The Effect of Forest Clear-cut Practices on Salamanders and their Habitat

BY

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Acknowledgements

First, I want to acknowledge that I carried out this project on the unceded territory of the Mi'kmaq and Wolastoqiyik (Maliseet) Peoples, covered by the "Treaties of Peace and Friendship." The treaties did not deal with the surrender of lands and resources. However, they recognized Mi'kmaq and Wolastoqiyik titles and established the rules for what was to be an ongoing relationship between nations. I recognize that their past, present, and future are intimately connected to the health and well-being of the land, waters, and all living beings. Reflecting on my research into forest management practices and their impact on forest health and salamander populations in New Brunswick, I recognize that this work takes place in the context of ongoing colonialism and systemic injustice. I commit to honouring this land's past and present knowledge keepers in pursuit of a more just and sustainable future by striving for closer relationships with Indigenous Peoples in New Brunswick.

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Abstract

The silvicultural practice of clear-cutting aims to emulate stand replacing natural disturbances, and its effects on forest ecosystems and processes have been well-studied. Along with clear-cutting, glyphosate-based herbicides are often applied to clear-cut forests in conjunction with replanting conifers. Yet, the impact of herbicide treatment on forest ecosystems is poorly known. One method of assessing the health of a forest ecosystem is to study the population health of a known bioindicator species. A well-known bioindicator species along the Atlantic coast of North America is the Eastern Red-backed Salamander (*Plethodon cinereus*). My research objective was to examine the impact of clear-cutting with and without herbicide treatment on *P. cinereus* populations across years post-harvest. I sampled a total of 30 sites and 243 salamanders within and around the Acadia Research Forest in Fredericton, New Brunswick. I found that control forests (i.e., unharvested plots) had the highest percent canopy cover, soil moisture, and lowest soil temperature. In the untreated clear-cut forests, soil temperature increased and soil moisture decreased as the forests aged. In contrast, soil temperature decreased and soil moisture increased in the treated clear-cut forests as they aged. I found the highest abundance of salamanders in the control forests; specifically, salamander abundance was 4 and 18 times more than treated clear-cut and untreated clear-cut forests, respectively. I found 3.2 times more salamanders in the treated clear-cut forests than in untreated clear-cut forests. Body condition was 1.1 times greater in the harvested forests than in the control forests. Overall, unharvested forests provide optimal environmental conditions for salamander populations. Within 26 years after clear-cutting, clear-cut forest treated with herbicides provide more suitable environmental conditions for salamander habitat than untreated clear-cut forests. The findings of this study provide valuable insight into the health of harvested forest ecosystems and highlights the importance of protecting the remaining older Acadian forests to mitigate global biodiversity loss.

Introduction

The degradation, conversion, and fragmentation of habitats by humans are main causes of biodiversity loss (Newbold et al. 2015). Specifically, large-scale spatial disturbance of forestry practices alters the forest age, structure, composition, and spatial-temporal dynamics of the forest ecosystem. These changes have a strong influence on a variety of species' habitats as some species are associated with new forest conditions and others are associated with old forest conditions (Keenan and Kimmins 1993; Özkan and Gökbulak 2017). Increasingly, forest science research is investigating strategies to promote a holistic approach to Euro-Canadian economic harvesting strategies that considers ecosystem functionality, called ecological forestry. One aim of this recent focus of forest research is to unite Western science approaches with traditional resource management values of Indigenous Peoples, such as New Brunswick's local Wolastoqiyik (Maliseet), Mi'kmaq and Peskotomuhkati (Passamaquoddy) Peoples (Blakney 2003). Yet, there is still a long way to go to apply this type of collaborative, holistic approach to forestry nationally. Overall, Canada is moving toward sustainable forest management that aims to balance resource and timber extraction with other ecological values such as biodiversity and ecosystem services. The general strategy for timber harvest and planning is the emulation of natural disturbances, in which the size, location, and timing of harvest mimics the dominant natural disturbance in a particular region.

The New Brunswick forest industry strives to balance the economic value of timber harvesting with maintaining biodiversity and ecological characteristics of the Acadian Forest (Government of New Brunswick 2009; SGS Belgium S.A. 2018). A dominant natural disturbance in the Acadian Forest of New Brunswick is insect outbreak, primarily Eastern Spruce Budworm, which results in large-scale conifer mortality. A commonly-used harvest method for large-scale tree removal is clear-cutting (Long 2009). Clear-cutting uniformly removes trees from a plot, leaving few behind post-harvest, causing the forest to regenerate at an even-age. Clear-cutting differs from natural disturbances in several ecological respects (Esseen 1997). Clear-cutting removes tree biomass (no living or dead trees left) rather than leaving standing dead trees or woody debris, and the resultant area is much more homogeneous than natural systems as all live trees are removed (Franklin et al. 2000). Environmental changes that arise from clear-cutting are largely due to complete canopy removal, which exposes the forest floor to an increased level of sunlight. These changes create warmer, drier microclimates and reduced leaf litter (Chen et al. 1993; Carlson and Groot 1997; Hashimoto and Suzuki 2004). By reducing downed woody debris, clear-cutting also disrupts the cycle of decomposing woody debris that is essential to the habitat of many forest organisms (Spies et al. 1988; Tinker and Knight 2000; Romano et al. 2018; Rose et al. 2001). These conditions remain during the early stages of forest regeneration (1-10 years), and the vegetation community is typically dominated by early herbaceous flora unless additional management occurs (e.g., replanting, silviculture, or chemical treatments).

In some replanted forests, silvicultural treatments are applied to guide stand regeneration to a desired state. When the desired state is a conifer dominated stand, the commonly used silviculture technique is the application of a glyphosate-based herbicide to suppress deciduous and shrub vegetation, resulting in earlier dominance of coniferous species (Mendes 2021). The herbaceous layer in forests contributes to nutrient cycling and provides cover against direct sun exposure to the forest floor microhabitat (Elliott et al. 2015). Herbicide application is therefore very likely to affect microhabitat conditions directly and indirectly, such as: coarse woody debris, soil temperature, and moisture. These are potential changes to microhabitat features that are important characteristics of wildlife habitat (Keenan and Kimmins 1993; deMaynadier and Hunter Jr. 1995; Nyland 2016).

Abundant and healthy salamander populations can reflect positive ecosystem health and integrity because they are a bioindicator species (Welsh JR. and Droege 2001). A bioindicator species is a living organism that is used to monitor and assess the health and quality of an ecosystem or environment. Specifically, salamanders regulate the capture of carbon and nutrient recycling and balance food webs, which both contribute to ecosystem stability (Wyman 1998; Keitzer and Goforth 2013; Walton 2013; Hickerson et al. 2017). In particular, plethodontid salamanders are considered bioindicators of forested environments because of their sensitivity to environmental change (Davic and Welsh 2004). Their presence and abundance changes along environmental gradients, for instance, the abundance of several species is positively related to soil moisture levels (Grover 2000; Fleishman and Murphy 2009). Additionally, previous research demonstrated that the abundance and body condition of plethodontid salamanders is closely tied to forest growth, development, and structural changes, thus classifying them as indicator species in forested environments (Welsh and Hodgson 2013). Eastern Red-backed Salamanders (*Plethodon cinereus*) are excellent study organisms because they are the most widely distributed and abundant vertebrate species in eastern North America, which allows for easily accessible populations for research (Petranka 1998; Welsh and Droege 2001). Considering salamander sensitivity to forest health, it is unsurprising to find that one of the threats to declining salamander populations worldwide is habitat loss from forestry activities (Collins and Storfer 2003).

Forestry alters several environmental variables that influence salamander abundance. Plethodontid salamanders are terrestrial, ectothermic, and lungless, relying mainly on their skin for respiration and hydration (Petranka 1998). Due to their natural history, salamanders require cool and moist habitat, like that under downed wood, rocks, and leaf litter in shaded forests (Grover 1998). The immediate environmental changes associated with clear-cutting increase the risk of desiccation to salamanders (deMaynadier and Hunter Jr. 1995; Rothermel and Luhring 2005; Kulmala et al. 2014). The effects from herbicide use may also change the environment salamanders rely on for survival. Herbicide application reduces the herbaceous midstory during forest regeneration, potentially enhancing or extending the environmental effects of canopy removal on salamander populations (Riedel et al. 2008). However, the removal of the herbaceous layer also allows a quicker succession of an overstory layer, which may increase the likelihood of fallen logs in the early stages of forest regeneration (Dempster 2022). Downed logs are a key component in salamander habitat, providing cool and moist refugia that are especially valuable in harvested forests with limited overstory (Kluber et al. 2009; Garcia et al. 2020). The contrasting effects of clearcutting and herbicide use on important salamander habitat variables may lead to complex and interacting effects on salamander populations.

The overall objective of this study was to examine the impacts of common New Brunswick forestry practices on salamander populations. I undertook this study in and around the Acadia Research Forest (ARF). There are three species of terrestrial salamander known to occur at the ARF: the Eastern Red-backed Salamander (*Plethodon cinereus*), the Yellow-spotted Salamander (*Ambystoma maculatum*), and the Eastern Newt (*Notophthalmus viridescens*) (iNaturalist 2023). I determined population health by quantifying three components of the salamander populations: community structure, indication of threats, and reproductive activity. Due to their bioindicator status and abundance in the study area, data collected from *P. cinereus* were used in all aspects of this study, whereas the other two species were only used to describe community structure. I hypothesized that the removal of the herbaceous layer by herbicides would enhance the negative effects of canopy cover removal from clear-cutting on salamander habitat between 5-26 years post-harvest. I used three lines of evidence to test my hypothesis:

1. I compared community structure of salamanders in different silviculture treatments along a seral gradient, comparing species richness and abundance of salamanders between clear-cut forests treated and untreated with a herbicide. I predict that species richness and abundance would be lower in treated clear-cut forests than untreated.

2. I investigated individual health of *P. cinereus* by documenting body condition, tail autotomy, and the presence of cuts and lesions. I predict that body condition would be lower in treated clear-cut forests than untreated, but the proportion of individuals with tails autotomized/injuries would exhibit the opposite trend.

3. I measured reproductive activity of *P. cinereus* by calculating juvenile-adult ratios and counting gravid females and egg masses in forests. I predict that more gravid females and egg masses will be found in untreated clear-cut forests than treated, as well as the juvenile-adult ratio will be higher reflecting more recruitment in these forests.

Though past observational and experimental studies have documented reduction and recovery in salamander abundance after clear-cutting, little research compares the effects on salamander population health from the interaction between clear-cut harvesting and herbicide application (Herbeck and Larsen 1999). My work will contribute to a growing body of research on the impacts of applying the world’s most used herbicide, glyphosate, on forest ecosystems.

Methods

Preliminary study

Deciduous and mixed forests provide the ideal combination of moisture, temperature, food, and cover that *P. cinereus* need to survive and thrive (Spotila 1972; Petranka 1998). Conifer forests may not provide the same habitat features because of reduced leaf litter, which increases evaporative losses (Williams et al. 1990). I conducted a preliminary study documenting the presence of salamanders in deciduous, coniferous, and mixed wood forest types to assess whether relative salamander abundance is comparable across forest types to assess the feasibility of studying salamanders in areas replanted with conifers. In May and June of 2023, I randomly sampled 11 forest plots in Gagetown, NB, and found salamanders in all forest types (Table 1). The observed salamander community was dominated by *P. cinereus*. However, I found two Eastern Newts and one Spotted Salamander in the coniferous forest type. The findings from the preliminary study indicated that salamander abundance is sufficient in coniferous forests to facilitate research on my main study objectives.

Table 1. Results from a preliminary study assessing salamander presence between deciduous, coniferous, and mixed wood forest types. I present the number of plots sampled, total number of salamanders counted, and species richness in each forest type.

Forest treatment	***Number of plots***	***Number of salamanders***	***Species richness***
Coniferous	5	12	3
Deciduous	3	10	1
Mixed wood	3	7	1

Main Study

Study Area

I conducted fieldwork in the ARF, approximately 25 km east of Fredericton, New Brunswick, on the traditional territory of the Wolastoqey Nation and the Wabanaki Confederacy. The ARF consists of about 9000 ha of coniferous, deciduous, and mixed forests of both the Grand Lake and Eastern Lowlands Ecoregions (Swift et al. 2005). Overall, 30 forest plots were selected; ten control plots, ten untreated clear-cut plots, and ten treated clear-cut plots (Figure 1).

All clear-cut plots were selected within an evenly distributed seral gradient; to be included, clear-cut plots had to be harvested within the past 5-26 years and be more than 3 ha in size.

The control forest plots are ecological reserves at the ARF and have no known history of silvicultural practices. These plots were included to act as an Acadian Forest reference condition. Control plots consisted of a diverse variety of mixed woods, including native tree species of boreal (e.g., Balsam Fir, Jack Pine, and White Spruce) and temperate forests (e.g., Sugar Maple, Yellow Birch, and Red Spruce). Ground cover of control plots consisted of woody debris, leaf litter, and a herbaceous layer of mainly ferns or moss. The untreated clear-cut forest plots I sampled varied in their time since harvest from 5 to 24 years. Dominant tree species in the untreated clear-cut plots were generally mixed wood. The ground cover in untreated clear-cut plots lacked a significant presence of leaf litter but contained some woody debris and a dense herbaceous layer. Limited herbicide treatment has occurred within the ARF, thus treated clear-cut forest plots were located north of the ARF on New Brunswick Crown land near Lower Durham, New Brunswick. Similarly to the untreated clear-cut plots, the treated clear-cut plots varied in time since harvest from 7 to 26 years. In addition, all the treated clear-cut plots were treated with a glyphosate-based herbicide five years post-harvest. Ground cover in the treated clear-cut plots had minimal volume of woody debris and a less dense herbaceous layer than the untreated clear-cut plots. Forest plots I sampled were chosen randomly from a Natural Resources database on Crown Land silviculture history in New Brunswick forests.

Field Surveys for Salamanders

Field surveys for salamanders occurred from July to August 2022. Each sampling day, one to three plots were sampled in a random order to account for the effect of time of day and season on salamander encounter rates. One survey was completed per forest plot. Standardized area-constrained searches within a circle with a 30 m radius were conducted by two observers during the day to locate salamanders. These standardized concentric surveys involved circular transects with the radius of 30 m, 20 m, 10 m, and 0 m from the center of the survey (Figure 1). As researchers walked through the survey, two observers hand-captured salamanders of all species by looking under cover objects such as rocks and logs within 5 m of the survey transect

(Houze and Chandler 2002). When determining the center point of a survey, I ensured that a minimum of a 50 m buffer of similar habitat surrounded the survey to avoid forest edge effects (Murcia 1995). After the center of the survey was established, the soil moisture (Campbell Scientific Hydrosense II Soil-water Sensor CS658), temperature (Master Chef Digital Instant Read Thermometer), and pH (Hanna Instruments Direct Soil pH Measurement Kit HI99121) was recorded at the center of the survey. Soil moisture was measured at an approximate depth of 20 cm, pH at 12 cm, and temperature at 10 cm into the soil. A 360° photo (Ricoh Theta 360) was captured of each survey to document canopy cover and ground cover. Active sampling surveys for salamanders, like those used herein, are constrained to the ground-surface and do not access the more complex, underground salamander habitats that are often unreachable with non-destructive methods.

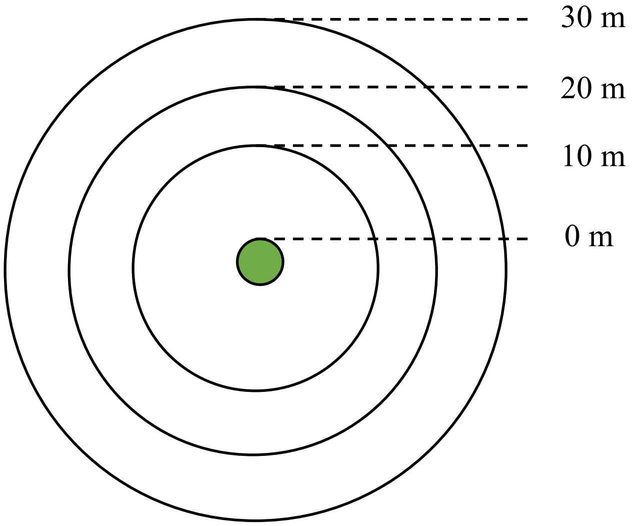

Figure 1. A diagram of the standardized concentric sampling method. The center point of the survey is represented by the green circle. The circular transects the researchers walked are represented by the black solid lines. Researchers looked under cover objects within 5 m on both sides of the survey transects.

Each time a salamander was captured, I identified and recorded the species. I also weighed all salamanders with a digital scale (US-Ranger) (± 0.01 g) and measured their snout–vent length (SVL) with digital callipers (Mastercraft) (± 0.1 mm) (i.e., the distance from the snout to the anterior boundary of the cloaca). Additional data was recorded for *P. cinereus*; each individual was categorized into two age classes: juveniles (≤ 32 mm) and adults (> 32 mm)

(Mazerolle et al. 2021). Adult *P. cinereus* were sexed as they were the only species with reliable secondary sexual characteristics outside of breeding season. This species was sexed using the candling method and examination of snout shape (Gillette and Peterson 2001). I also noted whether females were gravid, and any injuries such as cuts, lesions, and occurrence of tail autonomy. All salamanders were released at their point of capture immediately after processing. Salamanders were handled with nitrile gloves and measures to prevent disease spread were implemented in our data collection procedure (Canadian Herpetofauna Health Working Group 2017, Declining Amphibian Population Task Force 1998). I conducted this research under authorization from The Department of Natural Resources and Energy Development, Government of New Brunswick (Scientific Permit No. SP22--002) and my protocols were approved by the Mount Allison University Animal Care Committee (ACC # 103181).

Statistical Analysis

I used R software for my statistical analysis (v 4.2.2; R Core Team 2022) that consisted of a series of models (see details below). Before analyses, I explored the data and ensured there were no unexplainable outliers or collinearity between predictor variables (Zuur et al. 2010). For all models, α was set at 0.05. For each model, I checked model assumptions of normality of residuals, and homogeneity of variance using a histogram of the residuals and a normal probability plot (Q-Q plot) and a visual of the residuals versus the fitted values (Zuur et al. 2010). Data are presented as mean ± standard error (SE) in the text unless otherwise specified. For scatterplot data visualizations, I used coloured lines to represent a fitted linear trendline that was calculated using the function '*geom_smooth(method = lm)*' from the R package '*gpplot2*' (v3.4.1; Wickham 2016).

Environmental Factors

I tested for differences in environmental factors among forest treatments to evaluate my predictions about how forest harvest practices alter salamander habitat. First, I performed two generalized linear models (GLMs) using the function '*glm'* from the *'stats'* R package (R Core Team 2022) to test for an effect of forest treatment (categorical variable including control, untreated clear-cut, and treated clear-cut levels) and years since harvest (continuous variable ranging from 5 to 26 years) on canopy cover (%) and soil moisture (%). Canopy cover and soil moisture are percentages (ranging between 0 and 100), so I converted these data to proportions and used a binomial distribution for analysis (Altham 1978). I also ran two linear models (LMs) using the function '*lm*' from the '*stats*' R package (R Core Team 2022) to test for an effect of forest treatment and years since harvest on soil temperature (^{o}C) and soil pH. The interaction between forest treatment and years since harvest was initially included in both GLMS and LMs. If not significant, the interaction was removed and the model re-run to allow interpretation of the main effects (Gelman and Hill 2006). Significance of the interaction was assessed from the model output provided by the '*anova*' function from the '*stats*' R package for GLMs and LMs (R Core Team 2022). Test statistics were generated using the χ^2 test for GLMs and the F statistic for LMs (R Core Team 2022). I also examined significant interactions visually and data visualisations were made using the R package '*gpplot2*' (v3.4.1; Wickham 2016). Lastly, if the main effect of forest treatment was significant, I generated contrasts between each (three comparisons in total) using the function '*emmeans*' from the R package '*emmeans*' (Lenth 2022). *P* values generated for comparisons between forest treatment were corrected using Tukey's HSD multiplicity adjustment (Lenth 2022). I used the same approach for the inclusion of interactions and post-hoc multiple comparisons in models analysing community structure, indication of threats, and reproductive activity data (detailed below with some changes to the use of R packages for mixed effect models).

Community Structure

During each survey I documented species richness and salamander abundance. I describe differences in species richness found between each forest treatment, because there was too little variation in the data (range between 0 to 3) to allow statistical analysis. I summarized abundance of salamanders using data from *P. cinereus* only due to their common presence throughout each forest treatment. To test for an effect of forest treatment and years since harvest on *P. cinereus* abundance, I conducted a GLM with a Poisson distribution, because this response variable is count data (Consul and Jain 1973).

Indication of Threats

For each salamander captured, I noted weight, length, whether its tail was complete, and whether it had any other injuries. I do not describe findings regarding injuries because none were observed. I estimated adult salamander body condition using the scaled mass index (SMI) ordinary least squares (OLS) estimation (Peig and Green 2010). SMI which accounts for covariation between body size and body mass components during calculations by correcting body mass by a relative measure of body length (Peig and Green 2009). I chose the OLS estimation method based on predicted SMI curves using OLS regression of body mass against SVL and the robust scaled mass index (Figure A1) (Peig and Green 2010). I performed a linear mixed effect model (LMM) using the function '*lmer'* from the R package '*lmerTest'* (Kuznetsova et al. 2017) to evaluate *P. cinereus* body condition (SMI). This model included the fixed effects of forest treatment, year since harvest, sex (a categorical variable including male and female levels), and the random effect of plot identity (a categorical unique identifier for individual forest plots). I described findings of adult *P. cinereus* tail autotomy because very few autotomized tails were observed during my study.

Reproductive Activity

I described data regarding gravid females and egg masses because only a few were observed during my study. I used a GLM with a binomial distribution to evaluate the effects of forest treatment and years since harvest on the proportion of *P. cinereus* juveniles (out of the total number of individuals observed) during each survey (McCullagh and Nelder 1989).

Results

Environmental Factors

Canopy cover was significantly positively related to years since harvest ($\chi^2_{1, 24} = 169.47$, $p < 0.01$). On average, a forest will increase in canopy cover by 50% each year after it was clear-cut. Canopy cover also differed among forest treatments ($\chi^2_{2, 25} = 326.90$, $p < 0.01$). Control forests had the highest canopy cover of all forest treatments with an average of 77.5 ± 2.28%. In contrast, canopy cover was on average 83.5% lower in the treated clear-cut forest treatment, which averaged 10.7 ± 1.08%, and on average 89.1% lower in the untreated clear-cut forest treatment that averaged 7.61 ± 0.78% (Table 2a; Figure 2a).

For soil temperature, I observed a significant interaction between years since harvest and forest treatment ($F_{1, 25} = 6.23$, $p = 0.02$). Soil temperature was relatively constant as the forest ages after harvest in the untreated clear-cut forest treatment, whereas in the treated clear-cut forest treatment, the soil temperature was negatively related to years since harvest (Figure 2b). Although the interaction prevents interpretation of main effects, summary statistics suggest a difference in soil temperature between harvested and control forests; soil temperature was lower in the control (14.5 ± 0.36^{o}C) than the untreated clear-cut (16.1 ± 0.39^{o}C) and treated clear-cut (16.0 ± 0.61^{o}C) forest treatments.

I also observed a significant interaction between years since harvest and forest treatment for soil moisture ($\chi^2_{1, 29} = 211.18$, $p < 0.01$). The soil in untreated clear-cut forests become drier over time, whereas in treated clear-cut forests, the soil becomes slightly wetter as the forest ages (Figure 2c). Although the interaction prevents interpretation of main effects, summary statistics suggest a difference in soil moisture between harvested and control forests; soil moisture was greater in the control forest treatment (30.6 ± 4.16%) than the untreated clear-cut (21.3 ± 4.61%) and treated clear-cut (14.8 ± 2.03%) forest treatment. In contrast, soil pH was not related to years since harvest ($F_{1, 26} = 3.59$, $p = 0.07$) nor differed among forest treatments ($F_{2, 26} = 1.79$, $p = 0.19$; Table 2b; Figure 2d).

Table 2. Post-hoc pairwise comparisons testing for differences among the forest treatments (control, untreated clear-cut, and treated clear-cut) for canopy cover (%) and soil pH. For these environmental variables, an interaction was not significant in initial models (a generalised linear model and a linear model, respectively), which allows for interpretation of main effects (Figure A2). I present estimates (*β*) and their corresponding standard error (*SE)*, test statistics (*z*-values for GLMM or *t*-values for LMM), and corrected *p*-values (p_{corr}) for multiple comparisons. Bolded values indicate significance.

Response Variable & Contrasts	***β***	***SE***	***Test Statistic***	p_{corr}
(A) Canopy cover				
Control vs. Untreated	**41.81**	**8.95**	**17.44**	**< 0.01**
Control vs. Treated	**28.83**	**6.55**	**14.80**	**< 0.01**
Untreated vs. Treated	**0.69**	**0.08**	**-3.32**	**< 0.01**
(B) Soil pH				
Control vs. Untreated	-0.45	0.74	-0.62	0.81
Control vs. Treated	-1.35	0.78	-1.72	0.22
Untreated vs. Treated	-0.90	0.45	-1.99	0.14

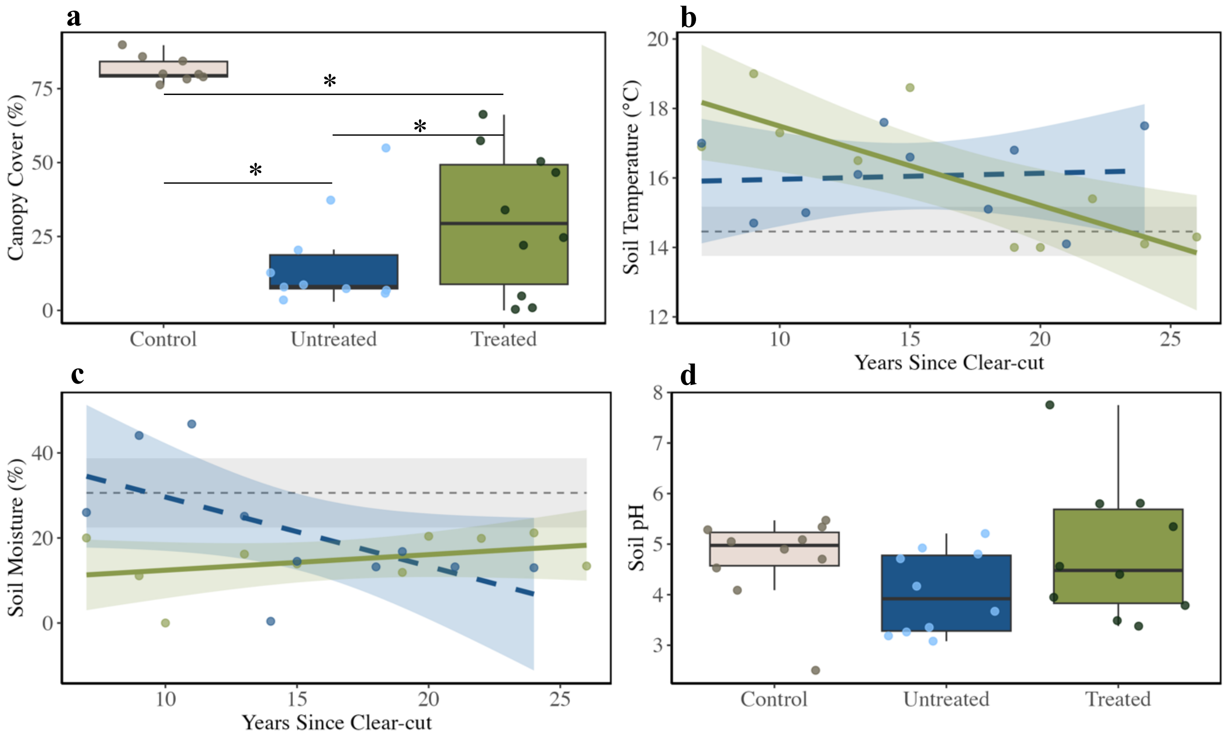

Figure 2. Plots depicting **(a)** a significant difference of canopy cover among forest treatments, **(b)** a significant interaction between years since harvest and forest treatment for soil temperature (°C) and **(c)** soil moisture (%), and **(d)** no significant difference in soil pH among forest treatments. Untreated clear-cut and treated clear-cut forest treatments are shown with a dashed blue line and a solid green line, respectively. Points represent raw data. Dotted grey lines represent the mean value in the control forests. For the boxplots, the box itself represents the interquartile range of the data (IQR) and the lower and upper limits of the box are the first and third quantile, respectively. The bottom/top whiskers that extend from the box are represent the minimum and maximum values of the data that are not considered outliers (1.5 times the IQR). Significant comparisons between forest treatments are shown using a bar connecting them and an asterisks (*).

Community Structure

I observed three salamander species: Eastern Red-backed Salamanders, Spotted Salamanders, and Eastern Newts. On average, species richness was 1.4 ± 0.2 in the control forest treatment, 1.1 ± 0.1 in the untreated clear-cut forest treatment, and 1.0 ± 0.0 in the treated clear-cut forest treatment. The observed salamander community was dominated by *P. cinereus*. More than one species was observed in only 5 plots of 30 plots: one Spotted Salamander in each of three control plots, one Eastern Newt (eft phase) in one control plot, and one Spotted Salamander in one untreated clear-cut plot.

In total, I captured 183 *P. cinereus* in the control forest treatment, 10 in the untreated clear-cut forest treatment, and 42 in the treated clear-cut forest treatment. *Plethodon cinereus* abundance was significantly, positively related to years since harvest ($\chi^2_{1, 29} = 244.10$, $p < 0.01$). On average, a forest will increase in salamander abundance by 0.9 ± 0.2 (~1) individual every 10 years, within 26 years post-clear-cut. *Plethodon cinereus* abundance also differed among forest treatments ($\chi^2_{2, 27} = 258.71$, $p < 0.01$). The control forest had the highest average abundance (47.0 ± 12.6). The abundance of *P. cinereus* was higher in the treated clear-cut forests (2.1 ± 0.6) than in the untreated clear-cut forests (0.8 ± 0.3) (Figure 3; Table 3a).

Table 3. (A) Outcomes of the generalized linear model that examined the effect of forest treatment and year since clear-cut on *P. cinereus* abundance. I present estimates (β) and their corresponding standard error (*SE)*, z-values, and p-values (p). **(B)** Post-hoc pairwise comparisons among the effect of forest treatment (control, untreated clear-cut, and treated clear-cut) on *P. cinereus* abundance. I present estimates (β) and their corresponding standard error (*SE)*, t-ratios, and p-values (p_{corr}) corrected for multiple comparisons. Bolded values indicate significance.

(A) Model Output				
Fixed effects	β	*SE*	z-value	p
Intercept (Control)	**2.91**	**0.07**	**39.32**	**< 0.01**
Forest treatment (Untreated)	**-4.10**	**0.52**	**-7.96**	**< 0.01**
Forest treatment (Treated)	**-3.10**	**0.52**	**-6.02**	**< 0.01**
Years since harvest	**0.09**	**0.02**	**3.64**	**< 0.01**
(B) Post-hoc Multiple Comparisons				
Forest treatment Contrasts	β	*SE*	z-ratio	p_{corr}
Control vs. Untreated	**60.37**	**31.10**	**7.96**	**< 0.01**
Control vs. Treated	**22.21**	**11.44**	**6.02**	**< 0.01**
Untreated vs. Treated	**0.37**	**0.12**	**-3.10**	**< 0.01**

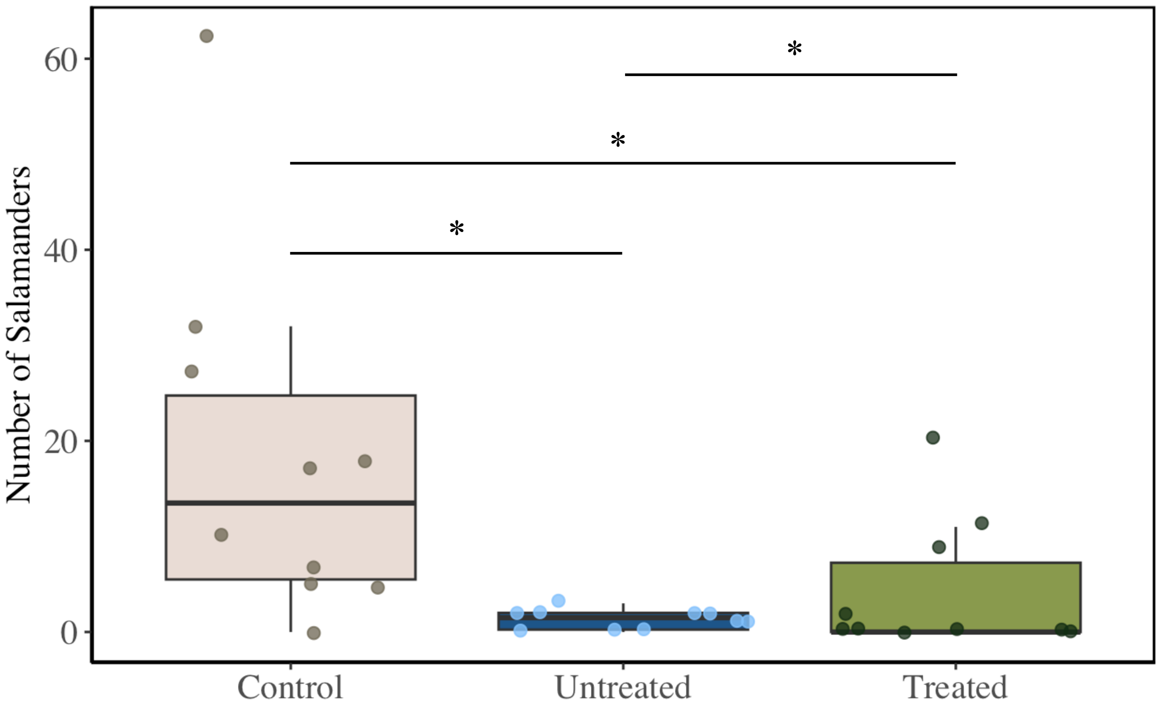

Figure 3. Salamander abundance in control (beige fill), untreated clear-cut (blue fill), and treated clear-cut (green fill) forest treatments. Median values are represented by the thick horizontal line inside the boxes. Raw data are represented by points. The box itself represents the interquartile range of the data (IQR) and the lower and upper limits of the box are the first and third quantile, respectively. The bottom/ top whiskers that extend from the box are represent the minimum and maximum values of the data that are not considered outliers (1.5 times the IQR). Significant comparisons between forest treatments are shown using a bar connecting them and an asterisks (*).

Indication of Threats

I investigated potential threats to individual salamanders by calculating their body condition and documenting tail autotomy and injuries (e.g., cuts and lesions) during each survey. *Plethodon cinereus* body condition was significantly, negatively related to years since harvest (Table 4a). Body condition was lowest in the control forests, averaging 0.74 ± 0.05 SMI (Table 4b; Figure 4). The untreated clear-cut and the treated clear-cut forests had an average of 1.26 ± 0.14 and 1.24 ± 0.15 SMI respectively. For both harvested forest treatments, body condition of salamanders was approximately 0.5 SMI higher than the control forests (Table 4b). Body condition did not differ between sexes (Table 4a). As for incidences of recorded tail autotomy, I only found 11 *P. cinereus* that were missing or regenerating their tail: 10 in control forests and 1 in the untreated clear-cut forests. Of these 11 salamanders, 9 were female and 2 were male. I did not observe any salamanders that had other injuries (e.g., cuts and lesions in addition to tail autotomy).

Table 4. (A) Outcomes of the linear mixed effects model that examined the effect of forest treatment, year since clear-cut, and sex on *P. cinereus* body condition (SMI). I present estimates (β) and their corresponding standard error (*SE)*, *t*-values, and *p*-values (*p*). For random effects, I present the variance (σ^2) and standard error (*SE*). **(B)** Post-hoc pairwise comparisons among the effect of forest treatment (control, untreated clear-cut, and treated clear-cut) on *P. cinereus* body condition (SMI). I present estimates (β) and their corresponding standard error (*SE)*, *t*-ratios, and *p*-values (p_{corr}) corrected for multiple comparisons. Bolded values indicate significance.

(A) Model Output				
Fixed effects	β	*SE*	*t*-value	*p*
Intercept (Control, Female)	**0.87**	**0.02**	**39.22**	**< 0.01**
Sex (Male)	-0.02	0.03	-0.78	0.44
Years since harvest	**-0.02**	**0.01**	**-2.32**	**0.02**
Forest treatment (Untreated)	**0.52**	**0.19**	**2.74**	**< 0.01**
Forest treatment (Treated)	**0.50**	**0.20**	**2.48**	**0.01**
Random effects	σ^2	*SE*		
Plot ID	0.00	0.00		
Residuals	0.03	0.04		
(B) Post-hoc Multiple Comparisons				
Forest treatment Contrasts	β	*SE*	*t*-ratio	p_{corr}
Control vs. Untreated	**-0.52**	**0.19**	**-2.73**	**0.02**
Control vs. Treated	**-0.50**	**0.20**	**-2.47**	**0.04**
Untreated vs. Treated	-0.02	0.07	0.28	0.96

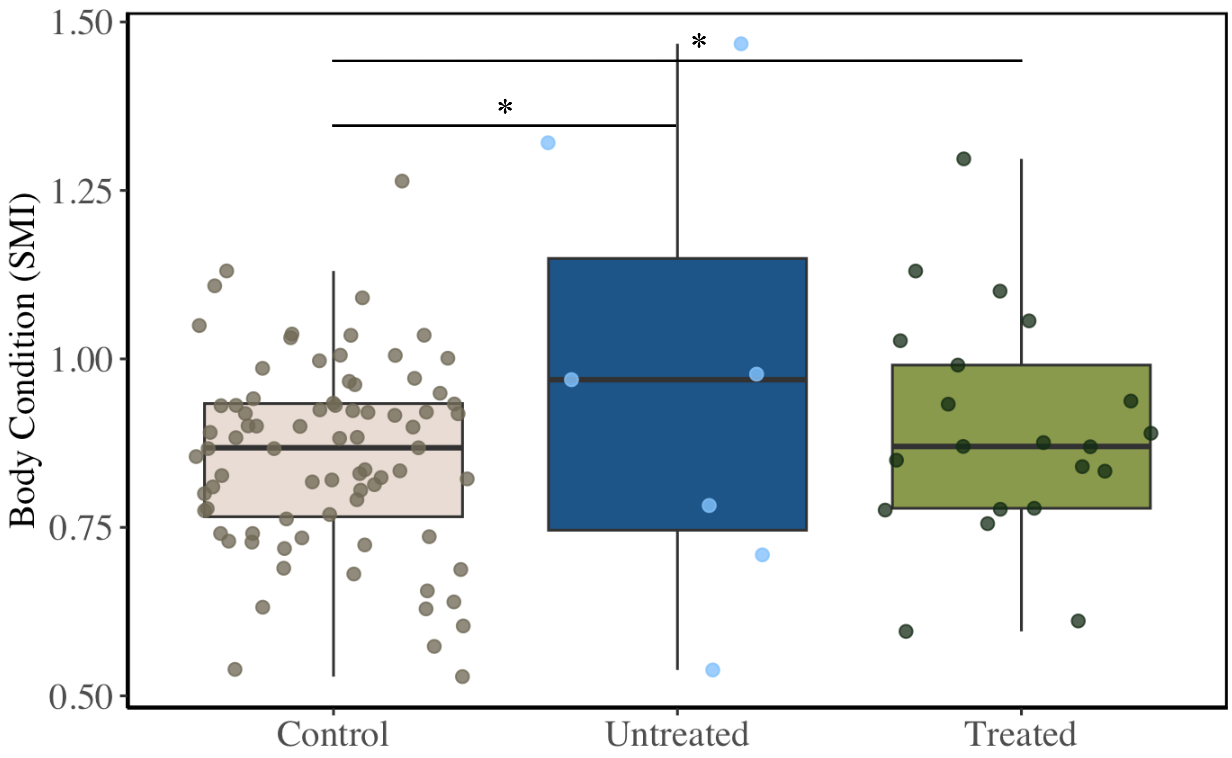

Figure 4. Salamander body condition in control (beige fill), untreated clear-cut (blue fill), and treated clear-cut (green fill) forest treatments. Median values are represented by the thick horizontal line inside the boxes. Raw data are represented by points. The box itself represents the interquartile range of the data (IQR) and the lower and upper limits of the box are the first and third quantile, respectively. The bottom/ top whiskers that extend from the box are represent the minimum and maximum values of the data that are not considered outliers (1.5 times the IQR). Significant comparisons between forest treatments are shown using a bar connecting them and an asterisks (*).

Reproductive Activity

I observed six gravid females during my surveys: five in control forests and one in treated clear-cut forests. I found 10 females guarding eggs in control forests, but none in clear-cut forests. The proportion of juveniles within a survey was not related to years since harvest nor differed significantly among forest treatments (Table 5).

Table 5. (A) Outcomes of a generalized linear model that examined the effect of forest treatment and year since clear-cut on the proportion of *P. cinereus* that were juveniles during surveys. I present estimates (β) and their corresponding standard error (*SE)*, *z*-values, and *p*-values (*p*). **(B)** Pairwise comparisons among the effect of forest treatment (control, untreated clear-cut, and treated clear-cut) on the proportion of juveniles found during surveys. I present estimates (β) and their corresponding standard error (*SE)*, *t*-ratios, and *p*-values (p_{corr}) corrected for multiple comparisons. Bolded values indicate significance.

(A) Model Output

Fixed effects	β	*SE*	*z*-value	*p*
Intercept (Control)	**-0.89**	**0.14**	**-6.51**	**0.02**
Forest treatment (Untreated)	0.33	1.71	0.19	0.85
Forest treatment (Treated)	1.49	2.48	0.60	0.55
Years since harvest	-0.10	0.12	-0.79	0.43

(B) Post-hoc Multiple Comparisons

Forest treatment Contrasts	β	*SE*	*t*-ratio	p_{corr}
Control vs. Untreated	-0.33	1.71	-0.19	0.99
Control vs. Treated	-1.49	2.48	-0.60	0.82
Untreated vs. Treated	-1.16	1.26	-0.92	0.63

Discussion

I investigated the effects of clear-cutting and herbicide treatment on salamanders and their habitat in the Acadian Forest. *Plethodon cinereus* abundance was highest and soil temperature and moisture were most favourable in the unharvested control forests that served as reference conditions of Acadian forests. With no previously known harvest history in the control forests, salamander abundance is higher in forests that have never been harvested than in forests with a harvest history, regardless of the type of forest harvesting method. Between 5-26 years after clear-cutting, *P. cinereus* abundance was 3.2 times higher in forests that received herbicide treatment than in untreated clear-cut forests, indicating a habitat better suited for salamanders in treated clear-cuts than untreated. Environmental conditions I measured in this study (soil temperature, soil moisture, and soil pH) more closely resembled preferred *P. cinereus* habitat in the treated clear-cut forests than in the untreated clear-cut forests during this early forest regeneration stage. In addition, I found that *P. cinereus* body condition in clear-cut forests was greater than in control (unharvested) forests and that reproductive activity did not differ among forest treatments. These results conflict with my initial hypothesis that the removal of the herbaceous layer by herbicides would enhance the negative effects of canopy cover removal from clear-cutting on salamander habitat and reduce salamander abundance, body condition, and reproductive success.

Environmental Factors

Herbicide treatment in a clear-cut forests is intended to increase reforestation of target tree species, recovering the forest canopy cover faster than clear-cutting without a herbicide treatment (Wagner et al. 2004). The current clear-cut cycle length for Crown land in New Brunswick is 35 years, with an average of 32 years (Government of New Brunswick 2014; Natural Resources Canada 2018). The results from my study indicate that the effect of herbicide treatment significantly increases the rate of canopy closure within 26 years after clear-cutting. However, canopy cover is not expected to fully return in treated clear-cut forests to canopy cover levels experienced by the control plots before the next timber harvesting event. This clear-cutting cycle leaves harvested forest ecosystems in a constant state of recovery, as they are never allowed to return to a form of their original state. Canopy cover removal impacted environmental features that influence the quality of salamander habitat.

Clear-cut forests generally have higher soil temperature and lower soil moisture than mature forests (Chen et al. 1993; Carlson and Groot 1997; Hashimoto and Suzuki 2004; Kulmala et al. 2014). However, the influence of the herbaceous layer on the forest environment is understudied. My results indicate a contrasting effect of canopy cover removal on soil temperature and moisture depending on whether a clear-cut forest is treated with herbicide post-harvest or not. Herbicides are used to suppress the growth and regeneration of the herbaceous layer and likely alter environmental factors that are important to salamanders, such as soil temperature and moisture. I expected that herbaceous layer suppression in treated clear-cut forests would increase soil temperature and lower soil moisture compared to treated clear-cut forests and that would, in turn, negatively impact salamander habitat. Instead, I found evidence that soil temperature in the treated clear-cut forests cooled faster as the coniferous forest recovered post-harvest than in the untreated clear-cut forest treatment. In the treated clear-cut forests, soil temperature was similar to the control forest approximately 20 years post-harvest. In contrast, soil temperature remained higher in the untreated clear-cut forests than the control forests throughout the post-harvest period studied here (26 years) (Figure 2b). Other environmental factors impacted by clear-cut and herbicide treatment in addition to soil temperature include soil moisture and soil pH.

In the untreated clear-cut forests, soil moisture in the forests that were clear-cut within the past 12 years was higher than the average soil moisture in the control forest, then became dryer as years since harvest increased. In the latter half of the study's seral gradient (~16-25 years), untreated clear-cut forests were becoming dryer than the treated clear-cut forests. In contrast, soil moisture in the treated clear-cut forest treatment remained lower than the control forests as the forests aged but gradually increased with years since harvest. Around 20 years after they are clear-cut, soil moisture in the untreated clear-cut forests and treated clear-cut forests had similar soil moisture levels (Figure 2c). These results suggest that in the early stages of forest regeneration (up to 26 years), *P. cinereus* habitat in treated clear-cut forest treatment gradually becomes more suitable for amphibians, whereas untreated clear-cut forests either becomes less suitable as the forest ages or has a slower recovery time. Previous studies have found salamander abundance remaining relatively high up to four years after clear-cutting if a high density of downed wood is left that conserves a cool and moist microhabitat on the forest floor (Grover

1998; Hocking et al. 2013; Garcia et al. 2020; Ochs et al. 2022). My findings of soil temperatures > 15°C and low *P. cinereus* abundance supports this literature that any downed woody debris left from clear-cutting no longer provides thermal refugia to salamanders in clear-cut forests within the first half of my study's seral gradient (~5-10 years). There was no significant effect of forest treatment on soil pH. However, previous research found that clear-cutting lowers soil pH (Johnson et al 1991). This study included clear-cut forests that were along a seral gradient, and a difference in soil pH may more likely be detected by comparing clear-cut forests from an early seral period (~0-10 years) with the control forests. Previous research has observed *P. cinereus* in a wide range of acidic soils from 3.0-5.5 and aligns with the presence of salamanders found in soil pH conditions in the present study (Federici and Raffaelli 2018; Moore and Wyman 2010; Wyman 1988).

I found a difference in the forest environments between the untreated clear-cut and treated clear-cut forest treatments, but only for the first 26 years after clear-cutting. I observed the effects of reduced competition in regenerating forests where herbicide was applied as the canopy cover was higher in treated clear-cut forests than the untreated clear-cut forests at similar years since harvest. The closed canopy of treated clear-cut forests likely contributed to soil temperatures lowering and soil moisture increasing as the forests aged in the treated clear-cut forests but increased and decreased in the untreated clear-cut forests. However as the treated clear-cut forests mature, they will likely experience drier soils than the mature untreated clear-cut forests as dominantly coniferous forests have reduced leaf litter, therefore a greater potential for evaporation (Li et al. 2014; Hou et al. 2019). Also, the treated clear-cut forests will likely have less large woody debris later in the forest regeneration cycle because young deciduous trees are removed early, before they become mature and die. Therefore, herbicide use may increase suitability earlier in regrowth at the cost of unsuitable habitat later in succession.

Community Structure

As the forest environment was impacted by harvest and herbicide treatment, it is unsurprising that salamander abundance and richness was higher in control forests than the two forest treatments that were harvested. These results support the findings of previous research that has also shown reduced terrestrial salamander abundance after clear-cutting (Tilghman et al.

2012). Interestingly, salamander abundance was 3.2 times higher in treated clear-cut forests than untreated clear-cut forests. This result refutes my initial hypothesis that canopy cover removal would have greater negative effects on salamander populations in clear-cut forests that received a herbicide treatment. These results may indicate that suppressing the herbaceous layer with herbicide after clear-cutting benefits *P. cinereus* within the first 26 years after clear-cutting. There is evidence from our control (not harvested) forests that a closed canopy provides protection for these salamanders against harsh sunlight. Thus, a faster closing canopy in the treated clear-cut forests may contribute to higher *P. cinereus* abundance compared to untreated clear-cut forests. These results provide information about how salamanders may respond to different forest harvesting practices in the first 26 years post-harvest. My research also provides evidence that Eastern Red-backed Salamanders strongly rely on protection from direct sunlight from canopy cover and/or increased cover objects, which has also been found in previous studies occurring in Virginia and Oregon, USA (Grover 1998; Garcia et al. 2020).

I used an active surveying technique to collect data on salamander populations among forest treatments. Literature supports the use of area-constrained surface surveys to document salamander abundances (Smith and Petranka 2000). That said, it was more difficult to find salamanders in the control forests compared with the clear-cut forests due to the greater structural complexity and the amount and size of cover objects (Welsh and Hodgson 2013). Therefore, the abundance of salamanders in the control forests likely under-represents true numbers when compared with the clear-cut forests, so differences between the control and harvested forest treatments should be considered conservative.

Indication of Threats

Similar to abundance, body condition of salamanders has also been shown to differ with varying forest management (Karraker and Welsh 2006; Homyack and Haas 2009; Ford 2011). Body condition was about 1.1 times lower in the control forests than in both the untreated clear-cut and treated clear-cut forests. These results oppose my hypothesis that salamander body condition would be greater in the control forest treatment due to the optimal habitat conditions. Greater body condition would be indicative of an individual having greater fat reserves, which could reflect greater foraging success or a lower rate of energy expenditure on avoiding threats like competitive interactions and predation (Werner and Anholt 1993; Grover 1998; MacCracken and Stebbings 2012). With a greater salamander density and richness in the control forests, intraspecies and interspecies competition for resources increases as more individuals and species are competing for the same resources and territory (Liebgold and Dibble 2011; Yackulic et al. 2014). Clear-cutting also alters *P. cinereus* predator (some birds, small mammals, and snakes) and prey (invertebrates) habitat, which can either be a benefit or detriment to these species populations depending on their habitat preferences (Wyman 1998; Walton 2013). For example, some bird species tend to avoid open areas like clear-cuts, and some embrace the human-altered habitat (Ram et al. 2023). A shift to an ecosystem with lower predator or higher prey density likely decreases energy expenditure avoiding predation or starvation and increases body condition. Body condition indices have been used to predict reproductive success in various species, but I did not find trends in my study that support a relationship between these variables (Milenkaya et al. 2015; Wise and Jaeger 2021)

Reproductive Activity

The reproductive activity of salamanders did not differ among forest treatments. However, there was limited data to evaluate reproductive condition. I only found gravid females and their egg masses in two forest treatments (control and treated clear-cut forests) and observations were limited ($n = 10$) within clear-cut forests. This restricts interpretation of this aspect of my research. In fact, previous research has found a different trend. Homyack and Haas (2009) found a higher proportion of juvenile *P. cinereus* in control forests than in clear-cut forests. Greenberg et al. (2016) found reproductive success varied among amphibian and reptile species with varying regeneration treatments. It is suggested that increased soil temperatures in clear-cut forests, like those found in this study, may require salamanders to expend more energy on metabolism and less on reproduction (Homyack et al. 2011). In future work, comparing both egg masses and abundance of juveniles may provide a more holistic picture of reproductive activity among forest treatments.

The finding of greater salamander habitat conditions (soil temperature and moisture) and densities in clear-cut forests that received a herbicide treatment indicates that a quicker canopy closure in treated clear-cuts is more beneficial to salamander habitat in regenerating forests than preserving the herbaceous layer. A study in the Southern Appalachian forest suggests clear-cut forests require 50-70 years to return to similar pre-disturbance salamander densities (Petranka et al. 1993). However, these harvested ecosystems are not expected to ever resemble those existing in this study's control (unharvested) forests, as they will likely be harvested again before these ecosystems fully recover.

Despite finding differences in salamander populations and their habitat among forest treatments, additional environmental and ecological variables should be considered in future research. Differences in woody debris between harvested forests in later regeneration stages also have the potential to impact salamander populations and their habitat. Decaying, coarse woody debris adds habitat complexity and protects salamanders from predation and desiccation by providing cool and moist environments. Similar to my study, results by Rothermel and Luhring 2005 found that Mole Salamanders (*Ambystoma talpoideum*) avoided desiccation in forests with more canopy cover and burrow availability. However, the forest homogeneity created using herbicides during forest regeneration likely limits the quality and quantity of downed woody debris. Another factor to consider in the future is soil pH, which is an environmental factor salamanders are sensitive to. Thus, it would also be beneficial to investigate whether glyphosate-based herbicide treatment to speed coniferous growth changes soil pH and, in turn, impacts salamanders (Wyman 1988; Wyman and Jancola 1992). This could be investigated at the landscape scale, because forestry preferences for coniferous forests resulted in conversion of deciduous and mixed wood stands throughout the Acadian Forest to conifer in the past 70 years (Noseworthy and Beckley 2020). Overall, continuing this research to include harvested forests in later regeneration stages would provide information about how different forestry practices may impact salamander populations and their habitat long-term. Lastly, data interpretation from the indication of threats and reproductive activity sections of this study was limited by samples size in clear-cut forests and a lack of data on egg masses. These adjustments would capture more information about salamander populations among forest treatments and provide insight into how clear-cutting and herbicide treatment impact forest ecosystems.

Conclusion

Using Eastern Red-backed Salamanders as bioindicators of ecosystem health, I explored how common clear-cut forestry practices impact a harvested ecosystem. I documented a relatively higher abundance of *P. cinereus* in clear-cut forests treated with a herbicide than untreated forests. The faster canopy cover closure in the treated clear-cut forests likely improved salamander habitat conditions, like cooler soil temperatures, and higher soil moisture earlier in the forest regeneration cycle than in the untreated clear-cut forests. My research encompassed the first 26 years after clear-cutting, and thus, my work did not investigate changes to salamander populations and their habitat in later stages of forest regeneration. Similarly, the long-term (greater than 26 years post-harvest) effects of treating clear-cut forests with herbicide on *P. cinereus* and their habitat is unknown. As forest regeneration continues past the scope of my study, one would expect that the canopy would continue to close in the untreated clear-cut forest treatment, and soil temperature and moisture are likely to correspondingly decrease and increase, respectively. This would likely benefit salamander populations at that time, yet future research that focuses on later stages of forest regeneration is needed to test this idea. It is important to note that in an unharvested (control) forest the environment was much more suitable for salamanders, and the abundance of *P. cinereus* was 4 and 18 times more than treated clear-cut and untreated clear-cut forests, respectively. Thus, older Acadian forests in New Brunswick likely provide a refuge for salamanders and should be prioritized for protection. Research that considers holistic approaches to sustainable forest management in a warming world will help protect important ecological species, like the Eastern Red-backed Salamander. Understanding the impacts of clear-cutting and the use of herbicides in forested ecosystems on wildlife remains an important conservation issue worldwide as global biodiversity loss continues.

Appendix

I have included my predictions plot that represents my method for choosing the scaled mass index (SMI) ordinary least squares (OLS) estimation (Peig & Green 2010) (Figure A1). I also included plots presenting insignificant interactions between years since harvest and forest treatment for canopy cover and soil pH (Figure A2). Lastly, I included plots visualizing differences in soil temperature and soil moisture among forest treatments where I found significant interactions and interpretation of main effects was not appropriate but can still be visualised to help understanding of these results (Figure A3).

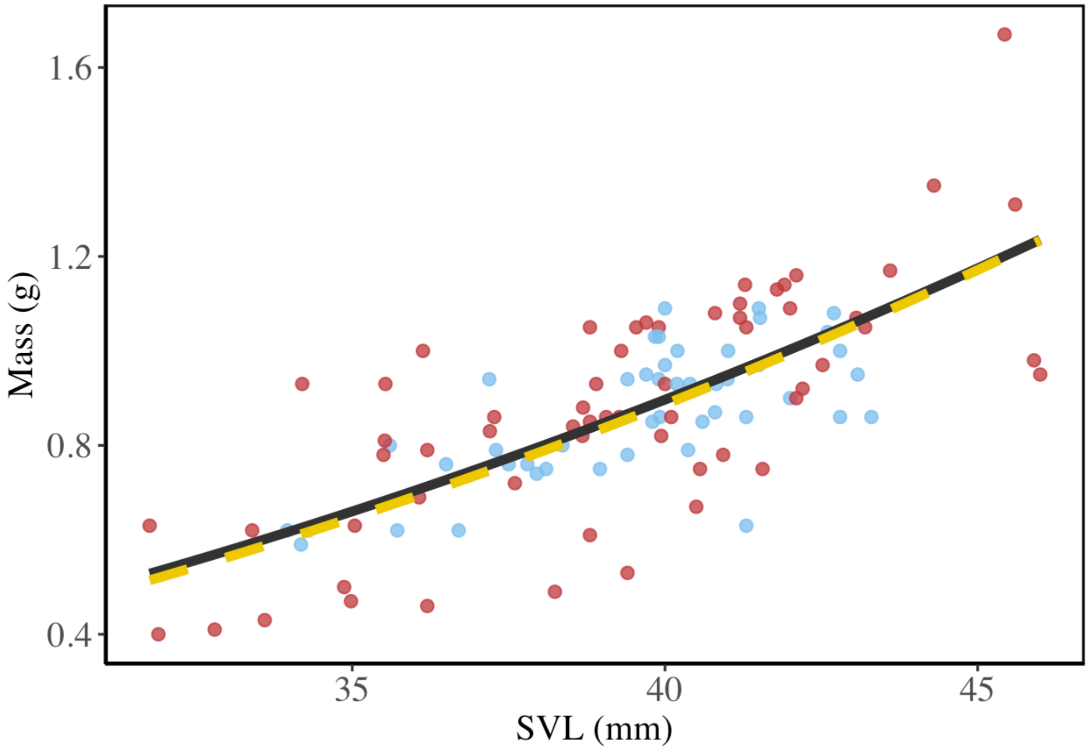

Figure A1. Predicted SMI curves using ordinary least squares (OLS, yellow dashed line) regression of body mass against snout-vent length (SVL), and the robust scaled mass index (black solid line) (Peig and Green 2010). Points represent raw data of female (red) and male (blue) *P. cinereus.*

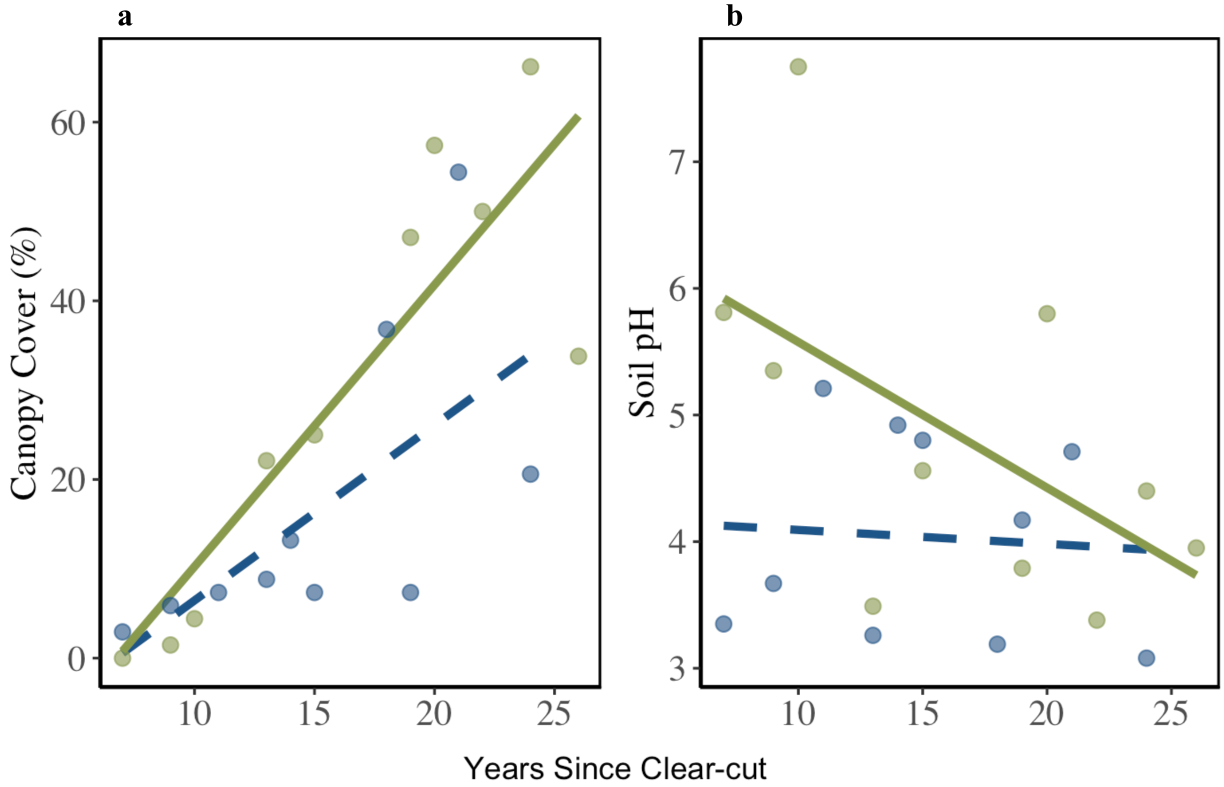

Figure A2. Plots of **(a)** canopy cover (%) and **(b)** soil pH of clear-cut (dashed blue) and treated clear-cut (solid green) forest treatments over time (yrs) since clear-cut. Coloured points are raw data.

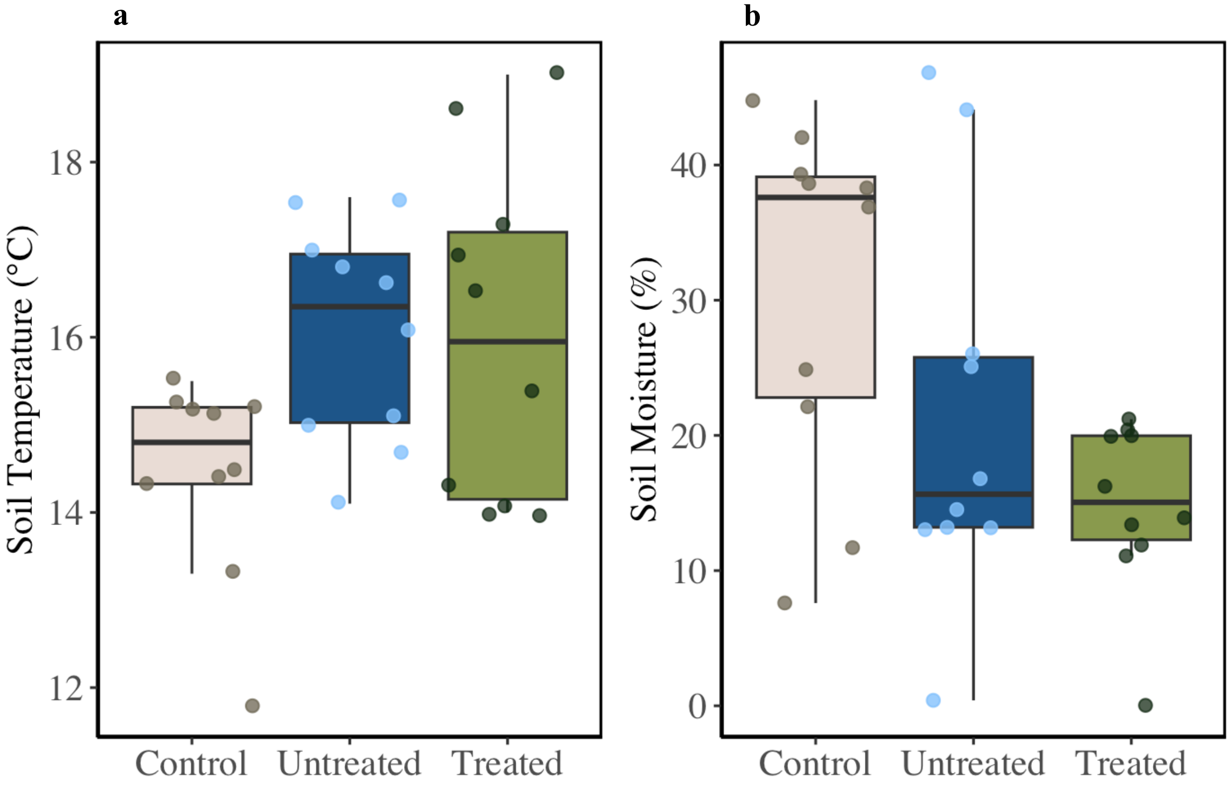

Figure A3. Environmental characteristics **(a)** soil temperature (°C) and **(b)** soil moisture (%) for the control, untreated clear-cut, and treated clear-cut forest treatments. These are represented in beige, blue, and green fill, respectively. Raw data are represented by coloured points. The box itself represents the interquartile range of the data (IQR) and the lower and upper limits of the box are the first and third quantile, respectively. The bottom/ top whiskers that extend from the box are represent the minimum and maximum values of the data that are not considered outliers (1.5 times the IQR).

Literature Cited

Altham, P.M.E. 1978. Two generalizations of the binomial distribution. Journal of the Royal Statistical Society Series C: Applied Statistics **27**(2): 162–167.

Blakney, S. 2003. Aboriginal forestry in New Brunswick: conflicting paradigms. Environments **31**(1): 61–78.

Canadian Herpetofauna Health Working Group. 2017. Decontamination protocol for field work with amphibians and reptiles in Canada.

Carlson, D.W., and Groot, A. 1997. Microclimate of clear-cut, forest interior, and small openings in trembling aspen forest. Agricultural and Forest Meteorology **87**(4): 313–329.

Chen, J., Franklin, J.F., and Spies, T.A. 1993. Contrasting microclimates among clearcut, edge, and interior of old-growth Douglas-fir forest. Agricultural and Forest Meteorology **63**(3): 219–237.

Collins, J.P., and Storfer, A. 2003. Global amphibian declines: sorting the hypotheses. Diversity and Distributions **9**(2): 89–98.

Consul, P.C., and Jain, G.C. 1973. A generalization of the poisson distribution. Technometrics **15**(4): 791–799.

Davic, R.D., and Welsh, H.H. 2004. On the ecological roles of salamanders. Annual Review of Ecology, Evolution, and Systematics **35**: 405–434.

Declining Amphibian Population Task Force (DAPTF). 1998. Fieldwork code of practice.

deMaynadier, P.G., and Hunter Jr., M.L. 1995. The relationship between forest management and amphibian ecology: a review of the North American literature. Environ. Rev. **3**(3–4): 230–261.

Dempster, W.R. 2022. Effects of planting, vegetation management, and pre-commercial thinning on the growth and yield of Lodgepole Pine regenerated after harvesting in Alberta, Canada. Forests **13**(6): 929.

Elliott, K.J., Vose, J.M., Knoepp, J.D., Clinton, B.D., and Kloeppel, B.D. 2015. Functional role of the herbaceous layer in eastern deciduous forest ecosystems. Ecosystems **18**(2): 221–236.

Esseen, P.-A. 1997. Boreal forests. *In* Ecol. Bull. pp. 16–46.

Federici, P., & Raffaelli, J. (2018). Habitat preference and activity patterns of the eastern red-backed salamander (*Plethodon cinereus*) in the southern Appalachian Mountains. Southeastern Naturalist, 17(3), 388-398.

Fleishman, E., and Murphy, D.D. 2009. A realistic assessment of the indicator potential of butterflies and other charismatic taxonomic groups. Conservation Biology **23**(5): 1109–1116.

Ford, J.K.B. 2011. Acoustic behaviour of resident killer whales (*Orcinus orca*) off Vancouver Island, British Columbia. Canadian Journal of Zoology. NRC Research Press Ottawa, Canada.

Franklin, J.F., Lindenmayer, D., MacMahon, J.A., McKee, A., Magnuson, J., Perry, D.A., Waide, R., and Foster, D. 2000. Threads of continuity. Conservation in Practice **1**(1): 8–17.

Garcia, T.S., Johnson, J., Jones, J., and Kroll, A.J. 2020. Experimental evidence indicates variable responses to forest disturbance and thermal refugia by two plethodontid salamanders. Forest Ecology and Management **464**: 118045.

Gelman, A., and Hill, J. 2006, December 17. Data analysis using regression and multilevel/hierarchical models. Cambridge University Press.

Gillette, J., and Peterson, M.G. 2001. The benefits of transparency: Candling as a simple method for determining sex in red-backed salamanders (*Plethodon cinereus*). Herpetological Review **32**: 233–235.

Government of New Brunswick. 2009. A balanced management approach for New Brunswick's Crown Forest pg 1-6.

Government of New Brunswick. 2014. Crown timber licence.

Greenberg, C.H., Moorman, C.E., Raybuck, A.L., Sundol, C., Keyser, T.L., Bush, J., Simon, D.M., and Warburton, G.S. 2016. Reptile and amphibian response to oak regeneration treatments in productive southern Appalachian hardwood forest. Forest Ecology and Management **377**: 139–149.

Grover, M.C. 1998. Influence of cover and moisture on abundances of the terrestrial salamanders *Plethodon cinereus* and *Plethodon glutinosus*. Journal of Herpetology **32**(4): 489–497.

Grover, M.C. 2000. Determinants of salamander distributions along moisture gradients. Copeia **2000**(1): 156–168.

Hashimoto, S., and Suzuki, M. 2004. The impact of forest clear-cutting on soil temperature: a comparison between before and after cutting, and between clear-cut and control sites. Journal of Forestry Research **9**(2): 125–132.

Herbeck, L.A., and Larsen, D.R. 1999. Plethodontid salamander response to silvicultural rractices in Missouri Ozark forests. Conservation Biology **13**(3): 623–632.

Hickerson, C.-A.M., Anthony, C.D., and Walton, B.M. 2017. Eastern Red-backed Salamanders regulate top-down effects in a temperate forest-floor community. Herpetologica **73**(3): 180–189.

Hocking, D.J., Connette, G.M., Conner, C.A., Scheffers, B.R., Pittman, S.E., Peterman, W.E., and Semlitsch, R.D. 2013. Effects of experimental forest management on a terrestrial, woodland salamander in Missouri. Forest Ecology and Management **287**: 32–39.

Homyack, J.A., and Haas, C.A. 2009. Long-term effects of experimental forest harvesting on abundance and reproductive demography of terrestrial salamanders. Biological Conservation **142**(1): 110–121.

Homyack, J.A., Haas, C.A., and Hopkins, W.A. 2011. Energetics of surface-active terrestrial salamanders in experimentally harvested forest. The Journal of Wildlife Management **75**(6): 1267–1278.

Hou, D., He, W., Liu, C., Qiao, X., and Guo, K. 2019. Litter accumulation alters the abiotic environment and drives community successional changes in two fenced grasslands in Inner Mongolia. Ecology and Evolution **9**(16): 9214–9224.

Houze, C.M., and Chandler, C.R. 2002. Evaluation of coverboards for sampling terrestrial salamanders in South Georgia. Journal of Herpetology **36**(1): 75–81.

iNaturalist. 2023. Acadia Research Forest. Retrieved April 11, 2023.

Johnson, C. E., Johnson, A. H., & Siccama, T. G. 1991. Whole-tree clear-cutting effects on exchangeable cations and soil acidity. Soil Science Society of America Journal **55**(2), 502-508.

Karraker, N.E., and Welsh, H.H. 2006. Long-term impacts of even-aged timber management on abundance and body condition of terrestrial amphibians in Northwestern California. Biological Conservation **131**(1): 132–140.

Keenan, R.J., and Kimmins, J.P. (Hamish). 1993. The ecological effects of clear-cutting. Environmental Reviews **1**(2): 121–144.

Keitzer, S.C., and Goforth, R.R. 2013. Spatial and seasonal variation in the ecological significance of nutrient recycling by larval salamanders in Appalachian headwater streams. Freshwater Science **32**(4): 1136–1147.

Kluber, M.R., Olson, D.H., and Puettmann, K.J. 2009. Downed wood microclimates and their potential impact on Plethodontid salamander habitat in the Oregon Coast Range. Northwest Science **83**(1): 25–34.

Kulmala, L., Aaltonen, H., Berninger, F., Kieloaho, A.-J., Levula, J., Bäck, J., Hari, P., Kolari, P., Korhonen, J.F.J., Kulmala, M., Nikinmaa, E., Pihlatie, M., Vesala, T., and Pumpanen, J. 2014. Changes in biogeochemistry and carbon fluxes in a boreal forest after the clear-cutting and partial burning of slash. Agricultural and Forest Meteorology **188**: 33–44.

Kuznetsova, A., Brockhoff, P.B., and Christensen, R.H.B. 2017. lmerTest Package: Tests in linear mixed effects models. Journal of Statistical Software **82**: 1–26.

Lenth, R. V. 2022. emmeans: estimated marginal means, aka least-squares means. R package version 1.8.5.

Li, X., Niu, J., and Xie, B. 2014. The effect of leaf litter cover on surface runoff and soil erosion in Northern China. PLOS ONE **9**(9): e107789.

Liebgold, E.B., and Dibble, C.J. 2011. Better the devil you know: familiarity affects foraging activity of red-backed salamanders, *Plethodon cinereus*. Animal Behaviour **82**(5): 1059–1066.

Long, J.N. 2009. Emulating natural disturbance regimes as a basis for forest management: A North American view. Forest Ecology and Management **257**(9): 1868–1873.

MacCracken, J.G., and Stebbings, J.L. 2012. Test of a body condition index with amphibians. Journal of Herpetology **46**(3): 346–350.

Mazerolle, M.J., Lapointe St-Pierre, M., Imbeau, L., and Joanisse, G. 2021. Woodland salamander population structure and body condition under irregular shelterwood systems. Canadian Journal of Forest Research **51**(9): 1281–1291.

McCullagh, P., and Nelder, J.A. 1989. Generalized Linear Models. Springer US, Boston, MA.

Mendes, K.F. 2021. Glyphosate: Agricultural uses, ecological impacts and potential carcinogenic effects. Nova Science Publishers, Incorporated.

Milenkaya, O., Catlin, D.H., Legge, S., and Walters, J.R. 2015. Body condition indices predict reproductive success but not survival in a sedentary, tropical bird. PLOS ONE **10**(8): e0136582.

Murcia, C. 1995. Edge effects in fragmented forests: implications for conservation. Trends in Ecology & Evolution **10**(2): 58–62.

Natural Resources Canada. 2018. National Forest Inventory: 2017 Technical Report. Ottawa: Natural Resources Canada, Canadian Forest Service.

Newbold, T., Hudson, L.N., Hill, S.L.L., Contu, S., Lysenko, I., Senior, R.A., Börger, L., Bennett, D.J., Choimes, A., Collen, B., Day, J., De Palma, A., Díaz, S., Echeverria-Londoño, S., Edgar, M.J., Feldman, A., Garon, M., Harrison, M.L.K., Alhusseini, T., Ingram, D.J., Itescu, Y., Kattge, J., Kemp, V., Kirkpatrick, L., Kleyer, M., Correia, D.L.P., Martin, C.D., Meiri, S., Novosolov, M., Pan, Y., Phillips, H.R.P., Purves, D.W., Robinson, A., Simpson, J., Tuck, S.L., Weiher, E., White, H.J., Ewers, R.M., Mace, G.M., Scharlemann, J.P.W., and Purvis, A. 2015. Global effects of land use on local terrestrial biodiversity. Nature **520**(7545): 45-50S.

Noseworthy, J., and Beckley, T.M. 2020. Borealization of the New England – Acadian Forest: a review of the evidence. Environ. Rev. **28**(3): 284–293.

Nyland, R.D. 2016. Silviculture: Concepts and applications, third edition. Waveland Press.

Ochs, A.E., Saunders, M.R., and Swihart, R.K. 2022. Response of terrestrial salamanders to the decade following timber harvest in hardwood forests. Forest Ecology and Management **511**: 120159.

Özkan, U., and Gökbulak, F. 2017. Effect of vegetation change from forest to herbaceous vegetation cover on soil moisture and temperature regimes and soil water chemistry. CATENA **149**: 158–166.

Peig, J., and Green, A.J. 2009. New perspectives for estimating body condition from mass/length data: the scaled mass index as an alternative method. Oikos **118**(12): 1883–1891.

Peig, J., and Green, A.J. 2010. The paradigm of body condition: a critical reappraisal of current methods based on mass and length. Functional Ecology **24**(6): 1323–1332.

Persson, T., & Wirén, A. 1995. Changes in soil chemistry in a Scots pine stand after clear-felling and slash burning. Scandinavian Journal of Forest Research, 10(1-4), 172-179.

Petranka, J. W. 1998. Salamanders of the United Sates and Canada. Smithsonian Institution Press, Washington DC, USA.

Petranka, J.W., Eldridge, M.E., and Haley, K.E. 1993. Effects of timber harvesting on southern Appalachian salamanders. Conservation Biology **7**(2): 363–370.

Ram, D., Lindström, Å., Pettersson, L.B., and Caplat, P. 2023. Farmland birds on forest clear-cuts: Liked by some, avoided by others. Forest Ecology and Management **529**: 120714.

R Core Team 2022. R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Riedel, B.L., Russell, K.R., Ford, W.M., O'Neill, K.P., and Godwin, H.W. 2008. Habitat relationships of eastern red-backed salamanders (*Plethodon cinereus*) in Appalachian agroforestry and grazing systems. Agriculture, Ecosystems & Environment **124**(3): 229–236.

Romano, A., Costa, A., Salvidio, S., Menegon, M., Garollo, E., Tabarelli de Fatis, K., Miserocchi, D., Matteucci, G., and Pedrini, P. 2018. Forest management and conservation of an elusive amphibian in the Alps: Habitat selection by the Golden Alpine Salamander reveals the importance of fine woody debris. Forest Ecology and Management **424**: 338–344.

Rose, C.L., Marcot, B.G., Mellen, T.K., Ohmann, J.L., Waddell, K.L., Lindley, D.L., and Schreiber, B. 2001 . Decaying wood in Pacific Northwest forests: Concepts and tools for habitat management.

Rothermel, B.B., and Luhring, T.M. 2005. Burrow availability and desiccation risk of Mole Salamanders (*Ambystoma talpoideum*) in harvested versus unharvested forest stands. Journal of Herpetology **39**(4): 619–626.

SGS Belgium S.A. 2018. Forest sustainability in the province of New Brunswick,. Canada. Project Number 180373 for client Engie-Electrabel.

Smith, C.K., and Petranka, J.W. 2000. Monitoring terrestrial salamanders: repeatability and validity of area-constrained cover object searches. Journal of Herpetology **34**(4): 547–557.

Spies, T.A., Franklin, J.F., and Thomas, T.B. 1988. Coarse woody debris in Douglas-Fir forests of Western Oregon and Washington. Ecology **69**(6): 1689–1702.

Spotila, J.R. 1972. Role of temperature and water in the ecology of lungless salamanders. Ecological Monographs **42**(1): 95–125.

Swift, D.E., Kilpatrick, B., and Pegler, S. 2005. Acadia research forest: a preliminary forest research facility for Natural Resources Canada, Canadian Forest Service - Atlantic.

Tilghman, J.M., Ramee, S.W., and Marsh, D.M. 2012. Meta-analysis of the effects of canopy removal on terrestrial salamander populations in North America. Biological Conservation **152**: 1–9.

Tinker, D.B., and Knight, D.H. 2000. Coarse woody debris following fire and logging in Wyoming Lodgepole Pine forests. Ecosystems **3**(5): 472–483.

Tschaplinski, T. J., Todd Jr, D. E., & Amthor, J. S. 1993. Soil chemistry and nutrient availability following clearcutting of a southern Appalachian forest. Soil Science Society of America Journal, 57(5), 1275-1283.

Wagner, R.G., Newton, M., Cole, E.C., Miller, J.H., and Shiver, B.D. 2004. The role of herbicides for enhancing forest productivity and conserving land for biodiversity in North America. Wildlife Society Bulletin **32**(4): 1028–1041.

Walton, B.M. 2013. Top-down regulation of litter invertebrates by a terrestrial salamander. Herpetologica **69**(2): 127–146.

Welsh, H.H., and Hodgson, G.R. 2013. Woodland salamanders as metrics of forest ecosystem recovery: a case study from California's redwoods. Ecosphere **4**(5).

Welsh JR., H.H., and Droege, S. 2001. A case for using Plethodontid salamanders for monitoring biodiversity and ecosystem integrity of North American forests. Conservation Biology **15**(3): 558–569.

Werner, E.E., and Anholt, B.R. 1993. Ecological consequences of the trade-off between growth and mortality rates mediated by foraging activity. Am Nat **142**(2): 242–272.

Wickham, H. 2016. ggplot2: elegant graphics for data analysis. Springer-Verlag New York.

Williams, C.E., Lipscomb, M.V., Johnson, W.C., and Nilsen, E.T. 1990. Influence of leaf litter and soil moisture regime on early establishment of *Pinus pungens*. The American Midland Naturalist **124**(1): 142–152.

Wise, S.E., and Jaeger, R.G. 2021. Maternal body size and condition predict measures of reproductive success and future reproductive allocation in territorial Eastern Red-Backed Salamanders. Ichthyology & Herpetology **109**(1):55-63.

Wyman, R.L. 1988. Soil acidity and moisture and the distribution of amphibians in five forests of Southcentral New York. Copeia **1988**(2): 394–399.

Wyman, R.L. 1998. Experimental assessment of salamanders as predators of detrital food webs: effects on invertebrates, decomposition and the carbon cycle. Biodiversity and Conservation **7**(5): 641–650.

Wyman, R.L., and Jancola, J. 1992. Degree and scale of terrestrial acidification and amphibian community structure. Journal of Herpetology **26**(4): 392–401.

Yackulic, C.B., Reid, J., Nichols, J.D., Hines, J.E., Davis, R., and Forsman, E. 2014. The roles of competition and habitat in the dynamics of populations and species distributions. Ecology **95**(2): 265–279.

Zuur, A.F., Ieno, E.N., and Elphick, C.S. 2010. A protocol for data exploration to avoid common statistical problems. Methods in Ecology and Evolution **1**(1): 3–14.