	0.3 – 5.1[b]
Amaranthus et al. (1985)	6.8[c]
North coast TMDL Studies	N/A[d]
PALCO: Bear Creek Sediment Source Assessment (source data from PWA 1998b)	11.5[e]
PALCO: Jordan Creek Sediment Source Assessment (source data from PWA 1999b)	3.0[f]
PALCO: Elk River Sediment Source Assessment (source data from PWA 1999a)	2.3[g]
PALCO: Draft Freshwater Watershed Analysis (source data from PALCO 2001 and PWA 1999)	2.3[h]
Green Diamond: Hunter Creek (unpublished)	1.0 – 1.7(max)

Notes

a: Includes older harvest practices. Impact of skid trails may not have been factored out. Uncertain whether landslide rates include delivered sediment volume or mobilized sediment volume.

b: Evaluates short-term impact of a large storm, likely not representative of long-term average. Ratios based on delivered sediment volume.

c: Includes older harvest practices.

d: Landslide rates are not normalized by harvest acreage; it is not possible to compute HR from these data.

e. Very high HR value reflects extraordinarily large debris slides that occurred in 1996/1997 in unusual storms on steep terrain shortly after harvest, and may therefore represent worst case scenario. Not all harvest areas in source data are clearcuts, most areas have some history of tractor harvest, and landslide rates are calculated for a 22-year period (1975-1997). Ratio calculated for delivered landslide volume. See also section 4 below.

f. Value represents the period 1975-1997. Not all harvest areas in source data are clearcuts and most areas have some history of tractor harvest. Ratio calculated for delivered landslide volume. See also section 4 below.

g. Value represents the period 1969-1997 (28-year period of record). Not all harvest areas in source data are clearcuts and most areas have some history of tractor harvest. Ratio calculated for delivered landslide volume. See also section 4 below.

h. Value represents the period 1969-1997 (28-year period of record). Not all harvest areas in source data are clearcuts and most areas have some history of tractor harvest. Ratio calculated for delivered landslide volume. The same ratio (to two significant digits) was computed for the period 1988-1997 in a comparison of landslide rates (not sediment delivery volume) in clearcuts and advanced second growth forest. See also section 4 below.

Modeling studies have also shown that understory vegetation often represents an important component of total root cohesion and that the retention of the understory canopy can substantially reduce the probability of slope failure (Schmidt et al. in review; Krogstad 1995; Sidle 1992). Because shallow landslides might opportunistically exploit gaps in the root network when partial harvesting is employed, uniform spacing of trees to minimize “gaps” that might develop in the root network between trees is important to provide the greatest root strength benefit (Burroughs and Thomas 1977; Schmidt et al. in review).

Based on the foregoing, it is appropriate to make adjustments in the clearcut HR to account for different stand densities and overstory retention resulting from partial harvest silviculture. Although the effect of tree roots is highly variable, it was assumed that on a regional level, the impact of harvesting can be related to overstory retention as a surrogate for the completeness of the root network and total root strength. The basic assumption is the more trees retained, the greater the root reinforcement.

Table F1-6 lists assumed corrections factors to the average long-term clearcut HRs for different levels of overstory retention. Vegetation retention assumes uniform or “square spacing” of conifers. Table F1-7 outlines overstory retention under pre- and post-Plan conditions, and forms the basis for estimating sediment delivery. For simplicity, it was assumed that all slopes within the riparian management zone (RMZ) are greater than the critical slope gradient (i.e., > 60 percent for Salmon Creek, > 65 percent for Little River, and >70 percent for Hunter Creek). Although this would overestimate the acreage of ground within the prescription zone, it is not expected to have a large impact on the estimate of sediment delivery. This is because at least 80 percent of the total volume of sediment delivered from streamside landslides is generated from landslides originating on slopes greater than the critical slope gradient.

Table F1-6. Assumed correction factors for different stand densities: overstory retentions compared to clearcut harvesting on shall landslide sediment delivery.

	Expected multipliers for landslide delivery rates relative to clearcutting		
Stand Density	***Lower***	***Most Likely***	***Upper***
85% to 100% Overstory Retention	100%	100%	100%
70% to 85% Overstory Retention	90%	90%	100%
50% to 70% Overstory Retention	60%	70%	80%
Selection Harvest	50%	60%	70%
Hardwood and Understory Retention	25%	35%	45%
Understory Retention	0%	10%	20%
Clearcut	0%	0%	0%

F1.2.1.6 Adjustments for Slope Position

Adjustments are needed to account for expected differences in the impact of harvesting on different MWPZs. MWPZs are broken down into Steep Streamside Slopes (RMZ and SMZ), Headwall Swales (SHALSTAB areas) and “Other” areas. The impact of harvesting is expected to be different in each of these areas. The impact of harvesting is likely slightly less than average along streamside slopes because some of the failures in this area are attributed to undercutting of the hillside by bank erosion and thus are likely to occur independent of vegetation cover. This is not to say that vegetation has no effect on hillslope stability in these areas, but rather the *relative* importance of vegetation in controlling overall hillslope stability along streamside slopes is less compared to the regional average.

Similarly, the impact of harvesting also appears to be slightly greater than average in headwall swale areas. The reported impact of clearcut harvesting in headwall areas in Freshwater Creek was 5.0 times background. The measured impact in Hunter Creek does not appear to be as large. Assumed correction factors for MWPZs are listed in Table F1-8.

Table F1-7. Summary of modeled streamside slope vegetation retention under existing and proposed Plan conditions.

	HPA Group[1]	*Slope Distance (feet)*[2]	*Slope Gradient*	*Name*		*Overstory Retention*	
				Existing	**Plan**	**Existin g**	**Plan**
CLASS 1	ALL	0-70	ALL[4]	WLPZ	RSMZ	70%	100%
	ALL	70-100	ALL[4]	WLPZ	RSMZ	70%	85%
	ALL	100-150	ALL[4]		RSMZ	0%	85%
	HUM	150-200	>60%		SMZ	0%	Selc
	KOR, SR	150-200	>65%		SMZ	0%	Selc
	CKLM	150-475	>70%		SMZ	0%	Selc
CLASS 2-2	ALL	0-30	ALL[4]	WLPZ	RSMZ	~70%	100%
	ALL	30-75	ALL[4]	WLPZ	RSMZ	~70%	85%
	ALL	75-100	ALL[4]		RSMZ	0%	85%
	HUM	100-200	>60%		SMZ	0%	Selc
	KOR,SR	100-200	>65%		SMZ	0%	Selc
	CKLM	100-150	>70%		SMZ	0%	Selc
CLASS 2-1[3]	ALL	0-30	ALL[4]	WLPZ	RSMZ	~70%	85%
	ALL	30-70	ALL[4]	WLPZ	RSMZ	~70%	75%
SHALSTAB	ALL	N/A	ALL[4]		SHALSTAB	0%	Selc

Codes
1 HUM Humboldt Bay and Eel River Hydrographic Planning Areas (HPAs)
KOR Mad River, Little River, Redwood Creek, Coastal Lagoons and Interior Klamath HPAs
CKLM Coastal Klamath and Blue Creek HPAs
SR Smith River HPA
2 Assumes 50% sideslopes to calculate horizontal distances
Assumes valley bottom width of 30' for Class 1, 20' for Class 2-2, and 10' for Class 2-1
Watercourse and Lake Protection Zone (WLPZ) distance assumes cable yarding
3 There is no Class 2-2 SMZ in Smith River
4 Assumes all slopes within the RMZ and SHALSTAB areas are greater than the critical slope gradient. This would overestimate the amount of ground in a prescription zone but is unlikely to have a large impact on associated sediment delivery. This is because at least 80% of landslide-derived sediment is from failures on slopes greater than the critical slope gradient.

Table F1-8. Assumed adjustments in the harvest ratio to account for different MWPZs.

	Multiplier Relative to Average		
Mass Wasting Prescription Zone	***Lower***	***Most Likely***	***Upper***
Streamside Slopes (WLPZ, RMZ)	80%	80%	100%
Headwall Swales (SHALSTAB)	100%	150%	150%
Other Areas	100%	100%	100%

F1.2.2 Deep-Seated Landslides

Deep-seated landslides are features with a basal slip plane that extends below the surficial mantle of weathered earth material and into bedrock. They include translational/rotational landslides and earthflows. Translational/rotational slides are characterized by a somewhat cohesive slide mass. In contrast, earthflows are characterized by slow progressive deformation or creep of the slide mass in a semi-viscous, plastic state. Combinations of the two are common. Most deep-seated failures move incrementally, with catastrophic failure being relatively rare.

F1.2.2.1 Methods

Most deep-seated landslides deliver sediment to the stream system by streamside erosion (bank erosion and streamside landslides). Sediment is delivered primarily along watercourses bounding the toes of and, to a lesser extent, by drainage from the interior of the slides. There are few studies, however, that have estimated sediment delivery rates from deep-seated landslides on a landscape scale.

Estimated average long-term deep-seated landslide delivery volumes were estimated for Green Diamond ownership within four pilot watersheds: Salmon Creek, Little River, Upper Mad River and Hunter Creek. It is assumed that sediment delivery from deep-seated landslides can be estimated by multiplying the length of stream channel bordering the toe and lateral margins of the slides by the average depth of the failure (approximate height of banks/gully walls) and average movement rate (Equation 4).

Equation 4: SED_{tot} **= Stream Length * Slide Depth * Rate of Slide Movement**

Because of the lack of data, estimates of sediment delivery from deep-seated landslides should be viewed as approximate. Moreover, because some of the sediment from deep-seated slides is a result of small shallow landslides (i.e., debris flows, debris slides, and channel bank failures) occurring along the toe of the larger landslide, it is likely that some “double counting” of sediment will occur when the results of deep-seated landslides are combined with shallow landslide volumes. At present, however, there is little data to differentiate between the two sediment sources.

The impact of harvesting on sediment delivery from deep-seated landslides was evaluated based on a review of published and unpublished reports, and using professional judgment.

F1.2.2.1.1 Landslide Acreage

Deep-seated landslides in Salmon Creek, Little River, and Hunter Creek were mapped from the historical set of aerial photographs using standard methodologies. Pertinent data associated with each mapped landslide were recorded into a database for further analysis. This information included landslide type (i.e., translational landsliding and earthflows), certainty of identification, and inferred level of activity. Limited field verification of mapped landslides was undertaken in Hunter Creek. Additional fieldwork in the other watersheds is pending.

The Upper Mad River pilot watershed is located upstream of Boulder Creek and encompasses the Boulder Creek Planning Watershed. Identification of deep-seated landslides in the Upper Mad River pilot watershed was initially based on published reconnaissance-level landslide mapping by the California Department of Water Resources (CDWR) (1982). The Mapping by CDWR revealed that roughly a third of the watershed is underlain by deep-seated failures. However, discussions with Green Diamond forestry staff revealed that the mapping of deep-seated landslides in pilot watershed by CDWR likely underestimates the landslide acreage and that as much as 60 percent of the watershed may be underlain by deep-seated landslides. For the purpose of this study it was assumed that 60% of the pilot watershed is underlain by deep-seated landslides.

CDWR (1982) did not differentiate between the two different classes of deep-seated landslides (translational landslides and earthflows). Review of aerial photographs and discussions with Green Diamond staff indicate that roughly 70 percent of the deep-seated landslides in Upper Mad River pilot watershed are earthflows.

Landslide acreage for each of the studied watersheds is summarized in Table 9. With the exception of the Upper Mad River pilot watershed, low and mid-range values were based on measured acreage for definite and probable landslides. For Little River and Salmon Creek, upper range values included acreages for questionable landslides. For Hunter Creek, questionable landslides were not mapped; therefore, upper range values were estimated. For Upper Mad River pilot watershed, the lower range was based on CDWR (1982) mapping; mid- and upper ranges were estimated based on qualitative field and air photo observations by Green Diamond staff.

F1.2.2.1.2 _Landslide Activity_

The range of landslide activity is classified as historically active, dormant, or relic. A slide with documented movement within the past 0 to 100 years (roughly the time frame of modern harvesting practices) is classified as a historically active landslide. In the field, these slides are recognized by some or all of the following features: recent scarps or cracks (>6 inches), leaning second growth trees, or sag ponds and/or offset road prisms (see appendix B for a more complete discussion). Slides with very low rates of movement that do not show signs of obvious movement within the past 50 to 100 years are classified as dormant or relic. It is assumed that harvest activities have the greatest relative impact on the more active slides and that impacts on dormant or relic slides are negligible.

It is usually not possible to accurately evaluate the level of deep-seated landslide activity using air photos alone. Therefore, estimates of slide activity were based on limited field observations, discussions with Green Diamond staff, review of completed geologic reports for timber harvesting plans (THPs), and professional opinion. Slide activity for each pilot watershed and landslide type is summarized in Table F1-9.

Table F1-9. Deep-seated landslide acreage, stream channel length, and level of activity.

Watershed		Range	TRANSLATIONAL/ROTATIONAL LANDSLIDE				EARTHFLOW LANDSLIDE			
			Slide Acres	Watercourse Length (miles)	Activity % of Slide Class		Slide Acres	Watercourse Length (miles)	Activity % of Slide Class	
					Historically Active	*Dormant/ Relic*			*Historically Active*	*Dormant/ Relic*
Salmon Creek		Lower	2880	11.7	5%	95%	61	0.6	5%	95%
		Most Likely	2880	11.7	10%	90%	91	0.6	15%	85%
		Upper	3447	14.5	20%	80%	91	1.1	25%	75%
Little River		Lower	6271	30.7	5%	95%	119	0.9	5%	95%
		Most Likely	6271	30.7	5%	95%	119	0.9	15%	85%
		Upper	7595	39.6	15%	85%	347	2.4	25%	70%
Upper Mad River[1]	**Conifer**	Lower	320	1.6	20%	80%	746	3.7	?	?
		Most Likely	575	2.8	20%	80%	1343	6.6	20%	80%
		Upper	815	4.0	30%	70%	1902	9.4	?	?
	Grassland/ Hardwood	Lower	594	3.1	65%	35%	1385	7.2	?	?
		Most Likely	1069	5.5	65%	35%	2493	12.9	65%	35%
		Upper	1514	7.8	75%	25%	3532	18.2	?	?
Hunter Creek		Lower	338	3.8	5%	95%	0	0	N/a	N/a
		Most Likely	338	3.8	5%	95%	0	0	N/a	N/a
		Upper	500	5.7	15%	85%	0	0	N/a	N/a

1: 49% of the ground in Upper Mad River pilot watershed is grassland or native oak and is not proposed to be harvested. Moreover, disproportionate percentage of the landslides in the watershed are located in these areas. Therefore, the conifer ground has been delineated out separately from grassland and oak.

About half of the Upper Mad River pilot watershed (49 percent) is grassland or native hardwood. Fifty one percent of the area is conifer. Green Diamond staff report that deep-seated landslides underlie about 60 percent of the pilot watershed, and that the slides, and particularly the more active earthflows, are preferentially located in the grassland and hardwood areas (65 percent versus 35 percent). As a result, sediment delivery from grassland/hardwood areas is significantly higher in comparison to conifer areas, and is considered the dominant source of sediment.

Sediment delivery from grassland/hardwood areas was evaluated separately from conifer ground. This is because 1) timber harvesting is not expected to occur in the grassland/hardwood areas and therefore there would be no management-derived sediment from harvesting occurring in these areas, and 2) the grassland/hardwood areas deliver a disproportionate amount of sediment to watercourses because of the high proportion of active earthflows, substantially overwhelming management-derived sediment generated from the conifer ground.

F1.2.2.1.3 <u>*Stream Channel Length*</u>

Sediment delivery from deep-seated landslides is assumed to correlate to the length of all watercourses bounding the toes and lateral margins of these features. This may slightly underestimate the length of stream channels delivering sediment from earthflows because it would not account for sediment eroded from streams draining the interior of the slide. Work by Kelsey (1977) indicates that well-developed gully systems on active earthflows could produce more sediment than erosion along the toe of the slide. However, this is in contrast to work presented by Nolan and Janda (1995) that suggests that less than 10 percent of the measured sediment leaving earthflows was delivered by fluvial processes operating in the small tributaries in the interior of the slide.

The length of streams bordering the toe and lateral margins of large landslides in Salmon Creek, Little River, and Hunter Creek were measured from watercourse maps available in Green Diamond's GIS database. Upper, mid-, and lower range values were based on the degree of certainty of landslide identification. The length of watercourses bounding the toe of large landslides in the Upper Mad River pilot watershed is not available at present and therefore was approximated based on average stream lengths measured in the other three pilot watersheds. Estimated stream lengths bordering landslides for all four pilot watersheds are summarized in Table F1-9.

F1.2.2.1.4 <u>*Slide Depth*</u>

The depth of deep-seated landslides is variable across the landscape depending on landslide size, local terrain, and processes. Swanston and others (1995) reported shear depths along earthflows and block glides in Redwood Creek to be between 12 and 40 feet. Past studies in the Eel River Basin found an average height of earthflow toes of 30 feet (SWS 1999; USACE 1980; USDA 1970).

Professional experience suggests that the depth of deep-seated translational landslides can vary considerably, from between 10 to greater than 100 feet. In general, translational landslides are much deeper than earthflows. An average slide depth subject to toe erosion of 40 feet was assumed for translational landslides, and 25 feet for earthflows. Upper and lower bounding depths were estimated at 10 feet deeper and 10 feet shallower, respectively.

F1.2.2.1.5 *Slide Movement Rates*

Deep-seated slide movement is highly variable and episodic, depending on storm history, underlying geology, and slide process. At present, very limited data are available for estimating average long-term movement rates of deep-seated landslides in northern California. In this preliminary analysis, the average creep rates on the west side of Redwood Creek was used.

Swanston and others (1995) monitored several sites in the Redwood Creek Basin to quantify natural creep and earthflow rates. A concerted effort was made to avoid areas of current, clearly definable active earthflows; however, Green Diamond's review suggests that several of these sites appear to have been on slides that may have been classified as historically active under the Plan's slope stability measures.

Progressive earthflows on the east side of Grogan Fault in Redwood Creek that are underlain by pervasively sheared sandstone and mudstone have movement rates from 3.0 to 131 mm/yr. These rates are assumed to be representative of active earthflows on Green Diamond property. Sites dominated by block slides displayed movement rates ranging between 2.5 and 16.4 mm/yr. These rates are assumed to be representative of active translational landslides on Green Diamond property. Progressive creep rates on the west side of the Grogan Fault in Redwood Creek that are underlain by sheared and foliated schists range between 1.0 to 2.5 mm/yr. These rates are assumed to be representative of natural soil creep and of dormant earthflows and translational landslides.

Regional data sources on active grassland earthflows report much higher average movement rates of 2.4 to 4 m/yr [Van Duzen River Basin (Kelsey 1980)] and 4 m/yr [Eel River Basin (Scott 1973, referenced in SWS 1999)]. It is doubtful that these rates are representative of all earthflows, because in these studies there was a bias toward monitoring the most active slides. Moreover, the rates are for earthflows in open grassland areas and not representative of forested slides where rates are much lower to support a timber stand.

Limited field reconnaissance of the deep-seated landslides in Hunter Creek, Little River, and Salmon Creek revealed that most of the large slides are dormant or relic, and have very low rates of movement. Where movement is observed, it is typically manifested by small discontinuous ground cracks along the head of slide blocks. Lobate toes or zones of accumulation are rarely present.

Estimated deep-seated landslide rates are summarized in Table F1-10. High and low range values are based primarily on data presented by Swanston and others (1995). Most likely values are from published data and were modified based on professional judgment. Most of the slides on Green Diamond property do not appear to be as active as those studied in the professional literature, as is indicated by the simple fact that most roads crossing large landslides are not disturbed by slide movement. Therefore, the most likely rate of movement on forested slides is assumed to be lower than the published average. Because few measurements of deep-seated landslides in northern

California exist, these rates should be viewed as very approximate. Additional research is required to refine these numbers and to increase the confidence in their accuracy.

Table F1-10. Average deep-seated landslide slip rates.

Slide Type	Activity	Average Slip Rate (mm/yr)		
		Lower	***Most Likely***	**Upper**
Translational/Rotational Landslide	Historically Active	2.5	4	16.4
	Dormant/Relic	0.5	2	2.5
Earthflow Landslide	Historically Active	3.0	20	130
	Dormant/Relic	0.5	2	2.5

F1.2.2.1.6 *Harvest-Derived Sediment*

Published work concerning the effects of timber harvesting (i.e., logging) on deep-seated landslide activity is sparse. Deep-seated landslides can theoretically be affected by hydrologic changes associated with reduced evapotranspiration and reduced canopy interception during rainstorms (California Department of Conservation 1997). Descriptions of conditions affecting deep-seated landslides have been discussed briefly by Swanston and Swanson (1977), Sidle and others (1985), and Miller and Sias (1998), but few studies exist that quantitatively address how timber harvesting affects deep-seated landslide stability.

Short-term increases in ground displacement following clearcutting have been documented on an active earthflow in southwestern Oregon (Swanston et al. 1988; Swanston 1981). Swanson and others (1988) report substantial short-term increases in ground displacement rates beginning the second year after harvesting, with movement rates returning to background rates in the third year following harvest. Post-harvest rates are reported to be more than two to four times the pre-harvesting rate (Swanston 1981). The short-term nature of the increase was probably the result of dry conditions and the small regolith blocks involved in accelerated displacement. In contrast, work by Pyles (1987) on the Lookout Creek earthflow in the central Cascades in Oregon concluded that timber harvesting was unlikely to induce a large increase in movement, primarily because the slide was well-drained.

Miller and Sias (1998) modeled the effect of timber harvest on groundwater conditions and slope stability of a large, deep-seated landslide in glacial lacustrine sediments adjacent to a large river channel in the western Washington Cascades. They predicted that timber harvest in the groundwater recharge area of the landslide would produce very small decreases in the factor of safety, suggesting that harvest would contribute to landslide movement only if the landslide were at or near the threshold of stability. This suggests that active deep-seated landslides are most likely to be affected by harvest-induced changes in groundwater, while inactive and dormant slides are less likely to be affected.

There may be some impact from clearcut harvesting on sediment delivery from deep-seated landslides; however, to what extent is difficult to quantify at present. For the purpose of this study it was assumed that harvesting will have an impact only on historically active slides and negligible impact on dormant or relic features, and that the level of impact will be proportional to the level of harvest. It was assumed that clearcutting the entirety of the slide will increase the rate of slide movement by a factor of two on historically active slides, diminishing linearly to pre-harvesting rates in 30 years. Based on this assumption, the average increase in deep-seated slide movement over the 50-year period of the Plan would be 1.3 times background if the slide were entirely clearcut.

It is assumed that the impact of harvesting on deep-seated slide activity is a function of percentage of canopy retained on a slide, which in turn is expected to be directly related to evapotranspiration rates. In this analysis, it was assumed harvesting will take place on the entirety of a slide. This is considered a worst-case scenario because many slides exceed the maximum 40-acre size of clearcuts under current California Forest Practice Rules, and harvest blocks would rarely have boundaries that coincide with slide boundaries. It is unlikely that all of a slide would be harvested at any given time; therefore, the impact of the harvest is expected to be less than modeled.

Under current conditions, vegetation retention results primarily from the required 70 percent overstory canopy retention along Class I and Class II WLPZs under Green Diamond's Owl HCP. The amount of vegetation retained on any given slide is quite variable, depending on the density and class of watercourses transecting or bordering the slide, existing stand density and composition, and silviculture prescriptions. Additional retention has often been provided on the more active slides in the interest of slope stability. On average, however, it is estimated that a minimum of 5 percent to 10 percent of the total canopy cover is currently retained on deep-seated landslides. Therefore, the sediment delivery under existing management conditions is estimated to be about 1.28 times background.

Under proposed Plan prescriptions, vegetation retention on historically active slides will be primarily from RMZ, slope management zone (SMZ), and SHALSTAB areas. Additional protection is provided by 25-foot no-cut zones along historically active toes and scarps (see Section 6.2). The proposed Plan prescriptions are estimated to be 15 percent effective in reducing the management component of sediment delivered from deep-seated landslides relative to existing conditions.

F1.2.3 Results

This section presents the results of a modeling effort designed to estimate average long-term landslide sediment delivery volumes to watercourses from the historical road network and from various silvicultural treatments. As previously mentioned, the information presented below is specific to sediment delivery from shallow and deep-seated landslides; sediment delivery from other processes, such as surface erosion, channel bank erosion, or erosion of watercourse crossings is not addressed in this appendix. The results represent long-term totals for each pilot watershed.

Average long-term sediment delivery volumes from shallow and deep-seated landslides were estimated for both existing and proposed Plan conditions for three pilot watersheds: Salmon Creek, Little River, and Hunter Creek. Sediment delivery from deep-seated landslides was also estimated in the Upper Mad River pilot watershed. Work in Ryan Creek and Tectah Creek was used to examine the effects of road building on landslides, but could not be used to examine the effects of silviculture at the time of the statistical analysis. Results from shallow-seated landslides are reported separately from deep-seated landslides.

F1.2.3.1 Shallow Landslide Results

Road-related and non-road-related shallow landslides were evaluated separately from one another. Shallow landslide data was gathered primarily from aerial photograph interpretation. Landslides that occur near roads were assumed to have been triggered by road construction (i.e., grading activity). Landslides in harvest areas were not assumed to be caused by harvest effects (e.g., loss of root reinforcement). Instead, the proportion of landslides in harvest areas that were likely triggered by harvest effects is estimated using the harvest ratio HR(n) (see Equation. 3). A spatial analysis of non-road- related landslides assesses the proportion of slides that originate in different Plan MWPZs. Finally, the expected sediment reductions resulting from the Plan's mass wasting prescriptions pertaining to harvest effects were estimated.

F1.2.3.1.1 Road-Related Landslides

Estimated shallow landslide delivery volumes from shallow landslides resulting from all grading activities are summarized in Tables F1-11 and F1-12. The data are presented in two forms. In Table F1-11, the average sediment delivery from shallow landslides is summarized for the entire (long-term) photoperiod. However, these values may not be representative of recent conditions because of improvements in road management and increased road densities. The relative impact of grading is most likely best represented by a more recent (1997) photoperiod, covering a roughly 7- to 12-year time span (Table F1-12). A summary of the relative percentage of each grading activity to the total volume of shallow landslide sediment delivered to watercourses is summarized in Table F1-13.

Table F1-11. Shallow landslide delivery from the long-term period of record.

Watershed	Period of Record (years)[1]	# of Shallow Landslides	Sediment Delivery (cy)				
			Total	Road and Landing	Skid Trail	Other[2]	Non-Grading[3]
Salmon Creek	58	756	156732	40398	1174	78	115082
Ryan Creek	46	1260	27903	6893	1248	1100	18663
Little River	64	419	139457	20230	2546	5714	110966
Hunter Creek	54	598	494523	216584	90167	0	187772
Tectah Creek	--	--	--	--	--	--	--

Notes

1. Landslides visible in the earliest set of air photos are assumed to have occurred within the previous 15 years based on the level of revegetation
2. Other includes failures along the old railroad lines and failures from non-harvesting-related grading activities.
3. Non-grading summarizes sediment not generated from grading activities

Table F1-12. Shallow landslide delivery from the 1997 photoperiod.

			Sediment Delivery (cy)				
Watershed	**Period of Record (years)**	**# of Shallow Landslides**	**Total**	**Road and Landing**	**Skid Trail**	**Other[1]**	**Non-Grading[2]**
Salmon Creek	6	329	55515	9241	333	0	45941
Ryan Creek	7	152	10014	3967	527	1100	4420
Little River	10	34	14525	5844	0	0	8681
Hunter Creek	13	301	29497	9729	1680	0	18088
Techtah Creek	?[3]	631	104121	18589	550	0	84982

Notes
1. Other includes failures along the old railroad lines and failures from non-harvesting-related grading activities.
2. Non-grading summarizes sediment not generated from grading activities
3. This period of record is uncertain because only one set of aerial photographs (1997) was examined

Table F1-13. Percentage of each grading activity relative to total shallow landslide delivery.

		Long-Term Period of Record			1997 Photoperiod		
Watershed	**Acreage**	Roads and Landings	Skid Trails	Other[1]	Roads and Landings	Skid Trails	Other[1]
Salmon Creek	7889	26%	1%	0%	17%	1%	0%
Ryan Creek	7590	25%	4%	4%	40%	5%	11%
Little River	28755	15%	2%	4%	40%	0%	0%
Hunter Creek	10126	44%	18%	0%	33%	6%	0%
Tectah Creek	12675	-	-	-	18%	1%	0%

Note
1 Other includes failures along the old railroad lines and failures from non-harvesting-related grading activities.

Roads and Landings

The data suggest that roads and landings (combined) are responsible for the majority of landslide-derived sediment that is generated from grading activities. Skid trail failures, in comparison, are infrequent. For the long-term period of record, landslide-derived sediment from roads and landings ranges between 15 percent and 44 percent of the total sediment delivered from shallow landslides. As expected, the impact of roads is greatest in the steeper gradient watersheds (e.g., Hunter Creek) and less in the lower gradient watersheds (e.g., Little River). In the 1997 photoperiod, road and landing failures comprise 17 percent to 40 percent of the shallow landslide delivery.

A decrease in the relative importance of road-related failures was observed in Salmon Creek and Hunter Creek, which have inherently high rates of landsliding, even though road densities have increased in both watersheds. The decrease in road-related failures (both volume and size) in these watersheds may be attributed to improvements in forest practices and the implementation of Forest Practice Rules over the past 25 years. Because of these regulations, new roads are more likely to be located on more stable ridge tops that have much lower rates of landsliding rather than less stable mid to lower slope areas, and constructed using end-haul construction techniques when steep slopes cannot be avoided. New roads and reconstructed (repaired) roads also have restrictions on fill depth, compaction of fill, more frequent cross drain and waterbar spacing, and increased culvert sizes. Steep ground is commonly cable yarded rather than tractor yarded, resulting in much less ground disturbance.

An increase in road and landing failures was observed in Ryan Creek and Little River; however, both of these watersheds have inherently low rates of slide activity. In both of these watersheds, it is believed the relative importance of shallow landslide processes to the total sediment budget is less than in the steeper watersheds such as Hunter Creek and Salmon Creek. In Little River, and to a lesser extent in Ryan Creek, it is also difficult to draw definitive conclusions on changes in sediment delivery over time because of the relatively small sample size in the 1997 photoperiod (see Table F1-2), and because much of the observed sediment from that period was generated from just a few slides.

Preliminary results show that mean landslide volumes for road and landing failures have decreased over time from 400 cy/slide in the long-term photoperiod to 275 cy/slide in the 1997 photoperiod. Additional work would be required to further evaluate whether the reduction is a result in improved road management or simply a product of storm history.

Skid Trails

Skid trail-related failures comprise a substantially smaller portion of the total volume of sediment delivered from landslides compared to roads and landings (Table 14). In the long-term period of record, skid trail failures comprise between 1 percent and 18 percent of the total volume of sediment delivered from shallow landslides. Additional unquantified sediment would be generated from surface erosion of the skid trail. The majority of this impact resulted from the early failures in the Hunter Creek watershed. Excluding Hunter Creek, the measured long-term impact of skid failures averages less than 2 percent of the total shallow landslide delivery volume.

In the 1997 photoperiod, skid trails comprise 0 percent to 6 percent of the landslide sediment delivered to watercourses. Mean landslide delivery volumes for skid trail failures have decreased from a long-term average of 275 cy/slide to a recent short-term average of 57 cy/slide. Again, the decrease in the size of slide may be due to changes in forest practices, such as a greater reliance on cable yarding rather than tractor yarding, or be a product of storm history. Skid trail failures were also substantially smaller than road failures, probably because skid trails tend to have smaller fill prisms.

Comparison of Road and Skid Trail Failures

One of the goals of this analysis was to gain insight into the relative importance of road failures compared to skid trail failures. In other words, how important are road failures to the total sediment delivery compared to skid trail failures? This is an important question when allotting resources to address legacy problems.

Comparing Table F1-14 summarizes the relative importance of road failures normalized against skid trail failures. This simple ratio was generated by dividing the volume of sediment delivered from road failures by the volume of sediment delivered from skid trail failures. The data is based on total landslide sediment delivered and has not been normalized against length of road or skid trail.

Table F1-14. Summary of sediment delivery from road and landing failures normalized against skid trail failures.

	Long-Term Period of Record		1997 Photoperiod	
Watershed	Road and Landing	Skid trail	Road and Landing	Skid trail
Salmon Creek	34.4x	1x	27.7x	1x
Ryan Creek	5.5x	1x	7.5x	1x
Little River	7.9x	1x	∞	1x
Hunter Creek	2.4x	1x	5.8x	1x
Tectah Creek	--	1x	33.8x	1x
AVERAGE[1]	**3.1X**	**1X**	**13.4X**	**1X**

Note
1 Average is calculated from the sum of all inventoried landslides with no weighting given to watershed area.

The ratio of road-derived sediment to skid trail-derived sediment is quite variable between watersheds. Much of this variability is likely attributed to relative differences in road and skid trail densities in each watershed. Nonetheless, the data do indicate for all watersheds there has been a sustainable decrease in sediment delivery from skid trails in comparison to road and landing failures (Table F1-14). One possible explanation for the measured reduction is the stricter forest practice rules that limit tractor yarding on slopes steeper than 65 percent. By avoiding tractor operations on such slopes, the potential for new skid trails to trigger slides has been greatly reduced, as documented in Table F1-14.

It is important to point out that the results in Table F1-14 are based on sediment volumes. A similar analysis based on frequency (number) of landslides would reveal that roads generate two to four times as many landslides as skid trails for both the long-term period of record and 1997 photoperiod, respectively. The difference between the analysis based on sediment volume and frequency of slides is a product of larger landslides occurring on roads compared to skid trails.

The results based on frequency of landslides are consistent with the results of the California Department of Forestry and Fire Protection's (CDF's) Hillslope Monitoring Program (1999), which documented 4.5 times as many large debris slides occurring on roads and landings compared to skid trails. Sediment volumes were not presented in the CDF report. The Hillslope Monitoring Program was based on a comprehensive field evaluation of erosion features identified on 292 random road transects (53 miles), 26 skid trail transects (33 miles), and 291 landing transects.

There are several possible explanations for the lower rate of skid trail failures compared to road failures. First, the majority of shallow landslides occur on slopes over 60 percent to 65 percent. This is ground that under the Forest Practice Rules must be cable or helicopter yarded rather than tractor yarded. By avoiding such steep slopes, the potential for future skid trails to trigger shallow landslides has been greatly diminished. Because Green Diamond began to employ cable yarding techniques on much shallower slopes than many of the other timber companies, the effect of skid trails may be much less than for other areas. Roads, on the other hand, often cannot avoid steep ground.

In addition, the landslide inventory suggests a reduction in skid trail failures compared to road and landing failures over time. One explanation for this is that many of the legacy skid trails that were located on steep slopes have since failed and comparatively few skid trails are constructed on steep slopes under present management practices. Many of the skid trail failures observed in the 1997 set of aerial photographs are associated with legacy skid trails. To address the potential for future skid trail failures, Green Diamond proposes to exclude tractor operations on slopes greater than 45%.

The lower rate of skid trail failures in relation to road failures may also be a product of the differences in the amount of ground disturbance required to cut a skid trail vs. a road. The average width of a skid trail is about 10 feet compared to a 20+ width for roads. A 10-foot-wide skid trail contouring across a 65 percent side slope would displace 0.7 cy of earth per foot of skid trail, resulting in a 1.8-foot-deep fill prism. A skid trail descending the same hillside at a steep gradient would generate much less fill. In comparison, a 20-foot-wide haul road contouring across the same slope on balanced cut and fill would generate four times as much sidecast, with a fill prism of over 4 feet. Moreover, thicker fill prisms on roads often exist at watercourse and swale crossings, which is where many of the larger fill failures originate.

F1.2.3.1.2 *Harvesting-Related Sediment*

Estimates of sediment delivery from shallow landslides are based primarily on a review of aerial photographs. The harvesting components (tree removal alone) of shallow landslide sediment delivery volumes were estimated for three pilot watersheds (Salmon Creek, Little River, and Hunter Creek) by applying non-road-related shallow landslide sediment delivery volumes measured from aerial photographs to several empirical models that relate management activities to increased erosion rates. Harvesting-related sediment delivery was estimated for existing and proposed Plan conditions. The results of this modeling effort are summarized in Tables F1-15 and F1-16.

Table F1-15. Non-road-related shallow landslide sediment delivery per mass wasting prescription zone under existing conditions.

			MWPZ			
WATERSHED	**ACRES**	**RSMZ Cy/yr %**	**SMZ cy/yr %**	**SHALSTAB cy/yr %**	**NONE cy/yr %**	**TOTAL cy/yr %**
Salmon Creek	7889	798	2	268	916	1984
		40.2%	*0.1%*	*13.5%*	*46.2%*	
Little River	28755	768	31	195	740	1734
		44.3%	*1.8%*	*11.2%*	*42.7%*	
Hunter Creek	10126	235	697	1190	1355	3477
		6.8%	*20.1%*	*34.2%*	*39.0%*	

Table F1-16. Non-road-related shallow landslide sediment delivery under existing and proposed Plan conditions.

WATERSHED	ACRES	BACKGROUND		HARVESTING		TOTAL NON-ROAD		Reduction in Management Component
				Existing Conditions	Proposed Plan	Existing Conditions	Proposed Plan	
		Cy/yr	Cy/ac/yr	Cy/yr	Cy/yr	Cy/yr	Cy/yr	%
Salmon Creek	7889	1174	0.15	810	523	1984	1698	35%
Little River	28755	1054	0.04	680	424	1734	1478	38%
Hunter Creek	10126	1693	0.17	1785	1109	3477	2802	38%

In Salmon Creek and Little River, non-road-related sediment delivery in the RMZ prescription areas is significantly greater than in SMZ or SHALSTAB areas. This contrasts notably with Hunter Creek, where the majority of sediment was generated from failures within SHALSTAB and SMZ areas. There are several possible reasons to account for the higher rate of sediment delivery in the Hunter Creek SMZ and SHALSTAB areas compared to either Salmon Creek or Little River. First, the majority of sediment in Hunter Creek is generated by very large slides that extend well outside the RMZ and therefore are not assumed to be controlled by conditions within the RMZ. Similar large slides are not as prevalent in either Little River or Salmon Creek, possibly because slopes are generally not as steep. Second, the watercourse mapping in Hunter Creek is relatively old and many Class III drainages in that drainage would be reclassified as Class II watercourses under current rules. In the analysis, this results in fewer RMZ slides than probably actually exist. Lastly, the terrain in Hunter Creek is much steeper than in either Little River or Salmon Creek, which results in a greater percentage of SHALSTAB areas.

The data also reveal that a significant volume of sediment (39 percent to 46.2 percent) is generated from failures located outside of any MWPZ. This might be partly explained by the inherent limitations of the existing 10-m digital elevation models (DEMs) used to generate slope gradients in the GIS. The DEM tends to underestimate slope gradients, especially in deeply incised drainages. Because this analysis relies on aerial photo interpretation and topographic and map data, fewer prescription zones may have been mapped compared to field-based mapping, potentially resulting in an underestimate of associated sediment delivery. Nonetheless, the results illustrate the inherent difficulties in identifying landslide hazard areas solely from a remote analysis. A greater level of prediction would be achieved based on site-specific field review.

Based on the HR equation (Equation 3) background, sediment delivery from shallow landslide processes averages between 0.04 and 0.17 cy/ac/year (see Table 16). The higher sediment delivery in Salmon and Hunter creeks likely results from steep streamside slopes (Salmon Creek) and headwall swale areas (Hunter Creek). Background sediment delivery rates in Little River are relatively low in comparison because of the relatively shallow slopes found throughout most of the watershed.

Harvesting (tree removal) over a 50-year period is estimated to be responsible for 39 percent to 51 percent of the total non-road-related shallow landslide sediment delivered to watercourses under existing conditions (1.6 to 2.1 times increase relative to undisturbed or advanced second growth forests). Implementation of the proposed Plan measures is expected to reduce the harvesting-related component of sediment by at least 35 percent to 38 percent. Significantly more sediment savings will be achieved by road upgrades (see Appendix F2).

F1.2.3.2 Deep-Seated Landslide Results

Estimated annual sediment delivery volumes from deep-seated landslides are summarized in Table F1-17. These estimates are based on the deep-seated landslide sediment source model presented earlier in this report. Average long-term sediment delivery from deep-seated landslides is estimated to range between 0.02 cy/ac/yr in Hunter Creek, where few landslides are present, to 0.44 cy/ac/yr in the Upper Mad River pilot watershed, where much of the watershed is underlain by deep-seated landslides, many of which are considered active.

In the Upper Mad River pilot watershed, sediment delivery rates are significantly higher in the oak and grassland areas compared to conifer ground. This is attributed to the much higher percentage of earthflows located in this terrain. In general, the open grassland and hardwood areas are less stable than the conifer ground, and many grassland areas are too active to support viable conifer forest. The impact of harvesting in the grassland areas is negligible because few trees grow in these areas.

For the purpose of this study it is assumed that the impact of harvesting is directly proportional to the amount of vegetation retained on a historically active slide. Based on this assumption, harvesting (tree removal) is estimated to be responsible for an increase of from 1.02 to 1.17 times the amount of sediment delivered by deep-seated landslides in conifer areas under existing conditions (harvesting is generally not proposed in grassland and hardwood areas). This may be an overestimate of the impact of harvesting, because it assumes that the slide block is located wholly within a harvest unit. More often, only a portion of a slide is cut at any given time.

Table F1-17. Deep-seated landslide sediment delivery under existing and proposed Plan conditions.

WATERSHED	ACRES	BACKGROUND		HARVESTING		TOTAL NON-ROAD (Background + Harvesting)		Assumed Reduction in Management Component
				Existing Conditions	Proposed Plan	Existing Conditions	Proposed Plan	
		cy/yr	cy/ac/yr	cy/yr	cy/yr	cy/yr	cy/yr	%
Salmon Creek	7889	706	0.09	42	35	748	741	15%
Little River	28755	1722	0.06	56	48	1778	1770	15%
Upper Mad River – Conifer	4658	767	0.16	135	115	902	882	15%
Upper Mad River – Grasslands/ hardwoods	4475	3309	0.74	0	0	3309	3309	N/a
Hunter Creek	10126	204	0.02	5	5	209	209	15%

The variability in landslide delivery between watersheds is primarily a function of the percentage of the watershed underlain by historically active landslides, particularly earthflows. Data indicate that sediment delivery rates on earthflows are much higher than for translational/rotational rockslides. Implementation of the proposed Plan measures is assumed to reduce the management component of sediment by at 15 percent.

Roads can affect the stability of deep-seated landslides by removing toe support and by concentrating and diverting runoff. However, at present there is little data on Green Diamond property to address the significance of roads on deep-seated landslide sediment delivery. Moreover, there are very few published studies that have addressed this question. This analysis does not separately address sediment delivery related to road construction on deep-seated landslides. It was assumed that any sediment delivered by deep-seated landslides as a result of roads is already indirectly addressed in either the shallow landslide section of this report or in the road inventory section presented in Appendix F2.

F1.2.3.3 Summary of Results

Road-related shallow landslides occurring in the most recent photoperiods range from 17 percent to 40 percent in the five watersheds investigated, with a watershed mean value of about 30 percent. The extent to which the Plan measures are expected to reduce road-related shallow landslides is discussed in Appendix F2.

Harvest-related shallow landslides were estimated to constitute 39 percent to 51 percent of non-road-related shallow landslides for the three watersheds investigated. The proposed Plan measures (MWPZs and associated prescriptions) are expected to reduce harvest-related shallow landslides by 36 percent to 44 percent. Shallow landslides occurring outside of MWPZs account for 39 percent to 46 percent of sediment delivery.

Timber harvest on deep-seated landslides is calculated (based on estimates) to increase sediment delivery to streams by 2 percent to 17 percent. Plan measures for harvest on deep-seated landslides are expected to be only 15 percent effective, resulting in small declines in harvest-related sediment delivery from deep-seated landslides. However, management-related sediment from deep-seated landslides is not considered to be a large component of the total volume of sediment delivered by landslides.

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Appendix F2. Road-Related Sediment Source Inventory of High and Moderate Priority Sites

CONTENTS

Figure

Tables

F2.1 INVENTORY METHODS

Since 1997, over 40 mi^2 of Green Diamond's forest lands have been inventoried for on-going and potential sediment sources that have the potential to deliver eroded sediment to stream channels. The inventories, funded by the CDFG Restoration Grant Program and by Green Diamond Resource Company, identified road-related sediment sources in the biologically high priority watersheds through a two-step process of air photo analysis and field inventories. An analysis of historic aerial photos was conducted to identify <u>all</u> the roads that were ever constructed in each of the inventoried watersheds, whether they were maintained and driveable, or abandoned and overgrown with vegetation. When possible, historic photographs from a number of years (perhaps one or two flights per decade) were selected to "bracket" major storms in the watersheds. This analysis led to the construction of detailed land use history maps for the watershed, specifically including road location and road construction history.

Field inventories and site analyses were employed to identify and quantify future road-related sediment sources and to develop defensible plans for erosion prevention in each of the five watersheds. From north to south these included Rowdy Creek (17.1 mi^2), McGarvey Creek (7.0 mi^2), Redwood Creek (11.0 mi^2), Little River (35.0 mi^2) and Salmon Creek (6.8 mi^2). The two most important factors used to evaluate the risk of road-related sediment delivery in these basins included: 1) an assessment of the probability of erosion or failure at all "susceptible" points along the alignment (termed "erosion potential") and 2) an estimation of the volume of potential sediment delivery to a stream (if no preventive work is done). The data that were collected were then employed to develop a defensible, cost-effective plan for mitigating or preventing road-related sediment delivery in each basin.

For the detailed field assessment, acetate overlays were attached to 9" x 9" aerial photographs and used to record site location information as it is collected in the field. A computer database (data form) was then completed for each site of potential sediment delivery identified in the field. Only sites of future sediment delivery were included in the inventory. Detailed inventories of all maintained and abandoned road systems were used to identify and determine future contributions of sediment to the stream system, and to define cost-effective treatments.

The most common sediment source sites generally included watercourse crossings, potentially unstable road and landing fills, and "hydrologically connected" road segments which exhibit surface erosion and sediment delivery. Once sites were identified and quantified, prescriptions for erosion control and erosion prevention were developed for each major source of treatable erosion that, if left untreated, would likely have resulted in sediment delivery to a stream. Prescriptions developed during the field inventory included types of heavy equipment needed, equipment hours, labor intensive treatments required, estimated costs for each work site and quantitative estimates of expected sediment savings.

F2.2 ROAD-RELATED SEDIMENT SOURCES

Three geomorphic processes are responsible for sediment delivery from roads. These include: 1) chronic surface erosion from bare soil areas, 2) landslides (mostly from the fill slope, but also including some cutbank failures), and 3) watercourse crossing failures (mostly gullying from washouts and diversions, but also including other types of crossing erosion). In sediment source inventories that have been performed on Green Diamond road networks in north coast watersheds over the last five years, these processes were found to deliver sediment to streams in different amounts and with differing efficiencies (Table F2-1).

F2.2.1 Chronic Erosion

In general, *chronic erosion* delivers sediment every winter, whether or not there are any large storms. The volume of fine sediment which is delivered to streams from the road system is a function of the type and amount of traffic on the road system, as well as the length of road and road ditches which drain directly to streams. Sediment delivery from chronic road erosion is generally greatest on roads that are open and used during the winter, and where ditches are connected to the streams. Roads which are abandoned and overgrown, and those where there is very little "connectivity" typically contribute far less sediment from chronic surface erosion than those which are well connected and used for commercial hauling.

In the inventories of Salmon Creek and Rowdy Creek, it was found that 12% and 21% of the road networks, respectively, are directly connected to the stream system through road side ditches. On average, over 30% of the inventoried road systems on Green Diamond lands were found to be hydrologically connected to the stream system. These road surfaces and ditches are delivering both runoff and fine sediment directly to streams. Although this represents a threat or risk to the aquatic system, it is not one which results in catastrophic sediment inputs.

F2.2.2 Episodic Sediment Sources

The other two types of sediment delivery that are derived from road-related landslides and watercourse crossing erosion are more episodic in nature (Table F2-1). Episodic mass wasting and watercourse crossing failures most commonly occur during large storm events. The more extreme the hydrologic event is, the more frequent and larger are the failures from these two sediment sources. These episodic sediment sources deliver relatively large quantities of sediment (including both fine and coarse grain sizes) to stream channels. Future episodic sediment sources represent a risk or threat to the aquatic system that tends to be more substantial as the storm size increases. All else equal, the risk is often greatest on old and/or abandoned roads which have culverts that may be unmaintained and/or undersized for the design (100-year) flow event. Newly constructed roads also exhibit increased risk of sediment production for the first several years following construction.

Table F2-1. Sources and magnitude of road-related sediment delivery in selected Green Diamond watersheds, north coastal California[1]

Site location	Process	Sediment delivery for road-related erosion sites			
		Delivery range for sites		Average delivery (yds^3)	Percent of road-related sediment delivery (range)[2]
		(%)	(yds^3)		
1. chronic surface erosion from bare soil areas (road surfaces, ditches and cutbanks)[3]	Surface erosion	NA	NA	NA	<5% - 15%
2. road-related landslide erosion	Mass wasting				15% - 80%
fill slope failures		5-100%	5 - 2,500	220	
landing failures		5-100%	5 - 2,000	385	
cut bank failures		50-100%	10 - 150	80	
hillslope landslides[4]		25-100%	10 - 10,000	3,500	
3. watercourse crossing erosion	Fluvial erosion				35% - 80%
watercourse crossing washouts		100%	5 - 3,000	225	
stream diversions (gullies)		80-100%	5 - 2,800	400	

[1] Data based on inventories of Salmon Creek and Rowdy Creek road systems; sediment delivery from stream diversions based on data from Jordan Creek (lower Eel River).

[2] Typically, watersheds with geologies like Salmon Creek and Rowdy Creek are dominated by fluvial processes, where road-related fluvial erosion (washouts and gullying at watercourse crossings) is expected to account for up to 85% of future sediment delivery. Road-related mass wasting is comparatively less in these watersheds. In steep, potential unstable watersheds on the north coast, such as those of the lower Eel River and Mattole, mass wasting may account for up to 65% of future road-related sediment delivery. In these watersheds, fluvial processes are relatively less important.

[3] Sediment delivery from road-related surface erosion occurs where the road is hydrologically connected to the stream system. Delivery volumes are based on contributing length of road reach, use levels, surface erosion rates and duration of analysis. Does not include surface erosion from non-road sources.

[4] Small to large hillslope slides triggered by road cuts, road fills or by altered hydrology (diversion or discharge)

F2.3 RESULTS

For this analysis, a total of 518 miles of forest road from five watersheds were included in the assessment. The watersheds spanned a number of the geologic types and geographical terrains of Green Diamond's north coast property. Just over 2,800 inventoried sites were judged to have a high or moderate priority for erosion prevention or erosion control treatment (Table F2-2). The average frequency of sediment delivery sites ranged from 3 sites/mile (Rowdy Creek) to over 7 sites/mile (Little River). Sub-watersheds in these basins displayed even greater variability in their potential for erosion and sediment delivery.

The field inventory employed standard inventory protocols developed by PWA and employed on forest and ranch lands throughout the north coast. Watercourse crossings represented the most common and volumetrically most important of the future sources of road-related sediment in most Green Diamond watersheds (Table F2-2). As future sediment sources, watercourse crossings were followed in importance by road-related landslides (mostly fill slope failures), and by "other" sediment sources (including ditch relief culverts and gullies). Non road-related landslides were not included in the road inventories (see Appendix F1).

Treatment costs were developed for all high and moderate priority sites in each of the five watersheds. These treatment costs were then analyzed according to each of the three main sediment sources (watercourse crossings, landslides and "other" sites). The breakdown of costs for erosion prevention treatments for these three sediment sources is depicted in Tables F2-3, F2-4 and F2-5, respectively. Total costs to treat all watercourse crossings (including both road upgrading (storm-proofing) and road decommissioning) is expected to exceed $9 million. Treatment of road-related landslide sites and "other" sites in these sample watersheds are expected to require $1.3 million and $0.5 million, respectively.

Basic treatment priorities and prescriptions were formulated concurrent with the identification, description and mapping of potential sources of road-related erosion and sediment yield.

Treatment priorities were evaluated on the basis of several factors and conditions associated with each potential sediment delivery site:

1) *Delivery volume* - the expected volume of sediment to be delivered to streams,

2) *Erosion potential* - the potential for future erosion (high, moderate, low),

3) *Access and access costs* - the ease and cost of accessing the site for treatments,

4) *Treatment costs* - recommended treatments, logistics and costs,

5) *Treatment immediacy* - the "urgency" of treating the site, and

6) *Treatment cost-effectiveness ($ spent per yd^3 "saved").*

Table F2-2. Analysis of inventoried road-related erosion sites in the Plan Area with high treatment priorities.

Watershed name	Assessment area (mi^2)	Road length analyzed (mi)	High and moderate priority sites (#)			Future sediment delivery from watercourse crossings				Future sediment delivery from landslides				Future sediment delivery from “other” sites			
			#	#/mi	#/mi^2	# of sites	yds^3	Yds^3/mi	yds^3/mi^2	# of sites	yds^3	yds^3/mi	yds^3/mi^2	# of sites	yds^3	yds^3/mi	yds^3/mi^2
Salmon Creek	6.8	36	183	5	27	153	43,472	1,208	6,393	19	7,023	195	1,033	11	364	10	54
Rowdy Creek	17.1	135	373	3	22	302	111,386	825	6,514	60	8,906	66	521	11	149	1	3
McGarvey Creek	7.0	63	383	6	55	195	110,115	1,748	15,731	181	49,330	783	7,047	7	84	1	12
Redwood Creek (PPZ)[1]	11.0	64	355	6	32	207	75,873	1,186	6,898	98	48,807	763	4,530	50	2,076	32	189
Little River[2]	35.0	220	1,533	7	44	939	248,390	1,129	7,097	315	60,994	277	1,743	279	6,454	29	184
Total	76.9[3]	518	2,827	5.5	37[3]	1,796	589,236	1,137	7,662[3]	673	175,060	338	2,276[3]	358	9,127	18	119[3]

[1] The Redwood Creek PPZ sediment source inventory is presently in progress. This data reflects only the inventoried roads on the west side of Redwood Creek.
[2] The Little River sediment source inventory is presently in progress. The data reflects all inventoried sites entered in the Access database as of 1/08/2001.
[3] Does not include data for Little River assessment area.

Table F2-3. Analysis of inventoried watercourse crossings in the Plan Area with high and moderate treatment priorities.

Watershed name	Assessment area (mi^2)	Road length analyzed (mi)	High and moderate priority sites (#)			Future sediment delivery From watercourse crossings				Estimated Cost ($)[1]			Uncorrected cost effectiveness ($/$yds^3$)	Cost per site ($/site)
			#	#/mi	#/mi^2	# of sites	yds^3	yds^3/mi	yds^3/ mi^2	$	$/mi	$/$mi^2$		
Salmon Creek	6.8	36	183	5	27	153	43,472	1,208	6,393	677,454	18,818	99,626	15.58	4,428
Rowdy Creek	17.1	135	373	3	22	302	111,386	825	6,514	1,456,251	10,787	85,161	13.07	4,822
McGarvey Creek	7.0	63	383	6	55	195	110,115	1,748	15,731	1,249,891	19,840	178,556	11.35	6,410
Redwood Creek (PPZ)[2]	11.0	64	355	6	32	207	75,873	1,186	6,898	986,364	15,412	89,670	13.00	4,765
Little River[3]	35.0	220	1,533	7	44	939	248,390	1,129	7,097	`4,695,622	21,344	134,161	18.90	5,001
Total	76.9[4]	518	2,827	5.5	37[4]	1,796	589,236	1,138	7,662[4]	9,065,582	17,501	117,888[4]	15.38	5,048

[1] Costs include low boy transportation, heavy equipment, labor, materials, and supervision. Costs are listed as though both high and moderate priority sites are to be treated. In reality, especially on decommission roads, all sites are treated at once. Additional costs have been included for endhauling and the use of dump trucks at upgrade watercourse crossing sites. It was assumed that for crossings greater than 200 yds^3 approximately 60% of the total volume excavated will have to be endhauled from the site during culvert installation or replacement.

[2] The Redwood Creek PPZ sediment source inventory is presently in progress. This data reflects only the inventoried roads on the west side of Redwood Creek.

[3] The Little River sediment source inventory is presently in progress. The data reflects all inventoried sites entered in the Access database as of 1/08/2001.

[4] Does not include data for Little River assessment area.

Table F2-4. Analysis of inventoried landslides in the Plan Area with high and moderate treatment priorities.

Watershed name	Assessment area (mi^2)	Road length analyzed (mi)	High and moderate priority sites (#)			Future sediment delivery from landslides				Estimated Cost ($)[1]			Cost effectiveness ($\$/yds^3$)	Cost per site ($/site)
			#	#/mi	$\#/mi^2$	# of sites	yds^3	yds^3/mi	yds^3/mi^2	$	$/mi	$\$/mi^2$		
Salmon Creek	6.8	36	183	5	27	19	7,023	195	1,033	66,953	1,860	9,846	9.53	3,524
Rowdy Creek	17.1	135	373	3	22	60	8,906	66	521	56,933	422	3,329	6.39	948
McGarvey Creek	7.0	63	383	6	55	181	49,330	783	7,047	263,447	4,182	37,635	5.34	1,456
Redwood Creek (PPZ)[2]	11.0	64	355	6	32	98	48,807	763	4,437	339,331	5,302	30,848	6.95	3,463
Little River[3]	35.0	220	1,533	7	44	315	60,994	277	1,743	572,758	2,603	16,364	9.39	1,818
Total	76.9[4]	518	2,827	5.5	37[4]	673	175,060	338	2,276[4]	1,299,422	2,504	16,898[4]	7.42	1,931

[1] Costs include low boy transportation, heavy equipment, labor, materials, and supervision. Costs are listed as though both high and moderate priority sites are to be treated. In reality, especially on decommission roads, all sites are treated at once.
[2] The Redwood Creek PPZ sediment source inventory is presently in progress. This data reflects only the inventoried roads on the west side of Redwood Creek.
[3] The Little River sediment source inventory is presently in progress. The data reflects all inventoried sites entered in the Access database as of 1/08/2001.
[4] Does not include data for Little River assessment area.

Table F2-5. Analysis of inventoried "other" sites in the Plan Area with high and moderate treatment priorities.

Watershed name	Assessment area (mi^2)	Road length analyzed (mi)	High and moderate priority sites (#)			Future sediment delivery from "other" sites				Estimated Cost ($)[1]			Cost effectiveness ($/$yds^3$)	Cost per site ($/site)
			#	#/mi	#/mi^2	# of sites	yds^3	yds^3/mi	yds^3/mi^2	$	$/mi	$/$mi^2$		
Salmon Creek	6.8	36	183	5	27	11	364	10	54	5,445	151	801	14.96	495
Rowdy Creek	17.1	135	373	3	22	11	149	1	3	8,376	62	490	56.21	761
McGarvey Creek	7.0	63	383	6	55	7	84	1	12	5,177	82	740	61.63	740
Redwood Creek (PPZ)[2]	11.0	64	355	6	32	50	2,076	32	189	63,224	988	5,748	30.45	1,264
Little River[3]	35.0	220	1,533	7	44	279	6,454	29	184	403,104	1,832	11,517	62.46	1,403
Total	76.9[4]	518	2,827	5.5	37[4]	358	9,127	18	119[4]	485,326	937	6,311[4]	53.17	1,314

[1] Costs include low boy transportation, heavy equipment, labor, materials, and supervision. Costs are listed as though both high and moderate priority sites are to be treated. In reality, especially on decommission roads, all sites are treated at once.
[2] The Redwood Creek PPZ sediment source inventory is presently in progress. This data reflects only the inventoried roads on the west side of Redwood Creek.
[3] The Little River sediment source inventory is presently in progress. The data reflects all inventoried sites entered in the Access database as of 1/08/2001.
[4] Does not include data for Little River assessment area.

Requiring proposed work to meet pre-established cost-effectiveness criteria is critical to developing a defensible and objective watershed protection and restoration plan. The cost-effectiveness of treating a restoration work site is defined as the average amount of money spent to prevent one cubic yard of sediment from entering or being delivered to the stream system. The cost-effectiveness of treating each of the sediment sources in each of the five Green Diamond watersheds is listed in the summary data tables. Cost-effectiveness values average \$15/yd^3 for watercourse crossings, \$7.50/yd^3 for road-related landslides, and \$53/yd^3 for “other” sites. “Other” sites are often less cost-effectively treated because of their relatively small delivery volume.

F2.4 LIMITATIONS AND ASSUMPTIONS IN SEDIMENT DELIVERY AND TREATMENT COST ANALYSES

The sediment production and delivery figures developed for Green Diamond lands in the five sampled watersheds have been extended to the remainder of the ownership (see Appendix F3). It is assumed that the sediment delivery volumes developed for the five watersheds are reasonable estimates of future sediment delivery from existing roads in the absence of future treatments (such as road upgrading and decommissioning, as described in the Plan).

As would be expected with a forward-looking sediment source assessment, the predictive data generated from such a field inventory of road systems have certain inherent limitations and uncertainties. The resulting data also display variability that is derived from a number of sources. Finally, some assumptions have necessarily been employed to derive “reasonable” values for future erosion and sediment delivery.

Sources of variability or uncertainty in the estimates are described below. Data are presented for four subject areas: 1) general procedures, 2) inventory volumes, 3) sediment delivery volumes, and 4) estimated treatment costs. The sources of variability are generally outlined in Table F2-6. The effects of these findings are expressed in Table F2-2 or have been incorporated in the final sediment delivery estimates for the Plan Area (Appendix F3).

F2.4.1.1 Assumptions Employed in General Road Sediment Analysis

1. All sediment delivery numbers generated for and applied to the remainder of the Green Diamond ownership assume that the sample data from the detailed inventories in the five watersheds correctly represents Green Diamond properties and road conditions. The broad range of geologic types represented by the five watersheds lends support to this assumption. Additional field inventories to be conducted in the first five years after implementation of the Plan will be examined to confirm these assumptions and estimates.

Table F2-6. Accounting for variability in sediment delivery and work estimates.

No.	Source of variability or potential error	Result	Possible action, solution or accounting	Proposed Analysis	Results and Findings
1	Not all inventoried sites will erode or fail	Overestimate of delivery volume and work requirement	Develop a reducing factor which assumes some sites will not fail in the analysis period	Determine how many sites on abandoned roads have failed (frequency) since abandonment. Go to past inventories to determine failure frequency (#/mi) landslides). Use P-L 4 watershed data of past delivery. Determine past erosion on inventoried watercourse crossings	Landslide delivery frequency & failure rates for PL 4-basin inventory: Past frequency = 1.09 - 2.47 slides/mile Past delivery = 760 - 3,300 yds^3/mi Future = 180 - 1,410 yds^3/mi (estimate appears reasonable) 53% of crossings on abandoned roads show sediment delivery (currently overestimated frequency - see below).
2a	Not all sites of future sediment delivery have been identified	Underestimate of future sediment delivery volumes and work estimate	Develop an inflating factor which assumes some new sites will develop and deliver that were not previously identified	Determine how well future failure sites can be identified. With RX get close to 100%. With LS maybe 75%? Give a range and work estimates from that range.	Past frequency = 1.1 - 2.5 slides/mile Future frequency = 1.2 - 2.6 slides/mile (some slides don't fail; some slides aren't recognized - generally balances)
2b	More sites have been identified than will fail	Overestimate for future sediment delivery volumes	Develop a reducing factor which assumes that not all sites that were identified will actually fail and delivery sediment.	Based on experience and field evidence on inventoried roads, estimate what percent of the mapped sites actually fail.	Past LS frequency = 1.1 - 2.5 slides/mile Crossing failure (erosion) frequency on abandoned roads = 53%
3	Erosion from stream diversions not fully accounted for (crossing volume used as surrogate)	Underestimate of delivery volumes and cost-effectiveness calculation	Review volumetric data for all diversions and compare against crossing volumes to develop corrected sediment savings estimate	Review crossing data from 4 P-L watersheds (determine # w/Dp and # diverted and average yield); review RNP Professional Paper findings; review USFS Furniss data; Compare all delivery data to watercourse crossing volumes.	31% of crossings have DP; range = 24% - 81%; Delivery from PL diversions averages 75% of crossing volume (range = 29% -130%). USFS estimates (KNF) 2x - 3x sediment delivery from 1997 diversions; RNP yields up to 10x

Table F2-6 (Continued). Accounting for variability in sediment delivery and work estimates.

No.	Source of variability or potential error	Result	Possible action, solution or accounting	Proposed Analysis	Results and Findings
4	Not all watercourse crossings will completely erode	Overestimate of delivery volumes	Develop a reducing factor, based on drainage area	Look at eroded watercourse crossings on abandoned roads. Look for crossings over 50% eroded and define minimum drainage area; Look at upgrade data for distribution of watercourse crossing drainage areas	53% of crossings have past delivery 68 % are 1% - 25% eroded 16 % are 25% - 50% eroded 9 % are 50% - 75% eroded 7 % are 75% - 100% eroded
5	Watercourse crossing erosion assumes 1:1 side slopes	Under estimate of long term delivery volumes	Develop a range of delivery volumes based on 0.5:1, 1:1 to 1.5:1 side slopes	Develop a range of delivery volumes based on 0.5:1, 1:1 to 1.5:1 side slopes.	There is an average 35% reduction or increase in volumes
6	Road surface erosion and delivery not included in delivery volume estimates	Underestimate of delivery volumes; treatment costs already included in estimates	Connectivity is already known for most inventoried areas; delivery volumes could easily be estimated	Define average connectivity numbers for inventoried roads and apply average erosion volumes for watercourse crossings.	Average connectivity = 33%; Range = 6% - 74% (Little River); Total sediment delivery = 123% of site erosion; Range = 102% - 146%
7	GIS does not identify all roads that could contribute to sediment delivery	Underestimate of delivery volumes, costs and work requirements	Include an inflation factor for unmapped roads; data already exists for this	Look at GIS road densities and actual road densities for Green Diamond watersheds. Determine unmapped road density.	Actual road mileage is an estimated 110% to 125% of GIS road mileage (mean = 120%)
8	New and upgraded roads have smaller and fewer sites with lower risk of failure	Estimates are for older roads; over time, unit volumes will decrease as roads are treated	Acknowledge risk is still present and determine new volumes for treated roads	Look at upgraded roads and new roads for reduction in watercourse crossing volumes and risk of failure. Estimate reduced risk and reduced volumes (no diversions, smaller volumes and less frequent failure).	NA
9	New and upgraded roads are not hydrologically connected (connection is minimized)	Current delivery estimate does not include road surface erosion	Measure or estimate new connectivity and estimate delivery volumes	Determine new connectivity and sediment delivery for upgraded roads (Assume an average connectivity of 100 feet for upgraded roads)	Past connectivity = 33% Future connectivity = 7% (Based on 100 feet per crossing @ 3.5 crossings/mile - 32.5 yds^3/mile/decade))

Table F2-6 (Continued). Accounting for variability in sediment delivery and work estimates.

No.	Source of variability or potential error	Result	Possible action, solution or accounting	Proposed Analysis	Results and Findings
10	Unknown if property-wide road building rate is greater or less than road closure (decom) rate	Total volume of deliverable sediment could be increasing or decreasing	Could be easily analyzed and projected into the future based on known road management plans	Not relevant	NA
11	Poor or inaccurate inventory will dramatically affect costs and sediment saving estimates	Could increase or decrease costs; reduced sediment savings (increased discharge)	Use trained inventory crews; employ peer review procedures for erosion assessment and erosion prevention prescriptions	Apply multiplier estimate for low, medium and high expertise and accuracy	Estimated that inventory crews, if contracted or held as long term employees, will achieve proficiency. Initial inaccuracy may increase costs by 5% - 15% for 3 years. Inefficiency may be reduced through technical oversight.
12	Inexperienced operators will increase costs and reduce effectiveness (sediment savings)	Increased costs; reduced sediment savings	Employ only trained, experienced operators; Train operators specifically for road work	Apply multiplier estimate for low, medium and high operator expertise	Estimated that equipment operators, if contracted or held as long term employees, will achieve proficiency. Initial inaccuracy may increase costs over skilled crews by 15% - 35% for first 3 years. Inaccuracy may be largely eliminated through technical training and oversight.
13	“Fluff factor” not included in excavation or endhaul volumes	Will increase costs for endhauling somewhat	Build in inflation factor for volume increases during excavation	Assume a 20% expansion factor for endhauling. Determine how much of total treatment costs in each watershed are for endhauling and increase costs by 20%	Endhauling the extra material (volume accounted for in the expansion of compacted soil) is estimated to increase endhauling costs by 24% and total project costs by 2%.
14	Unit costs (and total costs) for work will increase over time	Less work is done for fixed dollar amounts	Build in inflation factor to annual expenditure levels for road work	Inflation factor will be worked into overall cost and production estimate (see Plan text). Could tie it to fuel prices and general inflation rate	Not calculated

2. It is assumed that there are 10% to 25% more roads (mean 15*%)* than are documented in the Green Diamond GIS (based on field mapping projects already undertaken on Green Diamond lands). Most of these roads are abandoned and overgrown. Road-related erosion and sediment delivery will need to be adjusted to account for this.

3. Road inventories on Pacific Lumber Company lands have been used in place of Green Diamond inventories to determine some erosion and delivery estimates (e.g., past landslide frequency (slides/mile)) because PWA inventories in Green Diamond watersheds do not contain systematic data on past erosion and sediment delivery volumes. Inventories of Green Diamond roads contain data only on future and on-going sediment sources and only describe sediment delivery from High and Moderate priority sites.

F2.4.1.2 Assumptions Employed in Developing Sediment Production (Erosion) Volumes

F2.4.1.2.1 Future Landslide Volumes

Field inventories on Green Diamond and other industrial properties indicate that past landslide frequencies (1.1 to 2.5 slides/mile) are similar to future (predicted) landslide frequencies (1.2 to 2.6 slides/mile) that have been mapped in the recent field inventories. This appears reasonable for roads that are becoming more “seasoned” through time and lends support to the overall field estimate for the magnitude of future sediment delivery that could be derived from road-related landslides. Future (predicted) landslide volumes were estimated based on comparable features which have already failed in the vicinity of potentially active slides, as well as the location and physical dimensions of the potential slide as inferred from scarps and cracks within the road bed or on the fill slope. In almost all cases, there had to be physical evidence of a potential failure (scarps, cracks, etc) before a road or landing fill was classified as a potential road-related failure. Not all these sites will fail, but similarly, a limited number of other sites that have not yet developed overt signs of potential failure may end up failing and delivering sediment to the stream system.

F2.4.1.2.2 Future Watercourse Crossing Erosion Volumes

Watercourse crossing fill volumes can be measured fairly accurately in the field by employing simple measurements and applying double end-area calculating formulas. Initially, watercourse crossing washout volumes (predicted erosion) were geometrically calculated by assuming the stream would eventually cut through the fill exposing a natural channel bottom width and typically exhuming 1:1 (100%) sideslopes through the fill. Thus, in Table F2-2 it was assumed that if a culvert “failed” during a large storm event, the watercourse crossing fill would completely washout. This may be a reasonable assumption for crossings of large streams, or when it was standard practice to abandon roads between harvest rotations and leave them unmaintained for 50 years or longer. However, this is no longer a standard practice, and it cannot be assumed that all under-designed watercourse crossings will completely fail if they are not upgraded or decommissioned.

To determine what a reasonable erosion volume might be, a number of abandoned crossings were inventoried and characterized. Crossings on abandoned roads were studied because crossings on maintained roads are quickly repaired after storm events and data on erosion is no longer available. For abandoned crossings with no diversion potential, data from 707 inventoried watercourse crossings indicates that 53% show significant erosion. Generally, the older the crossing, and the larger the stream, the more erosion it exhibits. Table F2-7 outlines the erosion data for watercourse crossings on roads which have been abandoned for 10 to 50 years.

Table F2-7. Measured erosion of watercourse crossings on abandoned roads in the Plan Area.

Crossings showing erosion[1] (% of total number)	Amount of erosion (% of entire fill crossing)
36.0	1% to 25%
8.5	25% to 50%
4.8	51% to 75%
3.7	75% to 100%
53.0	_ = 14%

[1] A total of 707 abandoned watercourse crossing (none with diversion potential) were analyzed. Watercourse crossings had been abandoned for 10 to 50 years.

Based on field inventories, a more reasonable assumption of the actual frequency and volume of watercourse crossing erosion during a given 50 year period (assuming no upgrading or decommissioning treatments are undertaken) is outlined in Tables F2-8 and F2-9.

Table F2-8. Predicted watercourse crossing erosion in the Plan Area for a 50 year time period.

Crossings showing erosion (% of total number)	Amount of erosion (% of entire fill crossing)
40 %	10%
30 %	30%
20 %	50%
10 %	90%
Average erosion	32%

Table F2-9. Analysis of inventoried watercourse crossings in Plan Area with high and moderate treatments priorities.

Watershed name	Assessment area (mi^2)	Road length analyzed (mi)	Potential future sediment delivery from high and moderate priority watercourse crossings					Future sediment delivery (yds^3) using three calculation methods		
			# of sites	yds^3	yds^3/mi	yds^3/mi^2	Unit delivery volume (yd^3/site)	Complete crossing washout (yd^3)	Expected delivery 40% erode 10% 30% erode 30% 20% erode 50% 10% erode 90%	Abandoned xings 36.0% erode 13% 8.5% erode 38% 4.8% erode 63% 3.7% erode 88%
Salmon Creek	6.8	36	153	43,472	1,208	6,393	284	43,472	13,905	6,166
Rowdy Creek	17.1	135	302	111,386	825	6,514	369	111,386	35,660	15,813
McGarvey Creek	7.0	63	195	110,115	1,748	15,731	565	110,115	35,256	15,634
Redwood Creek (PPZ)[2]	11.0	64	207	75,873	1,186	6,898	367	75,873	24,310	10,780
Little River[3]	35.0	220	939	248,390	1,129	7,097	265	248,390	79,627	35,310
Total	76.9[4]	518	1,796	589,236	1,137	7,662[4]	328	589,236	188,508	83,592

[1] Costs include low boy transportation, heavy equipment, labor, materials, and supervision. Costs are listed as though both high and moderate priority sites are to be treated. In reality, especially on decommission roads, all sites are treated at once. Additional costs have been included for endhauling and the use of dump trucks at upgrade watercourse crossing sites. It was assumed that for crossings greater than 200 yds^3 approximately 60% of the total volume excavated will have to be endhauled from the site during culvert installation or replacement.

[2] The Redwood Creek PPZ sediment source inventory is presently in progress. This data reflects only the inventoried roads on the west side of Redwood Creek.

[3] The Little River sediment source inventory is presently in progress. The data reflects all inventoried sites entered in the Access database as of 1/08/2001.

[4] Does not include data for Little River assessment area.

The prediction of future watercourse crossing erosion on Green Diamond lands is based largely on a calculation of erodible fill volumes and an analysis of past erosion and delivery volumes from watercourse crossings on roads that have been abandoned for 10 to 50 years. Other than some data collected after singular flood events in northern California and Oregon, this is the best long term data set that is available for watercourse crossing erosion.

F2.4.1.2.3 *<u>Average Erosion</u>*

The watercourse crossing erosion data for abandoned roads is not unlike those that have been collected after a single large storm event (Figure 1). Furniss (2000) reported that hydraulic exceedence was not a major failure mechanism for watercourse crossings in large floods. Calculated peak flow and culvert capacity did not predict watercourse crossing failure where sediment and woody debris were the ultimate cause of failure and subsequent erosion.

It was thought that there would be a relationship between the degree of watercourse crossing erosion (washout) and the drainage area above the crossing (discharge), especially for the 53% of Green Diamond watercourse crossing fills that have already experienced some erosion. However, the observed relationship is weak and by itself, drainage area was not a good predictor of observed watercourse crossing erosion volumes.

Several other factors were considered in the evaluation of predicted sediment delivery from eroded watercourse crossings.

When watercourse crossings erode from overtopping, they typically develop head cuts and gullies across the road prism. Field observations suggest most gullies develop 1:1 side slopes. Initially some gullies will have steeper sides, and over time others (especially those in poorly consolidated, non-cohesive soils) will lay themselves back to a gentler angle. To account for the potential variability in watercourse crossing erosion volumes caused by variable side slope morphology, PWA employed a range of sideslope steepness values from 0.5:1 to 1.5 :1. This resulted in a potential ±35% range for watercourse crossing erosion volumes where gullying develops.

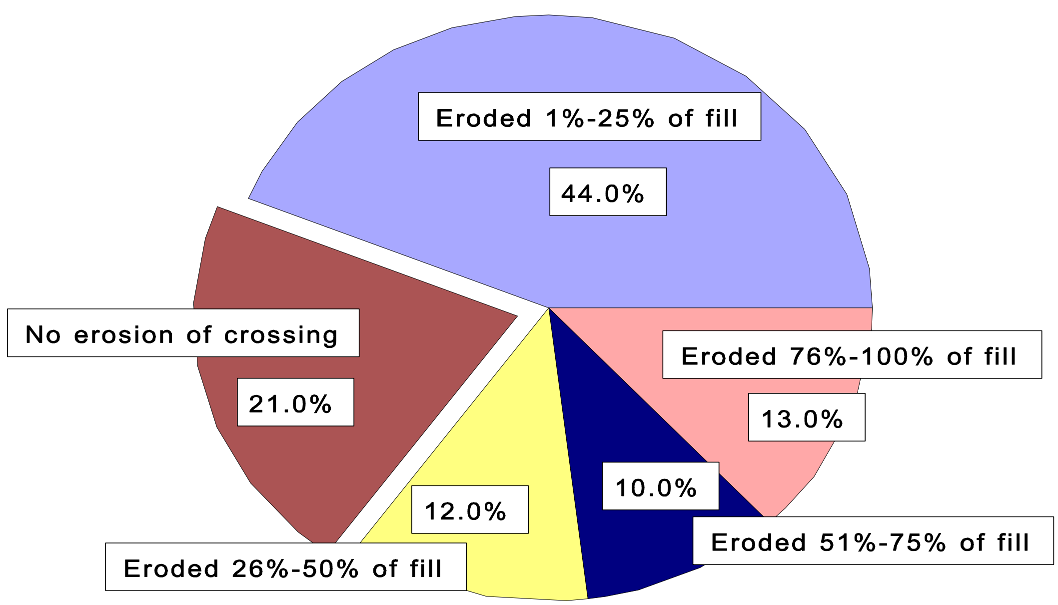
Erosion volumes calculated for watercourse crossing failures are “compacted” volumes. When excavation treatments (especially for decommissioning) are calculated, an expansion factor of 20% has been applied to these numbers. This expansion volume is not considered in developing estimates of future erosion volumes, only in developing cost estimates for heavy equipment treatments where soil is to be excavated and hauled in dump trucks.

F2.4.1.2.4 *<u>Future Erosion Volumes from “Other” Sediment Sources</u>*

“Other” sources of road-related erosion typically involve gullying at the outlets of ditch relief culverts and other road surface drainage structures. The calculation and estimation of future sediment delivery volumes from these sediment sources is largely a process of estimating the potential for continued enlargement of the existing gullies which remain active or appear to have the potential to enlarge.

Figure 1. Stream crossing erosion from single storm overtopping

Proportion of crossing eroded when overtopped



(Furniss, 2000)

Figure F2-1. Watercourse crossing erosion from a single storm overtopping.

F2.4.1.3 Assumptions Employed in Developing Sediment Delivery Volumes

It should be clearly stated that this analysis of road erosion in the five Green Diamond watersheds does not include an assessment of fine sediment contributions from road surface erosion. Only “site” data has been included. Volumetrically and ecologically, over the course of one or more decades of road use and log hauling, this sediment source can be a highly important source of impact to the aquatic system. Importantly, the treatments (and the resultant cost tables), have been developed under the assumption the road surface drainage is “disconnected” from the natural drainage network, to the extent that is feasible. Thus, although the fine sediment erosion volumes are not included in the analysis, the treatments required to eliminate chronic sediment delivery from the road systems have been included in the final cost tables.

F2.4.1.3.1 Future Landslide Delivery

Field inventories on Green Diamond and other industrial properties indicate that past landslide frequencies (1.1 to 2.5 slides/mile) are similar to future (predicted) landslide frequencies (1.2 to 2.6 slides/mile) that have been mapped in the recent field inventories, but that future (predicted) landslide delivery volumes (180 to 1,410 yd^3/mile) are 25% to 40% of past volumes (760 to 3,300 yd^3/mile). Future delivery volumes are estimated in the field based on physical measurements of potentially unstable fill materials (typically bounded by scarps and/or cracks) and sediment delivery rates. Sediment delivery rates (% of the slide mass that would be delivered to a stream if the fillslope failed) were estimated in the field by applying a reasonable delivery percentage that considers what other nearby slides have done, as well as specific site characteristics that typically influence slide run-out distances (e.g., slope gradient, distance to stream, slope shape, moisture, etc.).

A second method (analysis of sequential air photos) has been employed to determine road-related mass wasting and sediment delivery from the Green Diamond road network (Appendix F1). Air photo analysis is good at identifying moderate and large size features that break the forest canopy and deliver sediment to streams. Small slide features that cannot be seen on aerial photos are less likely to deliver substantial volumes of sediment to streams, but their potentially high frequency may still make them important to the aquatic system.

In three watersheds of the lower Eel River where there is good data on past mass wasting using both air photo analysis and field inventories, there was an additional 6% to 38% sub-canopy sediment delivery (average increase = 15%) from small features that could not be seen in the 1:12,000 aerial photos. The number of landslides in these project areas increased by 75% when the field inventory data was added to the air photo analysis, but the delivery volumes increased by only 15% (on average). Clearly, field inventories of road erosion pick up many smaller road-related landslides that do not show up on air photos. This suggests that if air photo analysis of past landsliding is used to estimate future sediment delivery from landsliding, landslide delivery volumes should be increased by 10% to 30% (average 15%) over the photographically-derived rate.

F2.4.1.3.2 *Future Sediment Delivery from Watercourse Crossings*

It has been assumed that 100% of all sediment that is eroded from a watercourse crossing is delivered to the stream network. It is further assumed that field inventories will identify all watercourse crossings and that no significant crossings will be overlooked in the inventory process. Based on past experience, these are valid assumptions.

F2.4.1.3.3 *Future Sediment Delivery from "Other" Sites*

In the analysis of sediment delivery from "other" sites, it has been assumed that 60% to 100% of the eroded sediment (mean = 75%) is delivered to the stream system. Most of the "other" sites consist of gullies that are well connected and integrated with the natural stream channel network. In general, connected gullies are very efficient at delivering eroded sediment.

F2.4.1.4 Assumptions Employed in Developing Erosion Prevention Treatment Costs

F2.4.1.4.1 *Covered Costs*

Costs for implementing erosion prevention work (road upgrading and road decommissioning) incorporate all relevant expenses, including equipment, labor and materials as well as technical oversight, monitoring and reporting. Costs for treatments in each of the five watersheds includes equipment mobilization (moving) costs, road opening costs (especially for overgrown roads), heavy equipment costs for treating sites and for addressing road drainage, endhauling costs, laborer costs for culvert installations, mulching and seeding, rock costs, culvert materials (including couplers and downspouts), planting and mulching materials, and professional costs for treatment layout, equipment oversight, supervision, documentation and reporting.

The costs that are summarized in Tables F2-3, F2-4 and F2-5 were developed from the detailed cost analyses for each road and each site in the five watershed erosion assessments, employing the assumptions listed above. The costs are based on competitive equipment rental and labor rates for the watershed areas. Based on recent road upgrading work, it has also been assumed that watercourse crossings exceeding 200 yd^3 in volume will require that 60% of the crossing volume be endhauled (because it is too wet to reuse) during the rebuilding process. The cost tables have been reworked to account for this added work effort.

F2.4.1.4.2 *Costs not Covered*

As the cost tables were developed for the five Green Diamond watersheds, and as experience in implementing road upgrading and road decommissioning has increased, additional cost categories have been added to better reflect actual on-the-ground expenses. It has become apparent that volume calculations which are based on in-place geometric shapes of fills (e.g., watercourse crossing fills) need to be increased to account for the expansion of the soil materials as they are excavated and loaded into trucks. Green Diamond has estimated that the increase in volume due to fluffing or expansion of excavated material will increase overall project costs by 2% over that which is stated in the cost tables. This increased cost is largely the consequence of increased endhauling requirements (these cost are added in Table F2-10).

Table F2-10. PWA treatment costs, as itemized and adjusted from Tables F2-3, F2-4, and F2-5.

Category Range	Watercourse crossings ($/mi)	Landslides ($/mi)	"Other" ($/mi)	Cost ($/mi)	Other costs (multiplier)	Total costs ($/mi)
Average	17,500	2,504	940	20,940	0.2	25,000
Minimum	15,000	420	60	15,480	0.2	18,000
Maximum	21,000	5,300	1,800	28,100	0.2	40,000

F2.4.1.4.3 *Additional Undefined Cost Variables*

Several cost elements cannot easily be estimated. These include: 1) operator experience and skill, and 2) the skill and experience of the road erosion inventory crews that ultimately identify problems and define treatment prescriptions. The data contained in the summary cost tables (Tables F2-3, F2-4 and F2-5)) assume that the inventory crews and the equipment operators are skilled, accurate and efficient in their work.

Technically and practically well trained inventory crews can have a large effect on the overall cost-effectiveness of the erosion prevention work that is undertaken. Poor problem identification or quantification can result in inaccurate or misguided prescriptions that either under or over estimate to scope of the necessary work. In addition, problems which are "missed" or mis-identified may end up resulting in environmental damage if necessary work is not correctly prescribed and undertaken. Similarly, well trained and experienced operators can save thousands of dollars in how they approach and conduct the prescribed work. A poor operator can doom a project to being significantly over budget.

As a result, it is anticipated that for the first three years of the road implementation program on Green Diamond lands, inventory crews and equipment operators will be training and improving in their skills and efficiency. As a result, equipment costs could be as much as 15% to 35% higher than listed in the data tables. Increased program costs associated with untrained inventory crews could similarly add up to 5% to 15% additional implementation costs. It should be noted that no estimates have been included in the cost tables to cover the actual erosion inventories of Green Diamond roads. Listed costs are only for the implementation of prescribed treatments (usually road upgrading and road decommissioning) as derived from the five sampled watersheds. Most of these increased costs could be eliminated by implementing an organized training and technical oversight program for quality assurance and quality control covering at least the first three years of the program.

The sediment data for the 76.9 mi^2 assessment area on Green Diamond property is summarized in Table F2-11. Sediment delivery from watercourse crossing erosion is expressed both as an uncorrected volume (assuming complete washout of untreated crossings at sometime during the term of the Plan) and as a corrected erosion and delivery volume. The "corrected" erosion volume assumes that watercourse crossings erode at frequencies and in proportion to the observed erosion characteristics listed in Table F2-9. In this manner, 50-year erosion and delivery volumes for untreated, under designed watercourse crossings would equal approximately 32% of the fill volume, on average.

Total (corrected) sediment delivery from the three main sediment sources is nearly equally divided between watercourse crossings and road-related landslides (~350 yd^3/mile) with only 3% (on average) attributable to “other” sediment sources (mostly gullies at ditch relief culverts). A range of potential sediment delivery volumes has also been developed based on the field inventory data (Tables F2-3, -4, and -5).

Average treatment costs for erosion prevention work, principally road upgrading and road decommissioning, is summarized in Table F2-10. Unit treatment costs are broken down by site type (crossing, landslide and “other”) and then summed as a single unit cost ($/mi). These have then been adjusted to account for the 2% increase in costs expected to result from additional endhauling where soil “expands” (or fluffs) during excavation. The range in treatment costs ($18,000 to $40,000/mile) assumes that operators are well trained and experienced in all implementation measures. These figures are in line with actual road upgrading and decommissioning costs encountered in recent erosion prevention projects.

Table F2-11. Summary data for inventoried erosion and sediment delivery volumes for 5 watersheds covering 76.9 mi^2.

Sediment Source	Sample size (number of sites of future sediment delivery, inventoried	Average potential sediment delivery (uncorrected assumes complete washout and failure) (yds^3/mi)	Range of potential sediment delivery volumes (among 5 inventoried watersheds) (yds^3/mi)	
			Low	High
Watercourse Crossings (uncorrected)	1,796	1,140	825	1,750
Watercourse Crossings (corrected)	1,796	364	264	560
Landslides	673	340	65	780
“Other”	358	20	0	30
Total site data (corrected)	**2,827**	**724**	**329**	**1,370**

F2.5 SUMMARY

Pacific Watershed Associates (PWA) conducted sediment source inventories in five watersheds on Green Diamond’s ownership. The inventories were designed to quantify the potential future sediment delivery associated with road-related landslides, watercourse crossing failures and “other” sites associated with Green Diamond’s road network. The results from these inventories for high and moderate priority treatment sites are shown in Table F2-2.

PWA also assessed the cost required to stabilize the potential sediment associated with these sites (Table F2-3). Although the summary data tables do not include potential sediment derived from road-related surface erosion, the costs outlined in Tables F2-3, F2-4 and F2-5 do include monies to address such sources of sediment. That is, although the sediment delivery from road surface erosion has not been quantitatively described in the previous inventory data tables, the treatment costs to address these sediment sources have been included in the cost tables. Thus, Green Diamond's Road Implementation Plan has this additional important benefit to the species covered by the Plan.

The PWA sediment inventory data were used extensively in the development of the sediment production model that is discussed in Appendix F3. The data were particularly helpful in developing sediment delivery estimates over the 50-year life of the Plan. A rather key result, based on PWA's investigations, is that much of the potential sediment associated with watercourse crossings may not deliver within the next 50 years even if left untreated (Table F2-9). The PWA data were also used to estimate the magnitude of the potential sediment issues associated with Green Diamond's road network which led to the development of an appropriate strategy to accelerate erosion control and erosion prevention efforts over the first 15 years of the Plan.

Appendix F3. Plan Area Sediment Delivery Estimates: A Model and Results

CONTENTS

Figures

Tables

F3.1 INTRODUCTION

A sediment delivery model was developed to:

- Consolidate information from the landslide assessment (Appendix F1) and road sediment source inventory (Appendix F2);
- Combine the findings from the above mentioned studies to produce an approximate sediment delivery estimate for the Plan Area;
- Compare sediment delivery for the "No Plan" versus Plan scenarios;
- Evaluate the statistical efficiency and effectiveness of the various conservation measures; and
- Assess the variation in sediment delivery due to the "uncertainty" or "ranges" associated with key assumption variables using Monte Carlo simulation techniques;

F3.2 A CONCEPTUAL SEDIMENT DELIVERY MODEL

A simple conceptual model was developed to integrate the various sources of data and to produce a partial sediment summary for the Plan Area (see Figure F3-1 below). The model does not include all sources of sediment. It only attempts to model the sediment produced from shallow and deep-seated landslides (see Appendix F1) and high and moderate priority sites associated with roads (see Appendix F2). These are (1) sources of sediment not directly addressed by the implementation of best management practices (BMPs), (2) sources of sediment that were studied in sufficient detail such that empirical models could be constructed, and (3) potential sediments that could be effectively addressed by the conservation measures proposed pursuant to this Plan to mitigate the impacts of the covered activities.

The sources of sediment not directly addressed in this simple model include sediment produced from surface erosion and sediment produced from stream bank erosion. It should be noted, however, that the Road Implementation Plan includes measures to address and correct potential surface erosion associated with high and moderate priority treatment sites. Thus, this potentially prolific source of fine sediment will be treated and its impacts to aquatic species largely eliminated by the end of the 50-year term of the Plan.

This simple property-wide model is based on expected 50-year (long-term) average sediment delivery rates. (The model was developed to assess property-wide sediment delivery issues. The model does not have a spatial component and, therefore, is not able to make site-specific sediment delivery predictions.) It is recognized that the annual variation in such rates may be large and lead to annual sediment delivery amounts that are much greater or much smaller than the averages contained within this model. A model that accounts for such variation would have been unwieldy (if not impossible) to construct and problematic to parameterize given the nature of the sediment delivery studies described in Appendices F1 and F2.

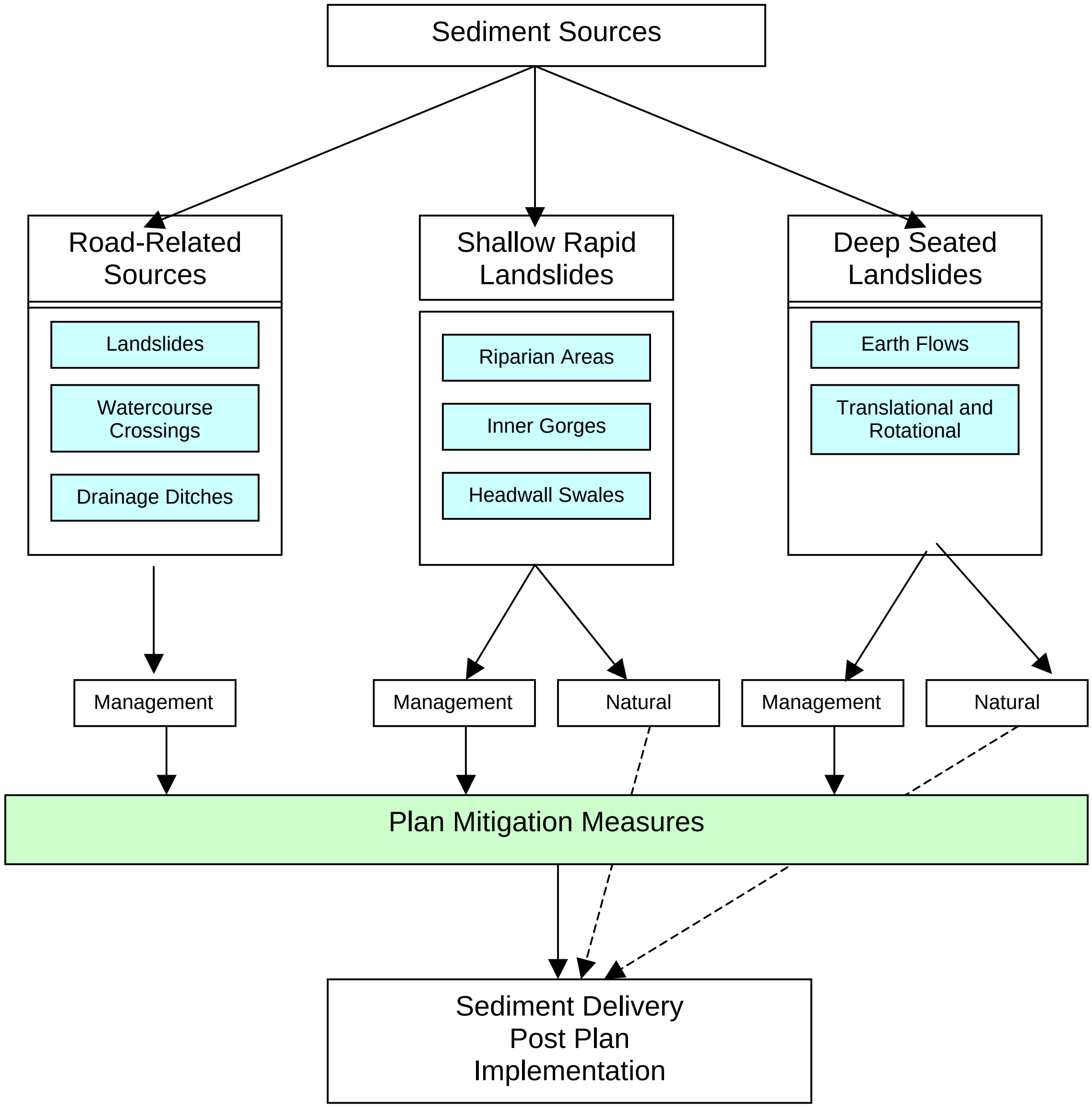


Figure F3- 1. Conceptual model of integration of data for partial sediment summary for Plan Area.

In any event, even if such a model could be constructed, its computed 50-year averages would be comparable to the output generated by the simple model described herein. Thus, the management options and conservation measures that evolve from the use of the model described in this appendix are entirely appropriate provided they are implemented over the 50-year term of the Plan to produce the desired results.

This conceptual model was used as the basis for developing a spreadsheet model that integrated the various data sets compiled for the Plan.

F3.3 ROAD-RELATED SEDIMENT SOURCE DATA

The road sediment source inventory conducted by PWA covered five watersheds: Salmon Creek, Rowdy Creek, McGarvey Creek, Redwood Creek, and Little River. The following table (Table F3-1) shows how the information from these watersheds (see Appendix F2 for watershed specific details) was combined to produce estimates for the Plan Area. The basic idea was to use an estimate of Plan Area road length (4,311 miles) as a multiplier to produce potential sediment totals for the Plan Area. For example, the current GIS estimate of road miles in the Plan Area is 4,311. Plan Area potential sediment from road-related landslides would be determined as follows:

$$1{,}456{,}862\ \text{yd}^3 = 4{,}311\ \text{miles} \times 338\ \text{yd}^3/\text{mile}$$

(Note: The spreadsheet model carries many digits beyond the decimal point so the math may not appear to “work out” properly in the equation above or the table below.) Only potential sediment from high and moderate treatment priority sites is used in the analysis, as it is these sites that are targeted for repair under the Road Implementation Plan.

As part of the sediment inventory, PWA provided Green Diamond with treatment costs (Table F3-2) that were used as the basis to determine the amount of sediment that could be stabilized using $2.5 million as specified under the Road Implementation Plan—approximately 204,000 cubic yards. An important consideration in this calculation is the efficiency that is realized by appropriately prioritizing the work and focusing on concentrations of high and moderate priority treatment sites. Such prioritization will allow Green Diamond to stabilize approximately 48% of the potential sediment during the first 15 years of the Plan with the $2.5 million annual commitment.

Several of the variables associated with the road sediment source inventory were assigned an appropriate range for purposes of conducting the Monte Carlo simulation exercise. These variables and their ranges are listed below in the VARIABLE RANGES section of this appendix. An example is the range associated with the miles of road contained within the Plan Area. Green Diamond recognizes that some roads have not been mapped and are not contained in Green Diamond’s GIS. To account for this understatement of Plan Area road miles, an assumption called the “road miles blow up factor” was devised. This factor was assigned a triangular distribution with a minimum increase of 10%, a most likely increase of 15%, and a maximum increase of 25%. The mean of this distribution, 16.7%, was used in the calculations to produce Tables F3-1 and F3-2. That is,

$$4{,}311\ \text{miles} = 116.7\% \times 3{,}695\ \text{miles},$$

where 3,695 miles is the length of roads according to Green Diamond’s GIS.

Table F3-1 Potential road-related sediment delivery from high and moderate treatment priority sites.[1]

	Road Length (mi)	Potential Sediment Delivery From Watercourse Crossings		Potential Sediment Delivery From Landslides		Potential Sediment Delivery From "Other sites"		Total Potential Sediment Delivery	
		yd^3	yd^3/mi	yd^3	yd^3/mi	yd^3	yd^3/mi	yd^3	yd^3/mi
Inventory Total from Five Watersheds	518	589,236	1,138	175,060	338	9,127	18	773,423	1,493
Estimate for the Plan Area	4,311	4,903,664	1,138	1,456,862	338	75,956	18	6,436,482	1,493

1 The inventory totals were extracted from Table F2-2 in Appendix F2. The Plan Area sediment delivery estimates are based on the inventoried rates (cubic yards per mile) multiplied by an estimate of the total miles of roads within the Plan Area.

Table F3-2. Calculation of the sediment stabilization effort for the Plan Area.[1]

	Watercourse Crossings	Landslides	Other	Total
Total sediment (yd^3)	4,903,664	1,456,862	75,956	6,436,482
Cost/yd^3	$15.69	$7.57	$54.24	$14.31
Total cost	$76,938,495	$11,028,445	$4,119,829	$92,086,769
48% of total sediment	2,329,708	692,148	36,086	3,057,943
Cost/yd^3	$13.45	$6.49	$46.49	$12.26
41% of total cost	$31,331,250	$4,491,054	$1,677,696	$37,500,000
Sediment stabilization effort (yd^3)	155,314	46,143	2,406	203,863
Cost/yd^3	$13.45	$6.49	$46.49	$12.26
Annual cost	$2,088,750	$299,404	$111,846	$2,500,000

1 The cost per cubic yard figures in this table is slightly larger than those shown Table F2-3. These cost adjustments were made to account for an underestimate in the basic data as described in Table F2-6.

Other road-related assumption variables that were assigned distributions (see Table F3-13) include:

- Delivery from road-related landslides
- Delivery from road-related watercourse crossings
- Delivery from road-related “other” sites
- Cost to fix watercourse crossing sites
- Cost to fix landslide sites
- Cost to fix “other” sites
- Road upgrade effectiveness factor

F3.4 WATERSHED SEDIMENT SUMMARIES AND PLAN AREA SEDIMENT DELIVERY ESTIMATES

Sediment delivery summaries for the Hunter Creek, Salmon Creek, Litter River, and Upper Mad River pilot watersheds are shown in Tables F3-3, F3-4, F3-5, and F3-6, respectively. These tables are based on the results of an assessment of long-term landslide sediment presented in Appendix F1. The sediment delivery summaries show how sediment is partitioned among three sources of sediment—roads, shallow landslides, and deep-seated landslides—contained in the conceptual model. (Note: The Upper Mad River watershed summary only shows sediment delivery estimates for deep-seated landslides.) The purpose of this section is to explain how these data were combined to derive appropriate sediment delivery estimates for the Plan Area.

Tables F3-3, F3-4, F3-5, and F3-6 are largely restatements of results presented in Appendix F1 (see Tables 15, 16, and 17) in a format that conveniently summarizes the modeled sources of sediment delivery and shows the reduction in sediment delivery that is expected to occur as a result of implementing the Plan's conservation measures. The road-related sediment delivery estimates, as discussed in detail below, are based on data presented in Appendices F1 and F2.

The data from these four pilot watersheds were combined to derive sediment delivery estimates for the Plan Area. This was accomplished by developing factors (or weights) that represent how much of the Plan Area is similar to each of the pilot watersheds. Such Plan Area factors were developed by examining the landslide processes acting within each of the unstudied sub-watersheds based on a review of terrain maps, geologic maps, available landslide maps, discussions with Green Diamond foresters, and observations made by a Registered Geologist during a year 2000 helicopter flyover of the Green Diamond property. The percentages of each pilot watershed were then assigned to each sub-watershed based on the criteria listed above. The results of this Delphi technique exercise are summarized in Table F3-7. The last row of Table F3-7 shows the Plan Area factors. This row was determined by multiplying the sub-watershed acreages by the pilot watershed percentages and then summing the results. Note that there are separate factors for shallow landslides and deep-seated landslides.

To illustrate the use of the Plan Area factors in Table F3-7 (see the last row of the table), consider the calculation of the expected sediment delivery that will come from Plan Area RMZs prior to implementation of the Plan (Pre-Plan estimates). To do this, the data from these three representative watersheds will be combined to develop an estimate for 394,675 timberland acres. From Tables F3-3, F3-4, and F3-5, the sediment delivery estimates for RMZ areas are 235 yd^3/yr, 798 yd^3/yr, and 768 yd^3/yr for the Hunter Creek, Salmon Creek, and Little River watersheds, respectively. The total acres within each of these watersheds, also shown in the tables, are 10,126 acres, 7,889 acres, and 28,755 for the Hunter Creek, Salmon Creek, and Little River watersheds, respectively. The appropriate equation, therefore, is

$$\begin{aligned} 13{,}200\ yd^3/yr = 394{,}675\ \text{acres} * [&0.312*(235\ yd^3/yr \div 10{,}126\ \text{acres}) \\ &+ 0.105*(798\ yd^3/yr \div 7{,}889\ \text{acres}) \\ &+ 0.583*(768\ yd^3/yr \div 28{,}755\ \text{acres})] \end{aligned}$$

Table F3-3. Hunter Creek sediment delivery summary. SMZ buffer widths are based on a cumulative sediment delivery volume of 80%. The sediment numbers in the table represent the total annual sediment delivery expected from the watershed. Note that natural and management related sediment delivery estimates are provided for both the pre- and post-Plan conditions.

	Sediment Split (roads vs. harvest)	Sediment Split	Mgt vs. Natural Sediment Under Current Practices	Effect of Plan Measures	Percent Acres in Zone	Acres in Zone	Pre-Plan			Post-Plan		
							Sediment Delivery (cu yds/yr)	Natural Sediment (cu yds/yr)	Mgt Sediment (cu yds/yr)	Sediment Delivery (cu yds/yr)	Natural Sediment (cu yds/yr)	Mgt Sediment (cu yds/yr)
Roads	54.8%	100.0%	100.0%	96.1%			4,465	0	4,465	173	0	173
Hillslope Shallow Landslides (extracted from Tables 15 and 16 in Appendix F1)	42.7%											
RMZs		6.8%	19.0%	95.4%	14.7%	1,489	235	191	45	193	191	2
SMZs		20.1%	50.0%	58.6%	3.5%	356	697	349	349	493	349	144
SHALSTAB		34.2%	60.0%	60.0%	13.1%	1,324	1,190	476	714	762	476	286
Other		39.0%	50.0%	0.0%	65.4%	6,621	1,355	677	677	1,355	677	677
Deep Seated Landslides (extracted from Table 17 in Appendix F1)	2.6%											
DSL Total		100.0%	2.6%	15.0%	3.3%	338	210	205	5	209	205	5
Total Sediment Delivery (Note that Total Acres is shown in one column)						10,126	8,153	1,898	6,255	3,184	1,898	1,287

Table F3-4. Salmon Creek sediment delivery summary. SMZ buffer widths are based on a cumulative sediment delivery volume of 60%. The sediment numbers in the table represent the total annual sediment delivery expected from the watershed. Note that natural and management related sediment delivery estimates are provided for both the pre- and post-Plan conditions.

	Sediment Split (roads vs. harvest)	Sediment Split	Mgt vs. Natural Sediment Under Current Practices	Effect of Plan Measures	Percent Acres in Zone	Acres in Zone	Pre-Plan Sediment Delivery (cu yds/yr)	Pre-Plan Natural Sediment (cu yds/yr)	Pre-Plan Mgt Sediment (cu yds/yr)	Post-Plan Sediment Delivery (cu yds/yr)	Post-Plan Natural Sediment (cu yds/yr)	Post-Plan Mgt Sediment (cu yds/yr)
Roads	23.6%	100.0%	100.0%	96.1%			842	0	842	33	0	33
Hillslope Shallow Landslides (extracted from Tables 15 and 16 in Appendix F1)	55.5%											
RMZs		40.2%	23.8%	99.8%	8.8%	698	798	608	190	608	608	0
SMZs		0.1%	50.0%	60.0%	0.3%	21	2	1	1	1	1	0
SHALSTAB		13.5%	60.0%	60.0%	3.0%	234	268	107	161	172	107	64
Other		46.2%	50.0%	0.0%	54.2%	4,279	916	458	458	916	458	458
Deep Seated Landslides (extracted from Table 17 in Appendix F1)	20.9%											
DSL Total		100.0%	5.6%	15.0%	33.7%	2,657	748	706	42	741	706	35
Total Sediment Delivery (Note that Total Acres is shown in one column)						7,889	3,574	1,880	1,693	2,471	1,880	591

Table F3-5. Little River sediment delivery summary. SMZ buffer widths are based on a cumulative sediment delivery volume of 60%. The sediment numbers in the table represent the total annual sediment delivery expected from the watershed. Note that natural and management related sediment delivery estimates are provided for both the pre- and post-Plan conditions.

	Sediment Split (roads vs. harvest)	Sediment Split	Mgt vs. Natural Sediment Under Current Practices	Effect of Plan Measures	Percent Acres in Zone	Acres in Zone	Pre-Plan			Post-Plan		
							Sediment Delivery (cu yds/yr)	Natural Sediment (cu yds/yr)	Mgt Sediment (cu yds/yr)	Sediment Delivery (cu yds/yr)	Natural Sediment (cu yds/yr)	Mgt Sediment (cu yds/yr)
Roads	40.4%	100.0%	100.0%	96.1%			2,377	0	2,377	92	0	92
Hillslope Shallow Landslides (extracted from Tables 15 and 16 in Appendix F1)	29.4%											
RMZs		44.3%	23.1%	99.4%	13.3%	3,815	768	590	177	592	590	1
SMZs		1.8%	50.0%	60.0%	0.3%	74	31	16	16	22	16	6
SHALSTAB		11.2%	60.0%	60.0%	2.5%	725	195	78	117	125	78	47
Other		42.7%	50.0%	0.0%	65.5%	18,830	740	370	370	740	370	370
Deep Seated Landslides (extracted from Table 17)	30.2%											
DSL Total		100.0%	3.2%	15.0%	18.5%	5,311	1,779	1,722	56	1,770	1,722	48
Total Sediment Delivery (Note that Total Acres is shown in one column)						28,755	5,889	2,776	3,113	3,340	2,776	564

Table F3-6. Upper Mad River sediment delivery summary. The sediment numbers in the table represent the total annual sediment delivery expected from the watershed. Note that natural and management related sediment delivery estimates are provided for both the pre- and post-Plan conditions. This is a "partial" summary because only sediment from deep seated landslides is included in the table.

	Sediment Split (roads vs. harvest)	Sediment Split	Mgt vs. Natural Sediment Under Current Practices	Effect of Plan Measures	Percent Acres in Zone	Acres in Zone	Pre-Plan			Post-Plan		
							Sediment Delivery (cu yds/yr)	Natural Sediment (cu yds/yr)	Mgt Sediment (cu yds/yr)	Sediment Delivery (cu yds/yr)	Natural Sediment (cu yds/yr)	Mgt Sediment (cu yds/yr)
Roads												
Hillslope Shallow Landslides												
RMZs												
SMZs												
SHALSTAB												
Non-Protected Areas												
Deep Seated Landslides (extracted from Table 17 in Appendix F1)	100.0%											
DSL Total		100.0%	14.9%	15.0%	41.2%	1,918	902	767	135	882	767	115
Partial Sediment Delivery (Note that Total Acres is shown in one column)						4,658	902	767	135	882	767	115

Table F3-7. Factors used to derive Plan Area sediment delivery estimates from the four pilot watersheds. The factors in this table represent that portion of the Plan Area that can be adequately characterized.

Road Planning Watershed	Acres	HPA Group	Shallow Landslide Division			Deep-Seated Landslide Division			
			SC	LR	HC	SC	LR	HC	MR
South Fork Winchuck	7,859	SR	50%	50%		100%			
Dominie	4,024	SR	50%	50%		100%			
Rowdy	8,342	SR	50%	50%		100%			
Little Mill	4,888	SR	50%	50%		100%			
Wilson	6,370	CKLM		50%	50%			100%	
Goose	10,250	CKLM			100%			100%	
Hunter	11,656	CKLM			100%			100%	
Terwer	21,592	CKLM			100%			100%	
Hoppaw	5,172	CKLM		100%				100%	
Waukell	2,815	CKLM		100%				100%	
McGarvey	4,867	CKLM		100%				100%	
Omagar	5,903	CKLM		50%	50%			100%	
Ah Pah	10,037	CKLM		50%	50%			100%	
Bear	6,199	CKLM		50%	50%			100%	
Surper	6,493	CKLM		50%	50%			100%	
Tectah	12,385	CKLM		25%	75%		25%	75%	
West Fork Blue	5,634	CKLM			100%			100%	
Blue	9,760	CKLM		50%	50%		75%		25%
Pecwan	15,692	KOR		50%	50%		75%		25%
Mettah	9,077	KOR		25%	75%		25%	75%	
Joe Marine	8,105	KOR		50%	50%		75%		25%
Roach	19,847	KOR		25%	75%		25%	75%	
Tully	12,727	KOR		25%	75%		25%	75%	
Panther	9,689	KOR		100%			75%		25%
Dolly Varden	13,543	KOR		100%			75%		25%
Noisy	9,719	KOR		100%			75%		25%
McDonald	2,040	KOR		100%			100%		
NF Maple	12,154	KOR		100%			100%		
Maple	18,236	KOR		100%			100%		
Coastal Tribs	7,756	KOR		100%			100%		
North Little River	6,846	KOR		100%			100%		
East Little River	7,658	KOR		100%			100%		
South Little River	11,535	KOR		100%			100%		
Lindsay	8,740	KOR		100%			100%		
Dry	9,487	KOR		50%	50%			100%	
Canon	13,566	KOR		100%			100%		
Basin	5,341	KOR		100%			100%		
Long Prairie	17,435	KOR		100%			100%		
Gosinta	5,418	KOR		100%			100%		
Boulder	17,711	KOR	50%	50%					100%
Jacoby	3,608	KOR		100%			100%		
Salmon	6,258	HUM	100%			100%			
Ryan	7,702	HUM	100%			100%			
Eel Van Duzen	7,932	HUM	100%			100%			
Plan Area Factors			10.5%	58.3%	31.2%	11.4%	44.6%	35.7%	8.3%

SC: Salmon Creek; LR: Little River; MR: Mad River, HC: Hunter Creek
SR: Smith River, CKLM: Coastal Klamath; KOR: Korbel; HUM: Humboldt Bay

Table F3-8. Pre- and post-Plan sediment delivery for the Plan Area. Sediment delivery figures represent cubic yards/year. Also included is an estimate of the sediment stabilization effort that can be achieved with an annual expenditure of $2.5 million. Road-related sediment "saved" differs from the stabilization effort because not all sediment from watercourse crossings and "other" sites is expected to deliver.

	Roads	RMZs	SMZs	SHAL-STABs	DSLs	Subtotal of All Zones	Outside of Zone	Total
Sediment Delivery--Pre-Plan	77,779	13,200	8,748	17,451	24,442	141,621	27,220	168,841
Percent of Total Sediment	46.1%	7.8%	5.2%	10.3%	14.5%	83.9%	16.1%	100.0%
Sediment Delivery--Pre-Plan/Acre[1]	4.43	0.25	1.74	0.75	0.37	0.97	0.11	0.43
Sediment Delivery--Post-Plan	3,012	10,276	6,182	11,169	24,201	54,840	27,220	82,060
Percent of Total Sediment	3.7%	12.5%	7.5%	13.6%	29.5%	66.8%	33.2%	100.0%
Sediment Delivery--Post-Plan/Acre[1]	0.17	0.20	1.23	0.48	0.37	0.37	0.11	0.21
"Natural" Sediment	0	10,241	4,374	6,981	22,832	44,428	13,610	58,038
Sediment Stabilization Effort	203,863							
Sediment "Saved"	97,648	2,924	2,566	6,282	242	109,662	N/A	109,662
Percent of Total	89.0%	2.7%	2.3%	5.7%	0.2%	100.0%	N/A	100.0%
Management Related Sediment (%)	100.0%	22.4%	50.0%	60.0%	6.6%			
Effectiveness	96.1%	22.1%	29.3%	36.0%	1.0%			
Do they fail with wood?	No	Yes	Yes	Maybe	Maybe			

[1] Calculations for roads are based on an estimate of "roaded acres" of 17,540 acres.

This simple calculation illustrates how the data in Tables F3-3, F3-4, F3-5, and F3-6 were combined to produce the non-road numbers shown in Table F3-9. Sediment delivery for roads is the next topic to be covered.

To derive an estimate of the sediment delivery associated with roads for the Plan Area it was necessary to integrate the road-related sediment delivery data provided in Appendices F1 and F2. Data presented in Appendix F1 were used to estimate road-related sediment delivery associated with shallow landslides. Data presented in Appendix F2 were used to estimate delivery from watercourse crossings as well as "other" sites. The calculations for the Plan Area are as follows:

The estimate based on Appendix F1 data (38,202 yd^3/year) only includes road-related sediment delivered from shallow landslides. This estimate was deemed to underestimate the contribution from road-related shallow landslides (not all shallow landslides can be observed on aerial photos) so a triangular distribution was developed to (1) account for this underestimate and (2) provide a range of estimates used in the Monte Carlo simulation exercise. The triangular distribution set up for the road-related shallow landslide component is shown in the VARIABLE RANGES section of this appendix (see the "Delivery from road-related landslides" assumption variable in Table F3-13) but is repeated in Table F3-9 to demonstrate the calculations. In summary, it was estimated that the road-related shallow landslide component was most likely under-represented by 15%. Thus,

$$43{,}933\ yd^3/year = 115\% \times 38{,}202\ yd^3/year$$

The minimum under-representation was thought to be 10% whereas the maximum under-representation was thought to be 30%.

Table F3-9. Road-related sediment delivery for the Plan Area.

	Watercourse Crossings (yd^3/year)	Shallow Landslides (yd^3/year)	Other Sites (yd^3/year)	Total (yd^3/year)
Minimum	16,672	42,023	911	59,607
Likeliest	31,383	43,933	1,139	76,456
Mean	31,383	45,206	1,190	77,779
Maximum	46,094	49,663	1,519	97,277
Estimate based on Appendix F1		38,202		

The expected delivery from watercourse crossings was assessed by PWA and is described in Appendix F2. PWA does not expect that all the sediment associated with high and moderate priority treatment sites (the 4,903,664 yd^3 shown in Table F3-1) will deliver within the 50-year term of the Plan. Their likeliest estimate was 32%. On an annual basis this equates to 31,383 yd^3/year. The calculation is as follows:

$$31{,}383\ yd^3/year = 32\% \times (4{,}903{,}664\ yd^3/50\ years)$$

The range associated with this variable (see the "Delivery from road-related steam crossings" assumption in the VARIABLE RANGES section of this appendix) may have a minimum of 17% and a maximum of 47%, which produces the range of estimates shown in Table F3-9 (16,672 yd^3/year to 46,094 yd^3/year). Furthermore, since watercourse crossing sediment delivery is thought to be correlated with shallow landslide sediment delivery, these variables were assumed to have a correlation coefficient of 0.75 for the purposes of conducting the Monte Carlo simulation exercise. (Rainfall often initiates landslides and causes watercourse crossings to fail.)

PWA also assessed the potential sediment delivery from "other" sites. Their review resulted in the values reported in the table above. In this case, PWA expects that 60% to 100% (with the likeliest at 75%) of this sediment may deliver within the 50-year term of the Plan. The calculation of the likeliest value is as follows:

$$1{,}139\ yd^3/\text{year} = 75\% \times (75{,}956\ yd^3/50\ \text{years})$$

Delivery from these "other" sites was also thought to be correlated with delivery from shallow landslides and so these variables were assigned a 0.75 correlation coefficient for the purposes of conducting the Monte Carlo simulation exercise.

Based on the mean estimates provided in Table F3-9, the total expected sediment delivery for the Plan Area from roads is the sum of three components:

Total sediment delivery from roads = sediment delivery from landslides
+ sediment delivery from watercourse crossings
+ sediment delivery from "other" sites

$$77{,}779\ yd^3/\text{year} = 45{,}206\ yd^3/\text{year} + 31{,}383\ yd^3/\text{year} + 1{,}190\ yd^3/\text{year}$$

The 77,779 yd^3/year is an important estimate and is a key figure in Table F3-8.

In addition to the variables already mentioned, several other variables associated with the landslide data and road-related sediment source studies and were assigned appropriate ranges for purposes of conducting the Monte Carlo simulation exercise. These variables and their ranges are provided in the VARIABLE RANGES section of this appendix.

Taken together, the various sources of data and sediment delivery assessments were combined to produce sediment delivery estimates for the Plan Area (Table F3-8).

From an efficiency and effectiveness perspective, the Road Implementation Plan offers a very efficient and effective means for reducing sediment delivery to watercourses (Table F3-8). It is efficient because it "saves" the greatest amount of sediment (89.0%) without setting aside merchantable trees. It is effective (96.1% effectiveness shown in Table F3-8) because approximately 90% of the high and moderate priority sites will be treated at some time during the term of the Plan and will no longer contribute sediment to Plan Area watercourses. It should be noted, however, that the Monte Carlo simulation model actually allows the effectiveness to vary between 94.2%1 and 96.1% (see the assumption variable called Road Upgrade Effectiveness Factor in Tables F3-13 and F3-14).

Due to the model's flexible structure, Green Diamond was able to compare the efficiency, effectiveness, and economic consequences of a wide range of conservation measures. It should be emphasized, however, that the conservation needs of the covered species were deemed to be of paramount importance and scenarios (sets of conservation measures) that did not adequately meet these needs were rejected by the Plan developers.

F3.5 BENEFITS OF THE PLAN PROPOSAL

Currently, Green Diamond stabilizes sediment associated with problematic legacy road sites at an annual rate of about 82,000 cubic yards. Based on Green Diamond's anticipated harvest levels over the next 15 years, an appropriate average annual projected stabilization rate would be 81,545 cubic yards. (Note: This assumes that the relationship between harvest level and sediment stabilization effort remains constant over this period.) The expenditure of $2.5 million on an annual basis for the first 15 years of the Plan will result in the stabilization of 203,863 cubic yards of potential sediment on an annual basis over the first 15 years of the Plan. These figures are summarized in Table F3-10.

Table F3-10. A comparison of road-related sediment stabilization efforts with and without the Plan.

Year	No Plan Sediment Stabilization Program (cubic yards)	Plan Proposal Sediment Stabilization Program (cubic yards)
2002	81,545	203,863
2003	81,545	203,863
2004	81,545	203,863
2005	81,545	203,863
2006	81,545	203,863
2007	81,545	203,863
2008	81,545	203,863
2009	81,545	203,863
2010	81,545	203,863
2011	81,545	203,863
2012	81,545	203,863
2013	81,545	203,863
2014	81,545	203,863
2015	81,545	203,863
2016	81,545	203,863
Total	1,223,177	3,057,943
% of "pile of dirt"	19%	48%

1 A 94.2% road upgrade effectiveness factor implies that 85% of the high and moderate priority sites were appropriately treated during the term of the Plan.

Over the next 15 years, the two scenarios produce vastly different results. The "No Plan" scenario only stabilizes 19% of the total (i.e., 1,223,177 cubic yards divided by 6,436,482 cubic yards) whereas the Plan Proposal stabilizes 48% of the total—a 250% improvement relative to the "No Plan" scenario.

The two scenarios also have dramatically different sediment delivery rates over the next 50 years. For example, in year 15 (2016) the "No Plan" delivery rate from roads is 76% greater than the Plan Proposal delivery rate (44,754 cubic yards per year as compared to 25,463 cubic yards per year). The differences become even larger as time passes. By year 30 (2031) the "No Plan" delivery rate is 174% greater than the Plan Proposal delivery rate (23,627 cubic yards per year as compared to 8,635 cubic yards per year).

The Plan curves shown in Figure F3-2 show the road-related sediment component approaching 3,000 cubic yards during the last decade of the Plan. This implies that the Road Implementation Plan will be 96.1% effective in controlling sediment associated with high and moderate priority treatment sites.

Table F3-11 summarizes the differences between the No Plan and Plan Proposal scenarios in terms of the number of Coho generations that may benefit from an accelerated road repair program.

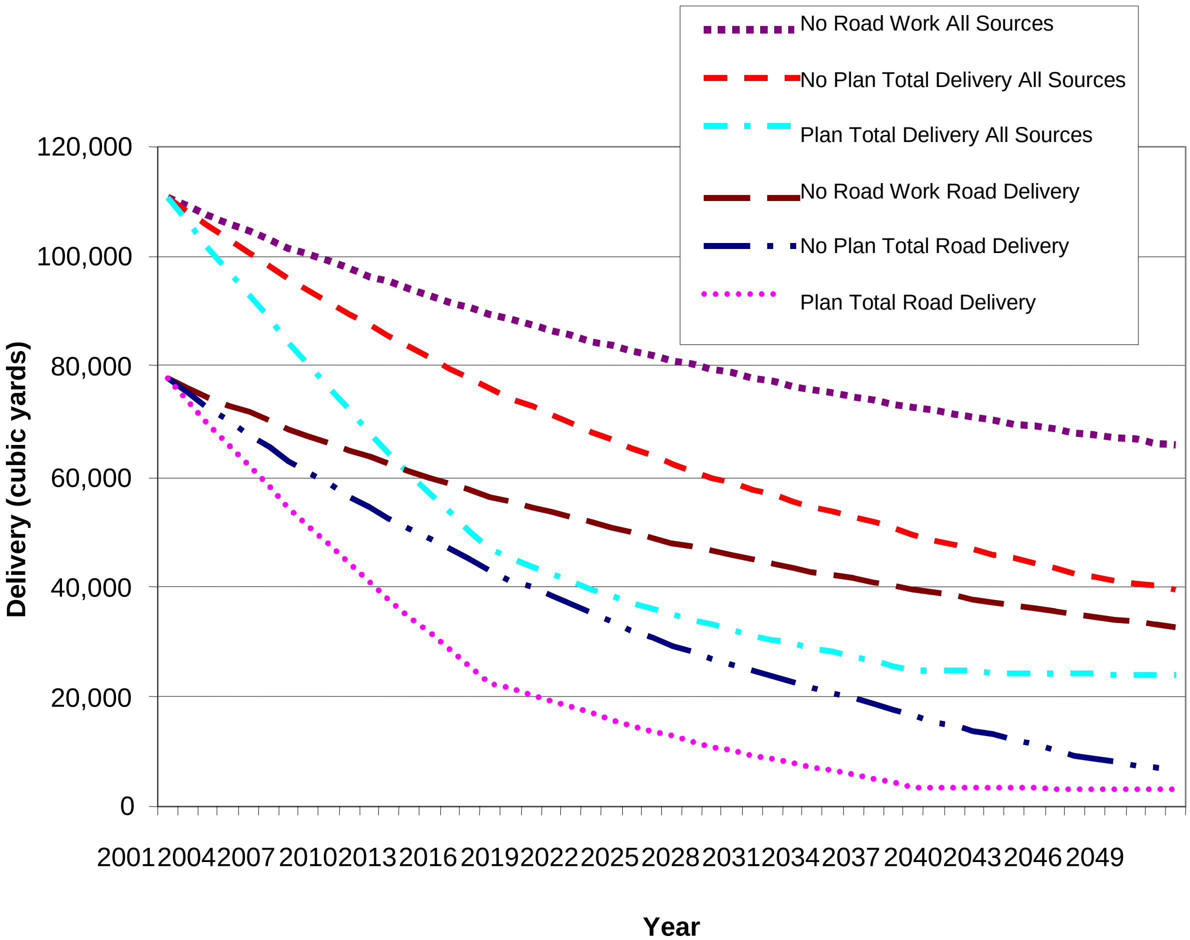
Table F3-11. Coho generations that benefit from the Plan's accelerated road repair and sediment stabilization program.

Scenario	% Pile of Dirt Stabilized	Timeframe (years)	Difference in years	No. of Coho generations that benefit
No Plan	48%	38.0		
Plan Proposal	48%	15.0	23	7.7

This type of analysis shows that the Plan's accelerated road repair and sediment stabilization program can provide benefits to approximately 7.7 generations (23 years divided by 3 years) of Coho salmon. Note that this is from road prescriptions alone. When coupled with the benefits of the other conservation measures, a greater number of fish generations benefit.

Finally, with respect to total sediment delivery from all sources, the No Plan delivery rate in year 50 is comparable to the Plan Proposal's delivery rate in year 15—a 35 year benefit (compare highlighted entries in Table F3-12).

Figure F3-2. Sediment delivery estimates over the term of the Plan. The "No Road Work" curves are based on the assumption that no money is spent repairing the high and moderate priority treatment sites over the next 50 years.



Note: Road-related sediment Includeds sediment from high and moderate priority sites only.

Table F3-12. Key sediment annual delivery rates at different points in time for both the "No Plan" and Plan Proposal scenarios.

	Year	Roads (1000 yd^3/yr)	Harvest Units (1000 yd^3/yr)	Natural (1000 yd^3/yr)	Total Delivery (1000 yd^3/yr)	Total as Compared to Background (i.e., Natural)	Roads Above Background
No Plan	0	78	33	58	169	2.9	1.3
No Plan	15	45	33	58	136	2.3	0.8
No Plan	50	7	33	58	**98**	1.7	0.1
Plan Proposal	0	78	33	58	169	2.9	1.3
Plan Proposal	15	25	24	58	**108**	1.9	0.4
Plan Proposal	50	3	21	58	82	1.4	0.1

F3.6 CALCULATION OF ACREAGE PLACED IN THE ADAPTIVE MANAGEMENT ACCOUNT

The acres within the Adaptive Management Reserve Account (AMRA) were established to address the risk associated with the management prescriptions for SMZs. Based on current GIS data, there are approximately 8,850 acres in SMZs. The acres contained within these zones will be managed using uneven-aged silviculture, defined within the Glossary of the Plan, as single tree selection. By applying single tree selection, Green Diamond will harvest approximately 65% of the conifer volume contained within these SMZs. Thus, approximately 35% of the volume will be retained within these zones to produce conservation benefits as the Plan is implemented over time. As proposed the prescriptions will represent approximately 3,100 acres (or 0.35 x 8,850 acres) of fully stocked timberland. To reduce the risk of potentially underestimating the protection needs of SMZs, Green Diamond will allow up to a 50% increase in the retained volume in SMZs. In terms of fully stocked acres, this will equate to 1,550 acres (0.50 x 3,100 acres = 1,550 acres) that can be applied to these zones. The opening AMRA balance t of 1,550 fully-stocked acres may increase or decrease in response to findings through the Effectiveness Monitoring programs outlined in Section 6.3.

F3.7 MONTE CARLO SIMULATION

The sediment delivery model for the Plan Area was subjected to a statistical procedure known as Monte Carlo simulation. This technique allows the analyst to assign ranges (or a probability density function) to key parameters (assumption variables) and to analyze the effects (the range of results) on forecast variables. The technique begins by randomly drawing parameter values from user-defined ranges and then the forecast variables are determined. This procedure is executed many times (10,000 for this exercise) and the results are saved so probability distributions can be displayed for the forecast variables. The ultimate purpose is to analyze how sensitive forecast variables are to changes in key parameters. The primary forecast variable in this exercise was an index of sediment "saved" (i.e., prevented from entering a watercourse) annually under the "No Plan" scenario as compared to the "With Plan" scenario. The benefit of using a tool like Monte Carlo simulation is that it allows the analyst to simultaneously vary a wide array of assumption variables to perform sensitivity analyses. Simplistic approaches to sensitivity analysis, like setting all assumption variables to their minimum or maximum values, may generate results in the forecast variables that are misleading because such an outcome is highly unlikely. Monte Carlo simulation produces forecast distributions that show which outcomes are most likely (the peaks in the distributions) and which outcomes are statistically unlikely (the tails of the forecast distributions).

F3.7.1 Monte Carlo Simulation Results and Variable Ranges

The complete output file from the Monte Carlo exercise is reproduced in Table F3-13. The table shows the results for the following six forecast variables:

1. Total Sediment Delivery

2. Total Sediment Stabilized

3. Road-Related Sediment Delivery

4. Road-Related Sediment Stabilized

5. No Plan Total Sediment Stabilized (compare to #2)

6. No Plan Road-Related Sediment Stabilized (compare to #4)

The first four forecast variables summarize results based on the implementation of the Plan measures. The last two forecast variables were included to provide some insight into what happens under the No Plan scenario. These No Plan forecast variables can be compared to their Plan counterparts to better understand the differences between the Plan and No Plan scenarios.

The table also includes a listing of 46 assumption variables and their ranges, some of which have been described above in this appendix. The entire output was reproduced here primarily to fully document the ranges associated with the assumption variables. The assumption variables listed in Table F3-13 are allowed to vary for a variety of reasons. The ranges associated with these assumption variables may be based on data, published literature, and/or professional judgment. Table F3-14 is included to indicate the basis for each of the assumption variables. Please review Appendix F1 and Appendix F2 for additional details.

Green Diamond assessed the differences in total sediment saved annually (over the next 15 years) under the No Plan scenario as compared to the Plan scenario. The appropriate forecast variables to inspect in Table F3-13 are “Total Sediment Stabilized” and “No Plan Total Sediment Stabilized”. A brief summary of these forecast variables is as follows:

Sediment Statistic	No Plan Total Sediment Stabilized (yd^3/year)	Plan Total Sediment Stabilized (yd^3/year)
Mean	42,575	114,973
Standard Deviation	1,534	4,801
Minimum	38,314	99,938
Maximum	47,093	129,822

These numbers indicate that the two scenarios are vastly different in a statistical sense. Note that the range of these two distributions does not overlap (i.e., the maximum No Plan value is less than the minimum of the Plan value). Thus, even considering the range (or uncertainty) of all the assumption variables, this key forecast variable shows that the Plan will result in significant sediment savings relative to the No Plan scenario.

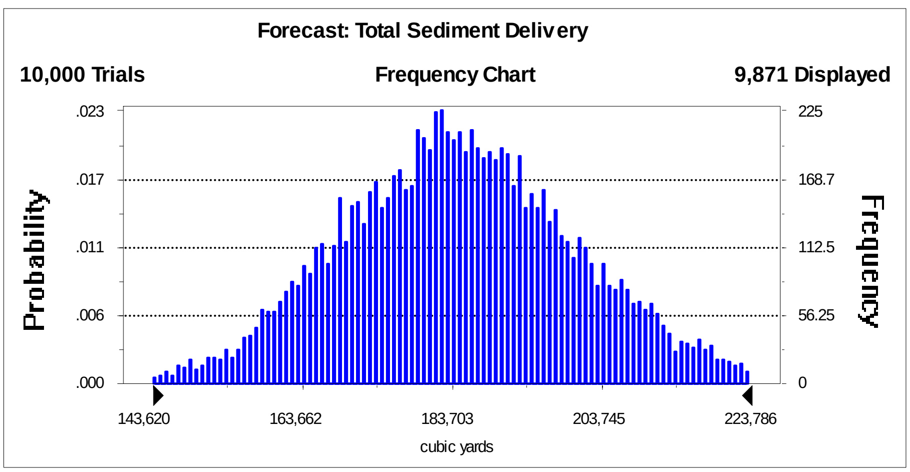
Table F3-13. Monte Carlo simulation results and assumption variable ranges. The program used to conduct the analysis is called Crystal Ball. The following is the unaltered output from that program.

Crystal Ball Report -- Option 1-SEL-b
Simulation started on 3/17/02 at 16:33:26
Simulation stopped on 3/17/02 at 16:38:31

Forecast: Total Sediment Delivery **Cell: K19**

Summary:
Display Range is from 143,620 to 223,786 cubic yards
Entire Range is from 131,750 to 263,258 cubic yards
After 10,000 Trials, the Std. Error of the Mean is 161

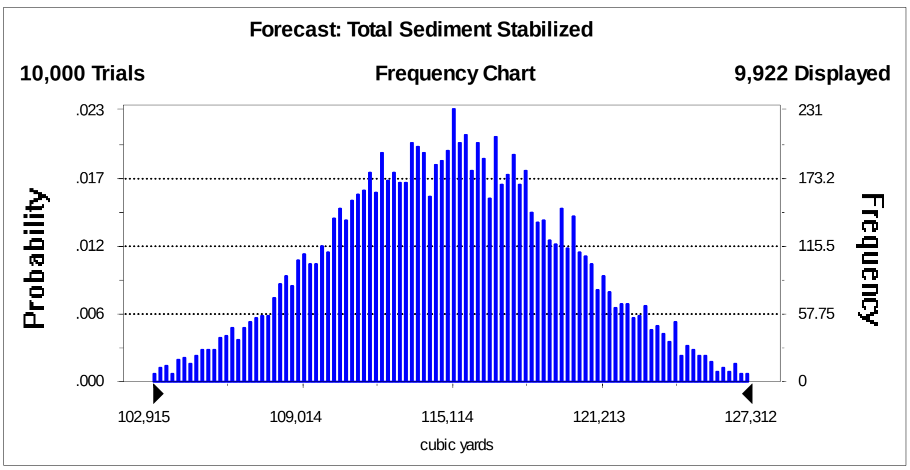
Statistics:	Value
Trials	10000
Mean	184,974
Median	184,520
Mode	---
Standard Deviation	16,070
Variance	258,234,756
Skewness	0.16
Kurtosis	3.01
Coeff. of Variability	0.09
Range Minimum	131,750
Range Maximum	263,258
Range Width	131,509
Mean Std. Error	160.70



Forecast: Total Sediment Stabilized **Cell: K25**

Summary:
Display Range is from 102,915 to 127,312 cubic yards
Entire Range is from 99,938 to 129,822 cubic yards
After 10,000 Trials, the Std. Error of the Mean is 48

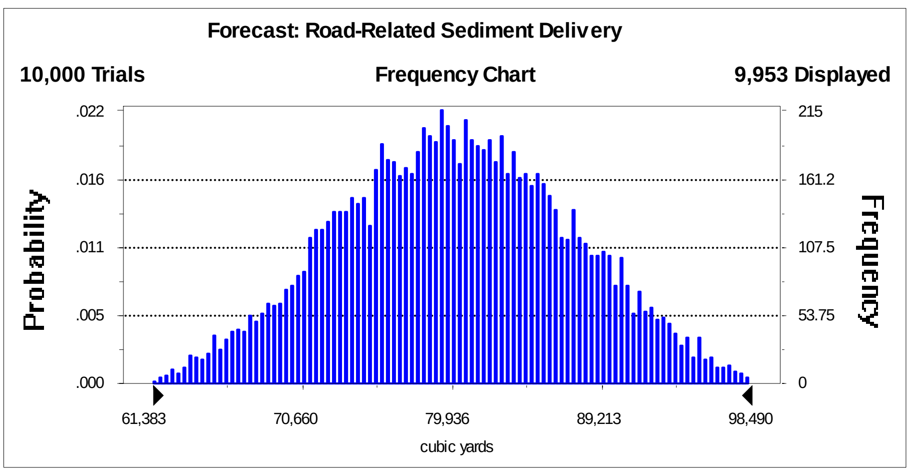
Statistics:	Value
Trials	10000
Mean	114,973
Median	115,016
Mode	---
Standard Deviation	4,801
Variance	23,047,670
Skewness	0.02
Kurtosis	2.77
Coeff. of Variability	0.04
Range Minimum	99,938
Range Maximum	129,822
Range Width	29,884
Mean Std. Error	48.01



Forecast: Road-Related Sediment Delivery **Cell: C19**

Summary:
Display Range is from 61,383 to 98,490 cubic yards
Entire Range is from 58,805 to 101,916 cubic yards
After 10,000 Trials, the Std. Error of the Mean is 73

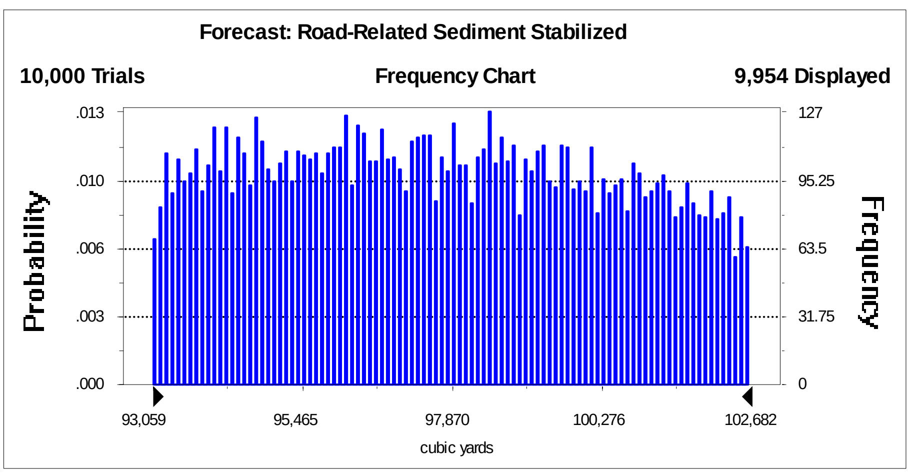
Statistics:	Value
Trials	10000
Mean	80,183
Median	80,142
Mode	---
Standard Deviation	7,258
Variance	52,676,578
Skewness	0.02
Kurtosis	2.61
Coeff. of Variability	0.09
Range Minimum	58,805
Range Maximum	101,916
Range Width	43,111
Mean Std. Error	72.58



Forecast: Road-Related Sediment Stabilized **Cell: C25**

Summary:
Display Range is from 93,059 to 102,682 cubic yards
Entire Range is from 93,026 to 102,745 cubic yards
After 10,000 Trials, the Std. Error of the Mean is 27

Statistics:	Value
Trials	10000
Mean	97,705
Median	97,638
Mode	---
Standard Deviation	2,695
Variance	7,261,524
Skewness	0.07
Kurtosis	1.86
Coeff. of Variability	0.03
Range Minimum	93,026
Range Maximum	102,745
Range Width	9,719
Mean Std. Error	26.95

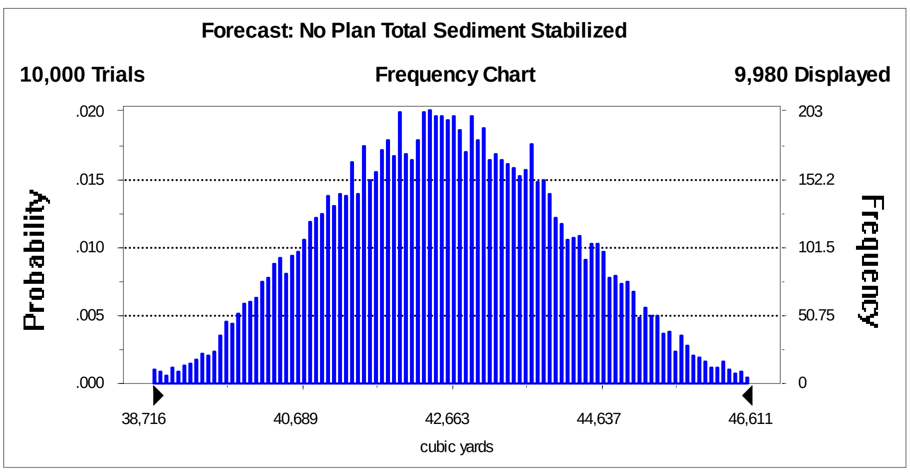


Forecast: No Plan Total Sediment Stabilized **Cell: K3**

Summary:

Display Range is from 38,716 to 46,611 cubic yards
Entire Range is from 38,314 to 47,093 cubic yards
After 10,000 Trials, the Std. Error of the Mean is 15

Statistics:	Value
Trials	10000
Mean	42,585
Median	42,569
Mode	---
Standard Deviation	1,534
Variance	2,353,559
Skewness	0.05
Kurtosis	2.52
Coeff. of Variability	0.04
Range Minimum	38,314
Range Maximum	47,093
Range Width	8,780
Mean Std. Error	15.34

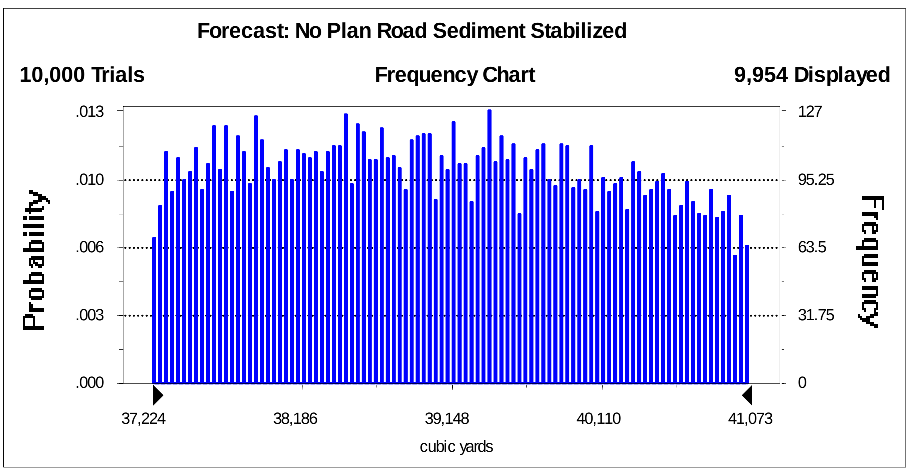


Forecast: No Plan Road Sediment Stabilized **Cell: K1**

Summary:

Display Range is from 37,224 to 41,073 cubic yards
Entire Range is from 37,210 to 41,098 cubic yards
After 10,000 Trials, the Std. Error of the Mean is 11

Statistics:	Value
Trials	10000
Mean	39,082
Median	39,055
Mode	---
Standard Deviation	1,078
Variance	1,161,844
Skewness	0.07
Kurtosis	1.86
Coeff. of Variability	0.03
Range Minimum	37,210
Range Maximum	41,098
Range Width	3,888
Mean Std. Error	10.78

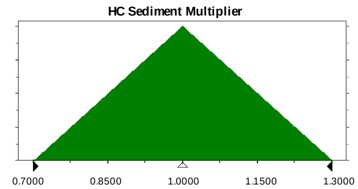


Assumption: HC Sediment Multiplier

[geology sediment model ver 7 best.xls]HC data - Cell: D26

Triangular distribution with parameters:

Minimum	0.7000
Likeliest	1.0000
Maximum	1.3000



Selected range is from 0.7000 to 1.3000

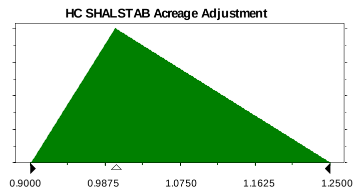
Assumption: HC SHALSTAB Acreage Adjustment

[geology sediment model ver 7 best.xls]HC data - Cell: G4

Triangular distribution with parameters:

Minimum	0.9000	
Likeliest	1.0000	
Maximum	1.2500	(=E4)

Selected range is from 0.9000 to 1.2500



Assumption: HC SMZ Acreage Adjustment

[geology sediment model ver 7 best.xls]HC data - Cell: G3

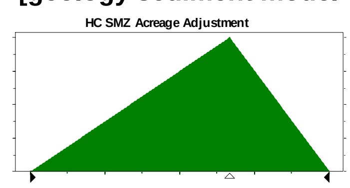
Triangular distribution with parameters:

Minimum	1.0000	
Likeliest	2.0000	
Maximum	2.5000	(=E3)

Selected range is from 1.0000 to 2.5000

Assumption: HC SMZ Acreage Adjustment (cont'd)

[geology sediment model ver 7 best.xls]HC data - Cell: G3



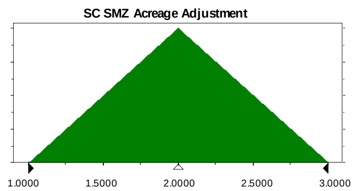
Assumption: SC SMZ Acreage Adjustment

[geology sediment model ver 7 best.xls]SC data - Cell: G3

Triangular distribution with parameters:

Minimum	1.0000	
Likeliest	2.0000	
Maximum	3.0000	(=E3)

Selected range is from 1.0000 to 3.0000



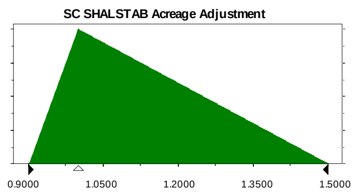
Assumption: SC SHALSTAB Acreage Adjustment

[geology sediment model ver 7 best.xls]SC data - Cell: G4

Triangular distribution with parameters:

Minimum	0.9000	
Likeliest	1.0000	
Maximum	1.5000	(=E4)

Selected range is from 0.9000 to 1.5000



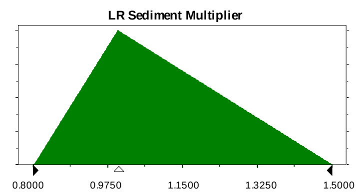
Assumption: LR Sediment Multiplier

[geology sediment model ver 7 best.xls]LR data - Cell: D26

Triangular distribution with parameters:

Minimum	0.8000
Likeliest	1.0000
Maximum	1.5000

Selected range is from 0.8000 to 1.5000



Assumption: LR SMZ Acreage Adjustment

[geology sediment model ver 7 best.xls]LR data - Cell: G3

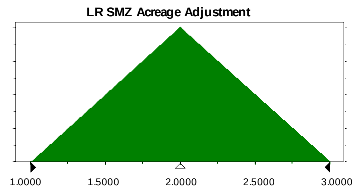
Triangular distribution with parameters:

Minimum	1.0000	
Likeliest	2.0000	
Maximum	3.0000	(=E3)

Selected range is from 1.0000 to 3.0000

Assumption: LR SMZ Acreage Adjustment (cont'd)

[geology sediment model ver 7 best.xls]LR data - Cell: G3



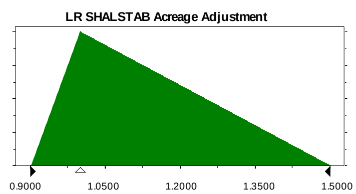
Assumption: LR SHALSTAB Acreage Adjustment

[geology sediment model ver 7 best.xls]LR data - Cell: G4

Triangular distribution with parameters:

Minimum	0.9000	
Likeliest	1.0000	
Maximum	1.5000	(=E4)

Selected range is from 0.9000 to 1.5000

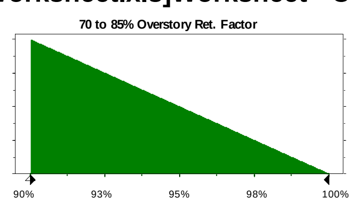


Assumption: 70 to 85% Overstory Ret. Factor

[EROSION RATES by BUFFER - Worksheet.xls]Worksheet - Cell: S7

Triangular distribution with parameters:

Minimum	90%
Likeliest	90%
Maximum	100%



Selected range is from 90% to 100%

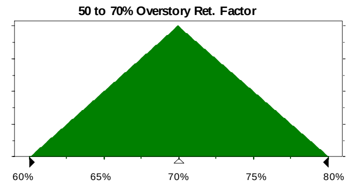
Assumption: 50 to 70% Overstory Ret. Factor

[EROSION RATES by BUFFER - Worksheet.xls]Worksheet - Cell: S8

Triangular distribution with parameters:

Minimum	60%
Likeliest	70%
Maximum	80%

Selected range is from 60% to 80%



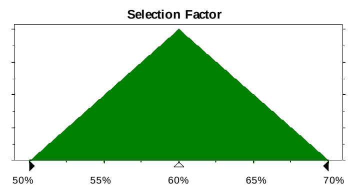
Assumption: Selection Factor

[EROSION RATES by BUFFER - Worksheet.xls]Worksheet - Cell: S9

Triangular distribution with parameters:

Minimum	50%
Likeliest	60%
Maximum	70%

Selected range is from 50% to 70%



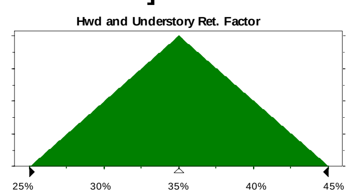
Assumption: Hwd and Understory Ret. Factor

[EROSION RATES by BUFFER - Worksheet.xls]Worksheet - Cell: S10

Triangular distribution with parameters:

Minimum	25%
Likeliest	35%
Maximum	45%

Selected range is from 25% to 45%



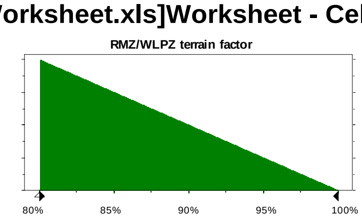
Assumption: RMZ/WLPZ terrain factor

[EROSION RATES by BUFFER - Worksheet.xls]Worksheet - Cell: S18

Triangular distribution with parameters:

Minimum	80%
Likeliest	80%
Maximum	100%

Selected range is from 80% to 100%



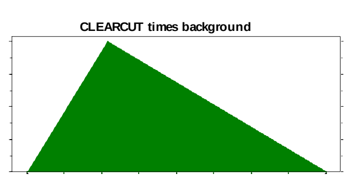
Assumption: CLEARCUT times background

[EROSION RATES by BUFFER - Worksheet.xls]Worksheet - Cell: V3

Triangular distribution with parameters:

Minimum	1.25	(=T3)
Likeliest	2.00	
Maximum	4.00	(=U3)

Selected range is from 1.25 to 4.00



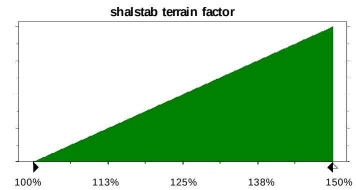
Assumption: shalstab terrain factor

[EROSION RATES by BUFFER - Worksheet.xls]Worksheet - Cell: S21

Triangular distribution with parameters:

Minimum	100%
Likeliest	150%
Maximum	150%

Selected range is from 100% to 150%



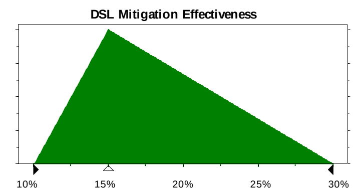
Assumption: DSL Mitigation Effectiveness

[EROSION RATES by BUFFER - Worksheet.xls]Worksheet - Cell: P27

Triangular distribution with parameters:

Minimum	10%
Likeliest	15%
Maximum	30%

Selected range is from 10% to 30%



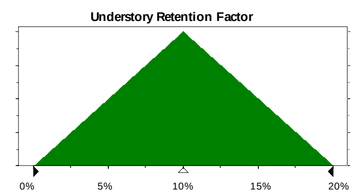
Assumption: Understory Retention Factor

[EROSION RATES by BUFFER - Worksheet.xls]Worksheet - Cell: S11

Triangular distribution with parameters:

Minimum	0%
Likeliest	10%
Maximum	20%

Selected range is from 0% to 20%



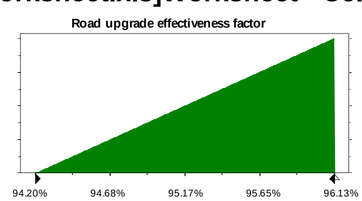
Assumption: Road upgrade effectiveness factor

[EROSION RATES by BUFFER - Worksheet.xls]Worksheet - Cell: S24

Triangular distribution with parameters:

Minimum	94.20%
Likeliest	96.13%
Maximum	96.13%

Selected range is from 94.20% to 96.13%



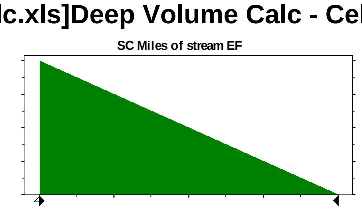
Assumption: SC Miles of stream EF

[Deep Volume Calc.xls]Deep Volume Calc - Cell: E17

Triangular distribution with parameters:

Minimum	0.6
Likeliest	0.6
Maximum	1.1

Selected range is from 0.6 to 1.1



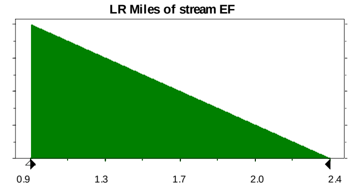
Assumption: LR Miles of stream EF

[Deep Volume Calc.xls]Deep Volume Calc - Cell: F17

Triangular distribution with parameters:

Minimum	0.9
Likeliest	0.9
Maximum	2.4

Selected range is from 0.9 to 2.4



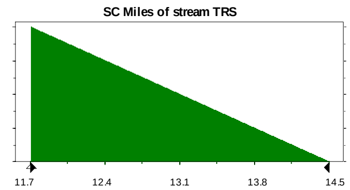
Assumption: SC Miles of stream TRS

[Deep Volume Calc.xls]Deep Volume Calc - Cell: E18

Triangular distribution with parameters:

Minimum	11.7
Likeliest	11.7
Maximum	14.5

Selected range is from 11.7 to 14.5



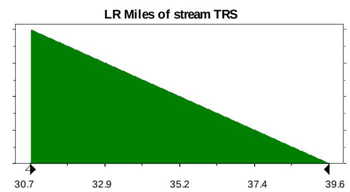
Assumption: LR Miles of stream TRS

[Deep Volume Calc.xls]Deep Volume Calc - Cell: F18

Triangular distribution with parameters:

Minimum	30.7
Likeliest	30.7
Maximum	39.6

Selected range is from 30.7 to 39.6



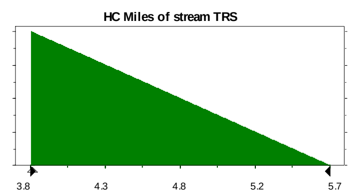
Assumption: HC Miles of stream TRS

[Deep Volume Calc.xls]Deep Volume Calc - Cell: G18

Triangular distribution with parameters:

Minimum	3.8
Likeliest	3.8
Maximum	5.7

Selected range is from 3.8 to 5.7



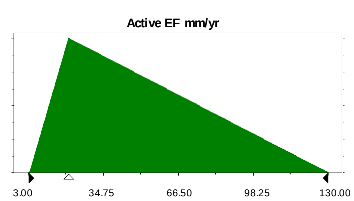
Assumption: Active EF mm/yr

[Deep Volume Calc.xls]Deep Volume Calc - Cell: E2

Triangular distribution with parameters:

Minimum	3.00	(=J2)
Likeliest	20.00	(=K2)
Maximum	130.00	(=L2)

Selected range is from 3.00 to 130.00



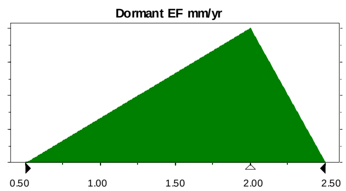
Assumption: Dormant EF mm/yr

[Deep Volume Calc.xls]Deep Volume Calc - Cell: E3

Triangular distribution with parameters:

Minimum	0.50	(=J3)
Likeliest	2.00	(=K3)
Maximum	2.50	(=L3)

Selected range is from 0.50 to 2.50



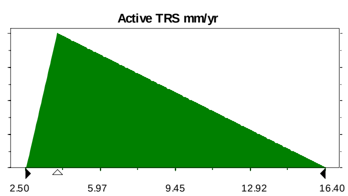
Assumption: Active TRS mm/yr

[Deep Volume Calc.xls]Deep Volume Calc - Cell: E4

Triangular distribution with parameters:

Minimum	2.50	(=J4)
Likeliest	4.00	(=K4)
Maximum	16.40	(=L4)

Selected range is from 2.50 to 16.40



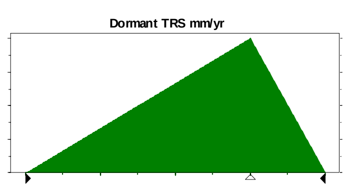
Assumption: Dormant TRS mm/yr

[Deep Volume Calc.xls]Deep Volume Calc - Cell: E5

Triangular distribution with parameters:

Minimum	0.50	(=J5)
Likeliest	2.00	(=K5)
Maximum	2.50	(=L5)

Selected range is from 0.50 to 2.50



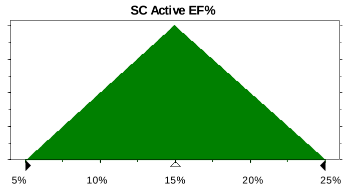
Assumption: SC Active EF%

[Deep Volume Calc.xls]Deep Volume Calc - Cell: E11

Triangular distribution with parameters:

Minimum	5%	(=J19)
Likeliest	15%	(=J20)
Maximum	25%	(=J21)

Selected range is from 5% to 25%



Assumption: LR Active EF%

[Deep Volume Calc.xls]Deep Volume Calc - Cell: F11

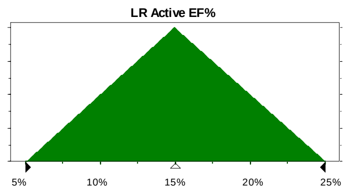
Triangular distribution with parameters:

Minimum	5%	(=K19)
Likeliest	15%	(=K20)
Maximum	25%	(=K21)

Selected range is from 5% to 25%

Assumption: LR Active EF% (cont'd)

[Deep Volume Calc.xls]Deep Volume Calc - Cell: F11



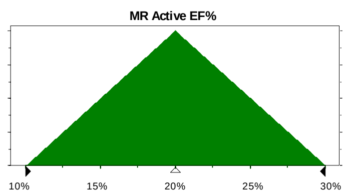
Assumption: MR Active EF%

[Deep Volume Calc.xls]Deep Volume Calc - Cell: H11

Triangular distribution with parameters:

Minimum	10%	(=M19)
Likeliest	20%	(=M20)
Maximum	30%	(=M21)

Selected range is from 10% to 30%



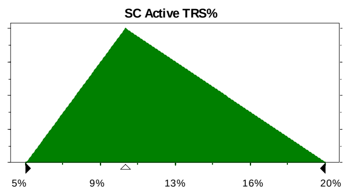
Assumption: SC Active TRS%

[Deep Volume Calc.xls]Deep Volume Calc - Cell: E14

Triangular distribution with parameters:

Minimum	5%	(=J25)
Likeliest	10%	(=J26)
Maximum	20%	(=J27)

Selected range is from 5% to 20%



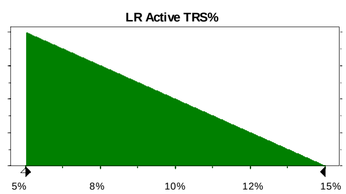
Assumption: LR Active TRS%

[Deep Volume Calc.xls]Deep Volume Calc - Cell: F14

Triangular distribution with parameters:

Minimum	5%	(=K25)
Likeliest	5%	(=K26)
Maximum	15%	(=K27)

Selected range is from 5% to 15%



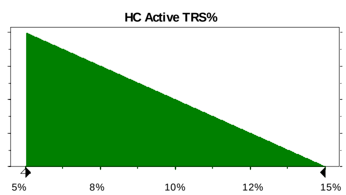
Assumption: HC Active TRS%

[Deep Volume Calc.xls]Deep Volume Calc - Cell: G14

Triangular distribution with parameters:

Minimum	5%	(=L25)
Likeliest	5%	(=L26)
Maximum	15%	(=L27)

Selected range is from 5% to 15%



Assumption: MR Active TRS%

[Deep Volume Calc.xls]Deep Volume Calc - Cell: H14

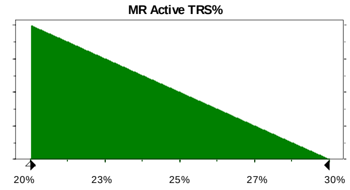
Triangular distribution with parameters:

Minimum	20%	(=M25)
Likeliest	20%	(=M26)
Maximum	30%	(=M27)

Selected range is from 20% to 30%

Assumption: MR Active TRS% (cont'd)

[Deep Volume Calc.xls]Deep Volume Calc - Cell: H14



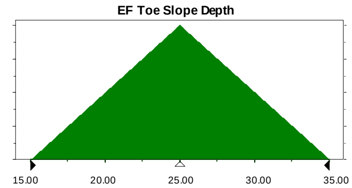
Assumption: EF Toe Slope Depth

[Deep Volume Calc.xls]Deep Volume Calc - Cell: B10

Triangular distribution with parameters:

Minimum	15.00	(=B14)
Likeliest	25.00	(=B15)
Maximum	35.00	(=B16)

Selected range is from 15.00 to 35.00



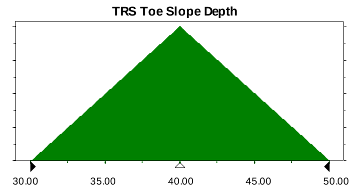
Assumption: TRS Toe Slope Depth

[Deep Volume Calc.xls]Deep Volume Calc - Cell: B11

Triangular distribution with parameters:

Minimum	30.00	(=B19)
Likeliest	40.00	(=B20)
Maximum	50.00	(=B21)

Selected range is from 30.00 to 50.00



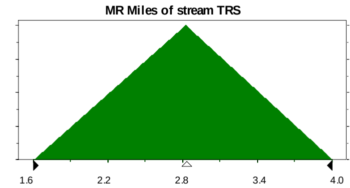
Assumption: MR Miles of stream TRS

[Deep Volume Calc.xls]Deep Volume Calc - Cell: H18

Triangular distribution with parameters:

Minimum	1.6
Likeliest	2.8
Maximum	4.0

Selected range is from 1.6 to 4.0



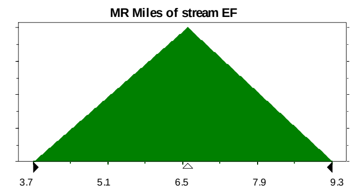
Assumption: MR Miles of stream EF

[Deep Volume Calc.xls]Deep Volume Calc - Cell: H17

Triangular distribution with parameters:

Minimum	3.7
Likeliest	6.6
Maximum	9.3

Selected range is from 3.7 to 9.3



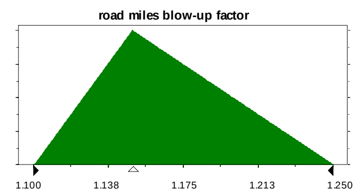
Assumption: road miles blow-up factor

[revised assessment summary ver 5.xls]data - Cell: I2

Triangular distribution with parameters:

Minimum	1.100
Likeliest	1.150
Maximum	1.250

Selected range is from 1.100 to 1.250



Assumption: Delivery from road-related landslides

[revised assessment summary ver 5.xls]removal and delivery - Cell: D22

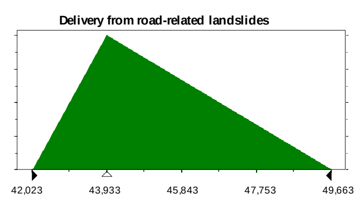
Triangular distribution with parameters:

Minimum	42,023	(=D24)
Likeliest	43,933	(=D25)
Maximum	49,663	(=D26)

Selected range is from 42,023 to 49,663

Correlated with:

Delivery from road-related other sites (F22)	0.75
Delivery from road-related stream xings (B:	0.75



Assumption: Delivery from road-related stream xings
[revised assessment summary ver 5.xls]removal and delivery - Cell: B22

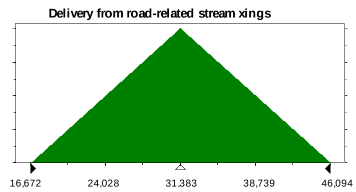
Triangular distribution with parameters:

Minimum	16,672	(=B24)
Likeliest	31,383	(=B25)
Maximum	46,094	(=B26)

Selected range is from 16,672 to 46,094

Correlated with:

Delivery from road-related landslides (D22) 0.75



Assumption: Delivery from road-related other sites
[revised assessment summary ver 5.xls]removal and delivery - Cell: F22

Triangular distribution with parameters:

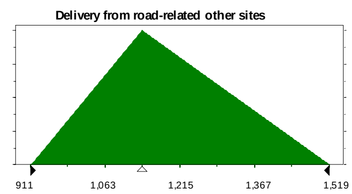
Minimum	911	(=F24)
Likeliest	1,139	(=F25)
Maximum	1,519	(=F26)

Selected range is from 911 to 1,519

Correlated with:

Delivery from road-related landslides (D22) 0.75

Assumption: Delivery from road-related other sites (cont'd)
[revised assessment summary ver 5.xls]removal and delivery - Cell: F22

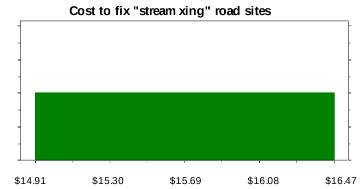


Assumption: Cost to fix "stream xing" road sites

[revised assessment summary ver 5.xls]removal and delivery - Cell: B5

Uniform distribution with parameters:

Minimum	$14.91
Maximum	$16.47



Correlated with:

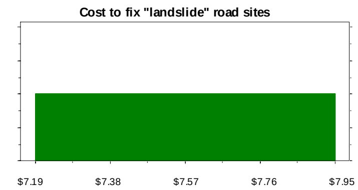
Cost to fix "landslide" road sites (D5)	0.75
Cost to fix "other" road sites (F5)	0.75

Assumption: Cost to fix "landslide" road sites

[revised assessment summary ver 5.xls]removal and delivery - Cell: D5

Uniform distribution with parameters:

Minimum	$7.19
Maximum	$7.95



Correlated with:

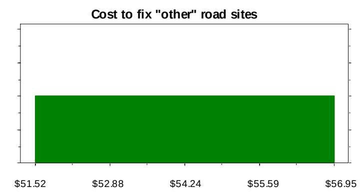
Cost to fix "stream xing" road sites (B5)	0.75

Assumption: Cost to fix "other" road sites

[revised assessment summary ver 5.xls]removal and delivery - Cell: F5

Uniform distribution with parameters:

Minimum	$51.52
Maximum	$56.95



Assumption: Cost to fix "other" road sites (cont'd)

[revised assessment summary ver 5.xls]removal and delivery - Cell: F5

Correlated with:

Cost to fix "stream xing" road sites (B5)	0.75

End of Assumptions

Table F3-14. The basis (i.e., data, literature, or professional judgment) used to determine the range of estimates for each assumption variable listed in Table F3-13. Much of the information pertaining to "hillslope" assumption variables was extracted from Appendix F1. For road-related assumption variables, information was taken from Appendix F2.

Variable No.	Assumption Variable	Hillslope or Road-Related	Basis Used To Determine Range	Comment
1	Hunter Creek Sediment Multiplier	Hillslope	Data and Professional Judgment	About 15% of the 1997 failures in Hunter Creek were field sampled to verify air photo interpretations and calibrate slide volumes and sediment delivery ratios. Range in landslide volumes estimated from 1) comparison of field and air photo measurements of landslide volumes and 2) professional judgment made from field reconnaissance and review of the historic aerial photographs.
2	Hunter Creek SHALSTAB Acreage Adjustment	Hillslope	Professional Judgment w/ limited supporting data	Range estimated from 1) comparison of the SHALSTAB map to aerial photograph interpretations of headwall swales and 2) field review of SHALSTAB areas on and off Green Diamond property.
3	Hunter Creek SMZ Acreage Adjustment	Hillslope	Professional Judgment w/ limited supporting data	The minimum is based on DEM measurements of slope gradient. Likeliest and maximum values have been increased to account for inherent underestimates of slope gradient by topographic maps and DEMs. The increase in SMZ acreage for likeliest and maximum values is estimated from 1) air photo observations, 2) limited field observations, and 3) discussions with Green Diamond forestry staff.
4	Salmon Creek Sediment Multiplier	Hillslope	Professional Judgment w/ supporting data	Limited field reconnaissance of the 1997 failures have been undertaken in Salmon Creek to verify air photo interpretations and calibrate slide volumes and sediment delivery ratios. Field reconnaissance has focused along steep streamside slopes. Range in landslide volumes is estimated from 1) comparison of field and air photo measurements of landslide volumes and 2) professional judgment made from field reconnaissance and review of the historic aerial photographs.
5	Salmon Creek SMZ Acreage Adjustment	Hillslope	Professional Judgment w/ limited supporting data	Range estimated from 1) comparison of the SHALSTAB map to aerial photograph interpretations of headwall swales and 2) field review of SHALSTAB areas on and off Green Diamond property.
6	Salmon Creek SHALSTAB Acreage Adjustment	Hillslope	Professional Judgment w/ limited supporting data	The minimum is based on DEM measurements of slope gradient. Mid and upper range have been increased to account for inherent underestimates of slope gradient by topographic maps and DEMs. The increase in SMZ acreage for likeliest and maximum values is estimated from 1) air photo observations, 2) limited field observations, and 3) discussions with Green Diamond forestry staff.

Table F3-14. (Continued)

Variable No.	Assumption Variable	Hillslope or Road-Related	Basis Used To Determine Range	Comment
7	Little River Sediment Multiplier	Hillslope	Professional Judgment w/ supporting data	Limited field reconnaissance of the 1997 failures have been undertaken in Little River to verify air photo interpretations and calibrate slide volumes and sediment delivery ratios. Field reconnaissance has focused along steep streamside slopes. Range in landslide volumes is estimated from 1) comparison of field and air photo measurements of landslide volumes and 2) professional judgment made from field reconnaissance and review of the historic aerial photographs.
8	Little River SMZ Acreage Adjustment	Hillslope	Professional Judgment w/ limited supporting data	Range estimated from 1) comparison of the SHALSTAB map to aerial photograph interpretations of headwall swales and 2) field review of SHALSTAB areas on and off Green Diamond property.
9	Little River SHALSTAB Acreage Adjustment	Hillslope	Professional Judgment w/ limited supporting data	The minimum is based on DEM measurements of slope gradient. Mid and upper range have been increased to account for inherent underestimates of slope gradient by topographic maps and DEMs. The increase in SMZ acreage for likeliest and maximum values is estimated from 1) air photo observations, 2) limited field observations, and 3) discussions with Green Diamond forestry staff.
10	Road Miles Blow-Up Factor	Road-Related	Data and Professional Judgment	Air photo analysis of Green Diamond and other property
11	Delivery From Road-Related Landslides	Road-Related	Data	Data from field inventories
12	Delivery From Road-Related Watercourse Crossings	Road-Related	Data	Data from field inventories
13	Delivery From Road-Related Other Sites	Road-Related	Data	Data from field inventories
14	Cost To Fix Watercourse Crossing Road Sites	Road-Related	Data	Field inventory, surveys, production rate estimates and standard cost rates
15	Cost To Fix Landslide Road Sites	Road-Related	Data	Field inventory, surveys, production rate estimates and standard cost rates
16	Cost to Fix Other Road Sites	Road-Related	Data	Field inventory, surveys, production rate estimates and standard cost rates

Table F3-14. (Continued)

Variable No.	Assumption Variable	Hillslope or Road-Related	Basis Used To Determine Range	Comment
17	70 to 85% Overstory Retention Factor	Hillslope	Professional Judgment w/ supporting data and literature	Adjustments to clearcut harvest ratio to account for different overstory retentions is based on professional judgment, supported by landslide inventories [e.g., ODF study on the impacts of 1995 and 1996 storms (Robison et al. 1999), PALCO Freshwater Creek Watershed Analysis (PALCO 2001a)], published literature (Megahan et al. 1978), shallow landslide modeling [e.g., (Krogstad 1995; Schmidt et al. in review; Sidle 1991; Sidle 1992; Ziemer 1981a, 1981b)], and experience.
18	50 to 70% Overstory Retention Factor	Hillslope	Professional Judgment w/ supporting data and literature	See # 17
19	Selection Factor	Hillslope	Professional Judgment w/ supporting data and literature	See # 17
20	Hardwood and Understory Retention Factor	Hillslope	Professional Judgment w/ supporting data and literature	See # 17
21	Road Upgrade Effectiveness Factor	Road-Related	Data and Professional Judgment	Data and observations from Green Diamond and other watersheds
22	RMZ/WLPZ Slope Position Factor	Hillslope	Professional Judgment w/ limited supporting data and literature	Adjustments in slope position (i.e., RMZ, SHALSTAB or other) are based on professional judgment supported by interpretations of regional landslide studies (PALCO Freshwater Creek Watershed Analysis (PALCO 2001a) and unpublished Hunter Creek landslide data) and professional experience.

Table F3-14. (Continued)

Variable No.	Assumption Variable	Hillslope or Road-Related	Basis Used To Determine Range	Comment
23	Clearcut Times Background	Hillslope	Professional Judgment and Literature	An average clearcut harvest ratio was estimated from a review of published and unpublished landslide inventories, including TMDL studies, the ODF study on the impacts of 1995 and 1996 storms (Robison et al. 1999), PALCO Sediment Source Investigations (PWA 1998a, 1998b, 1999a, 1999b), PALCO Freshwater Creek Watershed Analysis (PALCO 2001a), and Green Diamond's preliminary Mass Wasting Assessment for Hunter Creek. The results of these studies are summarized in Appendix F1, Table 5. A complete discussion of each study is included in Appendix F1 of this report. Range in clearcut ratio is based primarily on professional judgment.
24	SHALSTAB Terrain Factor	Hillslope	Professional Judgment w/ limited supporting data and literature	See #22
25	DSL Mitigation Effectiveness	Hillslope	Professional Judgment and data	The impact of harvesting on historically active deep-seated landslides is assumed to be a function of percentage of canopy retained. Landslides are mapped from the historic set of aerial photographs. The percentage of historically active slides is based on professional judgment (See #36, 37, 38, 39, 40, 41 and 42). Acreage of harvest on historically active slide determined from the GIS database. Analysis assumes clearcut harvesting on entirety of slide outside of prescribed retention areas (i.e. RMZ, SMZ, SHALSTAB, and active scarps and toes). Maximum and minimum based on professional judgment.
26	Understory Retention Factor	Hillslope	Professional Judgment	See # 17
27	Salmon Creek Miles of Stream Earth Flows	Hillslope	Data	Minimum and likeliest values based on length of streams on "Definite" and "Probable" landslides. Maximum value includes stream length on "Questionable" landslides. Certainty of landslide based on air photo observations.
28	Little River Miles of Stream Earth Flows	Hillslope	Data	See #27

Table F3-14. (Continued)

Variable No.	Assumption Variable	Hillslope or Road-Related	Basis Used To Determine Range	Comment
29	Salmon Creek Miles of Stream Translational/Rotational Landslides	Hillslope	Data	See #27
30	Little River Miles of Stream Translational/Rotational Landslides	Hillslope	Data	See #27
31	Hunter Creek Miles of Stream Translational/Rotational Landslides	Hillslope	Data	See #27
32	Active Earth Flow mm/yr	Hillslope	Literature and Professional Judgment	Maximum and minimum values based on range of measured rates of earthflow movement on the east side of the Grogan Fault in Redwood Creek (Swanson and others 1995). Likeliest value based on professional judgment supported by limited field review of slides on and off of Green Diamond property and professional experience.
33	Dormant Earth Flow mm/yr	Hillslope	Literature and Professional Judgment	Maximum and minimum values based on range of measured progressive creep rates on the west side the Grogan Fault in Redwood Creek (Swanson and others 1995). Likeliest value based on professional judgment supported by limited field review of slides on and off of Green Diamond property and professional experience.
34	Active Translational/Rotational Slides mm/yr	Hillslope	Literature and Professional Judgment	Maximum and minimum values based on measured rates of block glide movement in Redwood Creek (Swanson and others 1995). Likeliest value based on professional judgment supported by limited field review of slides on and off of Green Diamond property and professional experience.
35	Dormant Translational/Rotational Slides mm/yr	Hillslope	Literature and Professional Judgment	Maximum and minimum values based on measured progressive creep rates on the west side the Grogan Fault in Redwood Creek (Swanson and others 1995). Likeliest value based on professional judgment supported by limited field review of slides on and off of Green Diamond property and professional experience.
36	Salmon Creek Active Earth Flow %	Hillslope	Professional Judgment	Based on limited field reconnaissance of the watersheds, discussions with Green Diamond foresters and past experience.
37	Little River Active Earth Flow %	Hillslope	Professional Judgment	See #36

Table F3-14. (Continued)

Variable No.	Assumption Variable	Hillslope or Road-Related	Basis Used To Determine Range	Comment
38	Mad River Active Earth Flow %	Hillslope	Professional Judgment	See #36
39	Salmon Creek Active Translational/Rotational Slides %	Hillslope	Professional Judgment	See #36
40	Little River Active Translational/Rotational Slides %	Hillslope	Professional Judgment	See #36
41	Hunter Creek Active Translational/Rotational Slides %	Hillslope	Professional Judgment	See #36
42	Mad River Active Translational/Rotational Slides %	Hillslope	Professional Judgment	See #36
43	Earth Flow Toe Slope Depth	Hillslope	Literature and Professional Judgment	Depth based on professional judgment and experience, supported by published data on slide depth (e.g., Swanson and others 1995; SWS 1999; USACE 1980; USDA 1970).
44	Translational/Rotational Slide Toe Slope Depth	Hillslope	Literature and Professional Judgment	See #43
45	Mad River Miles of Stream Translational/Rotational Landslides	Hillslope	Data	See #27
46	Mad River Miles of Stream Earth Flows	Hillslope	Data	See #27

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Appendix G. Special Project to Enhance Coho Salmon Productivity by Utilizing Habitats Upstream of Anadromous Barriers

CONTENTS

G.1 PROJECT OVERVIEW

Green Diamond will undertake a special project that it anticipates will "jump start" the conservation of coho salmon by increasing the available habitat for spawning and rearing. Green Diamond will undertake one project to trap and transport adult coho salmon that are native to the respective stream system downstream of a barrier to anadromy. These spawners would then be allowed to spawn naturally in the previously unutilized habitats upstream of the barrier to anadromy. This project would be conducted and monitored over a ten-year period.

Small numbers (approximately 10 male/female pairs) of adult coho spawners would be carefully captured at weirs. These selected adults would then be anesthetized using MS-222 or CO_2, tagged (e.g. floytags), and gently placed into restraining and transport tubes made of PVC pipe. The selected fish then would be placed into large holding tanks on a flatbed truck. These holding tanks will be fitted with aeration or an oxygen supply to ensure adequate dissolved oxygen concentrations during transport. The water in the holding tanks should contain a therapeutic dose of 3% NaCl and/or an artificial slime agent such as PolyAqua® to reduce handling stress and loss of natural mucous. The selected adult coho salmon would then be transported to a release point upstream of the anadromous barrier and set free in a pool habitat following their recovery from capture. Several capture events and transport trips would be likely required to transport all the coho spawners selected for relocation. Release of these spawners would occur at a location upstream of the anadromous barrier that insures that the translocated spawners are not swept downstream over the barrier following their release.

The selected coho spawners will be monitored following their release to document any spawning success. If spawning is observed, subsequent surveys will be conducted during summer months to assess spawning success and the utilization of summer rearing habitats in the reaches upstream of the anadromous barrier by the juvenile fish. These summer surveys will also provide an opportunity to assess the potential interaction between the introduced coho population and any resident salmonids if present. Finally, out-migrant trapping will be conducted during the following winter/spring to document the number of coho smolts that emigrate from the system.

G.2 PROJECT GOALS AND OBJECTIVES

The goal of this project is to rapidly increase (within a few years) coho smolt production within the selected streams. This would occur concurrently but probably at a faster rate than the anticipated improved stream habitat conditions with the Plan Area. As analyzed below, although the capture of coho for movement around impassible barriers may technically constitute "take" of individuals, the project is not expected to cause unacceptable impacts to any Covered Species. An objective of the project would be to assist coho populations to fully maximize available spawning and rearing habitats within the selected streams or watershed.

G.3 PRE-PROJECT EVALUATION

Prior to selecting a stream to conduct the project, the stream will undergo a pre-project evaluation for its suitability. The project area will be evaluated in terms of the potential quality and quantity of coho habitat (i.e. spawning and summer and winter rearing). For the project to be effective, and meet the goal of rapidly increasing coho smolt production, many environmental conditions must be met. These include suitable water quality (temperature and clarity), adequate stream flows, velocities, and depths, appropriate spawning substrate quality (size) and quantity, sufficient food production, and a variety and complexity of cover for holding and refuge.

By carefully assessing the adequacy of a stream's habitats (quality) and its total spawning and rearing capacities (quantity), the worthiness of a stream will be determined. For example, if a projects' upstream location has high habitat quality but the quantity of habitat is small, in relation to the downstream area, it is likely that this location would not substantially increase overall smolt production. In that example, this stream would be ineffective in meeting the project goal (i.e. rapidly increase smolt production).

Secondarily, the stream will be evaluated in terms of its current use by resident salmonids and the potential for any negative impacts, especially to any of the other AHCP Covered Species. If there are existing conditions in the upstream areas of the project stream that likely would reduce the effectiveness of rapid smolt production (e.g. excessive predation on coho smolts), then this would negatively affect the overall effectiveness in meeting the project's goals. Such a stream location would not be selected for the project.

G.4 ANALYSIS OF PROJECT'S POTENTIAL IMPACTS

The following is a discussion of the potential impacts mechanisms and an examination of the likelihood of these occurring from the implementation of the project. It is likely that only a subset of any impact mechanisms would occur in a specific project location but for the purposes of this analysis, they will all be discussed.

G.4.1 Potential Impact Mechanisms

There are three principle groups of impact mechanisms, direct impacts, indirect impacts, and competitive (interactive) impacts. Many of these mechanisms overlap and will be discussed below.

The direct impacts to coho salmon that could occur from this project include:

- Death during transport and relocation of adults,
- Increased pre-spawning mortality,
- Increased egg mortality,
- Increased fry and juvenile mortality,
- Increased smolt mortality.

Indirect impacts to coho that could occur from this project include:

- Reduction in fry and juvenile growth rates,

- Delays in smolt emigration timing.

Competitive impacts that could occur include:

- Increased predation from resident species,
- Increased predation on other resident and anadromous aquatic species,
- Increased food competition with resident and other anadromous species,
- Displacement of one or more resident species or introduced coho salmon,
- Competition and predation on other anadromous species during out-migration.

G.4.2 Impact Analysis

G.4.2.1 Direct Impacts

The capture and transport of adult coho spawners has a large potential for direct losses (death) of the species and an increase in pre-spawning mortality following their release. Handling large fish can be awkward and potentially lethal to these fish if proper precautions are not taken. Fortunately, direct loss of adults during the capture, handling, transportation, and increased pre-spawning mortality following release can be minimized through planning and use of proven techniques.

If electrofishing is used as a collection method, attention to the power settings (i.e. 60 cps, D.C. pulsed power at low amperages and appropriate voltages) will eliminate adverse impacts to adult salmon during capture (K. Brown, FWS, pers. com). If capture is accomplished by use of picket weirs and traps, attention to construction (proper spacing of pickets and materials used), installation, and operations of the weir will eliminate losses and increased stress to adults during capture. Traps must be attended to frequently and trapped fish must be removed quickly and efficiently following their capture.

Handling techniques following adult capture including the judicial use of either MS-222 or carbon dioxide ($CaCO_2$) will result in safe anesthetization without risk or loss during handling. Risk of loss of adults during transportation also can be minimized and eliminated by use of techniques such as placement of captured fish into transportation “tubes”. These tubes are lengths of large diameter PVC pipe with flapper doors on their ends and a carrying handle in which a fish is placed and lifted into a holding tank from an anesthetization tank. These tubes are effective in handling large adult salmon and eliminating stresses and losses (K. Brown, FWS, pers. com.). Following the placement of fish into transport tanks with aeration and appropriate therapeutics (to minimize risk of infection and disease) fish can be quickly transported to their release sites. Adults then can be placed into receiving waters to acclimate thereby further minimize any life threatening stresses and losses including increased rates of pre-spawning mortality. In summary, using these and other techniques, risks to adult coho spawners will be minimized and direct losses to adults can be minimized.

An increase in *in vivo* egg mortality could occur as a result of stress from capture, handling, transport, and release of coho spawners. However, as discussed above, risks are likely minimized and significant losses from these activities will be eliminated by the use of proper techniques during capture, handling, transport and release of adult spawners. Increased rates of *in vitro* egg losses resulting from the project are also unlikely. By adequately characterizing substrate composition, water quality, and water quantity, in the project location prior to selecting this location, any risk of increased rate of egg losses through spawning in unfavorable upstream habitats will be minimized, if not eliminated.

Increased rates of fry and juvenile mortality could potentially occur from rearing in upstream habitats as opposed to downstream habitat. This could occur from a number of mechanisms including unfavorable habitat conditions in the rearing areas or increased competition for food, and increased predation from resident species. Two physical factors play a large role in the survival of the freshwater life history of coho fry and juveniles, water discharge (volume) and water temperatures (Sandercock 1991). Extreme floods are often detrimental to the survival of coho fry and fingerlings (Sandercock (1991). Additionally, low summer flow conditions with a corresponding rapid rise in water temperatures from less than 20°C to >25° C can result in high coho mortalities (Brett 1952 as cited in Sandercock 1991). Prolonged exposure to 0° C can be tolerated by coho during winter month but water temperatures sharply dropping to near 0° C from 5° C may result in mortality to coho juveniles (op. cite.). While the likelihood of these conditions would occur in watersheds within the Plan Area is low, these conditions would be more likely to occur in upper watershed areas than in lower watershed areas.

An especially important environmental condition that will be carefully considered is quality and quantity of over wintering habitat. An important factor in coho fry production is the stability of winter flows (Lister and Walker 1966 as cited in Sandercock 1991). Furthermore, the availability of winter habitat is often overlooked as a limiting factor in juvenile coho production (Nickelson et al. 1992). These authors found that during summer months juvenile coho salmon preferred trench, scour, and plunge pool habitats over other pools or riffle habitats. During winter months the authors found that alcove pools (“sidepools”) and beaver ponds which accounted for only 31% of the areas sampled, accounted for 66% of coho juveniles in surveys of coastal Oregon coho streams. Maximum pool depths for all pools types were highly correlated with juvenile coho density, but for alcove pools, pool depths were not an important correlate (op. cite.). Nickelson et al. (1992) concluded that it was likely for many Oregon coastal streams, coho salmon smolt production is probably limited by winter habitat availability. Larkin 1977 as cited in Sandercock 1991) states that coho abundance in a stream is limited by the number of suitable territories (rocks, LWD, and other structural elements within pool habitats). Careful consideration will be given to the volume of complex habitat available in the project stream including the availability and quality of coho winter habitat for fry and smolt production.

A discussion of the effects of competition and predation follows in the competitive impact section below. As discussed above, proper pre-project evaluation of habitat conditions, and careful consideration of incubation and rearing conditions will minimize risks for increased fry and juvenile mortality. Losses to rearing life stages can never be eliminated, but careful selection of an appropriate stream to conduct this project, will minimize the risk of incurring survival rates that would be lower than those for downstream habitat areas.

The potential risk of lower survival rates for smolts reared from upstream areas as compared to downstream areas are minimal but the rate of survival could be less for upstream reared smolts. This could occur because of greater travel distance to exit the stream (and corresponding increased rates of bird and fish predation) and possibly additional risks of injury and death during transit through the existing migration barriers. Factors that affect timing of smolt emigration include size of the fish, flow conditions, water temperature, dissolved oxygen, day length, and food availability (Shapovalov and Taft 1954, as cited in Sandercock 1991). There are no mitigation measures to avoid a risk to lower smolt survival during emigration other than through proper selection of the project location. Prior to selecting a project, the barrier to anadromy would need to be assessed as to its potential danger for successful smolt emigration. Prior to its selection, the upstream length of the project reach shall be evaluated as to its potential for stranding or injuring out-migrating smolts. If it were found a risk for successful smolt emigration (e.g., significant and deleterious loss during out-migration), this risk to the effectiveness of the project goal would be considered in the final selection of this stream for the project. However, due the nature of the coastal watersheds in the Plan Area (relatively short in total stream miles) it is unlikely that length of upstream reach would be a factor that would significantly affect smolt survival rates.

G.4.2.2 Indirect Impacts

Potential indirect adverse effects of the project could include reduction in fry and juvenile growth rates, and delays in smolt emigration. Fry and juvenile growth rates are primarily affected by water temperature and food availability. Given moderate water temperatures and abundant food supplies coho fry will grow from 30 mm at emergence to 100-130 mm by their second year (prior to emigration) (Roundsfell and Kelez 1949 as cited in Sandercock 1991). It is probable that in upstream stream reaches closer to the headwaters, a stream would have lower average water temperatures as compared to downstream reaches. With lower water temperatures a lower growth rate could occur for coho fry reared in upstream reaches. This however may be off set by greater food productivity in shallower, less turbid upstream stream reaches. In summary, it would be difficult to quantify the potential difference in growth rates of fry and juvenile coho without extensive data collection prior to the section of a project stream. An assessment of the temperature conditions and the food availability will be necessary prior to selecting the project stream and by doing so, the potential growth rates from upstream and downstream locations could be distinguished. It is unlikely that coho growth rates would significantly and adversely impact the effectiveness of the project.

Delays in smolt emigration may occur in upstream locations due to cooler water temperatures and slower growth rates (as discussed above). This could result in smaller overall size during peak out-migration months (March through May) as compared to smolts reared in downstream areas. Also, as previously discussed above delays in exiting the stream may occur with longer distances to travel from upstream rearing areas. However, it is unlikely that these factors would adversely impact the overall effectiveness of the project to rapidly increase the smolt production in the project stream.

G.4.2.3 Competition and Predation Interactions with Resident and Anadromous Species

Predation is a major source of mortality to juvenile coho salmon with the effects varying depending on the predator species present and the stream character (Sandercock

1991). In the Plan Area, cutthroat trout and steelhead/rainbow trout are the principle predators to juvenile coho. Sculpins are known to be important predators on coho fry from emergence (30mm) to approximately 45mm (Patten 1977 as cited in Sandercock 1991). In British Columbia, cutthroat trout were thought to be the principle predators of juvenile coho (Godfrey 1965 as cited in Sandercock 1991). However, in Oregon coho populations, Chapman (1965, as cited in Sandercock 1991) found that cutthroat trout were not significant in coho fry mortality as they were only occasionally taken by cutthroat trout even when coho fry were abundant. Coho fry and smolts are particularly vulnerable to predation when they are congregated in pools and side channels especially in years with high egg-to-fry survival rates (Sandercock 1991).

Predation in upstream project reaches could be a significant impediment to fry and smolt production if predator densities are high. Coho fry and juveniles may be a higher risk to predators in smaller habitat units that would typically be found in upstream reaches of a stream. However, if adequate refuge cover (LWD and SWD) is present in sufficient quantities, predation by trout and other species may be offset in these smaller habitat units as compared to larger less cover containing habitat units in larger downstream reaches. It will be necessary to determine the population densities of potential predator species in any areas in which coho may be introduced. Low densities of predator species such as cutthroat may not necessarily preclude successful and rapid fry and smolt production in project streams.

Coho fry feed principally on insect drift preferring to occupy slower moving sections of smaller streams (Sandercock 1991). Mason (1971, as cited in Sandercock 1991) found that 80% of food contents of coho stomachs was winged dipterans (true flies). Yearling coho may become predatory on fry of their own kind or of other species (Sandercock 1991). However, Shapovalov and Taft (1954, as cited in Sandercock 1991) found that in California coho and steelhead fry were not preyed upon because they emerged from the gravel after coho smolts had emigrated to sea. However, those authors did report that large numbers of chinook fry were preyed upon by coho smolts. Coho smolts would be expected to begin out-migration in March or April ending in June in most years (see Appendix C). Therefore, due their out-migration timing, it is unlikely that coho smolts in upstream reaches, to which their parents were introduced, would prey, to any significant level, on resident trout, steelhead, or coho young-of-the-year.

In these circumstances, it would also be unlikely that chinook fry would be present to be preyed upon by coho yearlings/smolts. Adult chinook are much less athletic than anadromous steelhead and cutthroat trout and therefore would not likely reach habitats upstream of the barrier to anadromy to spawn. Chinook fry that are rearing in downstream reaches however, may be preyed upon during active coho smolt emigration. However, if chinook or other salmonid fry production were sufficiently robust in the downstream reaches, the impacts of predation from out-migrant coho smolts would not likely be significantly large and have little adverse impact to those populations. If populations of these species were not sufficiently robust, then predation from out-migrating coho smolts may be deleterious.

Predation of tailed frog tadpoles by yearling coho produced from introduced adult coho would unlikely occur to any great extent. Rearing tailed frog tadpoles have been shown to prefer higher gradient riffles and faster flowing habitats (Diller, unpubl. report). Tailed frog eggs are deposited in the summer and hatch after four to six weeks (Brown 1990). In coastal regions, the tadpoles typically do not emerge from the nest site until later in the fall (Diller unpubl. report). In contrast, feeding activity of coho fry (which are not predatory on fish or non-insects) decreases in late summer. Young-of-the-year coho move into deeper pool habitats of a stream in the late summer and early fall months remaining in those habitats throughout the winter (Hartman 1965; Scott and Crossman 1973; and Bustard and Narver 1975 as cited in Sandercock 1991). Following winter, yearling coho may prey on tailed frog tadpoles to some extent if they are encountered but the period of predation would be rather short before the coho smolts emigrate out of the rearing stream beginning in March or April.

In summary, the likelihood that coho juveniles produced from introduced spawners would be significant and adversely impact other salmonid juveniles or tailed frogs is small. Predation of these species by yearling coho would be minimal and likely have little impact on those species' populations unless coho were to fully fill the available rearing areas within the stream reaches in which introduction occurred. Even in that event, structurally complex habitats (complexes of LWD, SWD, boulders, cobble, undercut banks and submerged vegetation) within these reaches would likely provide sufficient refuge for steelhead and trout fry and tailed frog tadpoles. Green Diamond will carefully consider existing populations of resident and anadromous species before selecting a stream for introduction will minimize negative impacts of coho predation on those species.

The potential for predation from avian and mammal (i.e. mink, otter, and fishers) species must be considered when selecting the project stream. Upstream areas without sufficient cover, either vegetative, or visual cover such as bubble curtains, would provide greater opportunities for predation by these species. Avian and mammal predation rates on coho may be lower in downstream reaches where fry and smolts may be dispersed over larger areas (lower densities) than that in upstream areas.

Coho fry demonstrate territorial behavior and once selected remain in a locality for relatively long periods (Hoar 1958 as cited in Sandercock 1991). Displacement from their preferred territory may come from a cohort or a competitor fish species. Conversely sufficiently large numbers of juvenile coho may displace other juvenile resident or anadromous species if habitat quality and quantity is inadequate. If sufficient habitat structural complexity is available and the competitor or coho cohort density is sufficient low the individual coho fry will remain in its chosen territory or it may choose to relocate. If excessive disturbance, harassment or displacement from its chosen territory occurs, coho will relocate and avoid the competitor. If displacement continues with no ability to rest on the stream bottom, juvenile coho will progressively be displaced downstream (Chapman 1962 as cited in Sandercock 1991). It that case displaced coho juveniles may, given sufficient numbers and habitat limitation, displace other resident and or anadromous juveniles in those downstream reaches. The essential parameters here are availability and abundance of structurally complex habitat and the density of competitors. Similar to predation, the effects of competition on coho fry and smolt productivity, is minimized if sufficient cover and territory is available and competitor density is not excessive; thus, the existence of sufficient cover and territory, as a ratio of competitor density, will be a project site selection criterion.

G.4.3 Conclusions

The most significant measure to ensure that the project will be effective in meeting its goal is the careful consideration in selecting a project location. Attention will be given to the necessary habitat conditions, the density and the species of potential competitor and predator species, and the limiting factor(s) to coho fry and smolt production in both the upstream and downstream areas being considered for introduction. These elements will be carefully weighed together and compared to the potential productivity that would occur in areas within the project area if coho introduction were not attempted. Green Diamond will only select a project stream location that meets all of the criteria necessary discussed herein. Accordingly, no unacceptable adverse impacts to coho salmon or other Covered Species in the Plan Area are expected to occur.

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