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Management and Conservation Article

Basin-Scale Surveys of Stream-Associated Amphibians in Intensively Managed Forests

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ABSTRACT Conservation and management of native species on landscapes managed for intensive wood production represents an ongoing challenge to forest managers. Previous research suggests that impacts of forest practices on stream-associated amphibians (SAA; giant [*Dicamptodon* spp.], torrent [*Rhyacotriton* spp.], and plethodontid [*Plethodon* spp.] salamanders and coastal tailed frogs [*Ascaphus truei*]) in Oregon and Washington, USA, vary spatially and temporally as a result of biotic and abiotic factors, some of which can be influenced by management treatments. Although individual harvest units can encompass multiple stream reaches and entire second-order basins, nearly all published research studies used stream reaches of various lengths as sample units. To address this discrepancy between research and operational scales, we sampled first-, second-, and third-order streams in 70 randomly selected third-order basins in Oregon and Washington in 2007 and 2008 to estimate detection and occupancy parameters for SAA and to develop basin-level density estimates for different species and genera. We estimated occupancy probabilities of 0.99 (95% CL = 0.96–1.00) for torrent and giant salamanders, 0.93 (95% CL = 0.76–0.92) for Dunn's salamanders (*Plethodon dunni*), and 0.60 (95% CL = 0.46–0.72) for tailed frogs. Our estimates can be compared with estimates for unmanaged third-order basins in Oregon and Washington to provide a relative measure of potential impacts of forest management on these taxa. In addition, our estimates provide baseline information with which to assess potential effects of future environmental changes on the 4 genera.

KEY WORDS *Ascaphus truei*, commercial forests, *Dicamptodon* spp., forest management, occupancy, Oregon, *Plethodon dunni*, *Rhyacotriton* spp., Washington.

Managers of multiple-use forest landscapes must respond to conflicting societal demands for increased levels of commodity production (Brooks 1995, Curtis and Carey 1996, Haynes 2003) and maintenance of biological resources and ecosystem services (Bunnell and Huggard 1999, Harrington and Nicholas 2007, Sedjo 2007). In Oregon and Washington, USA, widespread, unregulated historic logging, paired with a subsequent intensification of management (i.e., clear-cut harvesting when regenerated stands are 45–55 yr old, chemical or mechanical site preparation, followed by replanting), has altered the distribution, extent, and characteristics of certain forest types, primarily old-growth coniferous forest. These management regimes have led to widespread concern for the viability of native species on managed landscapes and the need to develop feasible conservation actions (Bunnell and Huggard 1999, Brockhoff et al. 2008). However, broad-scale restoration of late-successional forest conditions is an unlikely option on the >8 million acres of privately owned, intensively managed forest in Oregon and Washington (Risenhoover and Murden 2007). To enable long-term stewardship and positive conservation outcomes, landowners must have accurate assessments of both the negative and positive biological impacts associated with managing forests on short-term rotations (Bunnell and Huggard 1999, Binkley et al. 2005, Hayes et al. 2005).

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Forested streams in Oregon and Washington support a diverse array of endemic amphibians, including giant (*Dicamptodon* spp.), torrent (*Rhyacotriton* spp.), and plethodontid (*Plethodon* spp.) salamanders, and tailed frogs (*Ascaphus* spp.), collectively called stream-associated amphibians (SAA; Jones et al. 2005). Considerable research has been conducted on the distribution and ecology of these species, with an emphasis on their response to forestry practices (Corn and Bury 1989, Wilkins and Peterson 2000). Researchers have documented a range of SAA responses to forest management, including extirpations at the local scale to high occupancy probabilities in managed stands that have been harvested at least once (Welsh et al. 2006, Kroll et al. 2008). Although SAA are often abundant in streams within late-successional forests, SAA also can be abundant in streams occurring in intensively managed forests (Corn and Bury 1989, Diller and Wallace 1999, Adams and Bury 2002, Steele et al. 2003, Russell et al. 2004). Given these equivocal results and the absence of information about population sizes (Kroll 2009), considerable uncertainty exists about what management practices might maintain SAA population viability over successive rotations and how inherent differences (e.g., geology and climate) across the geographic distribution of SAA may mediate or exacerbate forestry impacts and influence effectiveness of alternative management practices (Olson et al. 2007).

Several factors hinder reliable inference about spatial and temporal variation in SAA responses to forest harvesting. First, nearly all published research reports on SAA have evaluated responses (either occupancy or relative abundance) at the individual stream reach (e.g., 10–100 m) even though harvest operations often include several stream reaches as well as entire first- and second-order basins (and despite calls for basin-level surveys to assess SAA responses; Corn and Bury 1989, Olson et al. 2007). Second, researchers have been unable to determine whether harvesting causes direct (mortality) or indirect (dispersal to more favorable within-stream conditions) impacts to SAA populations (Jackson et al. 2007) or whether absence of SAA from apparently suitable habitat indicates harvesting-induced extirpation or historical absence (i.e., SAA were never present; Ashton et al. 2006). Making inferences about relevant ecological processes from described patterns is difficult, in part due to variation in individual and species detection probabilities and a lack of information about inter- and intraspecies competition, habitat selection, and vital rates (Tilley 1980, Dodd and Dorazio 2004, Sagar et al. 2007, Kroll et al. 2008, Kroll 2009). Finally, individual species are thought to be associated with different portions of the stream network (Hayes et al. 2006, Olson et al. 2007). Sampling programs that target individual headwater streams (i.e., first-order basins) are unlikely to provide a unified understanding of SAA spatial distributions because species may be more or less prevalent across stream orders (i.e., first- to fourth-order streams) independent of any forest management effect.

We modeled basin occupancy for 7 species of SAA, including Dunn's salamander (*Plethodon dunni*); Pacific and Cope's (*Dicamptodon copei*) giant salamanders; Cascade (*Rhyacotriton cascadae*), Columbia (*Rhyacotriton kezeri*), and southern (*Rhyacotriton variegatus*) torrent salamanders; and tailed frogs (*Ascaphus truei*) as a function of landscape-level covariates to examine potential relationships between occupancy and intensive forest management. Our study differs from published SAA research in that we treated the third-order basin, rather than first-, second-, or third-order streams within the basin, as the sample unit (whereas Stoddard and Hayes [2005] sampled 16 third-order basins in OR and analyzed responses at the stream, stand, and basin scales, they did not treat the basin as the sample unit). Our objectives were to 1) determine how SAA detection probabilities varied with survey-specific covariates and 2) determine how SAA occupancy probabilities varied with basin-level covariates.

STUDY AREA

We conducted our sampling in third-order basins on Forest Capital Partners, Hancock Forest Management, Longview Timberlands, and Plum Creek ownerships in the Coast Ranges of Oregon and Washington and Cascades Mountains in southern Washington (Fig. 1). Sample basins were located in both the Western Hemlock (*Tsuga heterophylla*) and the Coastal Sitka Spruce (*Picea sitchensis*) zones (Franklin and Dyrness 1973). The landscape was dominated by second and third rotation stands differing in age from 0

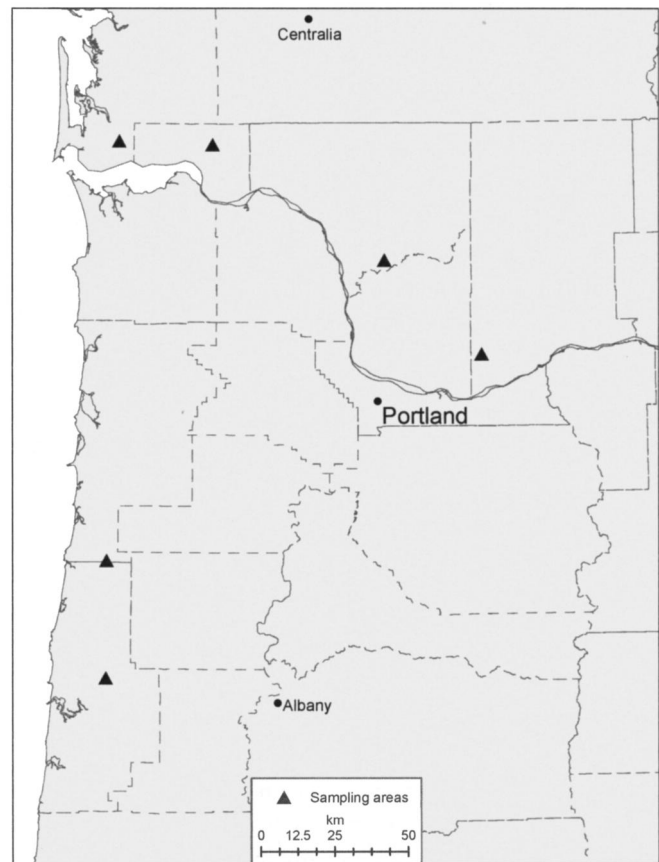


Figure 1. General locations of third-order basins sampled to estimate occupancy probabilities for stream-associated amphibians, western Oregon and southwestern Washington, USA, 2007–2008. Dashed lines indicate county boundaries.

to >70 years and composed primarily of western hemlock and Douglas-fir (*Pseudotsuga menziesii*). Some basins in the Washington Cascades contained some unmanaged second growth stands (<30% of total area).

Study basin management practices included clear-cutting followed by site preparation and planting with Douglas-fir, western hemlock, western red cedar (*Thuja plicata*), and true fir (*Abies* spp.) nursery stock as appropriate for site conditions. Precommercial and commercial thinning, aerial fertilization, and chemical control of competing vegetation were common. Douglas-fir was the dominant conifer, although western hemlock dominated stands closer to the Pacific Ocean in the Coast Range. Dominant deciduous trees included red alder (*Alnus rubra*), bigleaf maple (*Acer macrophyllum*), and vine maple (*Acer circinatum*). Dominant understory species included salmonberry (*Rubus spectabilis*), salal (*Gaultheria shallon*), huckleberries (*Vaccinium* spp.), and swordfern (*Polystichum munitum*).

METHODS

We used ArcGIS to select third-order basins for sampling using stream layers for Washington (Washington Department of Natural Resources 2007) and Oregon (Bureau of Land Management 2007). We intersected these layers with ownership layers so that all third-order basins on participating ownerships were available for selection (approx. 1,500

third-order basins). We assigned each third-order basin a numeric identifier and used a random number generator to select candidate basins for sampling.

We calculated basin-scale covariates using individual ownership Geographic Information System (GIS) data. To determine basin size, we used different digital elevation model (DEM) data for Oregon (Oregon Geospatial Enterprise Office 2008) and Washington (U.S. Geological Survey 2008). We used the ESRI Hydrology toolset (ArcGIS 9.2, ESRI 2007) to condition the DEM data, which distinguished true sinks (changes in elevation) from false sinks (mapping error) in the DEM (ArcGIS 9.2, ESRI 2007). We derived pour point locations (i.e., points at the downstream end of watersheds) from maps or electronic files containing coordinates received from landowners. We found those coordinates in GIS and created points for the beginning of each basin. We snapped pour points to the DEM-derived stream layer. We added enough points so that the software could delineate the desired basin and the surrounding watersheds to ensure boundaries were correct. We made any necessary edits to watershed boundaries on the shape file.

Amphibian Surveys

We conducted SAA surveys from July to September in 2007 and 2008, during which 2-person survey crews visited individual third-order basins up to 2 times. We used a removal-sampling design to sample each basin until we detected all target taxa or until we completed 2 surveys (MacKenzie and Royle 2005, MacKenzie et al. 2006). If we encountered all taxa in a basin during the first survey, then we removed that basin from further sampling (MacKenzie and Royle 2005). We searched for all target species at each visit, regardless of whether we previously detected a species on a visit. We included in the analysis basins we surveyed in both years only if occupancy status did not change across years, as a way to address the closure assumption (individuals do not move in or out of the sample units during the sampling period; MacKenzie et al. 2006) given that SAA seem to have low dispersal rates. Documented movements of SAA range from a few to several hundred meters in length (Welsh and Lind 1992, Nijhuis and Kaplan 1998, Johnston and Frid 2002, Burkholder and Diller 2007), so our decision to remove basins that did change occupancy status may have been more conservative than necessary. For those basins we sampled twice in the same year, we spaced visits 3–20 days apart.

We surveyed 6 100-m stream segments in each third-order basin. We placed one segment in the third-order stream 100 m below the second-order confluence, 2 segments in second-order streams immediately above the confluence with the third-order channel, and 3 segments in first-order streams immediately above the confluence with the second-order channel. Each third-order basin must include 4 first-order streams and 2 second-order streams (by definition); however, many basins contained numerous first- and second-order streams. In these basins, we randomly selected individual segments for sampling. We always sampled 3 first-order streams, 2 second-order streams, and 1 third-

order stream in each basin, and a maximal stream length of 600 m.

We searched under a time-constraint of 2.5 hours, or 2 individuals searching for 1.25 hours in each 100-m segment. We flagged sample reaches so that the same 100-m interval was sampled at each visit. We used the light-touch method to minimize disturbance to the stream segment (Hayes et al. 2006, Quinn et al. 2007, Kroll et al. 2008). The light-touch method included a visual search of the stream-bed and under cut-banks using high-power flashlights and commercial bait-nets. In addition, we moved all cover features <30 cm in diameter during each search and returned objects to their original positions. We did not disturb coarse woody debris of any size. We recorded species, life stage, and number for all SAA we detected during each survey. Our primary interest was species detection, and we released all captured animals after we identified them. We considered a basin to be occupied by a species if we detected any life stage of that species in any of the stream orders.

Most (>95%) giant salamanders we captured were larvae. Due to the difficulty of distinguishing between larval Cope's and coastal giant salamanders in the field, and similarities between the 2 species in habitat relationships reported by Steele et al. (2002), we combined those species for analysis. In addition, because our sample size in the Washington Cascades was limited, we combined Cascades, Columbia, and southern torrent salamanders for the occupancy analysis.

One of our objectives was to identify basin-level variables associated with SAA occupancy. For each third-order basin, we calculated the area (ha), average site index (average ht of Douglas-fir at 50 yr, an index of productivity), area-weighted stand age, total length of the stream network (m), total length of the stream network with a forested buffer (m), total length of road (m), and total area (m²) of road prism (i.e., road surface; Table 1). We measured air and water temperature (C) and recorded weather conditions (i.e., raining, overcast, clear, and whether it rained in the previous 24 hr) at the beginning of each survey.

Data Analysis

We used methods developed by MacKenzie et al. (2002, 2006) to estimate probability that a third-order basin was occupied by SAA as a function of the independent covariates we measured. We conducted analyses with program PRESENCE (PRESENCE, version 2.0, <www.mbr-pwrc.usgs.gov/software/doc/presence/presence.html>, accessed 1 Apr 2008).

We used a 2-step process to estimate occupancy parameters (ψ) for SAA. First, we modeled those covariates we thought would influence detection probabilities (p), including Julian date, rain during the survey, rain within 24 hours before the survey, and water and air temperature while holding occupancy constant (i.e., $\psi[.]p[covariate]$). Second, we examined occupancy models simultaneously with the best model for detection probability (Bailey et al. 2004, MacKenzie 2006).

We had no prior information regarding potential associations between basin covariates and occupancy for salaman-

Table 1. Definitions and summary statistics for covariates used to estimate third-order basin occupancy or detection probabilities for Dunn's salamanders, giant salamanders, torrent salamanders, and tailed frogs, western Oregon and southwestern Washington, USA, 2007–2008.

Covariate	Occupancy or detection	Definition and values	Average	95% CI
Date	Detection	Julian date	25 Aug	22 Aug–1 Sep
Rain before survey	Detection	Rain in the 24 hr before the survey	40/108 surveys	NA ^a
Rain during survey	Detection	Rain during basin survey	25/108 surveys	NA ^a
Water temp	Detection	Water temp (C) at the bottom of the third-order stream sample reach	11	10–12
Air temp	Detection	Air temp (C) at the beginning of the sample session	15	14–16
Site index	Occupancy	Average site index (average tree ht at 50 yr old) for stands in third-order basin	124	120–128
Area-weighted stand age	Occupancy	Area-weighted stand age for all harvest units in basin	25	23–28
Stream length	Occupancy	Total length (m) of stream in each basin	7,830	6,148–9,511
Buffered stream length	Occupancy	Total length (m) of buffered stream in each in basin	5,774	4,214–7,335
Road length	Occupancy	Linear length (m) of roads in basin	13,710	8,040–19,380
Road surface area	Occupancy	Total road area (m ²) in basin (length × average road width); average width: 4.58 m	62,792	36,823–88,761

^a No error is associated with this measurement.

ders. As a result, we evaluated only univariate occupancy models for these 3 genera (i.e., we evaluated each one of the 7 basin-scale covariates individually). However, for tailed frogs, we also evaluated an additional model that included a quadratic effect of basin area; smaller basins may be more likely to become dewatered during summer and larger basins may have fish in their third- and potentially second-order channels (Hayes et al. 2006). In total, we evaluated 5 detection models and 7 occupancy models for each genus except tailed frogs, for which we evaluated 8 occupancy models. We standardized all continuous covariates and Julian date (by subtracting the average and dividing by SD for each variable) to avoid convergence problems with parameter estimation in PRESENCE. We used Akaike's Information Criterion (Burnham and Anderson 2002) for model selection to account for small sample sizes. Finally, we stress that we chose our models to evaluate basin-level factors we thought would influence SAA parameter estimates. As such, our models represent generalizations that do not necessarily describe the full array of ecological factors that probably influence both SAA basin occupancy and species detection parameters (Box 1976).

RESULTS

We surveyed 14 basins in 2007 (i.e., we detected all species or 2 surveys occurred), 40 basins in 2008, and 16 basins in 2007 and 2008 (one survey in each yr). We sampled 70 third-order segments, 104 second-order segments, and 119 first-order segments. Three basins changed occupancy status for tailed frogs between 2007 and 2008 (i.e., we did not detect tailed frogs in 2007, but we detected them in 2008 in all 3 basins), and we excluded these basins from analysis for that species ($n = 67$). Fifty-six of the basins occurred within the range of Dunn's salamander. We detected tailed frogs in 8 of 10 (80%) basins in the southern Cascades, 22 of 30 (73%) basins in the Oregon Coast Range, and 8 of 27 (30%) basins in southwestern Washington (Willapa Hills). The average site indices for the 10 basins in the Washington Cascades, 30 basins in the Oregon Coast Range, and 27 basins in southwestern Washington, respectively, were 100

(range: 78–113), 122 (range: 106–143), and 137 (range: 106–153). We detected tailed frogs in 6 of 18 (33%) basins <50 ha (6 of 67; 9% of all basins sampled), 12 of 21 (57%) basins 50–100 ha (12 of 67; 18%), and in 20 of 28 basins (70%) >100 ha (20 of 67; 30%).

We detected both torrent and giant salamanders on most of our surveys (89 of 108 [82%] and 98 of 108 [91%], respectively). None of the detection covariates were associated with torrent or giant salamander detection probabilities. As a result, we used the intercept-only $p(\cdot)$ detection model to estimate occupancy probabilities for both of these genera. We detected tailed frogs on 40 of 103 of surveys (39%). None of the detection covariates were supported by the analysis, and we included the intercept-only $p(\cdot)$ detection model in all tailed frog occupancy models. Julian date was associated negatively with Dunn's salamander detection probability (Fig. 2) and we included the base model $p(\text{Julian date})$ in all Dunn's salamander occupancy models. Direction of the estimate for Julian date remained constant in all Dunn's salamander models we examined.

We found giant salamanders in 69 of 70 (99%) basins; 100% of detections occurred on the first visit. We found torrent salamanders in 69 of 70 (99%) basins; 96% (66/69) of detections occurred on the first visit. We found tailed frogs in 38 of 67 (57%) basins; 92% (35/38) of detections occurred on the first visit. In 2 of 3 (66%) basins excluded from the analysis, we found tailed frog larvae or metamorphs in 2008, indicating that the species was present but not detected during the survey in 2007; we detected an adult tailed frog in the remaining basin. We exclude these 3 basins from analysis because of the closure assumption (see Amphibian Surveys). We found Dunn's salamanders in 47 of 56 (84%) basins; 91% (43/47) of detections occurred on the first visit.

Due to high occupancy probabilities for giant and torrent salamanders, we did not evaluate their occupancy probabilities as a function of any of the covariates (Table 2). Instead, we based occupancy probabilities on the intercept-only model (Table 2). Tailed frog basin occupancy was most strongly associated (inversely) with average site index (Table 3; Fig. 3), and the 95% confidence interval did not

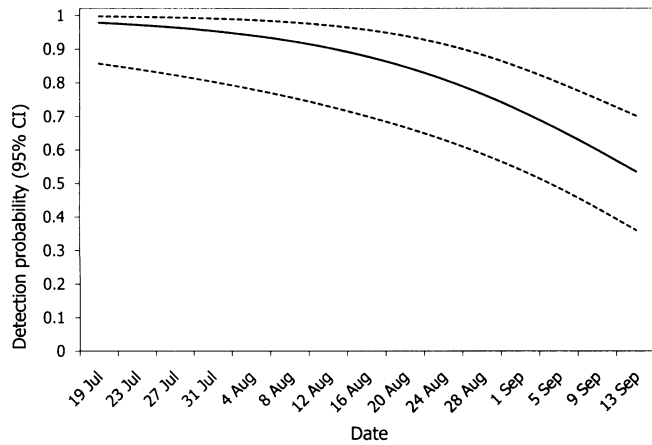


Figure 2. Relationship between Dunn's salamander detection probability and Julian date, western Oregon and southwestern Washington, USA, 2007–2008.

include zero (untransformed estimate = -0.91 , 95% CL = -1.56 , -0.26). Dunn's salamander basin occupancy was most strongly associated (inversely) with area-weighted stand age (Table 3; Fig. 4), and the 95% confidence interval included zero (untransformed estimate = -1.05 , 95% CL = -2.1 , 0.005), indicating that the association was not statistically significant. No other covariates were supported for basin occupancy of either tailed frog or Dunn's salamander.

DISCUSSION

Our results suggest that SAA occurrence probabilities in third-order basins within intensively managed forest landscapes in Oregon and Washington are high, which could reflect 1) extirpation of SAA from previously occupied basins (1% for the in-stream salamanders, 16% for Dunn's salamander, and 43% for tailed frogs, assuming 100% occupancy), 2) static occupancy over time, or 3) increases in occupancy probabilities. Without a comparable data set from unmanaged forests or data collected before logging on the sites we sampled, we cannot determine which of these possibilities is most likely. These data are baseline conditions that can be used to assess these possibilities in the future, and we stress the importance of management agencies collecting these types of data for public forests not managed for commercial wood or fiber production.

We sampled across a wide range of productivity gradients (i.e., site index), basin sizes, and total stream lengths (see Table 1); however, variation in area-weighted stand age,

road length, and road density were minor due to the similarity of management across ownerships. Weak support for any association among these characteristics and occupancy probabilities suggests that other factors (e.g., lithology, sediment deposition, and gradient) may regulate SAA occupancy on intensively managed forest landscapes (Wilkins and Peterson 2000, Kroll et al. 2008, Kroll 2009).

Although we did not design our study to address how in-stream characteristics interact with any potential forest harvesting impacts, the ubiquitous presence of giant and torrent salamanders and the high probability of estimated occupancy for Dunn's salamanders suggest that occupancy probabilities of these 3 genera are resilient to potential deleterious and prolonged effects of forest harvesting in the geographies where we sampled or that changes due to forestry are not limiting. The inverse association between Dunn's salamander occupancy and area-weighted stand age indicates a positive effect of younger forests. However, inclusion of zero in the confidence interval for this covariate suggests additional data are needed to confirm this relationship. In general, the null model (no to small covariate effects) of occupancy for all 3 salamander species is most representative for our data.

Tailed frog occupancy probability was inversely associated with average site index in each basin, although this result had a strong geographic component. Site index represents a combination of soil type, insolation and precipitation rates, and elevation (Oliver and Larson 1996). Given that other results suggest that tailed frogs respond positively to increased levels of productivity (Kiffney et al. 2004, Mallory and Richardson 2005), we suggest that site index should be considered a surrogate for geography in our study. Also, site index for upland stands may not be a reliable index of in-stream productivity, especially if forested buffers occur along much of the stream length in a basin (as we found in our study; Table 1).

Our occupancy estimate for southwestern Washington is lower than a previous finding of a naive tailed frog occupancy probability of approximately 0.6 for third-order basins in the Stillman Creek watershed in southwestern Washington (Hayes et al. 2006). However, we sampled across several watersheds in southwestern Washington that contained both soft and hard lithologies (Stillman Creek is dominated by basalt lithologies). Most forested area in southwestern Washington consists of second- and third-growth stands and our results could be taken as an indication that forest management on soft lithologies will have negative impacts on tailed frogs. We reiterate that our

Table 2. Average species detection and occupancy probabilities (based on the most supported models) for Dunn's salamanders, giant salamanders, torrent salamanders, and tailed frogs, western Oregon and southwestern Washington, USA, 2007–2008.

Genus or species	Detection probability	Detection 95% CL		Occupancy probability	Occupancy 95% CL	
		Lower	Upper		Lower	Upper
Dunn's salamanders	0.79	0.61	0.90	0.93	0.76	0.98
Giant salamanders	0.95	0.91	0.99	0.99	0.96	1
Torrent salamanders	0.89	0.81	0.97	0.99	0.96	1
Tailed frogs	0.86	0.67	0.95	0.60	0.46	0.72

Table 3. Summary of AIC model selection results for basin occupancy by tailed frogs and Dunn's salamanders, western Oregon and southwestern Washington, USA, 2007–2008. The symbol ψ indicates the occupancy portion of the model, and the symbol p denotes the detection portion of the model.

Model	$-2\log\text{-likelihood}$	K	AIC_c	ΔAIC_c	w_i^a
Tailed frog; $n = 67$					
$\psi(\text{average site index}), p(\cdot)$	112.2	3	118.2	0.00	0.74
$\psi(\text{area}), p(\cdot)$	116.2	3	122.2	4.02	0.10
$\psi(\text{stream length}), p(\cdot)$	118.0	3	123.9	5.94	0.04
$\psi(\text{area squared}), p(\cdot)$	116.1	4	124.1	7.69	0.04
$\psi(\text{road length}), p(\cdot)$	118.9	3	124.9	9.69	0.03
$\psi(\text{buffered stream length}), p(\cdot)$	119.5	3	125.5	9.69	0.02
$\psi(\text{road area}), p(\cdot)$	119.6	3	125.6	9.69	0.02
$\psi(\cdot), p(\cdot)$	121.9	2	125.9	9.69	0.02
$\psi(\text{area-weighted stand age}), p(\cdot)$	121.8	3	127.8	9.69	0.01
Dunn's salamander; $n = 56$					
$\psi(\text{area-weighted stand age}), p(\text{Julian date})$	87.7	4	95.7	0	0.49
$\psi(\cdot), p(\text{Julian date})$	92.1	3	98.1	2.45	0.14
$\psi(\text{road area}), p(\text{Julian date})$	91.1	4	99.1	3.46	0.09
$\psi(\text{road length}), p(\text{Julian date})$	91.5	4	99.5	3.86	0.07
$\psi(\text{buffer length}), p(\text{Julian date})$	92.0	4	99.9	4.33	0.06
$\psi(\text{average site index}), p(\text{Julian date})$	92.1	4	100.1	4.42	0.05
$\psi(\text{stream length}), p(\text{Julian date})$	92.1	4	100.1	4.43	0.05
$\psi(\text{basin area}), p(\text{Julian date})$	92.1	4	100.1	4.45	0.05
$\psi(\cdot), p(\cdot)$	115.0	2	119.1	23.39	0.00

^a K = the no. of parameters in the model; AIC_c = Akaike's Information Criterion adjusted for small sample sizes; ΔAIC_c = difference in AIC_c between top model and each subsequent model; w_i = Akaike wt.

results are descriptive and that the absence of information about historic distributions of tailed frogs in southwestern Washington and the inability to test the interaction between geology and forest management in this geographical region preclude a rigorous examination of this question (i.e., replicates of soft and hard lithologies across a range of unharvested and harvested stands are required; see Kroll 2009).

We did not find support for either linear or quadratic associations between basin area and tailed frog occupancy (Hayes et al. 2006, Kroll et al. 2008). We did not survey for fish in the sample basins, so we could not address the hypothesis by Hayes et al. (2006) that larger non-fish-bearing basins will have higher tailed frog occupancy probabilities.

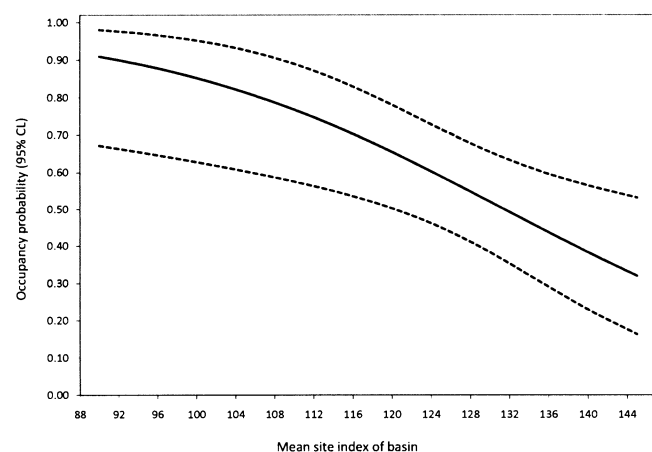


Figure 3. Relationship between tailed frog occupancy probability and average site index (average tree ht at 50 yr), western Oregon and southwestern Washington, 2007–2008.

MANAGEMENT IMPLICATIONS

We found high occupancy probabilities on intensively managed forests for Dunn's, giant, and torrent salamanders, and moderate occupancy probabilities for tailed frogs. We cannot determine whether current forestry practices in Oregon and Washington (e.g., forested buffers, equipment and herbicide exclusion zones) have maintained these high probabilities of occupancy or whether they are a function of residual conditions created before the onset of these management practices. However, alteration of entire third-order basins to stands with an average age of 25 years suggests that similar conditions and structures will only be created in riparian buffers in the future; if forest management has a prolonged deleterious effect on SAA and if the practice of buffering streams does not alleviate this effect

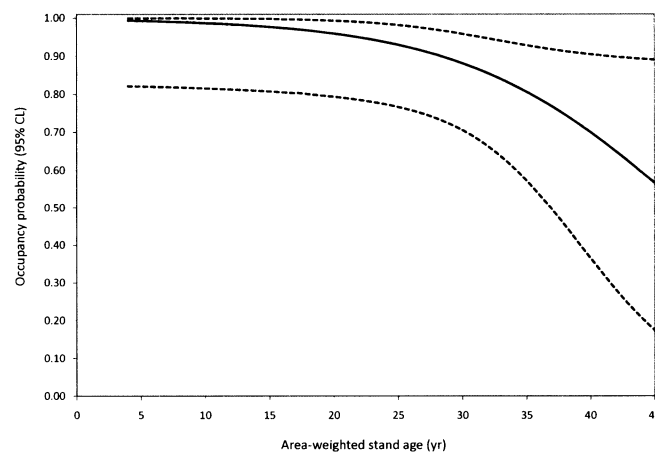


Figure 4. Relationship between Dunn's salamander occupancy probability and area-weighted stand age, western Oregon and southwestern Washington, USA, 2007–2008.

(although several studies suggest that they will), occupancy probabilities should decline over time. Although the absence of similar information for unmanaged third-order basins does not permit comparison to determine how occupancy probabilities might differ as a result of any prolonged impacts of forest management, our estimates do provide a basis for managers to monitor populations and evaluate the basin-scale effects of forest management, or other environmental changes, in the future.

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