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Impact of imidacloprid-based chemical control for hemlock woolly adelgid (*Adelges tsugae*) on the eastern red-backed salamander (*Plethodon cinereus*)

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Abstract

The hemlock woolly adelgid (HWA; *Adelges tsugae*) is an invasive insect causing widespread mortality of eastern hemlock (*Tsuga canadensis*) across eastern North America, altering forest structure and associated ecosystems. The insecticide imidacloprid is used for HWA control, but its persistence and potential for soil and water contamination raises concern for fossorial species. We used the eastern red-backed salamander (RBS; *Plethodon cinereus*) as an indicator species to assess how HWA infestation and imidacloprid treatment influence hemlock stands in southwestern Nova Scotia. We conducted line transect surveys across two HWA-infested sites—each containing four treated and four untreated 1-ha stands—and one uninfested control site with four mature hemlock stands. We applied a multi-model AICc framework to test the effects of treatment, infestation, and environmental covariates on RBS abundance, body condition, and abundance within a zone of potential residue exposure. Treatment and infestation were the only predictors of abundance: treatment increased abundance, whereas infestation reduced it. Canopy openness, soil pH, and soil moisture did not differ significantly among stands, and none were found to be predictors of body condition or abundance within the zone of potential exposure. These results reinforce the value of integrating chemical control with long-term monitoring while biological control strategies develop.

Keywords: Integrated pest management, invasive species, indirect effects, bark spray, Xytect 2F

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1. Introduction

Canada's forests are a vital ecological and economic resource. In recent years, these ecosystems have faced increasing challenges posed by both native and novel insect species outbreaks (Pureswaran et al., 2018). A recent threat to forests on the east coast of Canada is the hemlock woolly adelgid (Hemiptera: Adelgidae, *Adelges tsugae* Annand; hereafter, HWA), an aphid-like insect introduced to eastern North America from southern Japan, which has caused widespread hemlock mortality since its detection in Virginia, USA, in 1951 (Havill et al., 2016). Hemlock woolly adelgid were first detected in Nova Scotia, Canada in 2017 and have since spread throughout much of the western part of the province and caused extensive death and decline of eastern hemlock (*Tsuga canadensis* L.) which constitute much of the province's little-remaining mature old-growth forest (CFIA, 2017; Emilson et al., 2018). Noticeable tree mortality suggests that HWA populations have persisted in the area for at least a decade before they were detected (CFIA, 2017). Detecting HWA infestations is inherently difficult because populations are often cryptic and may persist for years before obvious signs of decline become visible (Emilson & Stastny, 2019). This delayed detectability has contributed to the rapid and widespread establishment of HWA across its invasive range, as infestations are typically well advanced by the time management interventions are implemented. Within HWA's native Asian and naturalized western North American ranges, hemlocks are resistant to HWA or natural enemies control the population (Lagalante & Montgomery, 2003; Havill et al., 2014, 2016; Crandall et al., 2022), both of which are lacking in eastern North America (McClure, 1991).

High HWA densities cause the death of impacted shoots and prevent the growth of new foliage, ultimately leading to inhibition of photosynthesis and tree mortality (McClure, 1991). Mortality of mature hemlock trees may occur in as little as four years after initial infestation,

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however the lower temperatures in northern latitudes limit dispersal, population density, and infestation intensity (Orwig et al., 2002; Skinner et al., 2003), and in the eastern hemlock's northern range—New England through Maritime Canada—HWA infestations cause > 90% tree mortality within 10-15 years (Emilson et al., 2018).

In Atlantic Canada, the eastern hemlock is an integral component of the region's last remaining old-growth forest and are highly valued for aesthetic reasons. This late-successional conifer is a foundation species that controls community structure by stabilizing ecological processes and environmental conditions (Ellison et al., 2005). Mature stands create a distinct microclimate and structurally diverse habitat characterized by low canopy light penetration, cool, moist soils, acidic litter deposition, and consistent daily temperatures that harbour unique communities of vertebrate and invertebrate taxa (Ellison et al., 2005; Eschtruth et al., 2006). Eastern hemlock loss has been shown, or predicted, to impact mammal, bird, and invertebrate communities (Tingley et al., 2002; Ingwell et al., 2012; Degrassi, 2018; Cox et al., 2022). Infestation by HWA and the associated decline of eastern hemlock have been predicted to facilitate changes in understory vegetation and the establishment of exotic plants (Orwig & Foster, 1998), and subsequent empirical work has documented increases in understory species richness and the colonization of non-native shrubs following hemlock loss (Eschtruth et al., 2006; Ford et al., 2012; Ellison, 2014). Although native tree species recolonize HWA-infested regions following mass hemlock mortality, the ecosystem services and habitat they provide are not comparable to those of the eastern hemlock (Orwig & Foster, 1998); therefore, the continued loss of mature stands is expected to cause further changes to wildlife abundance patterns in affected areas.

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The current Canadian HWA management plan focuses on the suppression of active infestations and short-term protection via insecticide use (Emilson et al., 2018). The goal of insecticide use is to slow the dispersal of HWA until effective biological control has been developed (Emilson et al., 2018). Systemic neonicotinoid insecticides are routinely used in the US and Canada to control HWA with several application methods: in the soil, via drench or injection; on the bark, via basal bark spray; or directly into the tree, via trunk injection, with the most commonly used active ingredient being imidacloprid (Havill et al., 2014; Emilson et al., 2018; Emilson & Stastny, 2019). Following application, imidacloprid can be translocated to the crown and reduce HWA populations within months; depending on application method, tree size, and infestation pressure, substantial suppression is often achieved within a year, and residues (and associated suppression) can persist for multiple years after a single treatment (Cowles et al., 2006; Coots et al., 2013; Benton et al., 2015; Turcotte et al., 2017).

The impact of chemical pesticide application on aquatic and riparian ecosystems, including imidacloprid, is a topic of extensive research (e.g. Coots et al., 2013; Benton et al., 2015; Morrissey et al., 2015); however, non-target effects of chemical pesticides on fossorial (adapted for burrowing or digging) species are often overlooked. Studies on acute and subchronic imidacloprid exposure in larval amphibians report minimal effects on survival and growth, but do reveal sublethal impacts on behaviour and the expression of genes controlling development (Danis & Marlatt, 2021; Sweeney et al., 2021). Importantly, these findings are based on aquatic exposure and are not directly relevant to the responses of terrestrial adult salamanders.

In the woodlands of northeastern North America, the terrestrial eastern red-backed salamander (*Plethodon cinereus*; hereafter, RBS) exists in high abundance and contributes a

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large source of biomass to forest food webs (Burton & Likens, 1975). It occupies multiple forest types but densities are highest in mature forests with low canopy openness and coarse woody debris in late stages of decomposition—eastern hemlock stands possess both these attributes, making them prime habitat for these salamanders (Ellison et al., 2005; Cox et al., 2022). *Plethodon* genus thrive in mature hemlock stands as the cool and moist soil conditions compliment their physiological requirements to prevent desiccation and facilitate gas exchange (Wells, 2007). If HWA infestations cause extensive mortality of canopy trees, increased soil temperatures and decreased soil moisture due to increased canopy openness could negatively impact RBS populations (Jaeger, 1971; Wyman, 1988). Soil pH has a strong influence on the spatial distribution of RBS and heightened soil acidity has been documented to cause reduced salamander growth rates and body condition (Wyman & Hawksley-Lescault, 1987; Sugalski & Claussen, 1997). Changes to understory vegetation due to canopy decline, and disruptions to nutrient cycling due to neonicotinoid-induced microbe decline (Akter et al., 2023), may indirectly alter soil pH. Collectively, an altered microclimate in hemlock stands caused by HWA infestation could reduce regional abundances and health of RBS populations.

Due to its heightened susceptibility to anthropogenic disturbances, RBS has become a commonly employed indicator to assess ecological change (Welsh JR. & Droege, 2001; Siddig et al., 2019), and serves here as a focal species for evaluating forest responses to HWA infestation and imidacloprid treatment. We compared RBS abundance and body condition across three forest conditions: (1) uninfested hemlock stands, (2) HWA-infested untreated stands, and (3) HWA-infested stands treated with imidacloprid via basal bark spray. To characterize how treatment and infestation influence both the habitat and salamander responses, we asked whether (1) environmental variables, (2) RBS abundance, and (3) RBS body condition differ across

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132 treatment and infestation status, and (4) whether RBS abundance differs between zones of
133 potential exposure to imidacloprid residues (0–2 m) and >2 m from treated hemlock trees.

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2. Methods

2.1. Study region and experimental design

We conducted the study at three sites in the provinces of New Brunswick and Nova Scotia, Canada (Figure 1). When this study was originally designed, uninfested control sites (Bennery and Moore) were chosen approximately 200 km away from the infested sites, which at the time were beyond the known HWA invasion front in Nova Scotia. However, subsequent surveys revealed infestation in the same provincial county making it likely that these sites were also infested. Consequently, we established an alternative control site in Odell Park, New Brunswick, where eastern hemlock stands remain uninfested. Although geographically distant from the Nova Scotia plots, this Odell Park site was selected to ensure the absence of active infestation while maintaining comparable old-growth hemlock forest structure. Because of this necessity, our study design is spatially imbalanced, with Odell Park functioning as the single true control site. We acknowledge that this limits the strength of comparisons between infested and uninfested forests. Nevertheless, retaining the control data was essential for assessing the effect of HWA infestation on RBS metrics and environmental variables, and we interpret these comparisons cautiously considering this constraint.

The uninfested control site comprised 4 plots (spaced ≥ 100 m apart) within Odell Park's hemlock stand. The other two sites, located in southwestern Nova Scotia (Sissiboo site, Digby County; McKay site, Shelburne County), were dominated (≥ 80%) by mature (80-150 years old) or old-growth (150+ years old) eastern hemlock stands (Lynds & LeDuc, 1995). Both Sissiboo and McKay sites were heavily infested with HWA and significant tree decline had occurred. At each of the Sissiboo Falls and McKay Lakes sites, 4 control (infested and untreated) and 4 insecticide treated (infested and treated) 0.5 ha radial forest plots were established. In each of the

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157 treatment plots the insecticide Xytect 2F (Canadian registration number: 34596, active ingredient
158 imidacloprid, 240 g/L) was applied to trees greater than 10 cm diameter at breast height (DBH)
159 at a rate of 1.16 ml Xytect 2F (0.278 g a.i) per cm DBH with a backpack sprayer (SP0 Triple
160 Function Bak-Pak Sprayer, Hudson, Lowell, Michigan, USA). Application began by treating
161 trees at the center of the plot and expanding outward until the maximum annual application rate
162 for imidacloprid (0.45 kg a.i/ha) was reached. Insecticide application occurred over two years
163 with 40–60 trees being treated in each treatment plot in October of 2020 and 2021 (Edge et al.
164 2025). In total, the experimental design consisted of 20 plots: 8 infested/treated, 8
165 infested/untreated, and 4 uninfested reference plots.
166 The three study sites are within the Acadian Forest Ecoregion (Rowe, 1972)
167 characterized by a combination of softwood, hardwood, and mixed-wood forests dominated by a
168 mix of eastern hemlock, eastern white pine (*Pinus strobus*), red spruce (*Picea rubens*), balsam fir
169 (*Abies balsamea*), american beech (*Fagus grandifolia*), sugar maple (*Acer saccharum*), white
170 birch (*Betula papyrifera*), and black spruce (*P. mariana*).
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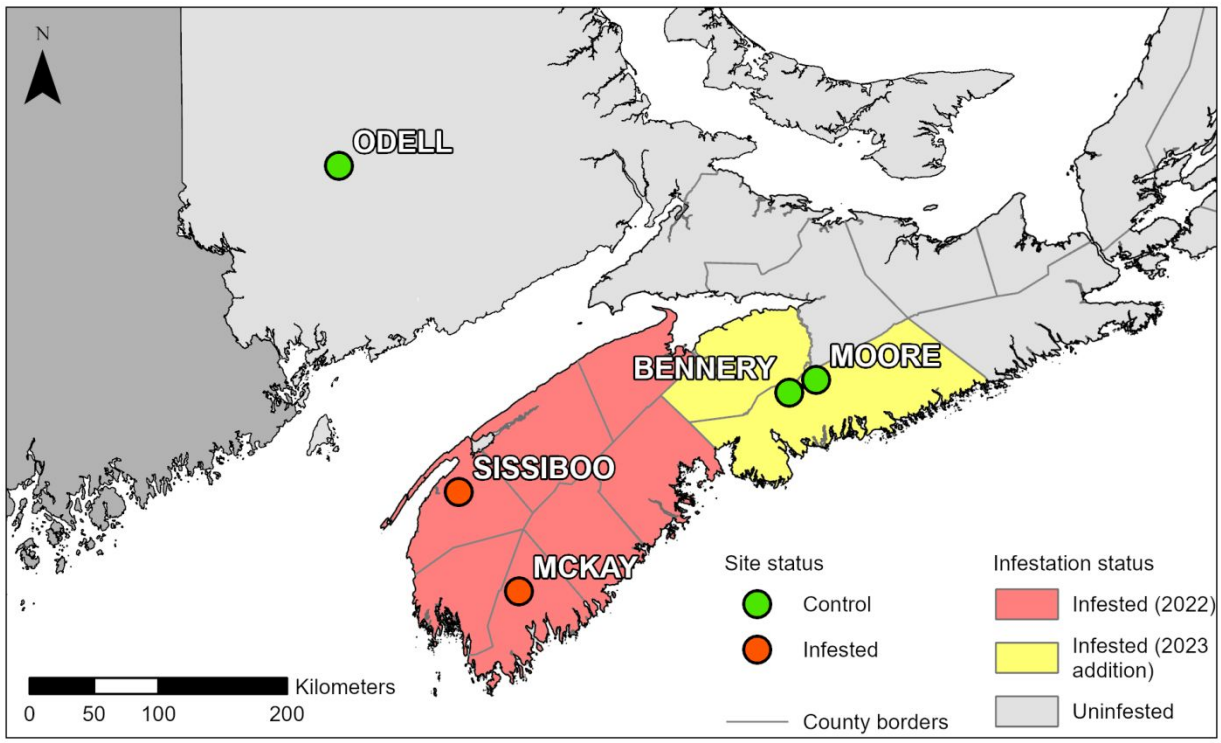


Figure 1. Study sites in Canadian provinces Nova Scotia and New Brunswick. Bennery and Moore were initially sampled as uninfested control sites before HWA infestation was detected in the county shortly after; therefore, these two sites are not included in this study. The figure was created using ArcGIS Pro version 3.3.0 and assembled from the following data sources: *North American Atlas—Political Boundaries* (Commission for Environmental Cooperation (CEC) 2022); and *Nova Scotia Topographic Database—County Boundaries* (Open Data Nova Scotia, Service Nova Scotia, 2023).

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180 **2.2. Salamander sampling**

181 We performed one visual encounter line transect survey for RBS, per plot, during summer 2023
182 (Sissiboo, June 28-29; McKay, June 26-27; Odell, September 5, 8, and 20). The 50 m L x 4 m W
183 transects were established at random direction from plot centre. To ensure consistent effort, a
184 maximum of 30 natural cover objects (e.g., coarse woody debris, rocks, etc.) were searched for
185 RBS along the length of each transect. To standardize abundance estimates, the total number of
186 cover objects were counted on each transect. Salamanders were identified to species, sexed
187 (male, female, or juvenile < 35 mm SVL; Leclair et al., 2006) and measured for total length
188 (nearest 0.1 mm), snout-vent length (SVL) at anterior angle (nearest 0.1 mm), and mass (nearest
189 0.01 g) then released at site. Sex was based on the presence of testes, eggs, and sexually
190 dimorphic external features (Quinn & Graves, 1999). RBS observed, but not captured, were
191 included in the plot's abundance count, but not in measurements, sex, or body condition.
192 Distances of RBS to nearest hemlock tree (nearest 0.25 m) and its treatment status (treated or
193 untreated) were recorded in the Sissiboo and McKay sites. Sampling periods avoided the mid-
194 summer heat, and we expect that seasonal variation in sampling time had minimal impact on the
195 observed abundances. Animal care and ethics was approved by the University of New Brunswick
196 protocol AUP 23009.
197

198 **2.3. Environmental sampling**

199 We measured environmental variables hypothesized to affect RBS abundance: soil pH (nearest
200 0.01 pH), via soil slurry from collected samples; soil moisture (nearest 0.1 %), via direct soil
201 probe (Campbell Scientific HydroSense II); and canopy openness (nearest 0.01 %), via
202 hemispherical canopy imagery. Soil temperature data were also collected; however, the data

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were excluded from all analyses due to a moderate correlation with canopy openness (r = -0.62) and to avoid overfitting the model (see results). Each variable was measured five times per plot in specific locations relative to the transect: the centre, at both ends, and 25 m from the centre perpendicular to the transect. To measure soil pH, we collected soil samples at a depth of 15-20 cm, dried them at 55 °C, and then processed them via soil slurry (10 g of soil to 25 ml of deionized H_2O; 1:2.5 ratio) and a benchtop pH meter (FiveEasy Plus Benchtop FP20). Hemispherical photos were collected with a Sony α NEX-7 24MP Interchangeable Lens Digital Camera fitted with a fish-eye lens (Regent Instruments Inc., Quebec, Canada) and mounted on a 1.1 m tall tripod. We calculated canopy openness using WinSCANOPY software (version 2019a; Regent Instruments Inc., Quebec, Canada). All images were set to the blue channel and processed using a two-step analysis with a 60° Zenith threshold and Standard Overcast Sky (SOC) light attenuation profile. After the first automatic canopy and sky pixel classification, we used interactive masks to remove incorrectly classified portions of the image before completing the analysis.

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2.4. Statistical analysis

We used multiple t-tests to compare environmental variables between plot treatment and infestation status. Soil pH, soil moisture, and canopy openness were compared between treated and untreated infested plots to isolate the impact of treatment, and between infested and uninfested untreated plots to isolate the impact of HWA infestation. We used the Bonferroni correction due to multiple comparisons; with 6 comparisons, the significance level was converted from $\alpha = 0.05$ to $\alpha = 0.0083$.

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Models were fit for RBS abundance and body condition independently. Each model included treatment status and infestation status as fixed categorical explanatory variables (two levels, TRUE/FALSE) and the mean of three environmental variables (soil pH, soil moisture, and canopy openness) were included as fixed continuous covariates. Covariate data was mean centred and scaled to the standard deviation to increase the interpretability of the results due to the varied units/scales. Plot served as the base unit of replication and residual term. We included site as a random factor to acknowledge the inherent variation among sites due to geographical location and to distinguish the experimental design's two levels of replication: the sites, and the plots within the sites.

We computed body condition using the scaled mass index proposed by Peig & Green (2009):

$$\hat{M}_i = M_i\left(\frac{L_0}{L_i}\right)^{b_{SMA}} \quad (1)$$

where M_i and L_i represent the body mass and SVL for an individual RBS, b_{SMA} (3.13; 95% CI = 2.75-3.51) is the scaling exponent estimated by the standard major axis regression of $\ln(M_i)$ on $\ln(L_i)$, L_0 is the mean SVL for all RBS across all three sites, and $\hat{M}_i$ is the predicted body mass for individual i when the linear body measure is standardized to L_0. Note that the calculation of body condition using the scaled mass index results in a measure with the same units as those used for body mass, M_i. Because we were interested in comparing the health of reproductive populations, we calculated the body condition for adults only ($N = 96$) and excluded gravid females.

We used linear mixed effects models to evaluate the impact of imidacloprid application for HWA control on RBS (1) abundance and (2) body condition. We used R (version 4.3.1; R Core Team, 2023) to conduct all statistical analyses and created the mixed effects models using

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248 the *lme4* package (version 1.1.34; Bates et al., 2015). Although only 30 cover objects were lifted
249 per transect, abundance was standardized by scaling counts to the total number of available cover
250 objects. Specifically, the number of salamanders recorded from the first 30 objects was divided
251 by 30 and multiplied by the total number of cover objects. For each of the two models—
252 abundance and body condition—we conducted an initial model subset comparison for random
253 effects using Akaike's Information Criterion corrected for small sample sizes (AICc). We
254 compared the fit of a random intercept/constant slope model against a random intercept/random
255 slope model. Subset models were originally fit using restricted maximum likelihood (REML) but
256 were refit with maximum likelihood (ML) during model comparison using the *anova* function
257 (*lmerTest* version 3.1.3; Kuznetsova et al., 2017). We continued with the best-fitting subset
258 model denoted by the lowest AICc value and then included all fixed effects and continuous
259 covariates. The abundance model favoured a random intercept/random slope model, whereas the
260 body condition model favoured a random intercept/constant slope model. Models with ΔAIC_c
261 values within 2 units of the top-supported model are considered competitive and may not differ
262 substantially in terms of model fit; therefore, we prioritized models with $\Delta AIC_c < 2$ (Arnold,
263 2010). Finally, we averaged all models with $\Delta AIC_c < 2$.

264 Analyses of soil imidacloprid fate data (Edge et al., 2025) found that residues were
265 detected frequently up to 2 m from treated hemlock trees and infrequently beyond 2 m from the
266 tree for up to 2 years post-application. Accordingly, we fit a third model to evaluate whether
267 RBS abundance differed between areas 0–2 m versus >2 m from hemlock tree bases in treated
268 and untreated plots, using these distances to represent a comparison within and outside a zone of
269 potential exposure. Because distance to nearest hemlock was not recorded for RBS caught in the
270 Odell reference site, they were excluded from this analysis. Site and abundance again served as

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271 the random effect and response variable, and treatment and radial distance (two levels: 0–2 m, >2
272 m) served as fixed categorical variables. Environmental continuous covariates were excluded
273 because the data were collected at the plot scale, not the scale of individual capture. We followed
274 the same steps during model building and selection as outlined above—this model favoured a
275 random intercept/constant slope model. Abundance measures used in this analysis were restricted
276 to counts from a total of 30 cover objects flipped during sampling; standardised abundance
277 counts based on the total number of cover objects found per transect/plot could not be used, as
278 distance from nearest hemlock was not recorded for all cover objects.

279 We used the *car* (version 3.1.2; Fox & Weisberg, 2019) and *MuMIn* (version 1.47.5;
280 Bartoń, 2023) packages to conduct mixed-effects modelling and facilitate model diagnostics and
281 selection.

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3. Results

3.1. Do environmental factors vary across treatment and infestation status?

Across all sites and plots, soils were strongly acidic (mean pH ~4.0), with relatively moist forest floor conditions (mean volumetric moisture ~33–36%). Canopy was consistently closed, with low openness values (~15–17% open sky), reflecting the dense hemlock overstory typical of these stands. Overall, baseline environmental conditions were comparable among the reference, infested–untreated, and infested–treated plots (Table 1).

No statistically significant differences were found in soil pH ($t_{(76.21)}$ = 0.75, p = 0.458), canopy openness ($t_{(75.40)}$ = -0.90, p = 0.372), or soil moisture ($t_{(77.92)}$ = 0.47, p = 0.640) when comparing treated versus untreated infested plots within the Sissiboo and McKay sites (Table 1). Similarly, when comparing infested versus uninfested untreated plots within all three study sites (Sissiboo, McKay, and the Odell reference site), no statistically significant differences were found in soil pH ($t_{(33.61)}$ = -2.11, p = 0.043), canopy openness ($t_{(48.16)}$ = -0.55, p = 0.586) or soil moisture ($t_{(52.38)}$ = -1.34, p = 0.185).

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296 Table 1. Experimental design, eastern red-backed salamander capture summary, and
297 environmental covariate comparisons across different site conditions in eastern hemlock
298 dominated (≥80%) forest plots in New Brunswick and Nova Scotia, 2023. Total RBS captures
299 and the adult/juvenile counts are counts from a maximum of 30 cover objects sampled per
300 transect, whereas abundance counts are standardised based on total cover object counts per
301 transect. Values are means ± SD for 200 m^2 transects.

	Site condition		
Habitat characteristic	**Reference**	**Infested/untreated**	**Infested/treated**
Site name(s)	Odell	Sissiboo/McKay	Sissiboo/McKay
Number of plots	4	8	8
Total RBS captures	51	66	82
Adult/Juvenile	31/20	31/35	47/35
Abundance	15.40 (±1.41)	10.10 (±6.09)	16.03 (±5.80)
Body mass (g)	1.16 (±0.37)	0.80 (±0.12)	0.78 (±0.16)
SVL (mm)	41.20 (±4.03)	37.57 (±1.72)	37.55 (±1.94)
Body condition	0.93 (±0.06)	0.85 (±0.07)	0.86 (±0.08)
Soil pH	4.19 (±0.30)	3.90 (±0.20)	3.97 (±0.25)
Soil moisture (% water)	36.43 (±2.50)	33.25 (±5.93)	34.36 (±4.73)
Canopy openness (% open sky)	16.74 (±1.77)	16.15 (±2.67)	15.31 (±2.39)

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3.2. Does abundance vary across treatment and infestation status?

Three models in the selection pool had $\Delta AIC_c < 2$ and were included in the model-averaging set (Table 2). Infestation and treatment were the only two factors included in the top three models: the top model included both, whereas the second and third best models solely included treatment and infestation, respectively. The averaged model indicated that both infestation and treatment had notable effects on RBS abundance, though in opposite directions (Table 3; Figure 2). HWA infestation was associated with a reduction of ~4.5 salamanders per plot relative to uninfested plots. In contrast, imidacloprid treatment was associated with an increase in ~4.6 salamanders per plot compared to untreated plots. Overall, model averaging highlighted infestation and treatment as the two most significant predictors of salamander abundance.

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Table 2. Summary of top models fit using Akaike's Information Criterion corrected for small samples sizes (AIC_c) estimating eastern red-backed salamander abundance in eastern hemlock dominated (≥80%) forest plots in New Brunswick and Nova Scotia, 2023. Models with $\Delta AIC_c < 2$ are included in the average model presented in Table 3 and are noted here with an asterisk (*). *K* represents the number of parameters and *w* represents model weight. N = 20 plots total.

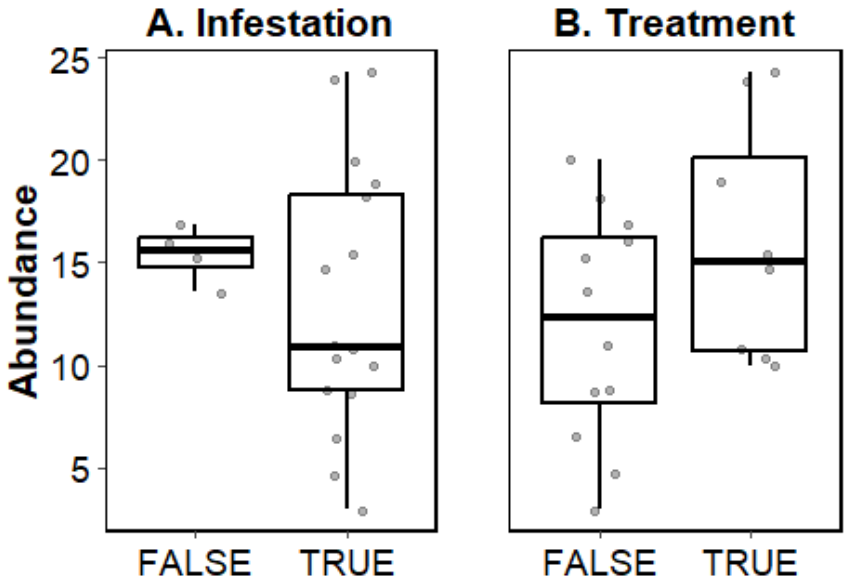
Model	***K***	**AIC_c**	**ΔAIC_c**	***w***	**Deviance**
A ~ infestation + treatment	3	126.26	0.00*	0.25	102.92
A ~ treatment	2	127.58	1.32*	0.13	109.12
A ~ infestation	2	127.87	1.61*	0.11	109.41
A ~ 1	1	128.88	2.62	0.07	114.59
A ~ infestation + treatment + canopy openness	4	129.65	3.39	0.05	100.56
A ~ infestation + treatment + soil pH	4	129.85	3.60	0.04	100.76
A ~ treatment + canopy openness	3	130.15	3.90	0.04	106.82
A ~ treatment + soil pH	3	130.16	3.90	0.04	106.83
A ~ infestation + treatment + soil moisture	4	130.29	4.03	0.03	101.20
A ~ infestation + canopy openness	3	130.49	4.23	0.03	107.16

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Table 3. Fixed effects model summary for the average model predicting eastern red-backed salamander abundance. This model is the average of the top model subset, which includes the three models with $\Delta AIC_c < 2$ (see model selection table; Table 2). Conditional model-averaged coefficients are presented for more accurate estimates of factors that consistently appear in the top model subset. The intercept refers to the uninfested-untreated reference condition.

Fixed effects					
			95% CI		
	Estimate	SE	Lower	Upper	z-value
Intercept	14.47	4.65	4.55	24.39	2.86
Infestation (TRUE)	-4.52	5.81	-17.00	7.96	0.71
Treatment (TRUE)	4.63	7.62	-11.71	20.98	0.56

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327 Figure 2. Effect of HWA infestation and imidacloprid treatment—the two factors included in the
328 averaged model from the top model subset (see Table 3)—on eastern red-backed salamander
329 abundance (N=20 plots total).

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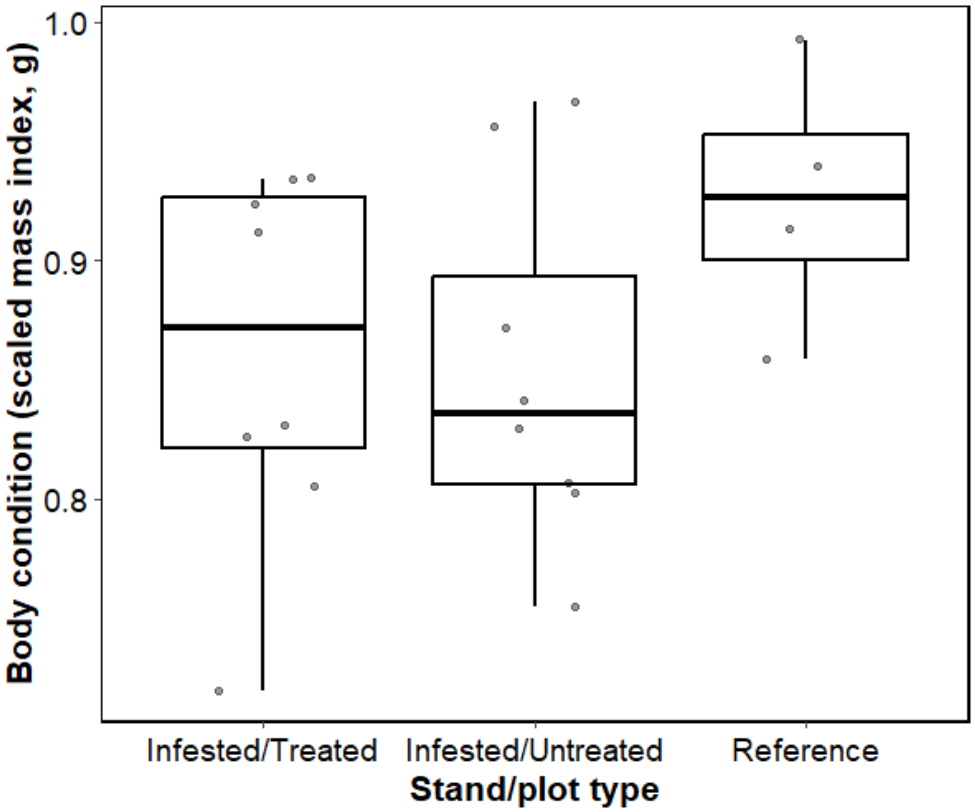
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330 **3.3. Does body condition vary across treatment and infestation status?**

331 Neither treatment status nor the presence of HWA had any detectable effects on RBS body
332 condition (Figure 3). The null (intercept-only) model was the only model with $\Delta AIC_c < 2$ and
333 received 84% of the total model weight (Table 4). All alternative models containing infestation,
334 treatment, or other covariates had substantially higher AICc values ($\Delta AICc \geq 5.7$) and negligible
335 weights, indicating little support for fixed effects on body condition. Estimated mean body
336 condition (non-gravid adults) was 0.91 g (95% CI = 0.86–0.94) in the Odell reference site, 0.89 g
337 (0.87–0.92) in the Sissiboo site, and 0.83 g (0.80–0.86) in the McKay site. When the two HWA-
338 infested plots were combined, mean body condition was 0.86 g (0.83–0.89).

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340 Figure 3. Effect of imidacloprid application on eastern red-backed salamander body condition

341 (N=20 plots total; n=8 for infested/treated and infested/untreated; n=4 for reference site type).

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342 Table 4. Summary of top models estimating eastern red-backed salamander body condition in
343 eastern hemlock dominated (≥80%) forest plots in New Brunswick and Nova Scotia, 2023.
344 Model fit assessed using Akaike's Information Criterion corrected for small samples sizes
345 (AIC_c). *K* represents the number of parameters and *w* represents model weight. N = 20 plots
346 total.

Model	*K*	AIC_c	ΔAIC_c	*w*	Deviance
B ~ 1	1	-37.82	0.00	0.84	-45.32
B ~ infestation	2	-32.14	5.68	0.05	-42.80
B ~ canopy openness	2	-32.10	5.71	0.05	-42.77
B ~ soil pH	2	-31.41	6.41	0.03	-42.07
B ~ treatment	2	-29.53	8.28	0.01	-40.20
B ~ soil moisture	2	-28.50	9.31	0.01	-39.17
B ~ infestation + canopy openness	3	-26.19	11.63	<0.01	-40.47
B ~ infestation + soil pH	3	-26.08	11.74	<0.01	-40.37
B ~ canopy openness + soil pH	3	-24.80	13.02	<0.01	-39.09
B ~ treatment + canopy openness	3	-23.75	14.07	<0.01	-38.03

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348 **3.4. Does abundance vary between zones of potential exposure (0–2 m) and >2 m from**
349 **hemlock trees?**

350 There was no detectable difference in salamander abundance within and outside the zone of
351 potential exposure. The null model was the top-supported model for predicting RBS abundance;
352 no alternative model had $\Delta AIC_c < 2$ (Table 5). Neither radial distance nor treatment influenced
353 salamander abundance (Figure 4). From the null model, abundance was estimated to be 15.58
354 (CI = 11.99–19.16) in the Sissiboo site and 10.92 (CI = 7.34–14.51) in the McKay site. Note that
355 modelled abundances are expressed per plot (per 200 m^2 transect).

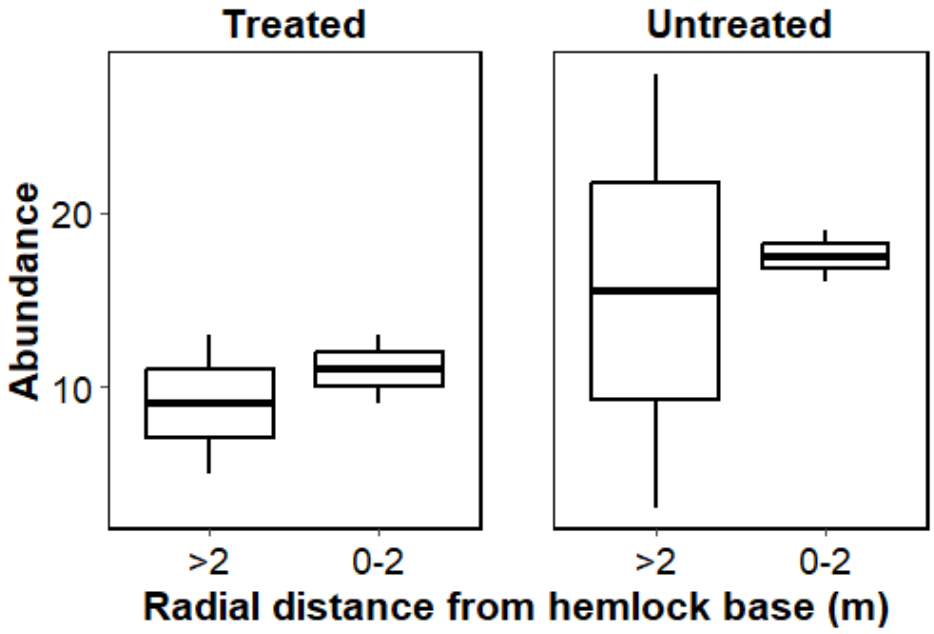
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356 Table 5. Summary of top models estimating eastern red-backed salamander abundance at radial
357 distances from eastern hemlock trees. Model fit evaluated using Akaike's Information Criterion
358 corrected for small samples sizes (AIC_c). *K* represents the number of parameters and *w*
359 represents model weight. N = 20 plots total.

Model	*K*	AIC_c	ΔAIC_c	*w*	Deviance
A ~ 1	1	62.68	0.00	0.71	50.68
A ~ treatment	2	65.28	2.59	0.19	43.94
A ~ distance	2	66.68	3.99	0.10	45.34
A ~ treatment + distance	3	78.70	16.01	<0.001	38.70

360



361

362 Figure 4. Effect of imidacloprid application and radial distance from hemlock base on eastern
363 red-backed salamander space use. Salamander abundance counts within the specified radial
364 distances (0–2 m and >2 m) are totals from n=8 treated and n=8 untreated plot transect surveys
365 (200 m^2 each).

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4. Discussion

The loss of hemlock trees as a result of the invasive HWA along the eastern coast of North America is having profound impacts on ecosystem condition (CFIA, 2017; Emilson et al., 2018). Attempts to reduce the impact and slow the spread of HWA involve the use of systemic insecticides which can pose risk to non-target species. Ultimately the choice to use insecticides compares the risk of HWA-induced ecosystem change against that of risk on non-target impacts from an insecticide (Benton et al., 2016; Cox et al., 2022). Here, we used eastern red-backed salamanders as an indicator species and found that the two most influential factors affecting their abundance were imidacloprid treatment (positive effect) and HWA infestation (negative effect). These findings suggest that although infestation reduces salamander abundance, chemical treatment partially offsets this effect, presumably because treated stands retain healthier hemlocks; however, the exact stand-level mechanisms driving this pattern remain unresolved, as significant differences in canopy structure and other environmental covariates were not observed in our results. We also found that body condition was not influenced by treatment or infestation status, and the zone of potential exposure 0–2 m from the base of treated hemlocks did not impact abundance.

Numerous studies suggest the use of plethodontid salamanders as indicators of forest disturbance due to their small home ranges, strong site fidelity, high densities, and ease of sampling (Welsh JR. & Droege, 2001; Davic & Welsh, 2004; Siddig et al., 2019). Previous work has shown that overstory loss reduces salamander abundances, with population recovery occurring as the understory regenerates (Brooks, 2001; Tilghman et al., 2012; Cox et al., 2022; Leslie et al., 2025). Our findings are consistent with this general pattern: HWA infestation reduced salamander abundance by ~4.5 individuals per plot, whereas imidacloprid treatment

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increased abundance by ~4.6 individuals per plot. However, because canopy openness was not a significant predictor of abundance and no measurable difference was found between treated and untreated stands nor infested and uninfested stands, we avoid attributing these effects to specific structural drivers. Instead, we interpret the opposing effects of infestation and treatment as evidence that salamander abundance responds to overall forest condition, and emphasize that alternative mechanisms not assessed in this study, such as prey availability or litter quality, may explain the effects we observed.

The lack of differences in canopy openness between the infested (treated and untreated) and uninfested sites could be due to insufficient time passing since infestation. There is a correlation of HWA infestation and hemlock mortality patterns with latitude (Orwig et al., 2002). Hemlock mortality can occur is as little as four years in the HWA's southern ranges, where milder temperatures facilitate rapid HWA population growth (McClure, 1991; Skinner et al., 2003). Conversely, at Nova Scotia's northern latitudes, lower temperatures restrict dispersal and infestation intensity (Orwig et al., 2002; Skinner et al., 2003), prolonging the period from initial establishment to hemlock mortality. Local infestations in southwestern Nova Scotia remain active in 2023—6 years post-discovery in 2017—and have likely been active for longer as HWA populations are difficult to detect when populations are low. Local reports document defoliation and decline in parts of the region, but robust canopy-structure assessments are not yet available, making it difficult to characterize the current extent of canopy deterioration. Given this uncertainty, it is plausible that our study sites have not experienced sufficient infestation progression to produce strong, detectable differences in canopy openness. It is possible that the small differences in abundance we observed could increase in size as infestations advance and canopy decline becomes more pronounced.

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We acknowledge that comparisons with the uninfested reference forest must be interpreted cautiously. Our Odell Park control site is geographically distant from the Nova Scotia infested plots, and surveys there occurred later in the season (September vs June). Seasonal shifts in salamander surface activity and body condition are well documented (Feder, 1983), meaning differences between infested and uninfested sites could partly reflect sampling time rather than forest condition. The necessity of selecting a control site outside the known infestation zone reflects the difficulty of detecting HWA in its early stages and highlights the broader challenge of establishing robust reference conditions for invasion studies. Although limited to a single site, Odell Park represents a mature hemlock stand in the region that is not infested, after our originally selected Nova Scotia "control" stands (Bennery and Moore; Figure 1) were subsequently confirmed to contain HWA. We therefore retained this site to provide essential qualitative context on salamander abundance in a reference, uninfested hemlock forest, while interpreting these comparisons with appropriate caution.

Further, our statistical approach is limited by the fact that the untreated category includes both uninfested and infested plots, creating partial dependence between the infestation and treatment factors. We nevertheless retained the factorial structure because it provides direct, mechanistic effect sizes for both HWA infestation and imidacloprid treatment—two distinct ecological processes central to our research questions. While collapsing the data into three site types would remove this dependence, it would also prevent separate estimation of each effect, so we interpret the results with this limitation in mind. Because of these limitations, our primary inference relies on the model-estimated effects of infestation and treatment across all plots, recognizing that the untreated group contains both uninfested and infested stands and that these comparisons should therefore be interpreted with caution.

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Soil moisture and soil pH did not differ between infested (treated and untreated) and uninfested plots and were not significant predictors of abundance, body condition, or abundance within the 0–2 m zone of risk. Precipitation levels preceding sampling days were consistent across all sites, reducing the potential influence of regional weather differences on abundance counts. Terrestrial salamanders are dependent on moisture for cutaneous respiration, and low soil moisture limits surface activity, potentially leading to regional extirpation if moisture levels decrease further (Grover, 1998). At our study sites, soil moisture and pH differences were minimal, suggesting that current forest conditions have not created limiting conditions for RBS activity or physiology.

Multiple mesocosm studies suggest that imidacloprid exposure may have sublethal effects on larval amphibians (Ade et al., 2010; Sweeney et al., 2021), and Crayton et al. (2020) reported bioaccumulation of imidacloprid and its metabolites in salamanders inhabiting streams adjacent to treated stands. In a study in the same treated blocks studied herein, imidacloprid residues were frequently detected within 2 m of treated trees at concentrations that could lead to negative effects on soil arthropods, but not beyond 2 m of treated trees (Edge et al., 2025). In the present study, RBS abundance did not differ significantly between the 0–2 m and >2 m radial distances from treated trees, suggesting that any residues present near the tree bases did not strongly influence salamander space use. While the application method used in this study's treatment plots (basal bark spray) differs from the soil-injection method used in Crayton et al. (2020), empirical comparisons of soil leaching or residue spread between these application types are limited, so we cannot definitively attribute the lack of a spatial effect to reduced environmental mobility of the insecticide under bark-spray conditions.

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Ecosystem change arising from HWA infestation having a negative impact on RBS abundance is consistent with Cox et al. (2022), who reported higher total salamander abundance and diversity in treated than untreated stands, alongside higher canopy closure. Our study corroborates this general pattern, though in our case treatment and infestation status were the sole predictors of abundance and canopy openness measurements revealed no detectable influence. This highlights that imidacloprid treatment may preserve salamander populations by buffering the broader consequences of infestation, even if those changes were not observed in canopy openness or our other tested environmental metrics. No significant differences in environmental variables between infested/uninfested and treated/untreated plots suggests the negative impact of HWA infestation on RBS abundance is likely due to changes in forest structure or forest floor microclimate not reflected in the data of this study. Additional variables that may impact abundance and could be altered via HWA infestation include downed woody debris availability and condition, prey availability, predation pressure, and disease susceptibility.

In summary, our models showed consistent and independent effects of treatment and infestation on salamander abundance: infestation was associated with lower abundance, whereas imidacloprid treatment was associated with higher abundance. Although canopy retention remains a likely mechanism, our results emphasize that treatment and infestation status—rather than direct canopy openness—were the only influential predictors. This highlights both the challenges of attributing effects to specific mechanisms and the importance of treatment in preserving forest-dependent fauna during early invasion stages.

Although RBS populations have remained relatively stable during the early stages of infestation, those located in untreated stands may experience further declines as hemlock mortality progresses. Further, we expect the influence of canopy openness to become more

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480 pronounced as the infestation grows. We recommend that managers prioritise conserving the
481 important ecological aspects of eastern hemlock forests, such as forest canopy and forest floor
482 microclimate, to minimize indirect effects of HWA infestation and canopy decline on RBS and
483 other forest biota. Our results reinforce the value of integrating chemical control with long-term
484 monitoring while long-term biological control strategies are developed to safeguard sensitive
485 species in North America's remaining eastern hemlock forests.

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Author Statements

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During the preparation of this work the authors used ChatGPT-4 OpenAI in order to improve readability. All content was originally written by the authors. After using this tool/service, the authors reviewed and edited the content as needed and take full responsibility for the content of the publication.

Competing interests statement

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

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Data availability statement

The data and R code for this study are available at request. Please email requests to David Fox at david.fox1@unb.ca or davidjfox@dal.ca.

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