AMPHIBIAN, FISH STOCKING, AND HABITAT RELATIONSHIPS IN SISKIYOU MOUNTAIN WILDERNESS LAKES, CALIFORNIA AND OREGON

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ABSTRACT—During 1999 and 2000, 10 of the 13 high-elevation lakes in and near the Red Buttes Wilderness in the Siskiyou Mountains were surveyed for amphibians. Five of the lakes had been stocked with non-native brook trout (*Salvelinus fontinalis*) for over 30 y, while fish were absent from the other 5 lakes. Pacific treefrog (*Pseudacris regilla*) larvae were observed in 40% of fish-bearing lakes and 80% of fishless lakes, rough-skinned newts (*Taricha granulosa granulosa*) were present in all study lakes, and coastal giant salamanders (*Dicamptodon tenebrosus*) were present in 40% of fish-bearing lakes. Pacific treefrog larvae were significantly more abundant in fishless lakes, while rough-skinned newt median abundances were identical between fish-bearing and fishless lakes. Differences in Pacific treefrog abundances and distribution between fish-bearing and fishless lakes were likely related to the presence of brook trout, but might also have been influenced by other factors such as lake morphometry and abundance of aquatic vegetation. Modifying the number or type of fish stocked in the Red Buttes Wilderness could reduce effects from fish stocking on Pacific treefrog populations.

Key words: Pacific treefrog, *Pseudacris regilla,* rough-skinned newt, *Taricha granulosa,* coastal giant salamander, *Dicamptodon tenebrosus,* brook trout, *Salvelinus fontinalis,* amphibians, fish stocking, wilderness lakes, Siskiyou, California, Oregon

Introduction of exotic species is a significant threat to wilderness ecosystems and a management issue that merits further study (Cole and Landres 1996). Exotic fish stocking is still permitted in many wilderness areas although it has created division among the public and management agencies (Duff 1995; Fraley 1996; Wiley 2003). Studies from several geographic regions show that stocking exotic fishes can negatively affect native lentic amphibian populations (Kats and Ferrer 2003; Dunham and others 2004). For example, lakes in California's John Muir Wilderness that were stocked with exotic salmonids contained fewer mountain yellow-legged frogs (*Rana muscosa*) and Pacific treefrogs (*Pseudacris regilla,* Crother and others 2003) than fishless lakes in adjacent Kings Canyon National Park (Bradford 1989; Knapp and Matthews 2000; Matthews and others 2001). In North Cascades and Mount Rainier National Parks, Washington, long-toed (*Ambystoma macrodactylum*) and northwestern (*A. gracile*) salamander larvae were less abundant in lakes stocked with trout (Liss and others 1995; Tyler and others 1998a; Larson and Hoffman 2002). Fish-bearing lakes in the Frank Church-River of No Return Wilderness in Idaho had significantly lower amphibian abundances than fishless lakes (Pilliod and Peterson 2001), and montane lakes stocked with trout in northeastern Oregon had fewer Pacific treefrogs than fishless lakes (Bull and Marx 2002).

Research on the effects of fish stocking on amphibian populations has been focused in the Sierra Nevada, Rocky, and Cascade Mountain ranges. Larson and Hoffman (2002) cautioned that ecological effects from fish stocking should not be broadly extrapolated among geographic regions. Consequently, this study sought to determine if similar relationships to those observed in other regions exist in high-elevation lakes in a relatively undisturbed area of the Siskiyou Mountains.

The Siskiyou Mountains in southwestern Oregon and northwestern California are part of the Klamath-Siskiyou region, which has exceptionally high floral and faunal diversity (DellaSala and others 1999) and which sup-

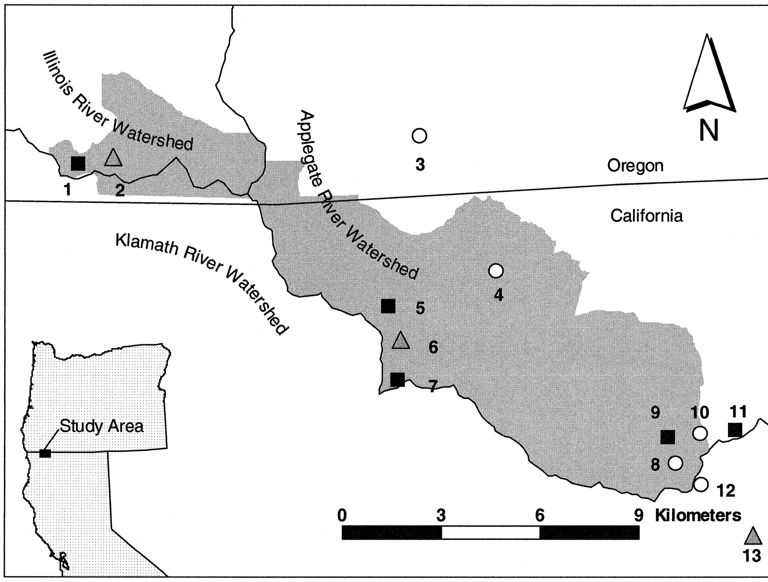

FIGURE 1. Map of the study area. Shaded area depicts the Red Buttes Wilderness Area, solid black lines are watershed boundaries, white circles represent fishless lakes, black squares represent fish-bearing lakes, and gray triangles represent lakes named on US Geological Survey 7.5 min quadrangle maps that were not surveyed in 1999 or 2000. Lake numbers are: 1 = Tannen, 2 = East Tannen, 3 = Hinkle, 4 = Frog Pond, 5 = Azalea, 6 = Snake, 7 = Lonesome, 8 = Towhead, 9 = Hello, 10 = Moraine, 11 = Echo, 12 = Lilypad, 13 = Goodbye.

ports a rich amphibian assemblage different from those found in other montane regions of the western United States (Bury and Pearl 1999). The Siskiyou Mountains possess relatively few high-elevation lakes because this area was largely unaffected by Pleistocene glaciation that helped form lake basins in other western regions (Johnson and others 1985). Subsequently, the 10 lakes surveyed in this study were the majority (76.9%) of the 13 named lakes within the Red Buttes Wilderness and adjacent area (US Geological Survey 7.5 min quadrangle maps: Figurehead Mountain, Grayback Mountain, Kangaroo Mountain, Oregon Caves).

Fish are naturally absent from surveyed lakes in the Red Buttes Wilderness. Trout, however, have been stocked in most of the larger lakes for over 30 y (Hoover 1969; USDA FS 1998). Little information exists describing effects of fish stocking on native amphibians in this region. In addition, many of the lakes in the region, including the lakes in this study, have not been formally surveyed for amphibians. Bury and Pearl (1999) described fish stocking as a threat to lentic amphibians in this region and stressed the importance of understanding the distribution and ecology of native Siskiyou Mountains amphibians to resource management and conservation. This study sought to compare amphibian assemblages and physical parameters between fish-bearing and fishless lakes in the Red Buttes Wilderness.

METHODS

This study was conducted in and immediately adjacent to the Red Buttes Wilderness, Siskiyou County, California and Josephine County, Oregon. This 8200-ha USDA Forest Service-administered wilderness area is located in the Inland Siskiyous ecoregion, a division of the Klamath Mountains ecoregion (Omernik and others 1997). Steep, rugged mountains with highly dissected drainages and patches of mixed conifers, hardwoods, and chaparral define the Inland Siskiyous ecoregion. The Red Buttes Wilderness divides the Applegate, Klamath, and Illinois River watersheds along the California-Oregon border (Fig. 1). Elevations in the wilderness area range from 850 to 2050 m above sea level; study lakes were 1450 to 1745 m above sea level (Table 1).

TABLE 1. Fish stocking rates and physical variables of lakes surveyed in 1999 and 2000.

Lake	Fish present[a]	Stocking rate (fish/ha/y)[b]	Elevation (m)	Maximum depth (m)	Surface area (ha)	Shoreline devel-opment	Vegetation abundance[c]	Dominant substrate
Azalea	Yes	786	1645	1.6	1.91	1.12	Low	Silt
Echo	Yes	770	1600	3.2	0.65	1.07	Low	Silt
Hello	Yes	—	1450	2.1	1.14	1.46	Moderate	Silt
Lonesome	Yes	505	1670	4.5	0.99	1.20	Low	Silt
Tannen	Yes	323	1555	9.0	3.10	1.04	Low	Silt
Frog Pond	No	—	1480	1.2	0.82	1.28	High	Silt
Hinkle	No	—	1705	0.4	1.03	1.09	Moderate	Silt
Lilypad	No	—	1745	1.1	0.44	1.05	High	Silt
Moraine	No	—	1720	4.5	1.27	1.47	None	Bedrock
Towhead	No	962	1645	4.4	0.26	1.07	Low	Sand

[a] Hello Lake was not listed on aerial stocking reports and Towhead Lake was listed on aerial stocking reports.
[b] Stocking rates were calculated by dividing fish numbers on stocking reports by lake surface areas calculated by Rogue River-Siskiyou National Forest GIS coverage.
[c] Vegetation abundance categories are described by Corn (1997), where none = 0%, low = 1–25%, moderate = 26–50%, and high = 51–100%.

Ten of the 13 lakes within the study area were surveyed. Eight of 10 lakes (4 fish-bearing and 4 fishless) were surveyed once between 27 July and 4 September 1999. Two additional lakes (1 fish-bearing and 1 fishless) were surveyed on 20 September 2000. Seven of the 10 study lakes were located within the wilderness and the other 3 lakes were 0.2 to 3.0 km outside the wilderness boundary. Lakes were deliberately selected from different drainages and elevations to provide broad coverage of the geographic area.

Fish status was assessed by consulting Oregon Department of Fish and Wildlife and California Department of Fish and Game aerial stocking reports. Shoreline surveys, hook-and-line sampling, and snorkeling were used to confirm fish presence or absence in all lakes. Fish presence was usually determined by using polarized sunglasses during amphibian visual encounter surveys because of the shallow depth and high clarity of most lakes. Hook-and-line sampling was used in lakes where fish were not observed during amphibian surveys by spending about 3 h (range: 2 to 4 h) angling with weighted flies. Daytime snorkeling (Thurow 1994) was used to determine fish presence in 4 deeper lakes where the bottom was not visible and fish were not detected during amphibian surveys or hook-and-line sampling. Snorkeling effort ranged between 15 and 30 min depending on lake size, and was focused in deeper areas of the lakes that were not visible from shore.

A combination of visual encounter surveys, dip net sweeps, and funnel trapping was used to detect amphibians (Bury and Major 1997). Only visual encounter surveys were used to measure amphibian abundance. Visual encounter surveys were conducted by walking in a zigzag pattern through littoral zones <1 m deep, around the entire perimeter of the lake, except where cliffs were present (<3% of surveyed lake perimeters). The observer reported species and lifestage (larva or adult) to a person on the shore recording data (Crisafulli 1997). Abundance was calculated by dividing the number of amphibians observed by the number of minutes searched.

D-framed dip nets (500 μm mesh) and funnel traps were used to complement visual encounter surveys for detecting amphibian species. Net and trap data were used to determine amphibian presence and were not used in abundance estimates. Dip nets were swept through benthic detritus and aquatic vegetation when present. Funnel traps were used in 8 of 10 lakes (5 fish-bearing and 3 fishless). Four 3-mm mesh collapsible minnow and 6 plastic bottle traps (Adams and others 1997) were placed on lake bottoms within 1.0 m of shore in 0.2 to 0.5 m of water. Five traps in each lake were baited with cured salmon eggs and 5 were left unbaited. Amphibian traps were equally spaced around the lake perimeter and set overnight (12 to 16 h), when possible, near cover objects such as wood and aquatic vegetation. Captured amphibians were identified using field guides (Nussbaum and others 1983; Stebbins 1985; Leonard and others 1993) and released at capture sites.

Maximum depth, Secchi disk transparency

(Wetzel 2001), substrate type, and percentage of lake supporting vegetation were assessed for each lake. Maximum depth and Secchi disk transparencies were measured by submerging a 20-cm-diameter Secchi disk from a float tube over the deepest part of the lake. Each lake was assigned a dominant substrate type based on the relative abundance of an estimated particle size (silt, sand, gravel, bedrock). Percent vegetation cover (combined submersed, floating-leaf, and emergent macrophytes) was visually estimated using categories described by Corn (1997), where none = 0%, low = 1 to 25%, moderate = 26 to 50%, and high = 51 to 100%. Rogue River-Siskiyou National Forest geographic information system (GIS) coverages were used to calculate lake perimeter, surface area, and shoreline development after Wetzel (2001): $D_L = L/2\sqrt{(\pi A_0)}$, where D_L = shoreline development, L = length of shore line, and A_0 = surface area of the lake. Fish stocking rates (fish/ha/y) were calculated by dividing numbers of fish listed in stocking reports by surface areas determined by GIS.

Mann-Whitney *W* tests of medians (Moore and McCabe 1999) were used to test for differences in survey effort, survey date, amphibian abundance, surface area, maximum depth, shoreline development, and elevation between fish-bearing and fishless lakes. A 2-sided Mann-Whitney *W* test was used for all comparisons except amphibian abundance because the alternate hypothesis predicted medians were unequal between fish-bearing and fishless lakes. When comparing amphibian abundances, 1-sided Mann-Whitney *W* tests were used because the alternate hypothesis predicted amphibian abundance in fishless lakes was greater than fish-bearing lakes.

Chi-square tests for homogeneity of proportions were used to compare amphibian assemblages between fish-bearing and fishless lakes. Fisher's exact test was used to relate fish presence with vegetation abundance or substrate composition with 2 × 2 contingency tables (Moore and McCabe 1999). The contingency table for vegetation abundance used 2 classes: none to low (0 to 25%) and moderate to high (26 to 100%); the contingency table for substrate composition used silt and other for classes. STATGRAPHICS Plus 5.0 (Statistical Graphics Corporation) was used to perform Mann-Whitney *W* and chi-square tests. Excel 2000 (Microsoft) with Analyse-it 2003 extension (Analyse-it Software Ltd.) was used to perform Fisher's exact test.

Results

The physical dimensions and associated aquatic habitat of study lakes varied relative to fish presence (Table 1). Differences in lake size were not significant; however, median surface area was 1.4 × greater and maximum depth was 2.7 × greater in fish-bearing lakes (1.14 ha, 3.2 m) than in fishless lakes (0.82 ha, 1.2 m) (surface area Mann-Whitney $W = 6.0$, $P > 0.2$; maximum depth Mann-Whitney $W = 6.5$, $P > 0.2$). Although association between vegetation abundance and fish presence was not significant (Fisher's exact test, $n = 10$, $P > 0.5$), 4 of 5 fish-bearing lakes had low abundance and 3 of 5 fishless lakes had moderate or high abundance of vegetation (Table 1). Median elevation of fishless lakes was 150 m higher than fish-bearing lakes (1750 > 1600 m, Mann-Whitney $W = 19.5$, $P > 0.1$). Substrate composition and shoreline development were similar between fish-bearing and fishless lakes (substrate composition Fisher's exact test, $n = 10$, $P > 0.4$; shoreline development = 1.12 and 1.09, Mann-Whitney $W = 13.5$, $P > 0.9$). Secchi disk transparencies could not be calculated and compared because the Secchi disk contacted lake bottoms before disappearing due to the high clarity and shallow depth of most lakes.

Brook trout (*Salvelinus fontinalis*) were present in all survey lakes listed in stocking reports except Towhead Lake, which was fishless (Table 1). Likewise, brook trout were absent from all surveyed lakes not listed in stocking reports with the exception of Hello Lake. Snorkeling and angling results confirmed that shoreline surveys accurately assessed fish presence or absence in all lakes.

Pacific treefrog, rough-skinned newt (*Taricha granulosa granulosa*), and coastal giant salamander (*Dicamptodon tenebrosus*) were the only amphibians detected. Pacific treefrog larvae were observed in 4 of 5 fishless lakes and 2 of 5 fish-bearing lakes, rough-skinned newts were found in all lakes, and coastal giant salamanders were observed in 2 of 5 fish-bearing lakes (Table 2). Adult and larval forms of all observed amphibian species were detected, although adult Pacific treefrogs and terrestrial coastal gi-

TABLE 2. Number of amphibians observed in 10 lakes during visual encounter surveys in 1999 and 2000. Abundances (individuals/min) are in parentheses. Amphibian acronyms are PSRE = Pacific treefrog, TAGR = rough-skinned newt, DITE = coastal giant salamander.

Lake	Fish present	Survey date	Sampling effort (min)	PSRE larvae	PSRE adult	TAGR adult[a]	DITE larvae[b]	Total amphibians
Azalea	Yes	07/27/99	65	5 (0.1)	0 (0.0)	45 (0.7)	0 (0.0)	50 (0.8)
Echo	Yes	07/30/99	35	0 (0.0)	0 (0.0)	40 (1.1)	0 (0.0)	40 (1.1)
Hello	Yes	09/20/00	36	0 (0.0)	0 (0.0)	5 (0.1)	2 (0.1)	7 (0.2)
Lonesome	Yes	07/26/99	45	20 (0.4)	0 (0.0)	20 (0.4)	0 (0.0)	40 (0.9)
Tannen	Yes	09/04/99	35	0 (0.0)	0 (0.0)	22 (0.6)	2 (0.6)	24 (0.7)
Frog Pond	No	08/15/99	55	250 (4.5)	12 (0.2)	450 (8.2)	0 (0.0)	712 (12.9)
Hinkle	No	08/10/99	45	500 (11.1)	0 (0.0)	25 (0.6)	0 (0.0)	525 (11.7)
Lilypad	No	07/31/99	37	20 (0.5)	0 (0.0)	20 (0.5)	0 (0.0)	40 (1.1)
Moraine	No	09/20/00	13	0 (0.0)	0 (0.0)	8 (0.6)	0 (0.0)	8 (0.6)
Towhead	No	08/11/99	35	40 (1.1)	0 (0.0)	99 (2.8)	0 (0.0)	139 (4.0)

[a] Rough-skinned newt larvae were detected during dip net sweeps in Azalea, Echo, Hello, Frog Pond, and Lilypad lakes.
[b] Value for Hello Lake includes 1 terrestrial coastal giant salamander.

ant salamanders were each only documented in 1 lake.

The same amphibian species and lifestages were captured in funnel traps as were observed during visual encounter surveys, although rough-skinned newt larvae in 3 of 5 fish-bearing and 2 of 5 fishless lakes were only captured in net sweeps. Visual encounter survey effort averaged 36 min (range: 13 to 65) per lake and was similar between fish-bearing and fishless lakes (Mann-Whitney $W = 11.5$, $P > 0.9$). Median calendar survey date was 5 August (217 out of 365) and not significantly different between fish-bearing and fishless lakes (Mann-Whitney $W = 16.5$, $P > 0.4$).

Visual encounter surveys detected 1585 amphibians: 1424 (89.8%) in fishless lakes, and 161 (10.2%) in fish-bearing lakes (Table 3). Median total amphibian abundance was 5 times higher in fishless lakes (4.0 individuals/min) than in fish-bearing lakes (0.8 individuals/min), although the difference was not significant (Mann-Whitney $W = 20.5$, $P > 0.05$). A significant difference in Pacific treefrog median larval abundances between fish-bearing (0.0 individuals/min) and fishless (1.1 individuals/min) lakes was detected (Mann-Whitney $W = 21.5$, $P < 0.04$). Median rough-skinned newt abundances were identical between fish-bearing and fishless lakes (0.6 individuals/min, Mann-Whitney $W = 16.0$, $P > 0.2$) as were the median abundances of coastal giant salamanders (0.0 individuals/min, no test output; Table 3).

The generalized amphibian assemblage in fish-bearing lakes contained a significantly higher proportion of rough-skinned newt adults and lower proportion of Pacific treefrog larvae than the generalized amphibian assemblage in fishless lakes ($\chi^2 = 51.3$, df = 1, $P < 0.001$). Rough-skinned newt adults were the dominant amphibians observed during visual encounter surveys in fish-bearing lakes, whereas rough-skinned newt adults and Pacific tree-

TABLE 3. Number and median abundances (individuals/min) of amphibians observed during visual encounter surveys in 5 fish-bearing and 5 fishless lakes in 1999 and 2000. Amphibians are PSRE = Pacific treefrog, TAGR = rough-skinned newt, DITE = coastal giant salamander.

	Fish-bearing (n = 5)			Fishless (n = 5)		
Amphibian	No. observed	Percent of total	Median abundance	No. observed	Percent of total	Median abundance
TAGR adults	132	82.0	0.6	602	42.3	0.6
PSRE larvae	25	15.5	0.0	810	56.9	1.1
PSRE adults	0	0.0	0.0	12	0.8	0.0
DITE larvae[a]	4	2.5	0.0	0	0.0	0.0
Total	161	100.0	0.8	1424	100.0	4.0

[a] Value includes 1 terrestrial salamander.

frog larvae were co-dominant in fishless lakes (Table 3). Coastal giant salamanders and Pacific treefrog adults comprised only 3.3% (16) of all amphibians observed during visual encounter surveys.

Discussion

Lentic amphibian species richness in the study area was low relative to regional amphibian richness (Bury and Pearl 1999) and overall biodiversity (DellaSala and others 1999). The study area also lacked lentic amphibians such as ranid frogs and ambystomatid salamanders, which occur in other areas within the Klamath-Siskiyou and other western regions (Bury and Pearl 1999). My results indicate that the presence of fish and the distribution and abundance of Pacific treefrogs in lakes of the Red Buttes Wilderness are negatively associated. However, differences in aquatic habitat characteristics between fishless and fish-bearing lakes also may have influenced Pacific treefrog demographics independently of fish presence. Fish presence or habitat variables did not appear to affect the abundance or distribution of rough-skinned newts. Coastal giant salamanders were absent from fishless lakes and observed in only 2 fish-bearing lakes.

Pacific treefrogs in fishless survey lakes were twice as common and significantly more abundant than in fish-bearing survey lakes. In comparison, Matthews and others (2001) found that Pacific treefrogs were 2.4 times more likely to occur in high-elevation Sierra Nevada lakes that lacked fish than in those stocked with trout. Several chorus and treefrog species (Family Hylidae) are less likely to occur in fish-bearing water bodies for reasons related to habitat, predation, or indirect effects of fish (Brönmark and Edenhamn 1994; Bradford and others 1998; Smith and others 1999; Adams 2000; Gillespie 2001; Matthews and others 2001; Bull and Marx 2002). In this study, predation on Pacific treefrogs seems likely because brook trout feed opportunistically on a variety of prey items (Dawidowicz and Gliwicz 1983) and prefer large, conspicuous prey (Allan 1978). Further, brook trout are fecund and can spawn in lentic habitats, which can lead to high-density populations of stunted individuals that can substantially reduce or eliminate conspicuous prey taxa in lakes (Reimers 1958; Bahls 1991). Bull and Marx (2002) found a significant negative relationship between Pacific treefrog larval abundance and the presence of exotic brook trout, but not stocked rainbow trout (*Oncorhynchus mykiss*). Pacific treefrogs may be especially susceptible to brook trout predation if they cannot detect brook trout chemical cues, as was observed in experiments with Pacific treefrogs and other exotic predators (Pearl and others 2003).

The abundance and distribution of Pacific treefrogs in fish-bearing survey lakes may also have been associated with breeding site selection. Most fish-bearing lakes were not optimal breeding habitat for Pacific treefrogs, and adults may have used alternate sites for reproduction. Skelly (1995, 1996) concluded from laboratory and field experiments that the related western chorus frog (*Pseudacris triseriata*) was better adapted to reproduce in shallow, ephemeral ponds than in deeper, more permanent water bodies. Pacific treefrogs also use small, shallow, often ephemeral water bodies for breeding, especially if aquatic vegetation is present (Nussbaum and others 1983; Leonard and others 1993).

On the other hand, Bull and Marx (2002) found that lake size, maximum depth, and amount of emergent vegetation were not significant predictors of Pacific treefrog larval abundance in northeastern Oregon lakes. However, they did find that brook trout presence and Pacific treefrog larval abundance were negatively related. In the Sierra Nevada Mountains, Pacific treefrogs were over twice as common in high-elevation fishless lakes than in fish-bearing lakes, after accounting for effects of significant habitat variables (Matthews and others 2001). In this study, Pacific treefrog larvae were more abundant in fishless lakes, which were generally shallower, smaller, and more vegetated than fish-bearing lakes.

The abundance and distribution of the rough-skinned newt was similar between fish-bearing and fishless survey lakes. Taylor (1984) also found no difference in rough-skinned newt abundance between lakes stocked with trout and fishless lakes in the central Cascade Mountains of Oregon. Coexistence of trout and rough-skinned newts may be possible because newt tetrodotoxin can reduce or prevent predation by fish (Brodie 1968; Efford and Mathias 1969). Although brook trout may not prey directly on rough-skinned newts, trout could potentially reduce newt fitness by competing with them for invertebrate prey (Efford and Tsumura 1973). Laboratory experiments examining fish-salamander interactions suggested that rainbow trout predation on and competition between larval northwestern and long-toed salamanders limits larval salamander growth and survival (Tyler and others 1998b). Also, because northwestern salamanders and rough-skinned newts share similar diets (Taylor 1984), it is possible that trout, in high-elevation lakes with low productivity, could affect newts through competitive interference. The similarity between the abundances and distribution of rough-skinned newts in fish-bearing and fishless survey lakes did not support this hypothesis.

This study was similar to others from different geographic areas that show fish stocking in montane lakes can reduce native amphibian populations (Kats and Ferrer 2003; Dunham and others 2004). These findings also agreed with 2 studies that found a negative relationship between stocked trout presence and Pacific treefrog abundance (Matthews and others 2001; Bull and Marx 2002). However, low sample size and the confounding variable of habitat quality complicated relationships between fish and amphibians in this study.

This study showed that the presence of introduced brook trout and the abundances and distribution of the Pacific treefrog in Siskiyou Mountain lakes were negatively associated. Yet, because the Pacific treefrog is a common, abundant, resilient species able to use habitats that cannot sustain fish (Nussbaum and others 1983; Leonard and others 1993), the effect of fish stocking on treefrog population dynamics in the Siskiyou Mountains remains unresolved. However, because brook trout may have greater negative impacts on native amphibians and other fauna than other salmonids (Dunham and others 2004), fishery managers should consider eliminating or reducing brook trout stocking in lakes where stunting is observed or adequate natural production exists and stocking different species that cannot reproduce in high-elevation lakes. All stocking rates in the study area exceeded 250 fish/ha/y, a rate Reimers (1958) suggested could reduce native fauna in high-elevation lakes. Sterilized brook trout could also be used to reduce impacts to native fauna (Lemoine and Smith 1980; Dubé and others 1991; Galbreath and Samples 2000). Finally, a thorough inventory of the distribution and availability of shallow, fishless wetlands within the study area would greatly enhance management for the continued presence of the Pacific treefrog in the Red Buttes Wilderness.

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