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Monitoring the short-term effects of prescribed fire on an endemic mollusk in the dry forests of the eastern Cascades, Washington, USA

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ABSTRACT

The restoration of natural fire regimes has emerged as a primary management objective within fire-prone forests in the interior western US. However, this objective becomes contentious when perceived to be in conflict with the conservation of rare or endemic species. We monitored the effects of two forest restoration treatments, spring- vs fall-prescribed burning, on the density of the endemic Tiny Canyon mountainsnail (*Oreohelix sp.*). We used a randomized block design with three replicates of each of the treatments and controls, and analyzed our data using multivariate repeated measures analysis of variance. We conducted pre-treatment surveys for mountainsnails and post-treatment surveys at three time periods: within two weeks of the treatment, the next snail season following the treatment (next spring or fall), and one year following the treatments. We did not detect any statistically significant differences in mountainsnail densities as a result of the spring-burn or fall-burn treatments, time of survey, or treatment × time interaction. The burns resulted in a fine-scale mosaic that included un-burned and lightly burned areas that acted as refuge for mountainsnails. We recommend that the application of prescribed burning as a restoration treatment within mountainsnail habitat be conducted under prescriptions that create a mosaic of burn conditions, including small unburned areas, and that prescribed fire return intervals mimic natural fire intervals (10–40 years).

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1. Introduction

Fire plays an important role in many ecosystems making fire management central to the conservation of biodiversity (Myers, 1997; Driscoll et al., 2010). Restoration of natural fire regimes has been suggested as a coarse filter for conservation (Agee, 2003; Prather et al., 2008), however, several authors have identified potential conflicts with the conservation of rare or endemic species with limited ranges (Myers, 1997; Agee, 2003; Prather et al., 2008; Gaines et al., 2010a). Forest restoration in dry forests often includes the use of mechanical thinning and prescribed fire to move patch structure and composition towards the natural range of variability (Harrod et al., 1999; Wright and Agee, 2004; Reynolds and Hessburg, 2005). Potential conflicts include the removal of habitat for species associated with dense, multi-layered forest structure (Gaines et al., 2007; Collins et al., 2010). The interactions between fire, fire management and maintaining habitat for rare or endemic species is of particular interest within fire-prone forests of the interior west (Agee, 1993; Brown et al., 2004; DellaSala et al., 2004; Hessburg et al., 2005; Noss et al., 2006; Prather et al., 2008).

The Tiny Canyon mountainsnail (*Oreohelix sp.*) (formerly known as the Chelan mountainsnail, Morales et al., in prep) is endemic to the dry forests of eastern Washington (Burke et al., 1999; Duncan, 2005; Weaver et al., 2010). The Tiny Canyon mountainsnail occurs within dry and mesic open-canopy forests that were historically dominated by ponderosa pine (*Pinus ponderosa*) and Douglas-fir (*Pseudotsuga menziesii*) cover types (Lillybridge et al., 1995) and low-mixed severity fire regimes (Hessburg et al., 1999, 2007). However, decades of fire exclusion, selective timber harvest, and grazing have resulted in dense dry and mesic forests and increased susceptibility to high severity fire (Wright and Agee, 2004; Hessburg et al., 2005). The potential of uncharacteristic high-severity fire has been identified as one of the risk factors for the conservation of the Tiny Canyon mountainsnail and restoration of dry forest habitats has been identified as an important management action to reduce the risk of high-severity fires (Burke et al., 1999; Duncan, 2005). However, managers currently lack an understanding of the effects that forest restoration treatments, such as prescribed fire, have on mountainsnail species (Burke et al., 1999; Lyon and Smith, 2000; Duncan, 2005; Gaines et al., 2005; Driscoll et al., 2010). As Driscoll et al. (2010) illustrate, fire management for biodiversity conservation requires knowledge of the mechanistic responses of species to fire regimes, which is the focus of this research.

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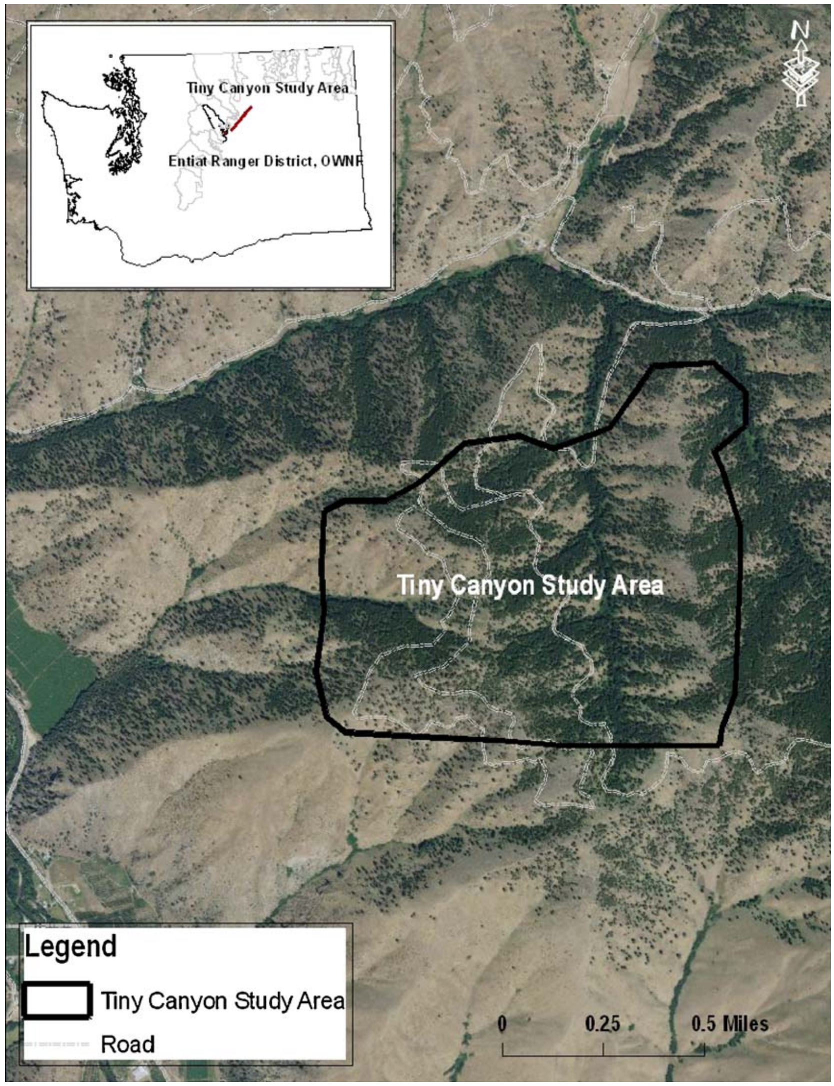

Fig. 1. Study area used to monitor the effects of spring and fall prescribed burning on the densities of Tiny Canyon mountainsnails, eastern Cascades, Washington.

Previous research on land snails has shown their populations to be resilient to both human induced (Watters et al., 2005; Strom et al., 2009) and natural disturbances (Kiss and Magnin, 2006) as long as habitats are allowed to recover following the disturbance. For example, Kiss and Magnin (2006) found that Mediterranean land snail communities were resilient to fires provided that the time lapse between two successive disturbances is longer than the time required for population recovery (about 5 years). Land snail population recovery following fire was aided by small unburned patches and cryptic microsites that served as refuges (Kiss and Magnin, 2003, 2006).

Because of the limited number, dispersal ability, and distribution of the Tiny Canyon mountainsnail, managers must carefully consider the effects of proposed management actions. Gaines et al. (2005) developed a predictive model to aid managers in determining where to focus surveys and carefully evaluate proposed management actions such as thinning or prescribed fire. However, because of the broad-scale focus of their analysis, they could not draw inferences about how management actions might affect the Tiny Canyon mountainsnail. They suggested that an experimental design with careful monitoring of different management actions be the next logical step in understanding the effects of dry forest restoration treatments (Gaines et al., 2005). In this paper, we present results from monitoring an adaptive management experiment designed to measure the effects of spring and fall prescribed fires on the Tiny Canyon mountainsnail. This information should prove useful in the development of a conservation strategy for this rare, endemic species, and other mountainsnail species in the area.

2. Methods

2.1. Study area

The study occurred within areas identified as dry forests in the vicinity of Tiny Canyon in the Entiat River watershed, Washington state, USA (Fig. 1). Average annual precipitation is 20–30 cm/year, falling mostly as snow. The elevations range from 670 to 790 m. This area is composed of ponderosa pine (*Pinus ponderosa*) and Douglas-fir (*P. menziesii*) forested plant associations (Lillybridge et al., 1995). The area has been the focus of dry forest restoration treatment planning efforts and previous surveys determined there were adequate populations of Tiny Canyon mountainsnails for study.

The study occurred in forested stands that are even-aged, about 90 years old composed of Douglas-fir with occasional, remnant older ponderosa pine. Tree cover was relatively high (40–60%) and average diameter at breast height (dbh) was about 15–18 cm. The understories are dominated by pinegrass (*Calamgrostis rubescens*) and arrowleaf balsamroot (*Balsamorhiza sagittata*).

2.2. Sampling design

We used a paired block design with three replicates of each of the treatments ($n = 6$ blocks; 3 spring-burn, 3 fall-burn) and controls ($n = 6$; 1 control for each treatment block) for a total of 12 blocks to monitor the effects of prescribed fire on the density of Tiny Canyon mountainsnails. The treatments were spring-prescribed fire and fall-prescribed fire. Treatments and controls were randomly assigned within each block. Each treatment block was 30 m × 30 m in size. The treatments were completed during October of 2006 and April of 2007. All blocks contained snails based on pre-treatment surveys. The treatment blocks were distributed over a 170 ha area, representing >50% of the known distribution of this species (Weaver et al., in prep).

2.3. Vegetation sampling and burn conditions

Each block was sampled to quantify the pre-treatment vegetation conditions. A 0.02 ha circular sample plot was established within the center of each block to determine the percent cover of grasses, Arrowleaf balsamroot, shrubs, moss/lichens, litter, bareground/rock, and tree cover. Arrowleaf balsamroot was specifically quantified because Burke et al. (1999) noted and our pre-treatment surveys suggested this plant to be important in determining where the mountainsnail occurred.

During the burning treatments information was collected on the duration of time that fire (flaming combustion) was present on the study plot, air temperature, relative humidity and % soil moisture. Two 150 g soil samples were collected from just under the duff layer at each block just prior to the burn and transported to the lab where they were weighed, dried and then re-weighed to determine the soil moisture.

2.4. Snail surveys

Snail surveys followed the protocol described in Duncan et al. (2003). Pre-treatment surveys were completed in the spring and fall 2005. Post-treatment surveys were implemented at three different times: within two weeks after the treatment, the next snail survey season following the treatment, and one year post-treatment. Our snail surveys were conducted across the entire sample block until all live snails and empty shells were located. All live snails and empty shells were left on site.

2.5. Data analysis

We used *F*-tests to determine if any differences occurred in vegetation and burn condition variables. Tests were run on percent grass cover and percent tree cover to determine if there were any pre-treatment differences by comparing treatment blocks ($n = 3$) to their paired control blocks ($n = 3$). Post-treatment tests were run on the duration of the burn and percent soil moisture by comparing blocks assigned to spring ($n = 3$) vs fall ($n = 3$) burns. Percentages were arcsin transformed prior to analysis. All significance tests were set *a priori* at $p = 0.05$.

Because land snails have very limited mobility (Cameron et al., 1980; Baur and Baur, 1990; Pfenninger, 2002), we assumed no immigration or emigration into our sample blocks. Therefore, we determined snail density for each treatment and control block at each sampling interval by simply dividing the number of live snails by the size of the treatment block (900 m^2). We assumed that detection probabilities did not change as a result of treatments based on the following: (1) snails have limited mobility; (2) we designed surveys and selected treatment block size to have a high probability of detecting all snails.

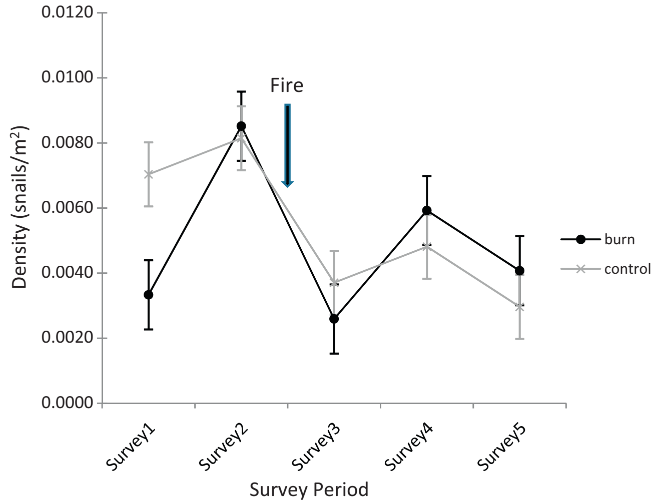

Fig. 2. Results (mean, SE) of the spring prescribed burn on the densities of Tiny Canyon mountainsnails, eastern Cascades, Washington. Surveys 1 and 2 are pre-treatment surveys, Survey 3 was conducted within two weeks of the treatment, Survey 4 was conducted the fall following the treatment, and Survey 5 was conducted one year following the treatment.

We used multivariate repeated measures analysis of variance (MANOVA) in SAS 9.2 (PROC GLM, REPEATED/PROFILE, SAS Institute 2008). The MANOVA determines the overall differences in snail densities (dependent variable) as a result of time or the treatment effect (independent variables), while the PROFILE analysis determines where the differences take place in the repeated measurements (Von Ende, 2001; Underwood and Quinn, 2010). We used this analysis approach because we measured the densities of the mountainsnails within the same sample block repeatedly so values were not independent (Zar, 1996). We tested our data for sphericity and used the Huynh–Feldt (HF) adjustment where necessary (Zar, 1996).

3. Results

3.1. Vegetation sampling and burn conditions

There were no significant pre-treatment differences in the grass cover ($p = 0.80$) or tree cover ($p = 0.92$) variables between the plots assigned to the fall-burn and the paired control plots (Table 1). There were also no significant pre-treatment differences between the assigned spring-burn and the paired control plots for grass cover ($p = 0.22$) or tree cover ($p = 0.052$) (Table 1).

There was no significant difference in the duration of time that fire was applied to the plots between spring and fall burns ($p = 0.36$). However, there was a significant difference in soil moistures ($p = 0.003$) (Table 2). Soil moistures were significantly drier during the fall-burn compared to the spring-burn (Table 2).

3.2. Snail densities

Pre-treatment surveys showed considerable variation in the densities of snails between the spring and fall survey periods (Survey 1 vs 2, Figs. 2 and 3). While we do not understand the causal mechanism of this variation, we anticipated this going into our study (Gaines et al., 2005) and it was one of reasons we incorporated multiple survey periods into the assessment of treatment effects.

We did not detect any statistically significant differences in mountainsnail densities that could be explained by the application of spring-burning ($p = 0.68$), time between surveys (0.22), or the interaction time × spring-burn treatment ($p = 0.73$) (Fig. 2).

Table 1
A summary of the pre-treatment vegetation conditions for each of the plots used in the Tiny Canyon mountainsnail study, eastern Cascades, Washington.

Plot number	Assigned treatment	% Cover						
		Grass	BASA[a]	Shrub	Moss/lichen	Litter	Bare Ground/rock	Tree cover
2	Fall-burn	58	0	13	3	23	1	80
3	Fall-burn	73	9	5	23	9	0	60
5	Fall-burn	23	15	15	0	2	18	25
Mean		*51*	*8*	*11*	*9*	*11*	*6*	*55*
1	Control	84	6	<1	28	5	0	30
4	Control	60	7	25	29	4	0	90
6	Control	42	35	7	1	1	4	60
Mean		*62*	*16*	*11*	*19*	*3*	*1*	*60*
10	Spring-burn	80	18	3	0	10	1	65
13	Spring-burn	15	14	4	0	25	0	30
15	Spring-burn	58	15	10	15	3	2	45
Mean		*51*	*16*	*6*	*5*	*13*	*1*	*47*
9	Control	36	21	5	0	0	0	35
14	Control	17	19	5	0	11	5	35
16	Control	38	10	17	33	0	15	40
Mean		*30*	*17*	*9*	*11*	*4*	*7*	*37*

[a] Arrowleaf balsamroot (*Balsamorhiza sagittata*).

Table 2
A summary of conditions at the time of the prescribed fire treatments, Tiny Canyon mountainsnail study, eastern Cascades, Washington.

Plot number	Treatment	Date treated	Duration of treatment	Air temp (°C)	% Relative humidity	% Soil moisture
13	Spring-burn	2007-4-19	45 min	9	40	11.0
15	Spring-burn	2007-4-19	35 min	11	35	11.5
10	Spring-burn	2007-4-19	50 min	12	35	13.5
Mean			*43*	*11*	*37*	*12.0*
5	Fall-burn	2006-10-3	45 min	14	38	4.3
2	Fall-burn	2006-10-3	45 min	15	37	4.0
3	Fall-burn	2006-10-3	45 min	16	34	7.2
Mean			*45*	*15*	*36*	*5.2*

Similarly, we detected no statistically significant differences in mountainsnail densities as a result of the fall-burn ($p = 0.12$), the time between surveys (0.65), or the interaction time × fall-burn treatment ($p = 0.38$) (Fig. 3). Sphericity was detected in the fall-burn analysis but the HF test was >1 so no adjustment of the P-values was necessary.

Pre-treatment densities were similar in the spring-burn block compared to the fall-burn block, ranging from 0.0011 to 0.0156/m^2. Though not statistically significant, mountainsnail densities immediately after the application of the spring-burn treatments were lower than control block densities (treatment $X = 0.0026/m^2$ vs control $X = 0.0037/m^2$). However, this trend was reversed in all subsequent surveys as densities were higher in treated plots the following fall (treatment $X = 0.0059/m^2$ vs control $X = 0.0048/m^2$), and one year following spring-burn treatment (treatment $X = 0.0041/m^2$ vs control $X = 0.0030/m^2$).

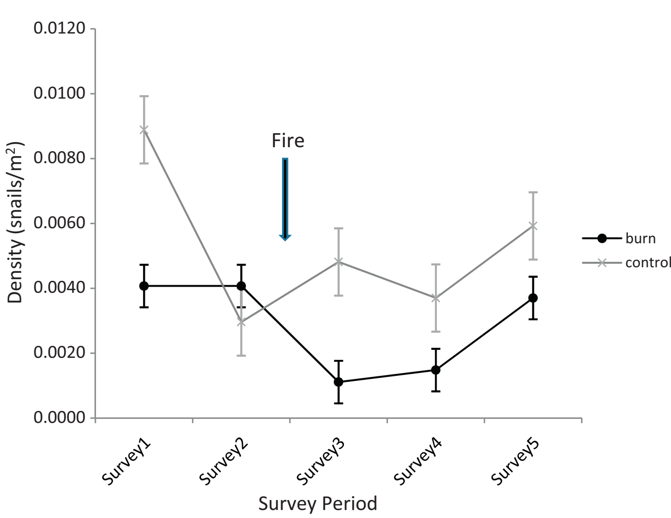

Fig. 3. Results (mean, SE) of the fall prescribed fire on the densities of Tiny Canyon mountainsnails, eastern Cascades, Washington. Surveys 1 and 2 are pre-treatment surveys, Survey 3 was conducted within two weeks of the treatment, Survey 4 was conducted the spring following the treatment, and Survey 5 was conducted one year following the treatment.

Pretreatment mountainsnail densities in the fall-burn blocks ranged from 0.0022 to 0.0124/m^2. While not statistically different, mountainsnail densities were generally lower during post-treatment surveys in the fall-burn treatment compared to the controls, though mountainsnail densities in both treatment and control blocks increased over time. This trend occurred in the surveys immediately following the treatments (treatment $X = 0.0011/m^2$ vs control $X = 0.0048/m^2$), during the surveys conducted the following spring (treatment $X = 0.0015/m^2$ vs control $X = 0.0037/m^2$), and the surveys conducted 1 year after treatments (treatment $X = 0.0037/m^2$ vs control $X = 0.0059/m^2$).

4. Discussion

Land snails are generally very sensitive to desiccation (Godan, 1983) and probably do not survive high temperatures that can be reached during fire (Kiss and Magnin, 2006). Tiny Canyon mountainsnails, unlike other *Oreohelix* land snails, live in the forest litter and duff layers with little or no protection from talus slopes (Burke et al., 1999) and are thus highly vulnerable to fire and changes to their habitats. However, we found no statistical relationship between the short-term (up to 1 year) effects of spring or fall prescribed fire treatments and the density of mountainsnails. We did find a consistent non-significant trend where fall-burning, conducted during lower soil moistures, resulted in lower densities of mountainsnails compared to controls.

So how did these mountainsnails survive the prescribed burns? The relatively large, thick shell common to the Oreohelices, is apparently an adaptation to arid areas and reduces the effects of desiccation and excessive heat (Pilsbry, 1939; Burke et al., 1999; Duncan, 2005). Studies of the effects of fires on land snails have shown the importance of a fire mosaic and small refuges that allow initial land snail survival and persistence after disturbance (Kiss and Magnin, 2003, 2006; Santos et al., 2009). This is consistent with the burn patterns that occurred in our study as both the fall and spring burns resulted in unburned or lightly burned patches that provided refuge for mountainsnails to survive. Kiss and Magnin (2006) also show that their Mediterranean snails resided in boulders and rock outcroppings, which led to lower intensity burns in those areas and higher snail survival. As the Tiny Canyon mountainsnail is found in dry forest, with little rock cover, unburned or lightly burned patches are even more important in our study area. A behavioral adaptation may also aid in Tiny Canyon mountainsnail survival in lightly burned areas. Although the Tiny Canyon mountainsnail does not live within protective talus slope environments like other *Oreohelix* species, individuals are often found aestivating deep within the leaf litter, and in the rootball of arrowleaf balsamroot. This depth may provide enough protection during low intensity fires. Thus, burn prescriptions that create a fine-scale mosaic of burn intensities, including unburned areas, are recommended to increase the resilience of mountainsnail populations to fire disturbances. Of particular concern are uncharacteristically severe disturbances that result from decades of fire exclusion in dry and mesic forests (Everett et al., 2000; Hessburg et al., 2007) and have the potential to dramatically reduce mountainsnail populations (Burke et al., 1999; Duncan, 2005; Santos et al., 2009). The use of prescribed fire to create a fine-scale mosaic of burn intensities is consistent with the idea of restoring a low-severity fire regime within dry forest landscapes (Agee, 1993; Brown et al., 2004; Hessburg et al., 2005; Gaines et al., 2010b).

The frequency of treatment application is another important consideration in the use of prescribed fire to restore dry forest ecosystems that include mountainsnail habitat. Kiss and Magnin (2006) found that land snail resilience was high as long as the time lapse between successive fires is longer than the time required for snail populations to recover. Kiss and Magnin (2006) found that about five years between fires was adequate to allow snail populations to recover. Recovery time for the Mediterranean snails may be faster than the Tiny Canyon mountainsnail because most of the species studied lay eggs, whereas *Orehelix* snails brood their young. Anderson et al. (2007) found that snails from another *Oreohelix* species (*O. cooperi*) likely have one clutch per year with an average of 3.3 ± 1.7 young per year (clutch size ranged from 1 to 11 young). Fire return intervals in the dry and mesic forests of the eastern Cascades generally range from 10 to 40 years (Agee, 1993; Everett et al., 2000; Wright and Agee, 2004). This length of time provides a high likelihood that populations would recover, assuming a low-severity fire similar to the fire behavior within our study plots.

The restoration of low severity fire regimes within dry forest to reduce the risk of uncharacteristic, large-scale high severity wildfires remains an important conservation consideration for the mountainsnails in our study area (Burke et al., 1999; Duncan, 2005; Gaines et al., 2005). Our information suggests that carefully applied prescribed fire implemented under proper conditions may be a valuable tool that can be used to restore dry forest habitats.

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