Effects of Woody Biomass Harvests on a Population of Plethodontid Salamanders in Southeast Indiana

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ABSTRACT.—Biomass harvesting removes unmarketable vegetative material from timber harvests for use as cellulosic bioenergy, leaving only leaf litter. To test whether biomass harvests negatively affect red-backed salamander (*Plethodon cinereus*) populations, we set up coverboard arrays at 10 sites (mean 3.29 ha, range 2.35–4.61 ha) with varying degrees of biomass harvesting at the Southeast Purdue Agricultural Center (SEPAC) in Jennings County, Indiana. We monitored salamander artificial cover object (ACO) arrays within each site from spring 2012 to fall 2015 and marked all salamanders with visible implant elastomers, generating capture histories for all individuals. Using Program MARK and Pollock’s robust design we developed 10 *a priori* candidate models to test salamander population parameters, with variations on capture probability, recapture probability, survival, emigration, and immigration, as well as a set of models comparing preharvest and postharvest data. To incorporate precipitation events, we classified sessions as wet or dry based on total rainfall prior to sampling. The best performing models were those that incorporated the year, season, and amount of precipitation when estimating capture probabilities. Linear regression results showed percentage of canopy cover and Downed Coarse Woody Derbies (DCWD) were significant predictors of salamander abundance. We also found no significant relationships between survival, DCWD, and canopy cover. Our results suggest DCWD has some impact on variations in population sizes of red-backed salamanders, although other factors are likely contributing as well.

INTRODUCTION

In response to increasing gas prices and the depletion of fossil fuels, the use of renewable biofuels as an alternative energy source is becoming a realistic option (Janowiak and Webster, 2010). The conversion of cellulosic ethanol from organic sources such as corn is common practice, and many institutions are now utilizing woody biomass as another source (Janowiak and Webster, 2010). Woody biomass harvesting is similar to a traditional timber harvest except the unmerchantable woody material, such as tree-tops, small-diameter trees, and pre-existing deadwood that would normally be left on site are removed for biofuel production (Riffel et al., 2011). Removal of harvest residues such as downed coarse woody debris (DCWD) can be detrimental for forest wildlife (Herbeck and Larsen, 1999; Riffel et

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In addition to its vital role in nutrient cycling as its subsequent decomposition recycles nutrients, DCWD also serves an ecologically important function as habitat for many species (Evans et al., 2010; Janowiak and Webster, 2010). Most states have implemented biomass harvesting guidelines to minimize the net loss of DCWD following intensive timber harvest (Evans et al., 2010; Berger et al., 2013); however, there is concern about long-term site productivity and stability of forest ecosystems if biomass harvesting is adopted as a common practice (Evans et al., 2010).

Intensive biomass harvesting could drastically interrupt the ecology of sensitive species (Riffel et al., 2011; Otto et al., 2013). Changes to forest ecosystems (i.e., timber harvesting) can alter the moisture composition of upper soil and leaf litter layers, decreasing the ability for forests to support sensitive species, such as salamanders (Welsh and Droege, 2001). Salamander species throughout North America are adversely affected by timber harvest (Semlitsch et al., 2009). Most often, salamander sensitivity to timber harvest is attributed to increasing variability in microclimate conditions (McKenny et al., 2006; Homyack et al., 2011). The response of salamanders to environmental disturbances, both natural and anthropogenic, can be used as a proxy for forest ecosystem health (Welsh and Droege, 2001). Plethodontid salamander abundance and trends are directly impacted by extreme fluctuations in temperature and precipitation on the forest floor, which can be altered by the depletion of canopy cover as a result of timber harvest (McKenny et al., 2006; Ruhl, 2014). When moisture levels in forest environments decrease, plethodontid salamanders rely more heavily upon DCWD for refuge than other salamanders (i.e., those in the family Ambystomatidae; Owens et al., 2008) due to the plethodontids reliance on sustained moist conditions for dermal respiration. DCWD serves as an interaction center for conspecifics (Jaeger, 1995), provides moisture throughout dry periods (Grover, 1998; McKenny et al., 2006), and concentrates small invertebrates used as a source of food by plethodontids (Jaeger, 1980a). However, biomass harvesting can negatively impact much of the suitable DCWD used by plethodontids (Grialou et al., 2000; Morneault et al., 2004). Therefore, decreases in well-decayed DCWD as a byproduct of harvest disturbance may have a net negative impact on plethodontid salamanders, which has yet to be investigated.

The eastern red-backed salamander (Plethodon cinereus) has been described as an ideal indicator species for mature forests for several reasons (Welsh and Droege, 2001). First, its distribution is nearly ubiquitous throughout the Eastern United States, which allows for ample comparisons between studies (Herbeck and Larsen, 1999; Welsh and Droege, 2001). Heatwole (1962) and Jaeger (1980b) estimated that populations of red-backed salamanders could reach 0.9–2.2 individuals/m², while Burton and Likens (1975) found that red-backed salamanders accounted for over 90% of the total vertebrate biomass in the Hubbard Brook Experimental Forest in New Hampshire. Second, due to its prevalence throughout its range, the species biology is well studied (Jaeger, 1980b; Feder, 1983). And third, it is known to be sensitive to environmental changes (Spotila, 1972; Rolstad et al., 2002). Pough et al., (1987) found there were fewer red-backed salamanders in forest stands recently harvested compared to old-growth forest stands. The authors suggest major changes in forest ecosystems may negatively affect salamander populations and therefore other wildlife populations that depend on salamanders as a food source. Though numerous studies have assessed the effects of traditional timber harvest on amphibians (e.g., clearcutting, group selection; DeMaynadier and Hunter, 1995; Morneault et al., 2004; Semlitsch et al., 2009) there have been no studies (DeMaynadier and Hunter, 1998; Otto et al., 2013) on the effects of biomass harvesting on plethodontid salamander abundance and survival.
The purpose of our study is to estimate the abundance and survival of eastern red-backed salamanders in sites with varying intensities of biomass harvesting and to assess relationships between salamander abundance and survival and microhabitat characteristics impacted by biomass harvests. Additionally, we estimated the effect of variation in wet and dry days on capture probability (by season and year) to improve our estimates of abundance. We predicted decreasing DCWD and canopy closure would result in reduced abundance and survival of salamanders. Finally, due to their strict dependence upon environmental moisture, we predicted the amount of precipitation (number of wet or dry days before sampling) would significantly impact salamander capture probabilities.

**METHODS**

**STUDY AREA**

We conducted our study at the Southeast Purdue Agricultural Center (SEPAC) located in Jennings County, Indiana (39°02'08.3"N, 85°31'44.6"W). The study area was composed of a 46-ha mature oak-hickory (*Quercus spp.*, *Carya spp*.) stand that was divided into 10 sites (Fig. 1). The majority of soils on the study area are Bonnel-Blocher-Hickory and Blocher-Cincinnati silt loams and a small percentage of Avonburg and Hickory-Grayford silt loams, Hickory loam, and Caneyville rock outcrop (Baillie, 2001). The 30 y average annual precipitation is 1185.67 mm, with May being the wettest month with 127.76 mm and February the driest month with 74.93 mm of precipitation (Scheeringa, 2001). Average snowfall is 358.14 mm, with February receiving the most snow averaging 129.54 mm annually (Scheeringa, 2001). Average temperature ranges from −0.5 C in January to 24.1 C in July (Scheeringa, 2001).
FIELD METHODS

**Salamander data collection.**—At each of the 10 sites (Fig 1), we established an array of artificial cover objects (ACO). Each ACO array consisted of twenty-five $30 \times 30 \times 5$ cm untreated poplar boards arranged in a $5 \times 5$ grid with 5 m spacing between the boards. Artificial cover objects have been shown to provide accurate estimates of relative abundance, as well as detect changes in plethodontid salamander population densities (Marsh and Goicochea, 2003). By using ACOs we standardized the amount of added cover across treatments and minimized habitat disturbance, sampling time, and between-observer variability that can be problematic with other sampling methods (e.g., natural cover object, leaf-litter, and transect searches; Fellers and Drost, 1994). In addition previous studies have found no significant differences in morphology or demography between salamanders captured using ACOs compared to natural cover objects (Monti et al., 2000; Marsh and Goicochea, 2003; Moore, 2009).

During Oct. of 2011, we placed ACO arrays near the center of each site to minimize edge effects (DeMaynadier and Hunter, 1998). This pretreatment sampling allowed adequate time for weathering prior to data collection. In addition to marking every individual ACO with a steel wire stake flag (Presco Products, Sherman, TX), we also georeferenced the locations of the 250 ACOs using a GPS (Garmin GPSMAP 76 CSX, Olathe, KS). We checked for the presence of salamanders once every 2 w from Mar.–Jun. of 2012 prior to biomass harvest. Following the preharvest field season, we removed all ACOs prior to the start of timber harvest in Sep. 2012. After timber harvest in Nov. 2012, we returned ACOs to their original positions in direct contact with the soil. Subsequently, we checked for the presence of salamanders once every 2 wk in the spring (Mar.–Jun.) and fall (Sep.–Nov.). Our six postharvest field seasons occurred between Mar. 2013 and Nov. 2015.

Using visible implant elastomers (Northwest Marine Technology, Inc., Shaw Island, WA, U.S.A.), we uniquely marked salamanders with body locations and colors determined by a code generator developed by MacNeil et al. (2011). Salamanders were classified into three stage classes according to their snout vent length (SVL): young of the year (<25 mm), juvenile (25–32 mm), and adult (>32 mm) (Marsh and Goicochea, 2003). Two and a half years of data was collected under Purdue Animal Care and Use Committee (PACUC) guideline protocol #1111000296, and we analyzed these data together with an additional year of data collected from the same sites using the same protocol requirements. Due to a rare oversight, this additional year of data was not reported to the Purdue Animal Care and Use Committee (PACUC). Data was collected per the research framework of protocol #1111000296 but were erroneously not reported. We contacted PACUC and received permission to use the data in this publication. During sampling we noted species other than red-backed salamanders and promptly released them. All salamanders were captured and handled in accordance with the “Guidelines for use of Live Amphibians and Reptiles in Field and Laboratory Research,” revised by the Herpetological Animal Care and Use Committee (HACC) of the American Society of Ichthyologists and Herpetologists (2004).

**Biomass harvesting.**—Timber harvest was conducted in Oct. 2012 and included the following treatments: three sites of biomass harvesting without DCWD retention, three sites with DCWD retention (20–30%), three control sites were left unharvested, and one site was harvested as a commercial clearcut. For a more complete description of site details see (Ruhl, 2014). Prior to harvest, we calculated DCWD and woody biomass in the 10 sites. Point relascope sampling (PRS; Gove et al., 2001) was used in a $30 \times 30$ m grid of 389 sample points equally distributed throughout the entire study area to measure volume of DCWD. Within each ACO array and in a 5m buffer around each array, we censused DCWD with a small-end
diameter of >10.2 cm, length >0.3 m. For a more complete description of the relascope sampling technique see Ruhl (2014).

Downed coarse woody debris measurements were repeated postharvest in the biomass removal sites and across a subset of points in the clearcut site. We assumed DCWD characteristics would remain unchanged for the unharvested control areas during the course of the study. We created a gradient of DCWD percent cover within each of the six manipulated biomass treatment sites in the study area by moving material between plots in Feb. of 2013 to create a gradient of DCWD across sites. We used the PRS point samples and the DCWD censuses within the ACO arrays to determine how much postharvest residual DCWD was moved between sites to create the desired gradient. The clearcut site served as the 100% retention array.

**DATA ANALYSIS**

We used Pollock’s robust design model in Program MARK (MARK 8.0, Fort Collins, CO) to analyze our mark-recapture data (Pollock, 1982). We had seven primary sessions (spring 2012–fall 2015) and five to eight secondary sessions within each primary session. A span of at least 3 mo (mean 4.3 mo) separated primary sessions, with the largest being 9 mo between the end of spring of 2012 and the beginning of spring of 2013. This largest gap occurred during the interval at which timber harvest and corresponding postharvest DCWD manipulations were implemented. When classifying secondary sessions, we disregarded days with ≤1 salamander capture across all sites in the construction of the salamander encounter histories.

We developed 10 *a priori* candidate models to test salamander population parameters with variations on P (capture probability), c (recapture probability), S (survival), and γ″ and 1-γ′ (emigration and immigration), as well as a set of models comparing preharvest and postharvest data. Twelve models are presented in table to include multiple best fit models for sites where the cumulative AIC weight was 0.97 or greater. We estimated emigration and immigration using Markovian temporary emigration based on the assumption environmental factors such as surface moisture and temperature may influence a salamander’s presence on the surface (Bailey *et al.*, 2004b). We classified sessions as wet or dry based on the total rainfall 2 d and 2 wk prior to each sampling day. We established precipitation threshold values of 0.59 mm for 2 d prior and 38.74 mm for 2 wk prior based upon 95th percentile of the total observed rainfall at the study site throughout the duration of our study. We acquired precipitation data from the Purdue University iClimate weather station located at SEPAC headquarters approximately 4.3 km SW of the study site. We utilized these threshold criteria to classify observations as wet or dry at each temporal scale, thereby incorporating precipitation events into our models of capture probability. We also incorporated differences in season and year into our models. We assumed the population size (N) would vary between primary sessions for all models and survival would be constant within primary sessions. Additionally, we compared the performance of models that estimated the influence of treatment site type upon n, P, c, S, γ″, 1-γ′. Despite being able to assign animals to age classes, we did not have enough captures and recaptures to run separate models for each of these categories, so salamanders were pooled together for the models we ran in MARK.

We compared and selected the best model based on Akaike’s Information Criterion (AIC) weights and their Delta AICs, with the best model having the AIC weight closest to one and the Delta AIC closest to 0.00 (Akaike, 1973). Using Program MARK, we estimated abundance and survival probabilities at all 10 sites. We ran multiple linear regressions to determine if
any relationships existed between salamander abundance or survival with environmental factors such as DCWD and canopy cover. We ran a one-way analysis of variance (ANOVA) to determine whether there were any significant differences between abundance and survival estimates among the treatment sites.

**RESULTS**

During this study we captured a total of 996 red-backed salamanders with 234 recaptured individuals. Out of 996 captured red-backed salamanders, 127 were young of year, 181 were juveniles, and 688 were adults. We captured 240 individuals (54 recaptures) in sites with no DCWD retention, 254 (56 recaptures) in sites with 20–30% DCWD retention, 385 (100 recaptures) in the control sites, and 117 (24 recaptures) in clearcut. Other species captured during our study included southern two-lined salamander (Eurycea cirrigera), northern slimy salamander (Plethodon glutinosus), eastern newt (Notophthalmus viridescens), marbled salamander (Ambystoma opacum), Jefferson salamander (Ambystoma jeffersonianum), spotted salamander (Ambystoma maculatum), long-tailed salamander (Eurycea longicauda longicauda), and Northern dusky salamander (Desmognathus fuscus).

The best fit models constructed for each site had varying capture probability at the wet/dry, seasonal and annual scales, survival was constant, and there was no emigration and immigration (Table 1). The best fit models were those that incorporated precipitation compared to those that did not. The best models established for each study site provided abundance estimates ranging between 57 to 223 salamanders (Fig. 2B) and survival estimates ranging from 0.59 to 1.00 (Fig. 2A). The mean abundance estimate was greatest at the
clearcut site with 211 salamanders ($\pm SE = 18.9$). Sites with DCWD retention had a mean abundance of 127 salamanders ($\pm SE = 29.9$), while site without DCWD retention had a mean of 113 ($\pm SE = 28.9$). The control sites had a mean abundance of 186 salamanders ($\pm SE = 15.7$; Fig. 2B). The mean annual survival estimate for red-backed salamanders was greatest at sites with DCWD retention (20–30%) with a value of 0.8 ($\pm SE = 0.10$). Sites without retention had a mean survival estimate of 0.63 ($\pm SE = 0.10$) and unharvested control sites had a mean survival of 0.774 ($\pm SE = 0.23$). Mean annual survival estimates for clearcut sites were 0.825 respectively ($\pm SE = 0.09$; Fig. 2A). The abundance and survival estimates were not significantly different between biomass harvest treatments ($F = 2.231$, df = 8, $P = 0.1887$).
Percentage of DCWD and amount of canopy cover were significant predictors of salamander abundance both as separate factors (DCWD + Canopy; $F = 9.16$, df = 7; P-value = 0.0111, $R^2 = 0.6446$), and when they interacted together (DCWD × Canopy; $F = 10.48$, df = 7, P-value = 0.0084, $R^2 = 0.7597$; Figs. 3A, B). There was no relationship between abundance and just DCWD ($F = 1.776$, df = 8, P-value = 0.2194, $R^2 = 0.0794$). We found no correlations between survival and percentage of DCWD ($F = 0.74$, df = 8, P-value = 0.4135, $R^2 = -0.0297$) and between survival and the interaction of DCWD and canopy cover ($F = 0.7323$, df = 6, P-value = 0.5695, $R^2 = -0.0980$) but observed a positive trend between survival and DCWD and canopy cover as separate factors ($F = 1.177$, df = 7, P-value = 0.3625, $R^2 = 0.0379$). The trends between survival and canopy cover were also positive ($F = 1.18$, df = 7, P-value = 0.3625, $R^2 = 0.26$), however, canopy cover accounted for a small percentage of variation in survival.

**DISCUSSION**

Anthropogenic disturbance can have a profound impact on the abundance and survival of red-backed salamanders (Noël et al., 2007; Marsh et al., 2008, Clipp and Anderson, 2014). Therefore, Otto et al. (2013) identified a need for more research on the effects of microhabitat changes on salamanders within a biomass harvesting context. In addressing this issue, our models emphasize the importance of maintaining adequate DCWD and canopy cover for salamander abundance. Our results demonstrate a correlation between salamander abundance, DCWD and canopy cover.

Given their value as a forest bioindicator species (Welsh and Droge, 2001), it is critical to monitor the response of plethodontid salamanders (e.g., the Eastern Red-backed salamander) to novel forest harvesting techniques (Welsh and Droge, 2001; Otto et al., 2013). We observed overall decreasing, albeit not statistically significant trends in eastern red-backed salamander abundance in response to biomass harvest in relation to the amount of DCWD within the treatment sites (Fig. 2a). This may be attributed to the short sampling turnover and duration after the timber harvests on our study sites. Pough et al. (1987) found no statistical significance between the abundance of red-backed salamanders in harvested and old-growth stands in New York and that microhabitat characteristics, such as leaf litter depth and soil moisture, were significantly correlated with the abundance estimates.
Plethodontid capture probabilities may still be high if favorable conditions are met in the short term (abundant moisture and prey availability). The results of our regression analysis support the hypothesis salamanders are sensitive to changes in abiotic factors (Bailey et al., 2004a; McKenny et al., 2006; Homayack et al., 2011). Red-backed salamander abundance was positively correlated with the interaction between DCWD and canopy cover. We attribute this relationship to plethodontid dependence on sensitive microhabitat requirements on the forest floor, such as soil moisture, leaf litter depth, and available cover (Heatwole, 1962; Jaeger 1980b, Pough et al. 1987). Such forest floor microhabitat characteristics are altered by opening up the canopy as a result of timber harvest (Zheng et al., 2000, Bailey et al., 2004b). As canopy cover is reduced, the amount of rainfall that reaches the forest floor increases, which floods the soil and forces salamanders to surface where they utilize DCWD (Taub, 1961; Sulgaski and Clauussen, 1997; Otto et al., 2013). This effect can even be magnified in biomass harvest situations due to the decreased evapotranspiration on harvested sites by the removal of vegetative material, which decreases the available surfaces for transpiration (Zheng et al., 2000).

Although survival was over 50% in all of the treatment sites, survival was not significantly correlated with DCWD and canopy cover. This observation suggests other variables, such as prey/predator densities or soil moisture, may have greater effects on survival. Harper and Guynn Jr. (1999) found salamander densities and survival were positively correlated with gastropod densities in the southern Appalachian Mountains. Incorporating prey densities into future models may provide better estimates of population trends, as well as further understanding of the effects of biomass harvest on salamander abundance and survival. Ruhl (2014) also found although leaf litter depth was higher in unharvested sites, humidity levels were similar across sites and temperatures under ACOs were higher in harvested stands. Higher temperatures and stable humidity levels underneath ACOs may still provide suitable above-ground conditions for red-backed salamanders in recently harvested stands. This may lead to increased capture probabilities and survival for red-backed salamanders, even in recently disturbed habitats. One study in Connecticut found red-backed salamanders, along with northern spring peepers, were the most tolerant to forest fragmentation compared to other woodland amphibians (Gibbs 1998). He attributed this to the species’ fossorial lifestyle, which enable them to evade predators more effectively, as well as their low-dispersal rates and high densities. Many studies on red-backed salamanders lack estimates of survival in disturbed sites, an aspect we provide some insight into.

Our study in southern Indiana was the first to analyze the effects of biomass harvesting on red-backed salamander abundance and survival. Additionally, we provide novel insights into the benefits of incorporating 2 d and 2 wk seasonal and annual precipitation scales into our capture probabilities. Assuming a uniform distribution of precipitation over a sampling period can produce inaccurate and biased population estimates for salamanders, which in turn leads to misguided management practices (O’Donnell et al., 2014). As precipitation events become more extreme due to climate change, it is important to consider rain pulses and patterns, as well as soil moisture, to better assess these effects on amphibian populations (O’Donnell et al., 2014; Walls et al., 2013). Extreme rain events distributed over long periods of little to no precipitation are becoming more common (Seneviratne et al. 2012). Relying solely on one replicate measure of total rainfall leads to the assumption precipitation is spread equally throughout the sampling period and therefore does not accurately capture the influence of variation in rain patterns. Our approach of classifying 2 d and 2 wk moisture levels provide more precise model estimates of demographic parameters of red-backed salamanders.
Due to their sensitivity to disturbance events and land modification, it is essential to design proper monitoring schemes for salamander populations in those areas (Hocking et al., 2013; Otto et al., 2014). Because research points to red-backed salamanders as a bioindicator species (Welsh and Droge, 2001; Herbeck and Larsen, 1999), their response to biomass harvesting can be used to monitor community response to anthropogenic activity (Otto et al., 2014). We suggest assessing the long-term effects of biomass harvest on red-backed salamanders as DCWD progresses into other stages of decay in the future.

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LITERATURE CITED


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