Canadian Science Publishing | Canadian Journal of Forest Research

# Herbicide Application Improves Plethodontid Salamander Habitat Conditions in Regenerating Clear-cut Forests

| | |
|---|---|
| Journal: | *Canadian Journal of Forest Research* |
| Manuscript ID | cjfr-2024-0294.R1 |
| Manuscript Type: | Research Article |
| Date Submitted by the Author: | 15-May-2025 |
| Complete List of Authors: | Leslie, Sara; Mount Allison University, Biology<br>Edge, Christopher; Canadian Forest Service, Canadian Forest Service<br>Riley, Julia; Mount Allison University, Biology |
| Is the manuscript for consideration in a Special Issue or Collection?: | Not applicable (regular submission) |
| Keyword: | Acadian Forests, indicator, clear-cutting, forest ecology, forest management |
| | |

# Herbicide Application Improves Plethodontid Salamander Habitat Conditions in Regenerating Clear-cut Forests

**Sara E. Leslie,** Department of Biology, Mount Allison University, Sackville, NB, Canada; Current: Faculty of Forestry, Department of Natural Resource Management, University of British Columbia, Vancouver, BC, Canada; sara.leslie@student.ubc.ca

**Christopher B. Edge*,** Natural Resources Canada, Canadian Forest Service, Fredericton, NB, Canada; christopher.edge@nrcan-rncan.gc.ca

**Julia L. Riley;** Department of Biology, Mount Allison University, Sackville, NB, Canada; jriley@mta.ca

*Corresponding Author: Christopher B. Edge (christopher.edge@nrcan-rncan.gc.ca)

**Abstract**

Forestry activities, including harvesting and herbicide application, alter both overstory and understory vegetation communities, reshaping ecosystem structure and condition. These changes likely impact wildlife sensitive to environmental change, such as the Eastern Red-backed Salamander (*Plethodon cinereus*). We compared canopy cover, soil temperature, soil moisture, soil pH, and salamander abundance across unharvested reference stands and clear-cut harvest blocks treated and untreated with a glyphosate-based herbicide. Overall, reference stands exhibited the highest canopy cover and soil moisture, and lowest soil temperature. Herbicide-treated blocks showed decreasing soil temperature and increasing moisture with time since harvest, whereas untreated blocks exhibited the opposite trend. Salamander abundance in reference stands was 4 and 18 times higher than in herbicide-treated and untreated blocks, respectively, and 3 times higher in herbicide-treated than untreated blocks. Greater canopy cover and soil moisture in herbicide-treated blocks likely improve habitat suitability, promoting higher salamander abundance compared to untreated blocks during forest regeneration. Our study suggests that herbicide application in clear-cut forests may accelerate the recovery of microhabitat conditions to preharvest levels, partially mitigating the impacts of harvesting on forest specialists like salamanders. We emphasize the need for holistic approaches in forestry management to sustain biodiversity and ecosystem integrity in increasingly changing landscapes.

**Keywords:** Acadian Forests, indicator, clear-cutting, forest ecology, forest management, glyphosate-based herbicide, Plethodontid, lungless salamander, silviculture, reproduction

## Introduction

The degradation, conversion, and fragmentation of habitats are some of the main causes of biodiversity loss worldwide (Newbold et al. 2015). Reversing past impact, limiting future impact, and potentially providing a benefit to forest habitats are considered within contemporary forest management planning and silviculture (Jung et al. 2023). Forestry operations often involve large-scale spatial disturbances and reshape forest dynamics, including age, structure, composition, and both biotic and abiotic conditions (Kuuluvainen et al. 2021). The biotic and abiotic changes that follow tree harvesting have a strong influence on the quality and quantity of habitat for a variety of species, especially those that make use of interior forest habitat (Keenan and Kimmins 1993). The direct (e.g., habitat loss) and indirect (e.g., changes to forest structure) effects of forestry on biodiversity have led to calls for ecosystem-centered approaches to forest management. Such approaches aim to balance timber harvesting with environmental conservation, including ecological preservation (Venn 2023). In navigating towards this balance, ecosystem-centered strategies must account not only for the impacts of harvesting and subsequent regeneration but also for the modifications to forest structure induced by post-harvest silvicultural interventions. Assessing these impacts requires reliable ecological indicators, particularly species that are highly sensitive to environmental change and are closely linked to forest ecosystem health.

Given the resource-intensive nature of evaluating multiple species, researchers often rely on bioindicator species, which are living organisms used to assess and monitor the health and quality of ecosystems or environments. The Eastern Red-backed Salamander (*Plethodon cinereus*), a terrestrial lungless salamander common in the Acadian Forests of eastern North

America, is an ideal bioindicator due to its sensitivity to environmental changes, strong site fidelity, high densities, and role in ecosystem processes (Welsh and Droege 2001; Davic and Welsh 2004; Homyack and Haas 2009). These salamanders require cool and moist microhabitats typically found under downed wood, rocks, and leaf litter for respiration and hydration (Petranka 2010). These requirements lead to their presence, abundance, and body condition being linked to environmental gradients (e.g. soil temperature, moisture, and pH) and forest structure (Wyman 1988; Sugalski and Claussen 1997; McKenny et al. 2006; Homyack et al. 2010). As mid-level consumers, they play an important role in maintaining ecosystem function and stability by regulating nutrient cycling and balancing food webs (Davic and Welsh 2004; Hickerson et al. 2017). Given their sensitivity to changes in microhabitat conditions, Eastern Red-backed Salamander populations are particularly vulnerable to disturbances that alter forest structure and microclimates, such as forestry practices (deMaynadier and Hunter Jr. 1995).

A common forest harvest system in eastern Canada is the sequence of clear-cutting, replanting, and herbicide application. Clear-cutting alters forest structure by removing the majority of tree biomass, including standing dead trees, woody debris, and leaf litter, resulting in a more homogenous landscape than naturally disturbed forests (Franklin et al. 2000). This loss of structural complexity reduces available cover objects and increases exposure of moisture-dependent species like salamanders to drier conditions, increasing desiccation risk (deMaynadier and Hunter Jr. 1995; Tilghman et al. 2012; Ochs et al. 2022). Additionally, canopy removal increases sunlight penetration, warming the forest floor, reducing humidity, and depleting leaf litter—negatively impacting key habitat components for salamanders (Keenan and Kimmins 1993; Tilghman et al. 2012).

After clear-cut harvesting, many forest blocks are replanted with coniferous species and treated with herbicides, primarily glyphosate-based, to suppress competing vegetation. Herbicide application facilitates the early dominance of coniferous species by reducing the abundance of deciduous and shrub vegetation, accelerating canopy closure (Wagner et al. 2006; Matute 2019). The suppression on deciduous and shrub vegetation enhances the growth and survival of desired tree species (Wagner et al. 2006); it also alters understory composition by reducing shrub abundance and shifting plant communities towards perennials (De Lombaerde et al. 2021; Xiao et al. 2024). These changes likely extend beyond the understory vegetation, potentially affecting other environmental conditions that are critical to various flora and fauna habitats, such as soil temperature, moisture, and pH. Thus, the combined environmental alterations resulting from herbicide application and clear-cutting may lead to cascading, ecosystem-wide impacts on biodiversity and habitat quality (Keenan and Kimmins 1993; deMaynadier and Hunter Jr. 1995; Tilghman et al. 2012; Ochs et al. 2022).

However, herbicide application may partially offset the impacts of clear-cutting by reducing competition from understory vegetation, accelerating overstory establishment, and promoting earlier canopy closure and downed log accumulation during forest regeneration (Dempster 2022). Since closed canopies and downed logs are key components in salamander habitat, providing cool and moist refugia that are especially valuable in harvested forests with limited overstory (Kluber et al. 2009). Previous research has documented salamander populations declining after clear-cutting, followed by gradual recovery as the forest regenerates (Tilghman et al 2012; Ochs et al. 2022). However, there is limited research on the cumulative impact of clear-cutting and herbicide use. The potentially diverging effects of clear-cutting 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 clear-cutting and herbicide use on salamander populations by assessing both environmental factors (canopy cover, soil moisture, temperature, and pH) and salamander-specific metrics across two scales: population-level effects (abundance and reproductive activity) and individual-level effects (body condition). We hypothesized that the removal of the herbaceous layer by glyphosate-based herbicides would exacerbate the effects of canopy cover removal from clear-cutting on salamander habitat post-harvest. We test the predictions that treating a clear-cut forest with glyphosate-based herbicides will (1) alter environmental conditions through reduced canopy cover, reduced soil moisture, increased temperature, and lower pH, (2) lower Eastern Red-backed Salamander abundance, (3) lower body condition and increase presence of injuries (including autotomized tails potentially due to increased predation), and (4) decrease the presence of gravid females and egg masses, and lower the ratio of juveniles to adults within the population.

## Methods

### ***Study Area***

The study was conducted in and around the Acadia Research Forest (ARF), located approximately 25 km east of Fredericton, New Brunswick (46°00'00"N, 66°20'24"W). The ARF encompasses 9000 ha of coniferous, deciduous, and mixed forests within the Grand Lake and Eastern Lowlands Ecoregions (Swift et al. 2005). Although previous research suggests Eastern Red-backed Salamanders exhibit a preference for deciduous and mixed wood forests over coniferous habitats (Degraaf and Rudis 1990), our preliminary study confirmed their presence across all three forest types in New Brunswick (see Supplementary Materials; Table S1). Therefore, we selected a total of 20 blocks that were at least 3 ha in size and harvested by clear-cutting between 5 and 26 years ago. All harvest blocks were planted with either a single coniferous species or a mix of coniferous species (Balsam Fir (*Abies balsamea*), White Spruce (*Picea glauca*), or Red Spruce (*Picea rubens*)) 1-3 years after harvest. Ten blocks did not receive herbicide application (hereafter untreated), and ten were treated with a glyphosate-based herbicide (hereafter treated). The treated blocks received an application of a glyphosate-based herbicide 5-6 years after harvest applied by helicopter. Limited herbicide use has occurred within the ARF, therefore treated blocks were located on crown land adjacent to the ARF (Figure 1). We qualitatively observed that treated blocks were dominated by planted coniferous species (height proportional to time since harvest) with sparse understory vegetation. In contrast, untreated blocks were generally mixed wood, consisting of naturally regenerating deciduous and coniferous species, along with planted conifers, and ground cover was characterized by some woody debris, minimal leaf litter, and a dense herbaceous layer.

To serve as controls, we selected 10 unharvested forest stands in the ARF that have no harvest history (Figure 1). These ecological reserve stands serve as reference conditions for the Acadian Forest, providing insight into the expected forest composition and structure of harvested blocks had they not undergone recent forestry practices (Mosseler et al. 2003). The control stands contained a diverse mix of tree species, including boreal species such as Balsam Fir, Jack Pine (*Pinus banksiana*), and White Spruce, as well as temperate species such as Sugar Maple (*Acer saccharum*), Yellow Birch (*Betula alleghaniensis*), and Red Spruce. Ground cover of control stands consisted of woody debris, leaf litter, and an herbaceous layer primarily composed of ferns or moss.

### ***Field Sampling***

Field surveys for salamanders occurred from July to August 2022. Although detectability is generally higher during nighttime or following rainfall, plethodontid salamanders are commonly detectable during daytime conditions as shown in previous research (O'Donnell and Semlitsch 2015) and our preliminary surveys in NB (Table S1). We therefore conducted all surveys during the day to ensure consistency across sites and accommodate logistical and safety constraints. To minimize potential biases related to weather conditions and time of day, one to three harvested blocks (treated or untreated) or unharvested control stands were randomly selected and surveyed each day. Each block or stand was surveyed once during the study period, with no repeated counts. Within each of the blocks or stands, we established a circular sampling plot with a diameter of 30 m. We established the center of each sampling plot to ensure a minimum 50 m buffer of similar habitat around its perimeter, minimizing potential edge effects (Murcia 1995). We conducted standardized area-constrained searches using circular transects at 30 m, 20 m, 10 m, and 0 m from the sampling plot's center (Figure S2). Two researchers searched under cover objects, rocks and logs, within 5 m of each transect and hand-captured salamanders. At the sampling plot's center, we measured 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) at depths of approximately 20 cm, 12 cm, and 10 cm, respectively. Additionally, we took a 360° photo using a Ricoh Theta 360 camera to document the canopy cover and ground cover of each survey plot. We estimated canopy cover by analyzing Ricoh Theta 360 images with ImageJ using the digital dot grid method (Goeking 2012).

Each time a salamander was captured, we identified and recorded the species, weighed the individual using a digital scale (US-Ranger) (± 0.01 g), and measured snout–vent length (SVL) with digital callipers (± 0.1 mm) (i.e., the distance from the snout to the anterior boundary of the cloaca). Species richness was low across plots, ranging from 0 to 3 species (see Supplementary Materials; Table S2). Most plots contained only one species, the Eastern Red-backed Salamander, which was captured in all plots. Given this limited variation, we restricted our analysis to Eastern Red-backed Salamander data. We categorized individuals into two age classes based on SVL: juveniles (≤ 32 mm) and adults (> 32 mm) (Mazerolle et al. 2021). Male Eastern Red-backed Salamanders were identified by the observation of testes using the candling method described by Gillette and Peterson (2001), and females were identified through the absense of testes and/or observation of eggs. We noted if females were gravid, recorded any injuries, including cuts, lesions, and tail autotomy. All salamanders were immediately released at the location of capture after processing. Sampling followed protocols to prevent disease spread as outlined by the Canadian Herpetofauna Health Working Group (2017) and that were approved by the Mount Allison University Animal Care Committee—ACC # 103181, as well research was conducted with authorization from The Department of Natural Resources and Energy Development, Government of New Brunswick under Scientific Permit No. SP22—002.

### ***Data Analysis***

All statistical tests were conducted in R version v 4.2.2 (R Core Team 2022). Before analyses, we examined the data to ensure the absence of unexplainable outliers or collinearity between predictor variables. The significance level (α) for all models was set at 0.05. For each model, we assessed the assumptions of normality of residuals and homogeneity of variance by examining a histogram and a normal probability plot (Q-Q plot) of the residuals, as well as a visual representation of the residuals plotted against the fitted values. Data are presented as mean ± standard error (SE) in the text unless otherwise specified.

#### Environmental Factors

We compared environmental factors among forest treatments (control, treated, and untreated) to evaluate our predictions regarding the impact of forest harvest practices on salamander habitat. First, using the ‘*glm*’ function from the ‘*stats*’ R package (R Core Team 2022) we performed two generalized linear models (GLMs). These models examined the influence of forest treatment (a categorical variable including control, untreated, and treated levels) and years since harvest (a continuous variable ranging from 5 to 26 years) on canopy cover (%) and soil moisture (%). Since canopy cover and soil moisture are expressed as percentages ranging from 0 to 100, we converted the data to proportions and used a binomial distribution for analysis (McCullagh and Nelder 1989). Second, we used the ‘*lm*’ function from the ‘*stats*’ R package (R Core Team 2022) to carry out linear models (LM) assessing the effect of forest treatment and years since harvest on soil temperature (°C) and soil pH.

Initially, the interaction between forest treatment and years since harvest was included in both GLMs and LMs. The significance of the interaction was evaluated based on the model output obtained from the '*anova*' function of the '*stats'* R package using the $\chi^2$-test statistic for GLMs and the $F$-test statistic for LMs. If not significant, we removed the interaction and re-ran the model to allow for the interpretation of main effects (Gelman and Hill 2007). For models where forest treatment had a significant main effect, we generated contrasts for each comparison (three comparisons total) using the '*emmeans*' function from the '*emmeans*' R package (Lenth 2022). *P* values generated for comparisons between forest treatments were corrected using a Tukey's HSD multiplicity adjustment (Lenth 2022).

### Abundance

We tested for differences in the relative abundance (i.e., the number of salamanders within our standardized survey area) of Eastern Red-backed Salamanders by performing a GLM with a Poisson distribution (McCullagh and Nelder 1989) using the same approach as describe for our analysis of environmental factors. The model included the fixed effects of forest treatment and years since harvest. We used the same approach to test interactions ($z$ = -1.45, $p$ = 0.15, not significant), then re-ran the model with the interaction removed to allow interpretation of the main effects (Gelman and Hill 2007). We performed post-hoc comparisons between forest treatments using the same approach as previously described.

### Body condition

We used the scaled mass index (SMI), derived from a standardized major axis (SMA)-adjusted ordinary least squares (OLS) regression of log-transformed body mass on log-transformed snout-vent length (SVL), as an index of body condition for adult Eastern Red-backed Salamanders (Schulte-Hostedde et al. 2005; Peig and Green 2009). SMA regression was used to account for allometric scaling, providing a scaling exponent that adjusts mass to a standardized SVL, making body condition estimates more size-independent (Peig and Green 2009). Since body condition serves as an indicator of energy reserves, particularly fat stores, higher SMI values indicate better condition (high mass relative to length) while lower values indicate poorer conditions. A similar, but slightly different approach, is to calculate SMI with a robust regression model to better account for outliers in the data (Maronna et al. 2019). Thus, we compared using OLS and robust regression estimators to assess which method best described the mass-SVL relationship using our data. We found that the OLS regression estimator met standard assumptions (independent and normally distributed residuals) and provided a better fit than the robust regression based on visual assessments, residual diagnostics, and RMSE and AIC values (see Supplementary Materials, Figure S2 and Table S3). As estimating SMI using robust regression did not improve model performance for our data, we selected to use the SMA-adjusted OLS regression to calculate our SMI (i.e., the body condition) of salamanders (as described by Peig and Green 2009).

To test for differences in body condition between forest treatments, we used a linear mixed-effects model (LMM) with the *'lmer'* function in the *'lmerTest'* R package (Kuznetsova et al. 2017). The model included forest treatment, years since harvest, and sex (a categorical variable including females and males) as fixed effects, and plot identity as a random effect. We used the same approach to test interactions ($F_{1,\ 101} = 1.50$, $p = 0.22$, not significant) and perform multiple comparisons as described previously. Due to the low numbers of instances of injury and tail autonomy observed, we report counts rather than performing statistical analyses.

Reproductive Activity

To analyze the effects of forest treatment and years since harvest on the proportion of juvenile Eastern Red-backed Salamanders relative to the total number of individuals observed, we performed a GLM with a binomial distribution (McCullagh and Nelder 1989). This metric serves as a proxy for demographic structure, providing insight into juvenile recruitment and potential population dynamics under different forest management conditions (Homyack and Haas 2009). While it does not directly measure reproductive output, it captures potential variation in juvenile survival and recruitment. The model included forest treatment and years since harvest as fixed factors. We used the same approach for testing interactions ($\chi^2_{1,\ 15} = 0.46$, $p = 0.50$, not significant) and for comparisons between treatment levels as described previously. Due to the low number of gravid females and egg masses observed during the study, we report counts rather than performing statistical analysis.

## Results

### *Forest Environmental Factors*

Canopy cover increased significantly with years since harvest ($\chi^2_1 = 169.47$, $p < 0.01$), and varied among forest treatments ($\chi^2_2 = 326.90$, $p < 0.01$). Control plots had the highest average canopy cover percentage (77.5 ± 2.28%), followed by treated blocks (10.7 ± 1.08%) and untreated blocks (7.61 ± 0.78%) (Figure 2a). Post-hoc contrasts confirmed that canopy cover was significantly greater in control plots compared to both treatment types ($p < 0.01$ for both) and was also greater in treated than untreated blocks ($p < 0.01$) (Table 1a).

We observed a significant interaction between years since harvest and forest treatment regarding soil temperature ($F_{1,\,25} = 6.23$, $p = 0.02$). Visual assessment suggested that soil temperature remained relatively constant across years since harvest in untreated blocks, but declined with increasing years since harvest in treated blocks (Figure 2b). Summary statistics suggest that control stands had lower soil temperatures (14.5 ± 0.36°C) compared to untreated (16.1 ± 0.39°C) and treated (16.0 ± 0.61°C) blocks.

There was also a significant interaction between years since harvest and forest treatment regarding soil moisture ($\chi^2_1 = 25.57$ $p < 0.01$). Based on visual assessment, soil moisture declined with years since harvest in untreated blocks, while remaining relatively stable in control and treated blocks (Figure 2c). Summary statistics indicate that on average, soil moisture was higher in control stands (30.6 ± 4.16%) than in untreated (21.3 ± 4.61%) and treated (14.8 ± 2.03%) blocks.

In contrast, soil pH did not differ significantly with years since harvest ($F_{1,\,26} = 3.59$, $p = 0.07$) nor among forest treatments ($F_{2,\,26} = 1.79$, $p = 0.19$; Table 1b; Figure 2d). Although estimated marginal means indicated that soil pH was lowest in control stands, intermediate in treated blocks, and highest in untreated blocks, post-hoc comparisons revealed no significant differences among treatments (all $p > 0.05$; Table 1b).

### *Salamander Population Structure*

Within our 30 m$^2$ sample plots, we captured a total of 183 Eastern Red-backed Salamanders in the control stands, 10 in the untreated blocks, and 42 in the treated blocks. Abundance of Eastern Red-backed Salamanders increased significantly with years since harvest ($\chi^2_1 = 14.60$, $p < 0.01$) (Table 2a). Based on comparisons across sites, salamander abundance was predicted to increase by 9.4% per year, corresponding to an approximate 2.45-fold increase over 10 years within the 26-year post-harvest range sampled. Additionally, Eastern Red-backed Salamander abundance differed significantly among forest treatments ($\chi^2_2 = 205.46$, $p < 0.01$) (Table 2a). Average salamander abundance was highest in control stands, followed by treated blocks, and lowest in untreated blocks (Figure 3; Table 2b).

### *Salamander Body Condition*

Eastern Red-backed Salamander body condition, quantified using a SMI (Peig and Green 2009), was significantly and negatively related to years since harvest (Table 3a). Body condition was lower in the control stands (0.74 ± 0.05 g) than in untreated (1.26 ± 0.14 g) and treated (1.24 ± 0.15 g) blocks, but did not differ between treated and untreated blocks or between sexes (Table 3b; Figure 4). Body condition of Eastern Red-backed Salamanders was approximately 0.5 g higher in the treated and untreated blocks than the control stands (Table 3b). During our study, we identified 11 Eastern Red-backed Salamander individuals with missing or regenerating tails: 10 in the control stands and one in the untreated blocks. Among these, nine were female and two were male. No Eastern Red-backed Salamanders with additional injuries, such as cuts or lesions, were observed.

### *Salamander Reproductive Activity*

We observed six gravid Eastern Red-backed Salamander females during the surveys: five in control stands and one in treated clear-cut blocks. Additionally, we observed 10 females guarding eggs in control stands, but none in harvested blocks. The proportion of juveniles within a survey was not related to years since harvest, and did not differ significantly among forest treatments (Table 4).

## Discussion

Herbicide application following clear-cutting altered key abiotic conditions of salamander habitat, which significantly influenced salamander abundance in regenerating stands. Contrary to our prediction that herbicide would exacerbate the negative effects of canopy removal on Eastern Red-backed Salamanders, treated clear-cut blocks instead exhibited more favourable conditions than untreated blocks. These differences were associated with greater canopy cover and higher abundance of salamanders in treated blocks. While the impacts of clear-cutting on salamander populations are well documented (e.g. DeMaynadier and Hunter 1995, Tilghman et al. 2012; Ochs et al. 2022), our study adds new insight into how herbicide application interacts with canopy removal to shape post-harvest habitat.

We identified significant interactions between forest treatment and years since harvest, indicating that the effects of canopy removal on soil temperature and moisture differ depending on whether a clear-cut forest was treated with herbicide (Figure 2). Treated blocks showed a decreasing trend in soil temperature and a slight increase in soil moisture, gradually approaching the cooler and wetter conditions observed in control stands (Table 2). In contrast, untreated blocks maintained consistently warmer soil temperatures and a negative trend in soil moisture over time, diverging from control conditions (Table 2). The findings for untreated blocks align with previous research showing that clear-cutting elevates soil temperature due to increased solar radiation and initially increases soil moisture as reduced tree root respiration limits water uptake (Keenan and Kimmins 1993). Though herbicide-treated blocks exhibit a return of herbaceous species to pre-herbicide levels within 3-5 years, abundance remains lower than in similarly aged untreated blocks for at least 10 years after application (Xiao et al. 2024). These long-term shifts in understory composition may contribute to the observed differences in microclimate conditions. Together, these findings suggest that herbicide-induced acceleration of canopy closure may help mitigate post-harvest abiotic changes, promoting a faster return to habitat conditions favourable for salamanders.

In contrast to soil temperature and moisture, we found no significant differences in soil pH among forest treatments. Short-term effects of clear-cutting (0-3 years post-harvest) typically lead to soil acidification (Johnson et al. 1991). Therefore, shifts in soil pH that may impact salamander populations are likely more evident in early seral blocks (approximately 0-5 years) compared to control stands, considering the progression of forest development. However, our study did not include harvest blocks within this early seral range, limiting our ability to assess these changes directly. Additionally, glyphosate strongly binds to soil particles and has a relatively short half-life in soil, 45-60 days (Feng et al. 1990), suggesting minimal long-term impact on soil pH and other chemical properties within the timescale of our study. While pH remained consistent across forest treatments, the changes observed in other abiotic factors, such as temperature and moisture, reflect how herbicide application can influence salamander habitat quality. Nonetheless, our study highlights the resilience of Eastern Red-backed Salamander populations, demonstrating their ability to withstand the challenging post-harvest and herbicide-exposed period.

This resilience was also reflected in salamander body condition. Contrary to our prediction that Eastern Red-backed Salamander body condition would be highest in unharvested (control) stands, we observed the opposite trend: body condition was lower in control stands than in both treated and untreated harvest blocks, where it remained similar. This finding contrasts with Welsh et al. (2008), who reported higher body condition in the Siskiyou Mountains salamander (*Plethodon stormi*) in mature versus young forests. However, our results align with those of Gade et al. (2023), where Eastern Red-backed Salamander body condition was lower in mature forests than in early successional stands. The disparity between these findings may be explained by differences in the competitive regimes under which each population exists, which may impact their relative fitness. Body condition, an indicator of fat reserves, reflects density-dependent processes such as foraging success and energy expenditure in response to competition and predation pressures (Homyack et al. 2011). Lower body condition in high-density populations, as observed in the control stands, may indicate resource limitation on a per individual basis due to higher salamander abundance or reduced food availability, both of which can increase intraspecific competition (Getz 1996). Meanwhile, the similar body condition observed in treated and untreated blocks suggests that energy availability or competition may be comparable between salamanders in both environments, despite differences in overall abundance. The link between forest management and salamander body condition likely involves multiple indirect pathways, requiring further investigation, particularly regarding the differences in inter- and intra-specific interactions across various management practices.

Although salamander abundance and body condition varied among forest treatments, we observed no significant differences in reproductive activity, measured by the proportion of juveniles, number of gravid females, and egg masses. While our dataset was sufficient to assess the proportion of juveniles in the population, the low number of gravid females and egg masses constrained the strength of our inferences for these indicators. To improve detection of gravid females and egg masses, future studies could increase sampling frequency during the reproductive season and target periods following recent rainfall when salamander surface activity is highest (O'Donnell and Semlitsch 2015). While previous research has reported a higher proportion of juvenile Eastern Red-backed Salamanders in unharvested stands compared to harvested blocks (Homyack and Haas 2009), we found no evidence that these differences were reflected in reproductive activity in our study.

Unsurprisingly, control stands, which serve as reference stands for mature Acadian Forest, had highest canopy cover, soil moisture, and lowest soil temperatures than harvested blocks. These conditions are optimal for salamanders (Wyman 1988; Petranka 2010) and likely explain our finding of higher Eastern Red-backed Salamander abundance in control stands compared to harvested blocks, regardless of whether they were treated with an herbicide. While our results support this pattern, we caution that variation in cover object availability was not accounted for in our sampling design, which may have an influence on salamander detectability across plots and confounded abundance estimates (O'Donnell and Semlitsch 2015). Differences in cover availability could influence salamander estimates across all forest treatments. Nonetheless, our findings align with Homyack and Haas (2009), who reported significantly lower salamander abundance in harvested forests where the canopy has been removed compared to unharvested stands.

Historical forest management practices in the Acadian Forests of New Brunswick have increased the amount of conifer and decreased the amount of deciduous and mixed forests (Noseworthy and Beckley 2020). The increase in planting of conifer trees has led to increased use of glyphosate-based herbicides, up to 30 % of forested lands in some watershed have been treated over the last 10 years (Edge et al. 2023). Changes in forest composition could have significant ecological implications, particularly for forest-dependant species like the Eastern Red-backed Salamander. Our findings indicate that while herbicide-treated clear-cut blocks support higher salamander abundance compared to untreated ones, their recovery to levels comparable to unharvested control stands within the typical harvest rotation length (40-60 years) for crown land in New Brunswick remains uncertain (Frego et al. 2014). Given that older Acadian Forests serve as critical refuges for salamander and other forest interior species, prioritizing the protection of these stands is essential for maintaining biodiversity in managed forest landscapes.

As the regeneration cycle advances beyond the scope of this study (26 years post-harvest), the environments of harvested forests continue to change. While our results demonstrate that herbicide application may enhance salamander habitat conditions and support higher abundance during forest regeneration, longer-term consequences for salamander populations and habitat quality remain uncertain and are in need of further study. This study offers a foundational step toward understanding how post-harvest silvicultural practices interacts with forest harvesting to influence forest floor microhabitats and forest specialist species. However, the short-term and single-season design of this work limits the strength of our conclusions for generalization. Future studies incorporating long-term, repeated sampling and continuous environmental monitoring across successional and climatic gradients are needed to fully understand the ecological consequences of forest management practices. Integrating these insights into forestry planning will support the development of more adaptive and ecologically informed strategies that balance timber production with sustaining biodiversity and ecological integrity.

## Acknowledgments

We sincerely thank David Fox for his dedicated efforts during the sampling process. We also appreciate the support provided by Anna Varty, Jordan Green, Chelsea Burnham, Gregor Wilson, Andrew Lewis, and Shane Heartz throughout the summer of 2022. We are grateful to Dr. Joshua Kurek for his valuable comments and suggestions on earlier drafts of this manuscript.

## Article Information

### *Data availability*

The data and R code for this study are available from Open Source Framework (OSF), which can be found at the following link:

https://osf.io/5hc6s/?view_only=8a428e3052a94760a295ffedb1bfbd31

## Author Information

### *Author ORCIDS*

SEL: 0009-0008-0442-8801; CBE: 0000-0002-4713-0035; JLR: 0000-0001-7691-6910

### *Author contributions*

Conceptualization: SEL, CBE, JLR. Data curation: SEL, JLR. Formal analysis: SEL, JLR. Funding Acquisition: CBE, JLR. Methodology: SEL, CBE, JLR. Project administration: SEL, CBE, JLR. Supervision: CBE, JLR. Visualization: SEL, JLR. Writing – original draft: SEL. Writing – review & editing: SEL, CBE, JLR.

***Competing interests:*** The authors declare there are no competing interests.

### *Funding information*

This work was funded through a Mount Allison University Independent Student Research Grant (SEL), Natural Sciences and Engineering Research Council of Canada (Discovery Grant to JLR), and Natural Resources Canada – Canadian Forest Service (CBE).

## References

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

Davic R.D., Welsh, H.H. 2004. On the ecological roles of salamanders. Annu. Rev. Ecol. Evol. Syst. **35**: 405–434.

De Lombaerde E., Baeten L., Verheyen K., Perring M.P., Ma S., Landuyt D. 2021. Understorey removal effects on tree regeneration in temperate forests: a meta-analysis. J. Appl. Ecol. **58**: 9–20.

Degraaf R.M., Rudis D.D. 1990. Herpetofaunal species composition and relative abundance among three New England forest types. For. Ecol. Manag. **32**: 155–165.

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**: 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**: 929.

Edge C.B., Haines W., Blaney M., Noël M. 2023. Low detection of glyphosate in rivers following application in forestry. Pest Manag. Sci. **79**: 2951–2958.

Feng J.C., Thompson D.G. 1990. Fate of glyphosate in a Canadian forest watershed. 2. Persistence in foliage and soils. J. Agric. Food Chem. **38**: 1118–1125.

Franklin J.F., Lindenmayer D., MacMahon J.A., McKee A., Magnuson J., Perry D.A., Waide R., Foster, D. 2000. Threads of continuity. Conserv. Pract. **1**: 8–17.

Frego K., Haughian S., Mathieson K., Mcintosh L., Mudge J., Pokorski M., Witkowski A. 2014. Forest management on New Brunswick's Crown Lands: The bryophyte perspective. N. B. Nat. **41**: 40–47.

Gade M.R., Gould P.R., Wilk A.J., Donlon K.C., Brown M.L., Behan M.L., Roseman M.A., Tutterow A.M., Amber E.D., Wagner R.B., Hoffman A.S., Myers J.M., Peterman W.E. 2023. Demography and space-use of Eastern Red-backed Salamanders (*Plethodon cinereus*) between mature and successional forests. Ecol. Evol. **13**: e9764.

Gelman A., Hill J. 2007. Data analysis using regression and multilevel/hierarchical models. Cambridge University Press, Cambridge [England]; New York.

Getz W.M. 1996. A hypothesis regarding the abruptness of density dependence and the growth rate of populations. Ecology **77**: 2014–2026.

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

Goeking S.A., Liknes G.C., Lindblom E., Chase J., Jacobs D.M., Benton R. 2012. A GIS-based tool for estimating tree canopy cover on fixed-radius plots using high-resolution aerial imagery. ***In*** Moving from status to trends: Forest Inventory and Analysis (FIA) symposium 2012. Compiled by Morin R.S., Liknes G.C. USDA For. Serv. Gen. Tech. Rep. NRS-P-105. pp. 237-241.

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

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

Homyack J.A., Haas C.A., Hopkins W.A. 2010. Influence of temperature and body mass on standard metabolic rate of Eastern Red-backed Salamanders (*Plethodon cinereus*). J. Therm. Biol. **35**: 143–146.

Homyack J.A., Haas C.A., Hopkins W.A. 2011. Energetics of surface-active terrestrial salamanders in experimentally harvested forest. J. Wildl. Manag. **75**: 1267–1278.

Johnson C.E., Johnson A.H., Siccama T.G. 1991. Whole-tree clear-cutting effects on exchangeable cations and soil acidity. Soil Sci. Soc. Am. J. **55**: 502–508.

Jung M., Lesiv M., Warren-Thomas E., Shchepashchenko D., See L., Fritz S. 2023. The importance of capturing management in forest restoration targets. Nat. Sustain. **6**: 1321–1325.

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

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

Kuuluvainen T., Angelstam P., Frelich L., Jõgiste K., Koivula M., Kubota Y., Lafleur B., Macdonald E. 2021. Natural disturbance-based forest management: moving beyond retention and continuous-cover forestry. Front. For. Glob. Change **4**.

Kuznetsova A., Brockhoff P.B., Christensen R.H.B. 2017. lmerTest package: Tests in linear mixed effects models. J. Stat. Softw. **82**: 1–26.

Lenth R. 2022. emmeans: estimated marginal means, aka least-squares means. R package (version 1.7.2).

Maronna R., Martin D., Yohai V., Salibian-Barrera M. 2019. Robust statistics: theory and methods (with R). 2nd ed. John Wiley & Sons, Hoboken, N.J.

Matute P. 2019. Review of glyphosate use in British Columbia forestry. FPInnovations. Project No. 301013763.

Mazerolle M.J., Lapointe St-Pierre M., Imbeau L., Joanisse G. 2021. Woodland salamander population structure and body condition under irregular shelterwood systems. Can. J. For. Res. **51**: 1281–1291.

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

McKenny H.C., Keeton W.S., Donovan T.M. 2006. Effects of structural complexity enhancement on Eastern Red-backed Salamander (*Plethodon cinereus*) populations in northern hardwood forests. For. Ecol. Manag. **230**: 186–196.

Mosseler A., Lynds J.A., and Major J.E. 2003. Old-growth forests of the Acadian Forest Region. Environ. Rev. **11**.

Murcia C. 1995. Edge effects in fragmented forests: implications for conservation. Trends Ecol. Evol. **10**: 58–62.

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., Purvis A. 2015. Global effects of land use on local terrestrial biodiversity. Nature, **520**: 45-50.

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

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

O'Donnell K.M., Semlitsch, R.D. 2015. Advancing terrestrial salamander population ecology: The central role of imperfect detection. J. Herpetol. **49**(4): 533–540.

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

Petranka J.W. 2010. Salamanders of the United States and Canada. Smithsonian Books, Washington, D.C.

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

Schulte-Hostedde A.I., Zinner B., Millar J.S., and Hickling G.J. 2005. Restitution of mass–size residuals: validating body condition indices. Ecology, **86**(1): 155–163.

Sugalski M.T., Claussen D.L. 1997. Preference for soil moisture, soil pH, and light intensity by the salamander, *Plethodon cinereus*. J. Herpetol. **31**: 245–250.

Swift D.E., Kilpatrick B., and Pegler S. 2005. Acadia Research Forest: a primary forest research facility for Natural Resources Canada, Canadian Forest Service – Atlantic. **1**.

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

Venn T.J. 2023. Reconciling timber harvesting, biodiversity conservation and carbon sequestration in Queensland, Australia. For. Policy Econ. **152**: 102979.

Wagner R.G., Little K.M., Richardson B., McNabb K. 2006. The role of vegetation management for enhancing productivity of the world's forests. For. Int. J. For. Res. **79**(1): 57–79.

Welsh H.H. Jr., and Droege S. 2001. A case for using plethodontid salamanders for monitoring biodiversity and ecosystem integrity of North American forests. Conserv. Biol. **15**: 558–569.

Welsh H. H. Jr., Pope K. L., Wheeler C. A. 2008. Using multiple metrics to assess the effects of forest succession on population status: a comparative study of two terrestrial salamanders in the US Pacific Northwest. Biol. Conserv. **141**: 1149–1160.

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

Xiao J., Yakimowski S., Brown M., Heartz S., Parachnowitsch A.L., Edge C.B. 2024. Changes to the understory vegetation community of the Acadian Forest a decade after herbicide use. For. Int. J. For. Res. **97**: 414–423.

## Tables

**Table 1.** Post-hoc pairwise comparisons testing for differences among the forest treatments (control, untreated clear-cut, and treated clear-cut) for **(A)** canopy cover (%) and **(B)** soil pH. Canopy cover was analyzed using a generalized linear model (GLM), while soil pH was analyzed using a linear model (LM). Since interactions were not significant in initial models, main effects could be interpreted directly (Figure 2). We present estimates ($\beta$) and their corresponding standard error (*SE)* on the response scale, test statistics ($z$-ratios for GLM or $t$-ratios for LM), 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** | | | $z$-ratio | |
| 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** | | | $t$-ratio | |
| 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 |

**Table 2.** **(A)** Outcomes of the generalized linear model (GLM) that examined the effect of forest treatment and year since clear-cut on Eastern Red-backed Salamander abundance. I present estimates ($\beta$) and their corresponding standard error (*SE)* on the latent scale, $z$-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 Eastern Red-backed Salamander abundance. We present estimates ($\beta$) and their corresponding standard error (*SE)* on the response scale, $t$-ratios, and $p$-values ($p_{corr}$) corrected for multiple comparisons. Bolded values indicate significance.

| **Eastern Red-backed Salamander Abundance** | | | | |
|---|---|---|---|---|
| **(A) Model Output** | | | | |
| *Fixed effects* | $\beta$ | *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* | $\beta$ | *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** |

**Table 3. (A)** Outcomes of the linear mixed effects model (LMM) that examined the effect of forest treatment, year since clear-cut, and sex on Eastern Red-backed Salamander body condition (g). We present estimates ($\beta$) and their corresponding standard error (*SE)* on the latent scale, *t*-values, and *p*-values (*p*). For random effects, we present the variance ($\sigma^2$) and standard error (*SE*). **(B)** Post-hoc pairwise comparisons among the effect of forest treatment (control, untreated clear-cut, and treated clear-cut) on Eastern Red-backed Salamander body condition (g). We present estimates ($\beta$) and their corresponding standard error (*SE)* on the response scale, *t*-ratios, and *p*-values ($p_{corr}$) corrected for multiple comparisons. Bolded values indicate significance.

| **Eastern Red-backed Salamander Body Condition** | | | | |
|---|---|---|---|---|
| **(A) Model Output** | | | | |
| *Fixed effects* | $\beta$ | *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* | $\sigma^2$ | *SE* | | |
| Plot ID | 0.00 | 0.00 | | |
| Residuals | 0.03 | 0.16 | | |
| **(B) Post-hoc Multiple Comparisons** | | | | |
| *Forest treatment Contrasts* | $\beta$ | *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 |

**Table 4. (A)** Outcomes of a generalized linear model that examined the effect of forest treatment and year since clear-cut on the proportion of Eastern Red-backed Salamander that were juveniles during surveys. We present estimates ($\beta$) and their corresponding standard error (*SE)* on the latent scale, *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. We present estimates ($\beta$) and their corresponding standard error (*SE)* on the response scale, *t*-ratios, and *p*-values ($p_{corr}$) corrected for multiple comparisons. Bolded values indicate significance.

| **Eastern Red-backed Salamander Juvenile Proportion** | | | | |
|---|---|---|---|---|
| **(A) Model Output** | | | | |
| *Fixed effects* | $\beta$ | *SE* | *z*-value | *p* |
| Intercept (Control) | **-0.89** | **0.14** | **-6.51** | **< 0.01** |
| 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* | $\beta$ | *SE* | *t*-ratio | $p_{corr}$ |
| Control vs. Untreated | 0.72 | 1.23 | -0.19 | 0.98 |
| Control vs. Treated | 0.23 | 0.56 | -0.60 | 0.82 |
| Untreated vs. Treated | 0.31 | 0.40 | -0.92 | 0.63 |

## Figure Captions

**Figure 1**. Map showing the locations of the study survey sites within the Acadia Research Forest, New Brunswick, Canada. The grey circles indicate control stands, the blue squares indicate untreated harvest blocks, and the green triangles indicate treated harvest blocks. Figure was created using ArcGIS Pro version 3.3.0 and assembled from the following data sources: *North American Atlas – Political Boundaries* (Commission for Environmental Cooperation [CEC], 2022); *Acadia Research Forest Boundary* shapefile (Natural Resources Canada); and *Global Oceans and Seas*, version 1 (Flanders Marine Institute [VLIZ], 2021)

**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. Dashed blue lines represent untreated clear-cut blocks and solid green lines show treated clear-cut forest treatments. Points represent raw data. Dotted grey lines represent the mean value in the control stands. 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).

**Figure 3.** Salamander abundance in control (grey 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. (*).

**Figure 4.** Salamander body condition in control (grey 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 (*).

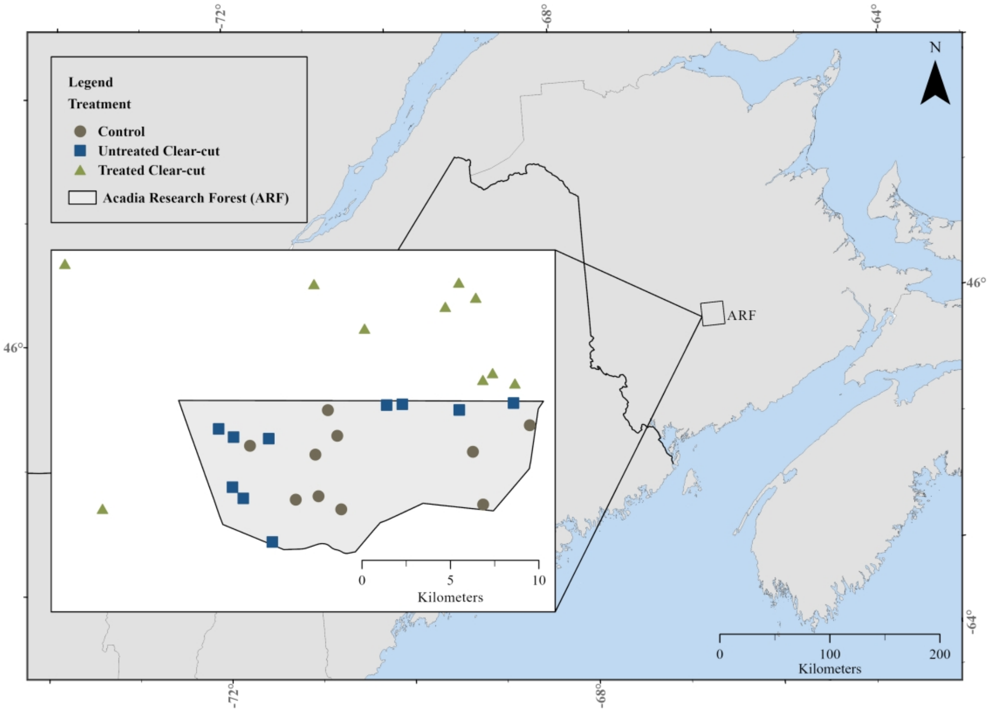
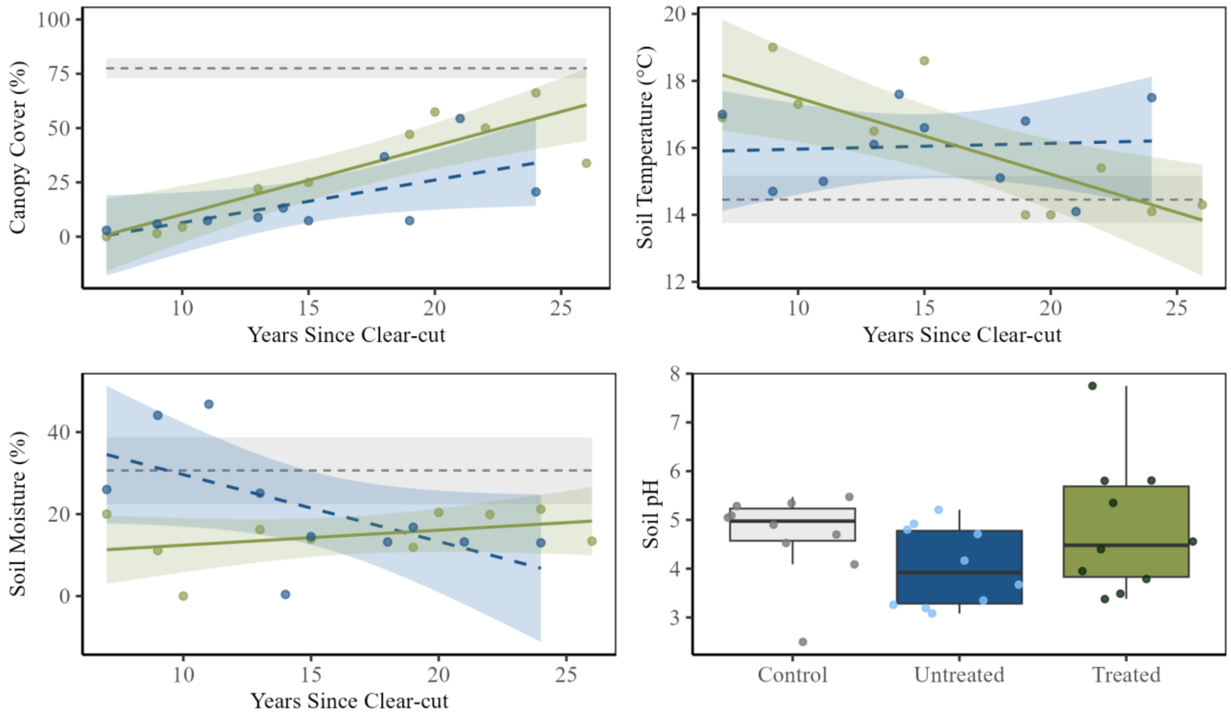
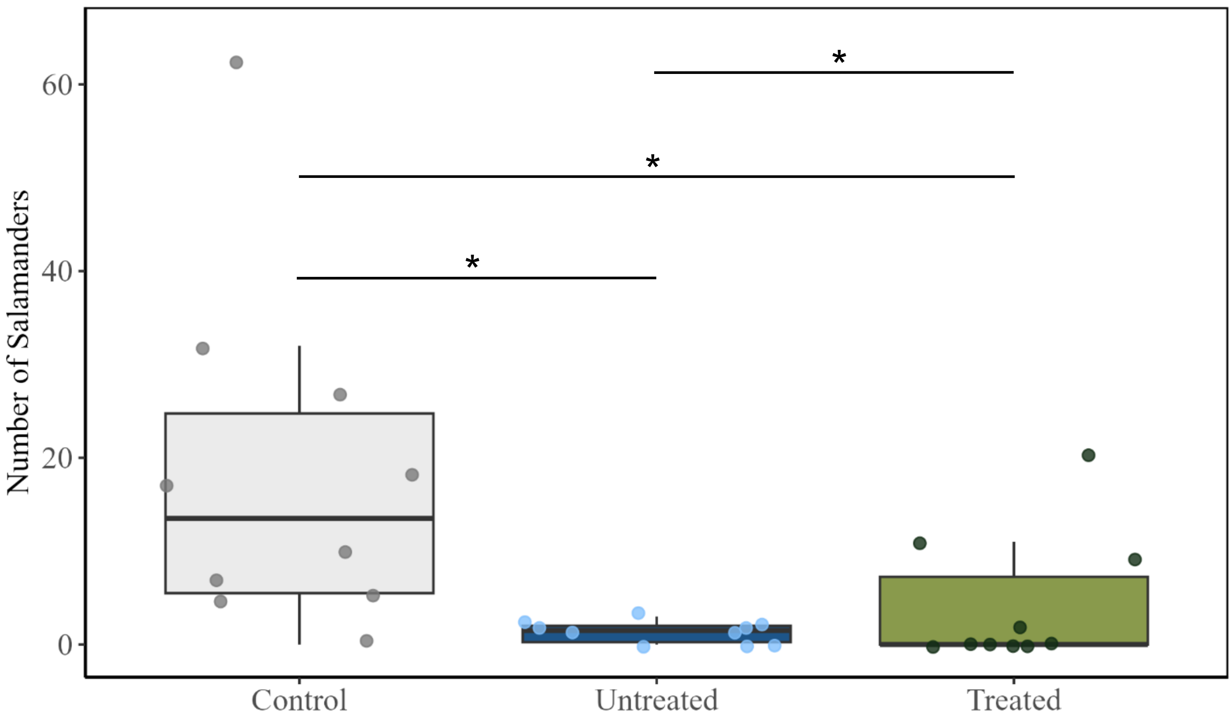
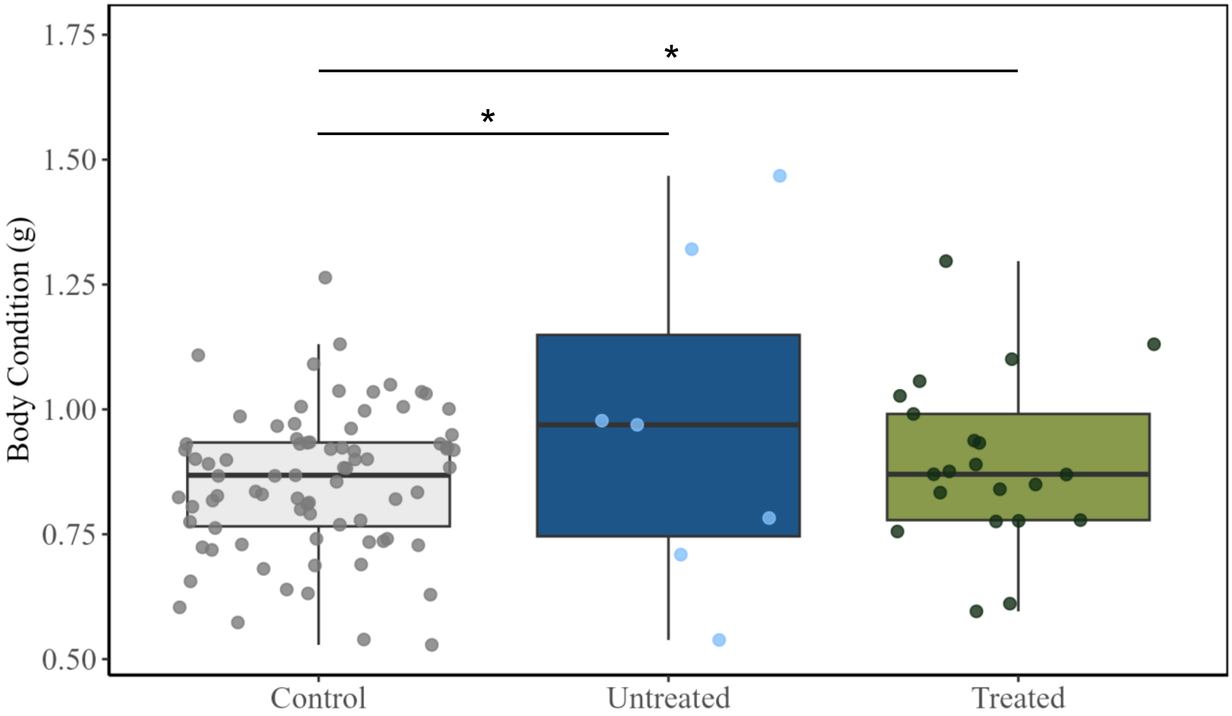


Fig 1

297x209mm (300 x 300 DPI)


Canopy Cover (%)
100
75
50
25
0
10
15
20
25
Years Since Clear-cut
Soil Temperature (°C)
20
18
16
14
12
10
15
20
25
Years Since Clear-cut
Soil Moisture (%)
40
20
0
10
15
20
25
Years Since Clear-cut
Soil pH
8
7
6
5
4
3
Control
Untreated
Treated


Number of Salamanders
0
20
40
60
Control
Untreated
Treated
*
*
*


*
*
Body Condition (g)
1.75
1.50
1.25
1.00
0.75
0.50
Control
Untreated
Treated