Appendix C11. Headwater Amphibian Studies and Monitoring

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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 codominant 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.

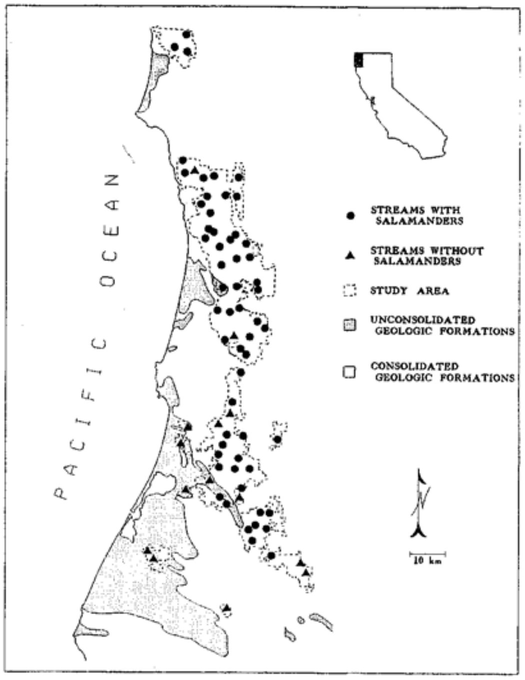

FIG. 1. Map of study area showing consolidated and unconsolidated formations, and streams with and without the southern torrent salamander (*Rhyacotriton variegatus*), north coastal California.

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).

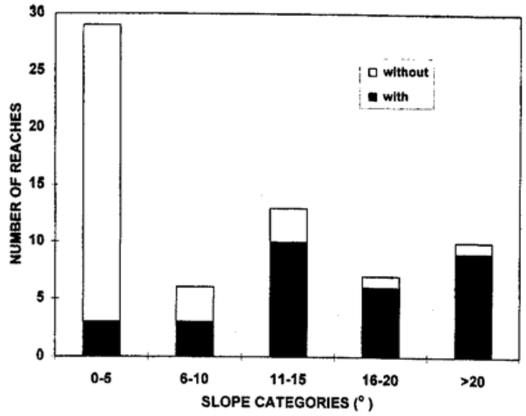

FIG. 2. Number of stream reaches with and without *Rhyacotriton variegatus* in five categories of stream slope (gradient), north coastal California.

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) $\bar{x}$	Sites with (N = 31) SD	Sites without (N = 33) $\bar{x}$	Sites without (N = 33) SD	*P*
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 $5.5/m^2$ (overall average = $0.83/m^2$).

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 stastitical 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

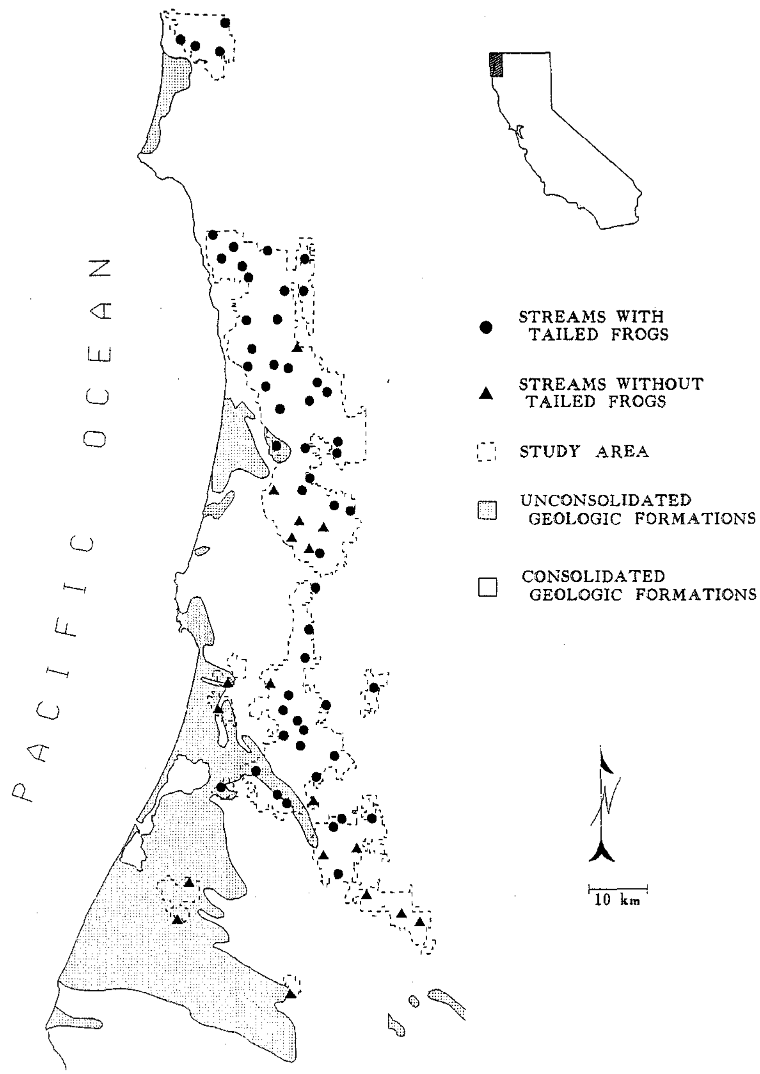

FIG. 1. Map of study area showing consolidated and unconsolidated geological formations, and streams with and without *Ascaphus truei,* north coastal California. Data obtained from landscape survey.

= 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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