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THE EFFECTIVENESS AND APPLICABILITY OF AMPHIBIANS AS INDICATOR SPECIES FOR LONG-TERM MONITORING OF ECOLOGICAL CHANGES IN NEW ENGLAND FORESTS

AHMED SIDDIG

UMASS AMHERST, asiddig@eco.umass.edu

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CHANGES IN NEW ENGLAND FORESTS**

A Dissertation Presented

by

AHMED A.H. SIDDIG

Dissertation submitted to the Graduate School of the University of Massachusetts
Amherst in partial fulfillment of the requirements for the degree of

DOCTOR OF PHILOSOPHY

May 2015

Environmental Conservation

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Approved as to style and content by:

Curtice R. Griffin, Chair

Aaron M. Ellison, Member

Bethany Bradley, Member

Scott Jackson, Member

Curtice R. Griffin, Department Head

Environmental Conservation

DEDICATION

To my home Sudan, the country that I wish to see one day among the well developed nations and its people in peace, harmony and prosperity.

To my parents who always support me to achieve my goals

To my beloved wife Israa and children Ali and Jasmine for being excellent support

To my mentor Aaron who taught me the value of selflessness and smiling when helping people

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ABSTRACT

THE EFFECTIVENESS AND APPLICABILITY OF AMPHIBIANS AS INDICATOR SPECIES FOR LONG-TERM MONITORING OF ECOLOGICAL CHANGES IN NEW ENGLAND FORESTS

MAY 2015

AHMED A. H. SIDDIG, B.Sc. FORESTRY, UNIVERSITY OF KHARTOUM

M.Sc. DESERTIFICATION STUDIES, UNIVERSITY OF KHARTOUM

Ph.D. UNIVERSITY OF MASSACHUSETTS AMHERST

Directed by: Professor Aaron M. Ellison

The objective of this study is to assess the potential of two amphibians species, Eastern Red-backed Salamander (*Plethodon cinereus* (Green)) and Eastern Red-spotted Newt (*Notophthalmus viridescens viridescens* Rafinesque), as indicator species of forest disturbances at Harvard Forest, located in Petersham, Massachusetts, United States. Specifically, I 1) assess the impacts of these focal species to decline of hemlock forests in Harvard Forest; 2) calibrate abundance indices of *P. cinereus* based on artificial and natural objects surveys with a population size estimator based on depletion sampling; and 3) assess the potential of these salamanders as indicator species by developing an objective and multimetric method.

My results showed that decline of Eastern Hemlock (*Tsuga canadensis*) forests due to invasive insects has increased the occupancy of *P. cinereus* but significantly reduced its estimated abundance and detection probability. Similarly, the estimated abundance of *N. v. viridescens* also declined dramatically after hemlock decline. The anticipated transition from forests dominated by *T. canadensis* to mixed-hardwood may alter the abundance and detection probability of both salamander species by up to 50%.

Abundance indices based on both cover board and natural object surveys were able to be calibrated using density estimates of *P. cinereus* derived from depletion (removal) surveys. The cover-board abundance index was eight times higher than the estimated density of *P. cinereus*, whereas the natural object survey was half the size of the density estimator. I introduced the Indicator Species Potential (ISP) index – a multi-metric method to quantify the efficacy of indicator species in classifying sites, monitoring ecological changes, and assessing desired management conditions. When applied to salamanders as potential indicators of changes in forests in Massachusetts, the ISP suggests that *P. cinereus* is a reasonable indicator for ecological change in hemlock stands whereas *N. viridescens* is a potential IS in mixed hardwoods. Overall, the ISP shows promise as a method for summarizing ecological and statistical information about potential IS in a single value.

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CHAPTER 1

INTRODUCTION TO THE USE OF INDICATOR SPECIES APPROACH FOR MONITORING ENVIRONMENTAL CHANGES: ITS IMPORTANCE AND POTENTIAL LIMITATIONS

1.1. Background

Recent reports have indicated that the excessive utilization of natural resources and frequency of occurrence of natural disturbances will continue to increase worldwide, which may subsequently lead to environmental changes such as climatic changes, biodiversity decline, habitat loss and fragmentation, and outbreaks of invasive species (IPCC, 2013). These environmental changes could be viewed as persistent changes in the state and trend of the environment (biotic or abiotic (or both) components) at different time points (e.g. years) as result of human or natural stressors (Spellerberg, 2005). A current relevant example of such changes in the northeastern United States is the response of the decline in eastern hemlock forests (*Tsuga canadensis* (L.)) as result of massive infestations of the invasive insect, the hemlock wooly adelgid (HWA) (*Adelges tsugae*) (Ellison et al., 2005; Orwig et al., 2013). As hemlocks decline, mixed deciduous species (e.g. red maple and various birch species) will replace it, and thus many ecological processes are expected to be changed along with the floral and faunal abundances and diversity.

Amphibian populations are declining worldwide due to climatic changes, habitat loss and alteration, invasive species, diseases, and environmental pollution. The anticipated habitat disturbances in the northeastern US may negatively influence the abundance, diversity and population structure of amphibians populations (Lanoo, 2005;

Becker et al., 2007; Dodd, 2010; Siddig et al., *in prep*-chapter 2). However, regardless the nature of the drivers of these changes and whether they are anthropogenic or natural, worldwide there is a consensus and urgent need for conservation plans based on solid environmental adaptation and mitigation measures.

Among many reasonable strategies, long-term ecological monitoring (LTEM) has been recommended as best way to providing diverse and detailed temporal ecological information that better improve our understanding to build better plans to combat these environmental challenges (Likens, 1989; Morison, 2009; Gitzen et al., 2012; Peters, 2013). In its simplest form, LTEM is defined as *a systematic process of repeated field-based empirical measurements of some ecological state variables that are collected continuously over a reasonably long time (at least 10 years) then analyzed for the purposes of assessing the state of the environment and drawing inferences about changes in state over time* (Yoccoz et al., 2001; Lindenmayer and Likens, 2010). Not surprisingly, LTEM has many significant benefits:

- Detects the change in important ecological process.
- Detects the change in important ecosystems.
- Improves ecological early warning and forecasting.
- Facilitates local, regional and global comparisons.
- Informs management and decision making.
- Helps to establish new facilities (or improves already existing facilities), recruits funding and creates training opportunities.

Despite the appeal of using LTEM, the selection and use of the state variables to monitor (hereafter ecological indicators) has been a challenging task for ecologists and conservation biologists for a long time (Cairns and Pratt, 1993). There also are many other issues related to the application and validity of ecological indicators in real world monitoring schemes. For instance, ecologists and conservation biologists commonly use selected populations as indicator species (IS) for long-term monitoring of environmental changes, assessing the efficacy of management, and providing warning signals for impending ecological shifts (Siddig et al., *in review* EI). This cost-effective and intuitive biological surveillance approach relies on the assumption that the responses of living organisms to local environmental changes indicate the condition of their habitat or even the trends of ecosystems that they are occupying (Cairns and Pratt, 1993; Bartell, 2006; Burger, 2006).

According to Noss (1990), Cairns and Pratt (1993), Carignan and Villard (2002), Dale and Beyeler (2001), U.S. EPA (2002), Doren et al. (2009) and Siddig et al. (*in review* EI), the best indicator species for monitoring ecological changes not only should have known responses to natural and anthropogenic disturbances and a range of variability of these responses, but also (1) strongly and immediately reflect cause-and-effects relationships in ecosystem changes; (2) predict the management interventions and/or alteration; (3) integrate as much as possible the key environmental features (e.g. vegetation type and climate conditions) but also give information about unmeasured variables; (4) have low variability in abundance in space and time and a stable population structure (e.g. sex ratio); (5) be easy to detect and measure in terms of logistics and accessibility; and (6) be socially relevant and of value to local communities.

Terrestrial salamanders, such as eastern red back salamander (*Plethodon cinereus* (Green) - hereafter red backs) and Eastern Red-spotted Newts (*Notophthalmus viridescens viridescens* (Rafinesque) - hereafter red efts) are ecologically important species in northeastern forest ecosystems due to their abundance and position in the middle of the food web (Burton and Likens, 1975; Welsh and Droege, 2001). At the Hubbard Brook Experimental Forest, for example, terrestrial salamanders account for as much biomass as small mammals and twice the biomass of breeding birds (Burton and Likens, 1975). It has been hypothesized that as top-level predators of soil invertebrates who share the leaf litter with them, terrestrial salamanders have important impacts on ecosystem functions such as soil decomposition rates (Hairston, 1987; Wyman, 1998; but see Hocking and Babbitt, 2014; Best and Welsh, 2014). In addition to being important predators of soil fauna, salamanders are also prey items for higher level predators including birds (Coker, 1931; Eaton, 1992) and snakes (Uhler et al., 1939; Arnold, 1982). Their ecological importance along with a number of other traits characteristic of effective indicator species, including: sensitivity to changes in the local environment; site fidelity; low, known variability in space and time; and ease of sampling and detection using simple, non-destructive, and cost-effective methods (e.g. artificial cover boards or visual encounter surveys), make terrestrial salamanders ideal candidates to monitor ecological changes such as those anticipated as hemlock-dominated forests decline and transition into mixed deciduous forests (Welsh and Droege, 2001; Siddig et al., *in prep – chapter 2*).

Despite the popularity and potential of amphibians (or other species) as indicators, the use of indicator species in general has been critiqued in ways that may weaken the

effectiveness of selected indicators. These arguments include: a single population never reflects the whole complexity of the environment; selection criteria for indicator species are subjective; associations between IS and the environmental contexts (goals) are vague; abundant species are not always good indicators; there are influences of other biological interactions at the community level (e.g. predation/parasitism); sampling and methodological difficulties (e.g. detectability issue and sampling protocols) are not accounted for; and the lack of robust quantitative methods to quantify the effectiveness of indicators species; the lack of knowledge about expected effects of future climatic changes on indicator species (Carignan and Villard, 2002; U.S. EPA, 2008; Lindenmayer et al., 2000; Lindenmayer and Fisher, 2003; Lindenmayer and Likens, 2011; Siddig et al., *in review* EI).

1.2. Research objectives

In this study, I assess the effectiveness and applicability of indicator species for long-term monitoring of environmental changes. Specifically, I investigate the potential of two terrestrial salamanders, red backs and red efts, as indicators for monitoring forest changes in New England region where hemlock forests are declining due to infestation of invasive insects. I have the following objectives for this research:

- (1) Assessing and predicting the impacts of losing hemlock forests on occurrence and abundance of both red backs and red efts in Harvard Forests.
- (2) Calibrating red backs abundance indices with population estimators to more accurately and effectively monitor the population dynamics in New England forests.

(3) Developing a multimetric index to assess the potential of terrestrial salamanders as an indicator species for long-term monitoring of forest ecosystem dynamics.

1.3. Study location

This study was conducted at the Harvard Forest, in Petersham, Massachusetts USA (42.47° – 42.48° N, 72.22° – 72.21° W; elevation 215–300 m a.s.l.) within the Harvard Forest Hemlock Removal Experiment (HF-HeRE). HF-HeRE broadly aims to assess the long-term and large scale impacts of decline of eastern hemlock on forest dynamics and biodiversity (Ellison et al., 2010). Full details of the design and analysis of HF-HeRE are provided by Ellison et al. (2010); only the relevant details are given here. The experiment is a replicated block design with two blocks and four treatments within each block. One block is located on a relatively dry ridge and the other in a lower, wetter valley; the two blocks are separated from each other by \approx 500 m. In each block, four \approx 0.81 ha-plots were sited. Two of the plots in each block had different manipulations applied to them – girdling or logging – and two were controls. In the girdled plots, all *T. canadensis* individuals, from seedlings to mature trees, were girdled to kill them and mimic the slow death seen as a function of the adelgid (see also Yorks et al., 2003). In the logged plot, all hemlock trees $>$ 20 cm diameter at breast height (dbh: 1.3 meters above ground), along with merchantable hardwoods, were logged and removed, simulating salvaging as a common management alternative for obtaining economic gain from hemlock stands before the adelgid rendered them economically worthless. In both the girdled and logged plots, *T. canadensis* accounted for at least 65% of the basal area before treatments were imposed in early spring of 2005. Hemlock control plots adjacent

to the logged and girdled plots were unmanipulated (and likewise consisted of $\geq 65\%$ basal area of *T. canadensis*), and hardwood control plots in each block consisted of young hardwoods that represent the expected future of the forests when hemlock totally disappears from the ecosystem.

1.4. Dissertation outline

Chapter 2 describes the responses of red backs and red efts to the decline of hemlock habitats and the subsequent local environmental changes at Harvard Forest. After presenting some background information about the decline in eastern hemlock forests in response to a massive outbreak of HWA, and the subsequent effects on faunal biodiversity at the forest ecosystem. Following this introduction, I stated the objectives and specific research hypothesis. Then, I elaborate on the research methodology, which includes details about the settings, experimental design, amphibian sampling, habitat sampling, and methods of data analysis. I used occupancy modeling based on hierarchical models to incorporate ecological and sampling processes important to amphibians in the area. After presenting the results, the chapter ends with a short discussion and conclusion about suitability of occupancy, abundance, and detection probability as metrics for assessing performance of indicators species.

Chapter 3 describes the calibration of relative abundance indices of red backs based on cover boards and natural objects surveys with the total population size estimated by the depletion method. As background, I briefly review the concept of calibration of abundance indices and its importance in ecology and conservation. In the core part of the chapter I describe a calibration experiment to estimate the population size of red backs at

Harvard Forest, including the design and sampling methods, data collection, and analysis. I conclude with a short discussion about the implication of this method for conservation and monitoring of amphibians populations in the northeastern United States.

Chapter 4 develops a new method for selecting and using indicator species or set of indicators in sites classifications, monitoring ecological changes, and assessing desired management conditions. I introduce the Indicator Species Potential (ISP) index; a multimetric method for quantifying the efficacy of indicator species in ecological monitoring with special emphasis on these both terrestrial amphibians of Harvard Forest. This multimetric index assesses the effectiveness of a focal species as an ecosystem indicator based on five metrics, including occupancy, specificity, detection probability, the strength of the association between a species and environmental correlates (covariates), and the spatial and temporal variability of the abundance for each single indicator species within each group of sites. The index ranges from 0.0 (weak indicators) to 1.0 (powerful indicator species). I conclude this chapter by discussing the advantages, potential applications, and limitations of the ISP index, along with future research needs to enhance the index.

Chapter 5 summarizes the thesis by emphasizing the significance of the research questions in the context of management and conservation of biodiversity, and also discusses the possibility of extrapolation of these various methodological frameworks to other circumstances (e.g. different taxa and ecosystems). Furthermore, I clearly wanted to point out the limitations of the outcomes of this research and suggesting some potential future directions to fill such gaps.

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CHAPTER 2

ASSESSING AND PREDICTING THE IMPACTS OF THE DECLINE OF EASTERN HEMLOCK (*TSUGA CANADENSIS*) STANDS ON THE OCCURANCE AND ABUNDANCE OF TERRESTERIAL AMPHIBIANS IN NEW ENGLAND FORESTS

2.1 Abstract

Forest disturbances such as an outbreak of invasive pests can cause devastating damage to many unique habitats and may directly contribute to the loss of biodiversity in these forests. Throughout its range, the foundation tree species *Tsuga canadensis* (eastern hemlock) is declining due to infestation by the nonnative insect *Adelges tsugae* (hemlock woolly adelgid). The decline and eventual elimination of hemlock from northeastern US forests is expected to cause changes in forest structure and function, and in the assemblages of associated species, including amphibians. I assessed changes in occupancy, relative abundance, and detection probability of two species of terrestrial salamanders, *Plethodon cinereus* (red back salamander) and *Notophthalmus viridescens viridescens* (eastern red spotted newt), to the experimental removal in 2005 of *T. canadensis* at Harvard Forest. Salamanders were sampled under cover boards and using visual encounter surveys in replicate 0.81-ha plots. Sampling was done between May and July in 2004 (pre-manipulation), and between May and July in 2005, 2013, and 2014 (post-manipulation). In 2004, occupancy of *P. cinereus* was 35% lower in stands dominated by *T. canadensis* than in associated mixed-hardwood control stands, whereas detection probability and estimated abundance of *P. cinereus* were, respectively, 60% and 100% greater in *T. canadensis* stands. Estimated abundance of *N. v. viridescens* in 2004

was 50% higher in *T. canadensis* stands. Removal of the *T. canadensis* canopy by girdling or logging increased the occupancy of *P. cinereus* but significantly reduced its estimated abundance and detection probability. Similarly the estimated abundance of *N. v. viridescens* also declined dramatically after canopy manipulations. The anticipated transition from forests dominated by *T. canadensis* to mixed-hardwood may alter the abundance and detection probability of both salamander species by up to 50%. Overall I provided glimpses into three possible futures for populations of terrestrial salamanders in current Hemlock-dominated stands in central New England. The first two views are of what populations may look like ten years following two methods of hemlock loss – direct mortality caused by the adelgid and pre-emptive salvage logging. The third is a vision of what populations of salamanders will look like fifty to seventy years from now, once former hemlock-dominated stands have become mid-successional mixed deciduous stands typical of the region. My results suggest all three future forests will have fewer salamanders which suggest establishing better monitoring and conservation programs.

Keywords: *abundance, Adelges tsugae, detection probability, Harvard Forest, indicator species, monitoring, Notophthalmus viridescens, occupancy, Plethodon cinereus, Tsuga Canadensis*

2.2 Introduction

Foundation species are species that control the distribution and abundance of associated species and modulate important ecosystem processes (Dayton, 1972; Ellison et al., 2005a). In terrestrial environments, foundation species tend to be large, abundant, occupy basal positions in local food webs, and control ecosystem processes and dynamics principally through non-trophic interactions (Baiser et al., 2013). *Tsuga canadensis* (L.) Carrière (Eastern hemlock) is a foundation tree species in northeastern North American forests (Ellison et al., 2005a; Orwig et al., 2013; Ellison et al., 2014). Throughout its range, stands dominated by *T. canadensis* are both structurally and functionally different from surrounding mixed deciduous stands (Orwig et al., 2002; Ellison et al., 2005a). Hemlock-dominated stands are dark, cool, and moist (Rogers, 1980; Benzinger, 1994; D'Amato et al., 2009; Lustenhouwer et al., 2012); have acidic, nutrient-poor soils with slow rates of nutrient cycling (e.g. Orwig and Foster, 1998; Orwig et al., 2013), and generally species-poor assemblages of associated plants and animals (e.g. Ellison et al., 2005b; Rohr et al., 2009; Orwig et al., 2013).

Tsuga canadensis is declining throughout its range for two reasons. First, the nonnative insect *Adelges tsugae* Annand (Hemlock woolly adelgid), introduced to the US from Japan in the early 1950s, kills hemlock seedlings, saplings, and mature trees (Ellison et al., 2010). Second, many landowners and land managers have been logging *T. canadensis* prior to the arrival of the adelgid (Orwig et al., 2002; Foster and Orwig, 2006). In New England, as *T. canadensis* declines or is logged out, it has been replaced by deciduous species including *Acer rubrum* L. (Red maple) *Betula lenta* L. (Black

birch), and *Quercus rubra* L. (Northern red oak) (Orwig and Foster, 1998; Orwig, 2002; Brooks, 2004).

Although the faunal assemblages of *T. canadensis* stands generally have fewer species than nearby mixed hardwood stands, the former provide habitat for a number of associated arthropods (Ellison et al., 2005b; Rohr et al., 2009), birds (Tingley et al., 2002), and salamanders (Mathewson, 2009; Mathewson, 2014). Although the loss of *T. canadensis* from eastern North American forests is predicted to result in a cascade of associated faunal changes (Ellison et al., 2010; Ellison et al., 2014), less is known about how different animals will respond to the different ways in which *T. canadensis* is lost from stands that it currently dominates.

Terrestrial salamanders such as *Plethodon cinereus* (Green) (Eastern red-backed salamander; henceforth “red-backs”) and the juvenile phase of *Notophthalmus viridescens viridescens* Rafinesque (Eastern red-spotted newt; henceforth “red efts”) are abundant and centrally located in food webs of northeast forest ecosystems (Burton and Likens, 1975; Welsh and Droege, 2001). For example, at the Hubbard Brook Experimental Forest, terrestrial salamanders accounted for as much biomass as small mammals and twice the biomass of breeding birds (Burton and Likens, 1975). As predators of soil invertebrates, salamanders also have important effects on soil decomposition rates (e.g. Hairston, 1987; Wyman, 1998; but see Best and Welsh, 2014; Hocking and Babbitt, 2014). Red-backs also are prey for snakes (Uhler et al., 1939; Arnold, 1982) and birds (Coker, 1931; Eaton, 1992) although the toxins in the skin of red efts make them unpalatable to most potential predators (Hurlbert, 1970; Uhler et al., 1939). Their abundance, site fidelity, and ecological importance suggest that terrestrial salamanders

are ideal indicators of ecological changes in many systems, including forests (Welsh and Droege, 2001; Best and Welsh, 2014).

Here, I ask how experimental removal of *T. canadensis* either through logging or simulated infestation by the adelgid (Ellison et al., 2010) affects the detectability, occupancy, and abundance of red-backs and red efts over a ten-year period. I also examine plausible cause-and-effect relationships between hemlock decline and associated changes in habitat characteristics on salamander abundance. My results provide additional insights into the use of salamanders as indicator species for ecological changes in eastern North American forests.

2.3 Materials and Methods

2.3.1 Study site and experimental design

I studied red-backs and red efts within the Harvard Forest Hemlock Removal Experiment (HF-HeRE), located at the Harvard Forest Long Term Ecological Research Site in Petersham, Massachusetts, USA (42.47° – 42.48° N, 72.22° – 72.21° W; elevation 215–300 m a.s.l.). HF-HeRE was designed to assess long-term, large scale effects of the decline and loss of *T. canadensis* on forest dynamics and biodiversity (Ellison et al., 2010). Full details of the design and analysis of HF-HeRE are given in Ellison et al. (2010); key details are repeated here. HF-HeRE is a replicated block design with two blocks and four treatments within each block. Both blocks are located within the \approx 150-ha Simes Tract of the Harvard Forest (Ellison et al., 2014); The northern “ridge” block and the southern “valley” block are separated from each other by \approx 500 m. Each block contains four \approx 90 × 90-m (\approx 0.81) ha-plots. Three of the plots in each block were initially

dominated ($> 65\%$ basal area) by *T. canadensis* whereas the fourth was dominated by young (< 50 -year-old) mixed hardwoods. Plots were sited in 2003; in 2005, canopy manipulations were applied to two of the *T. canadensis*-dominated plots in each block. One of the plots was “girdled”: the cambium of all *T. canadensis* individuals, from seedlings to mature trees was cut through with chainsaws or knives to kill the trees slowly but leave them standing in place, as would happen following adelgid infestation (see also Yorks et al., 2003). The other manipulated plot was “logged” in a simulation of a commercial pre-emptive salvage cut: all *T. canadensis* trees > 20 cm DBH, along with merchantable *P. strobus* L. (White pine) and hardwoods (primarily *Q. rubra*), were logged and removed. The remaining *T. canadensis*-dominated plot in each block was left as a control to await adelgid infestation (which occurred in 2010: Kendrick et al., 2015), and the plot dominated by mixed-hardwoods represented the expected future condition of the forest after *T. canadensis* has been lost from the landscape.

2.3.2 Amphibian sampling

I counted red-backs under artificial cover objects (ACOs) on three (in 2004), five (in 2005 and 2014), or two (in 2013) sampling dates from May through July in 2004 (before canopy treatments were applied) and 2005 (post-treatment but before the infestation of the plots by the adelgid), and from June through July 2013 and 2014 (post-treatment and during the now ongoing adelgid infestation). During the second and third weeks of September 2003 four $1 \times 0.25 \times 0.02$ -m rough-sawn *T. canadensis* boards (ACOs) were placed at randomly selected points along 75-m transects each of the eight HF-HeRE plots. These ACOs were removed in 2006; new ACOs were put in the plots in

early May 2013 and sampled in mid-June and July of 2013 and 2014. All ACOs were placed at least 15 m from the edge of the plots. In all years, we usually sampled all of the ACOs in all of the plots on the same day; if not, at least one plot of every treatment type was always sampled on the same day. The 1–2-week interval between sampling individual ACOs was sufficient to ensure that repeated sampling did not impact detection probability (Marsh and Goicocchea, 2003).

Red efts were sampled using visual encounter surveys only in 2004 (pre-treatment) and 2014, both concurrently with sampling ACOs for red-backs. The 2004 visual-encounter survey was done along two 75×1 m transects randomly-positioned > 15 m from edge of each plot (Mathewson, 2014). The 2014 survey was done along two 60×1 -m parallel transects far 30 m from each other. For 2013 and 2014 data collection scheme see figure 2.1.

All amphibian sampling methods were approved by Harvard University's Institutional Animal Care and Use Committee (File 13-02-144 – last updated June 02, 2014).

2.3.3 Habitat characteristics

Habitat characteristics and local environmental conditions, including understory vegetation and relative humidity were used as covariates in the analysis. Understory vegetation (seedling density and percent cover) has been measured annually in two sets of five 1-m² plots equally spaced along 30-m transects in each canopy manipulation plot (detailed methods and data in Orwig et al., 2013). Relative humidity data were acquired from the Fisher metrological station at Harvard forest.

2.3.4 Data availability and statistical analysis

All raw data from this study are available from the Harvard Forest Data Archive (<http://harvardforest.fas.harvard.edu/data-archive>), datasets HF075 (2004, 2005 salamander data), HFXXX (2013, 2014 salamander data), HF106 (understory vegetation) and HF001-10 (relative humidity).

All statistical analyses were done using R version 3.1.2 (R Core Team, 2013). Based on single-species, single-season occupancy modeling that developed by Mackenzie et al. (2002) and implemented in the R's unmarked package (Fiske and Chandler, 2011), I estimated detection and occupancy probabilities of red-backs and red efts. Site covariates in the occupancy and detection probability models included block location (ridge, valley) and relative humidity on the days when salamanders were sampled. Although I could estimate detection and occupancy for red-backs in all four sampling years, I could only estimate these quantities for red eft for 2014, as sample sizes for this species were inadequate in 2004 ($N = 4$), 2005 ($N = 0$), and 2013 ($N = 0$). My estimate of abundance was based on counts of individuals and whether observed under ACOs for red-backs or within the area of the strip transect ($60 - 75 \text{ m}^2$) for red efts. Raw counts of red backs (average number individuals/ACO) was converted to density/ m^2 given the area of each ACO is 0.25m^2 .

I examined potential relationships among decline of *T. canadensis*, other elements of habitat change, and abundance of salamanders using analysis of covariance (ANCOVA). Because habitat variables (understory vegetation) were strongly correlated,

I first calculated a multivariate (principal component) score for them using the `prcomp()` function in R. I then used the ANCOVA model described in Ellison et al. (2010) and Orwig et al. (2013) to test for effects of canopy manipulation, habitat characteristics (principal axis scores – first axes, PC1), climate (relative humidity), and sample year (reflecting pre-treatment, post-treatment but pre-adelgid, and post-treatment and post-adelgid) on red-backs abundance. For red efts I used the same ANCOVA model except it only assessed the year and treatments, given the data deficit for other terms. Summary of the methodology of data collection and analysis is presented in Table 2.1.

2.4 Results

Prior to applying the treatments, the occupancy probability of red-backs in the hemlock-dominated plots (what would become the logged, girdled, and hemlock control plots) was lower (mean = 76%, range = [0.65 – 0.82]) than in the hardwood plots (1.0) (Figure 2.3). In contrast, detectability of red-backs was twice as high in the Hemlock-dominated plots (mean = 57 %, range = [0.55 – 0.63]) as in the hardwood plots (0.30) (Figure 2.3). The average relative abundance of red-backs in the Hemlock-dominated plots was slightly higher in the hardwood plots (2.0 individuals/m² vs. 1.2 individuals/m² respectively) (Figure 2.3).

One year after the canopy-manipulation treatments had been applied, the occupancy probability of red-backs had substantially increased – to almost 100% in all plots. The associated standard errors of these occupancy estimates were 0.03 in Hemlock control plot and 0.05 in the hardwood plots, but they were wider in the girdled and logged plots (0.13 and 0.29, respectively). In contrast, the detection probability of red-backs

significantly dropped in all plots following canopy manipulation, although the magnitude of change was lower in the hardwood plots. The relative abundance of red-backs also dramatically decreased in all plots after the manipulations – from 2.0 to 0.12 individuals/m² in the logged plots, from 2.4 to 0.6 individuals/m² in the girdled plots, and from 1.6 to 0.2 individuals/m² in the hemlock control plots, and from 1.2 to 0.2 individuals/m² in the hardwood control plots.

Ten years after the treatments, red-backs still occupied nearly 100% (SE = 0.53) of the logged plots (Figure 2.3), 70% of the girdled and hardwood plots, and 62% of the hemlock control plots. The detection probability was about the same in the logged plots ten years following canopy treatment as it was one year following canopy treatment. Over the same time, detection probability decreased three-fold in the girdling plots (12% to 4%) but increased almost nine-fold in the hemlock control plots (from 7% to 62%) and five-fold in the hardwood plots (from 5% to 25%). Likewise, the relative abundance of red-backs increased slightly in the logged plots (0.12 individuals/m² to 0.2 individuals/m²), seven-fold in the hemlock control plots (from 0.2 individuals/m² to 1.4 individuals/m²), and four-fold (from 0.2 individuals/m² to 0.84 individuals/m²) in the hardwood stands. Overall, the current relative abundance of red-backs in Hemlock control plots is 1.6 individuals/m², five times higher than in the girdled plots, seven times higher than in the logged plots, and about two times higher than in the hardwood plots.

Analysis of covariance revealed that there were no significant direct effects of canopy treatment, understory density, understory cover and relative humidity on the relative abundance of red-backs (Table 2.2), but the interaction of canopy treatments with

relative humidity significantly affected the relative abundance of red-backs ($F_{1,15} = 4.05$, $P < 0.05$).

Similar to abundance pattern of red-backs the relative density/ m^2 of red efts before treatments was 0.07 individuals/ m^2 (range = [0.05 - 0.08]) in the hemlock-dominated plots and 0.04 individuals/ m^2 in the hardwood stands (Figure 2.4). However, the precision of the density estimates as indicated by the standard errors, was higher in the hardwood plots (SE = 0.002) than in the hemlock plots (SE=0.005).

Ten years after canopy manipulations, the relative density of red efts was significantly lower in the logged and girdled plots ($F_{3,8} = 4.07$, $P = 0.04$; Figure 2.4 and Table 2.2). However, all plots were occupied by red efts equally and fully. Detection probability in the Hemlock control sites was twice that of the logged and girdled plots and 1.5 times greater than in the hardwood stands (Figure 2.4).

Similar to red-backs, analysis of covariance revealed no significant direct effects of canopy treatments on the abundance of red efts ($F_{3,8} = 0.66$, $P > 0.05$) or interactions between year and treatment ($F_{3,8} = 2.22$, $P > 0.05$, but direct effects of sample year ($F_{1,8} = 22.22$, $P < 0.01$; Table 2.2 and figure 2.4).

2.5 Discussion and Conclusion

This study provides glimpses into three possible futures for populations of terrestrial salamanders in current Hemlock-dominated stands in central New England. The first two views are of what populations may look like ten years following two methods of Hemlock loss – direct mortality caused by the adelgid and pre-emptive salvage logging. The third is a vision of what populations of salamanders will look like

fifty to seventy years from now, when once former Hemlock-dominated stands have become mid-successional mixed deciduous stands typical of the region. My results suggest all three future forests will have fewer salamanders.

It appears that ten years following the loss of Hemlock via either adelgid infestation or pre-emptive salvage logging, the relative abundance of both red-backs and red efts will be significantly lower, and that the method of Hemlock loss will have little impact on the severity of the decline of either species (red-backs experienced a decline of 90% in logged plots and 88% in girdled plots; red efts experienced a decline of 75% in logged plots and 67% in girdled plots – numbers based on comparisons between years 2004 and 2014) (Figure 2.3 and Figure 2.4). These declines are greater than any declines seen in a meta-analysis of twenty-four studies examining the effect of timber removal on the relative abundance of terrestrial salamanders (Tilghman et al., 2012). However, Hocking et al., (2013) observed similar declines, approximately 85%, to those we observed, in group cuts, patch cuts, and clear-cuts. The decline in the relative abundance of red-backs was immediate and drastic in the logged plots. In contrast, a decline in the relative abundance of red-backs in the girdled plots was not seen in the first year following treatments, which is not surprising as little foliar loss was seen in the first months following the girdling treatment (Orwig et al., 2013).

My results suggest that the relative abundance of red-backs will take approximately fifty more years to recover to the pre-logged relative abundance if the 40% rate of increase observed between one-year post-treatment and ten years post-treatment continues (Figure 2.3c). The partial recovery in the logged plots already seen in the logged plots could have been due to availability of dense understory vegetation in these

plots, although it does not show significant statistical effects on salamander abundance (Tables 2.2). It is also possible that no recovery has occurred in the logged plots at all, as the relative abundance of red-backs in the logged plots was 40% lower than in the control plots in 2005, but was 86% lower in 2014 (Figure 2.3c). Further sampling in the girdled plots would be required to determine whether red-backs are increasing or decreasing in the these plots, since the treatment had little effect in 2005, and there is therefore no way to tell whether relative abundance is still declining or whether it reached its lowest point somewhere between one year post-treatment and ten years post-treatment.

Hardwood control plots provide a glimpse into what the relative abundance of red-backs and red efts will be like between fifty and seventy years following the loss of Hemlock. It appears as if populations will never return to the levels seen in Hemlock-dominated stands prior to infestation, perhaps declining by as much as 50%. Given the significant contribution terrestrial salamanders make to the overall vertebrate biomass in forests, decline in the relative abundance of terrestrial salamanders may impact populations of vertebrates that prey on them.

Declines in the relative abundance of both species of salamanders in both the Hemlock and hardwood control plots between 2004 and 2014 suggests that populations of both species may be experiencing declines caused by factors other than logging or simulated adelgid infestation. This finding coincides with a far-ranging study assessing populations of *Plethodontidae* species in twenty-two states in eastern North America that reported declines in the relative abundance of 180 out of 205 populations, with only 22 of these being attributed to destruction of habitat (Highton, 2005). The author of that paper speculated that soil acidification is one potential explanation for these declines (Highton,

2005). Increases in temperatures or changes in precipitation are other potential causes. Another potential explanation, at least in the Hemlock control plots, is that adelgid infestation is causing changes in habitat that may be impacting the relative abundance of each species. An alternative explanation may be that sampling in control plots was not conducted far enough away from logged and girdled plots so as to eliminate edge effects. Hocking et al. (2013) report the abundance of salamanders can be impacted up to 34m into the surrounding forest.

Surprisingly, I failed to detect any impacts from the treatments on habitat characteristics; one reason for this may be that sample sizes were too small. Future investigations should consider using larger sample sizes and different statistical technique(s), such as structured path analysis explicitly suited for revealing cause-effect relationships between the disturbance-habitat and amphibians species.

I successfully assessed the changes in the occupancy, relative abundance, and detectability of red-backs and red efts before and after the arrival of the adelgid in a New England forest. This seems important not only by showing their utility as state variables in monitoring salamander populations but also by improving our understanding about sampling efforts needed for salamanders given this hemlock decline(Mackenzie et al., 2002). Since the habitat factors affected by the decline of hemlock are not clear yet, I suggest that future efforts should consider monitoring these set of variables in finer scales (e.g. exact salamander sampling locations) with larger sample sizes so that more robust conclusions may be reached((Mackenzie et al., 2002 and 2003)).

Salamanders are sensitive to forest disturbances and because of their position in the middle of the food web as both prey and predator, they are thought to be efficient and

effective indicator species that can be used to monitor local environmental changes (Welsh and Droege, 2001; Best and Welsh, 2014). Consistent with this, both species showed significant responses to hemlock treatments that simulated the habitat disturbances. Certainly this ability to quickly predict these changes in the physical environment are good signs for the potential of both red-backs and red efts as indicator species for monitoring ecosystem changes that are consistent with the recommended criteria by Welsh and Droege (2001), Carignan and Villard (2002), Welsh et al. (2008), Lindenmayer and Likens (2011) and Siddig and Ellison (*in prep*-chapter 4).

Of course the use of indicator species in ecological monitoring needs to be achieved through careful identification of set of metrics that best describe the potential of the candidate indicator species and their relevance to a particular ecosystem context (Dale and Beyeler, 2001; Siddig et al., *in review*- JE). Along this line it is remarkable that the hemlock forest disturbances as simulated by canopy manipulations can be predicted well by the substantial changes in abundance and detection probability of these amphibians' species. Abundance of an indicator species is usually considered the best predictor of the quality of its habitat (Urban et al., 2012). Dufrene and Legendre (1997) have demonstrated that abundance is a major component of computing the indicator value index (IndVal) that measures the species importance in the ecosystem. As demonstrated by Urban et al. (2012), accurate estimates of the IndVal index of a given species in a certain habitat type(s) are strongly related to its detection ability, which translates into abundance (i.e. specificity) and occupancy (i.e. fidelity) of the indicator species. Both salamanders species have demonstrated a high level of occupancy in both habitat types – hemlock stands as current and hardwood as future habitat – which suggests their

persistence and stability, and hence their potential as indicator species in long-term monitoring ecosystem changes of these forests in New England (Mackenzie et al., 2002 and 2003; Siddig and Ellison, *in preparation* – chapter 4). Overall, the future changes in these species given this baseline information may indicate to some changes in the entire forest ecosystem. Therefore, long-term, systematic and integrated assessment to red-backs and red efts populations, together with selected relevant habitat variables in focal areas across New England (e.g. Harvard Forest), not only will be a cost-effective and intuitive approach to inform conservation and environmental management decisions but also may anticipate predicted environmental changes in the region.

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Table 2.1: Summary of the methodology of assessing the impacts of Hemlock decline on terrestrial amphibians in the Harvard Forest Hemlock Removal Experiment (HF-HeRE), including local climate conditions during summers of 2004, 2005, 2013 and 2014.

Species	Year / Period	Sampling method	Sampling Occasions	Average relative humidity % during the sampling period	Variables assessed
<i>P. cinereus</i>	2004 May - July	CB	3	68	Abundance, occupancy and detectability
	2005 May - July	CB	5	69	Abundance, occupancy and detectability
	2013 May - June	CB	2	84	Abundance, occupancy and detectability
	2014 May - July	CB	5	74	Abundance, occupancy and detectability
<i>N. viridescens</i>	2004 May - July	VES	3	68	Abundance
	2014 May - July	VES	7	74	Abundance, occupancy and detectability

- CB = hemlock Cover board 1× 0.25 ×0.02 m; VES = visual encounter surveys along 1× 60 m strip transects.

Table 2.2: The results of Analysis of Covariance (ANCOVA) that assessed the direct effects of canopy treatments, local habitat characteristics changes, and interaction terms on the abundances of *P. cinereus* and *N. viridescens* in Harvard Forest

<i>P. cinereus</i> ~ Treatment * (density + cover + RH) + year					
	DF	Sum Sq.	Mean Sq.	F-value	P-value
Treatment	3	5.516	1.839	1.516	0.2510
Year	1	4.049	4.049	3.339	0.0876
Understory density	1	1.457	1.457	1.201	0.2904
Understory percent cover	1	4.242	4.242	3.498	0.0811
Relative humidity	1	0.157	0.157	0.129	0.7244
Treatment : Understory density	3	2.780	0.927	0.764	0.5317
Treatment : Understory % cover	3	2.790	0.930	0.767	0.5302
Treatment : relative humidity	3	14.766	4.922	4.059	0.0269 *
Residuals	15	18.191	1.213		
<i>N. viridescens</i> ~ Treatment * year					
	DF	Sum Sq.	Mean Sq.	F-value	P-value
Treatment	1	0.002500	0.0025000	22.22	0.59572
Year	3	0.000225	0.0000750	0.667	0.00151 **
Treatments: year	3	0.000750	0.0002500	2.222	0.16306
Residuals	8	0.000900	0.0001125		

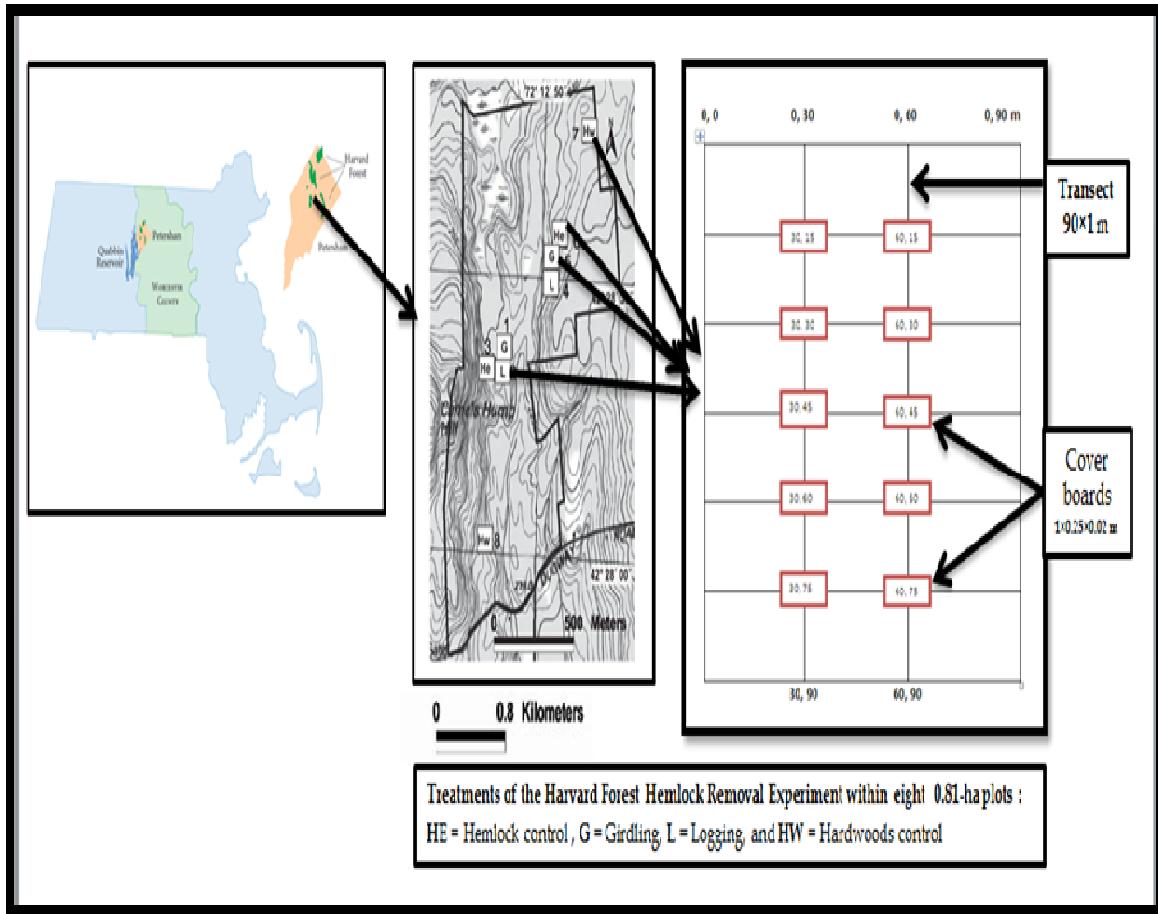


Figure 2.1: Study locations, treatments of the Harvard Forest Hemlock Removal Experiment (HF-HeRE) and sampling transects within HF-HeRE in the Simes Tract at the Harvard Forest, Massachusetts, USA.

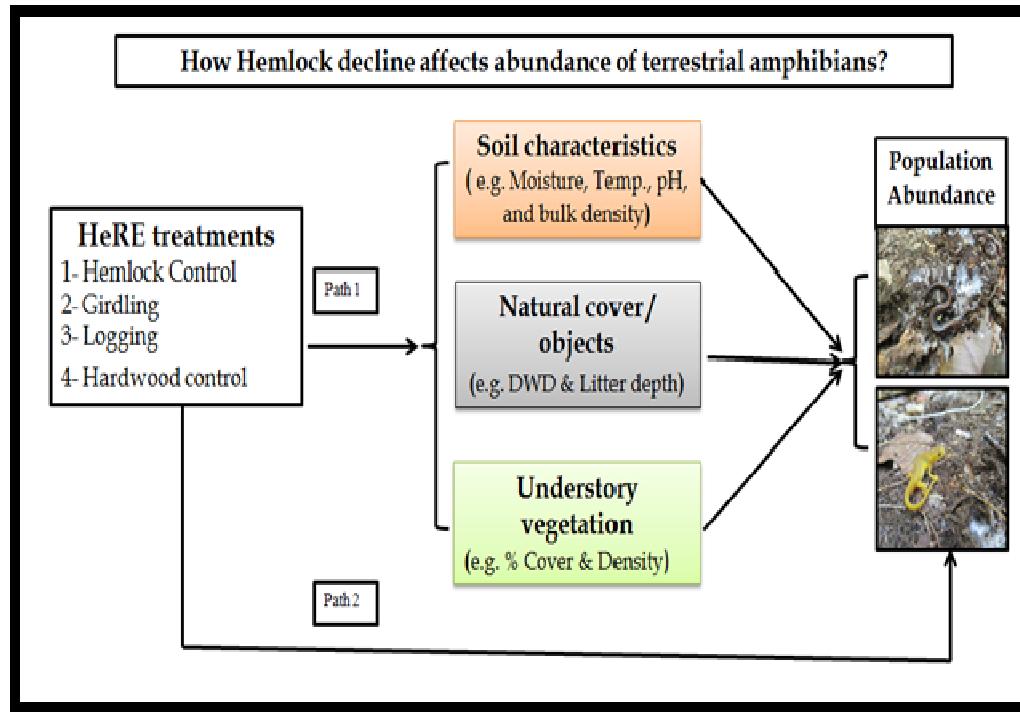
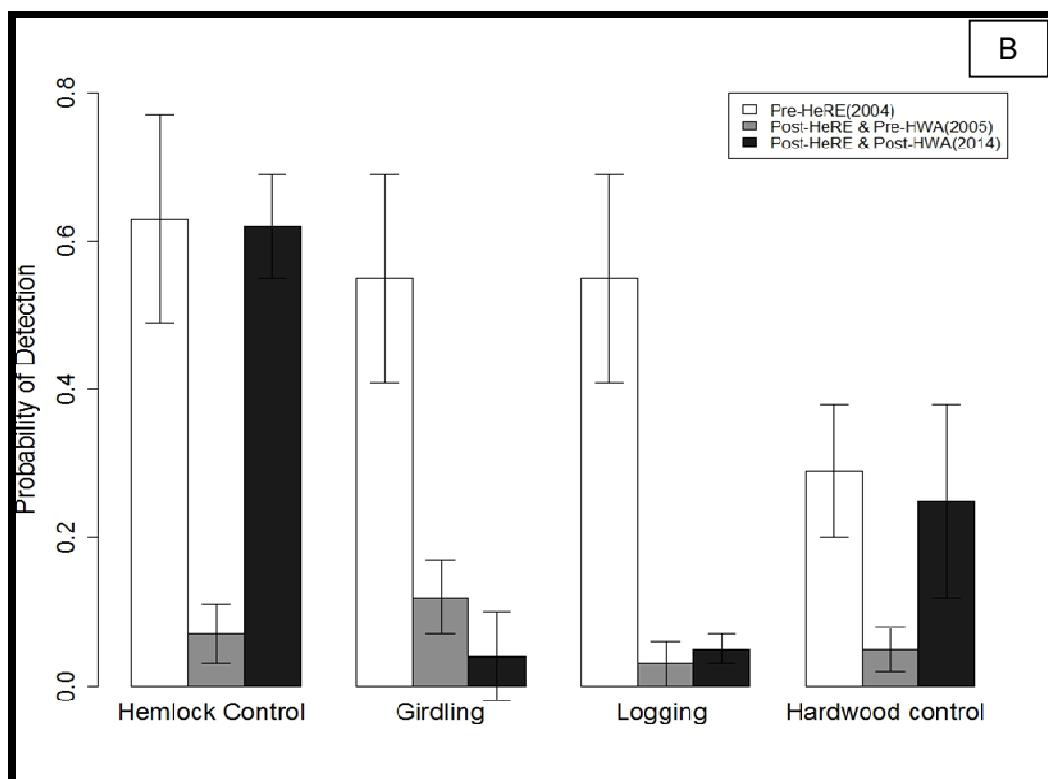
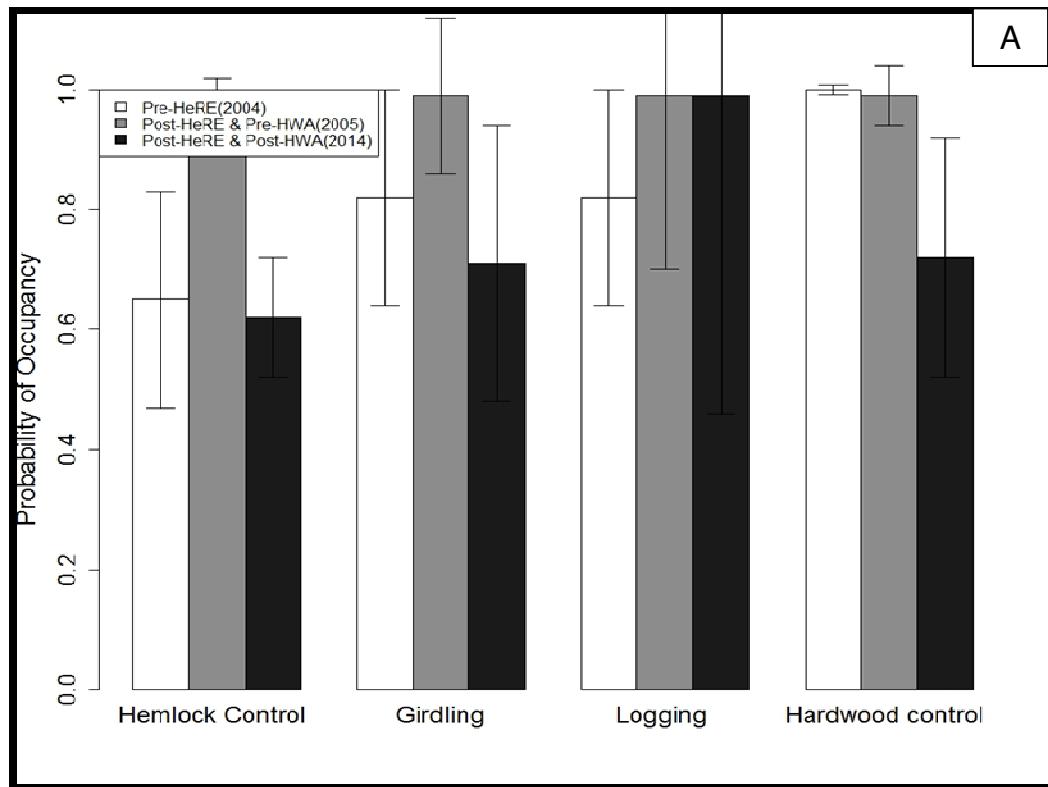


Figure 2.2: Conceptual model of causal linkages between the HF-HeRE treatments, habitat /environmental variables and *P. cinereus* and *N. viridescens* occupancy, detectability, and abundance



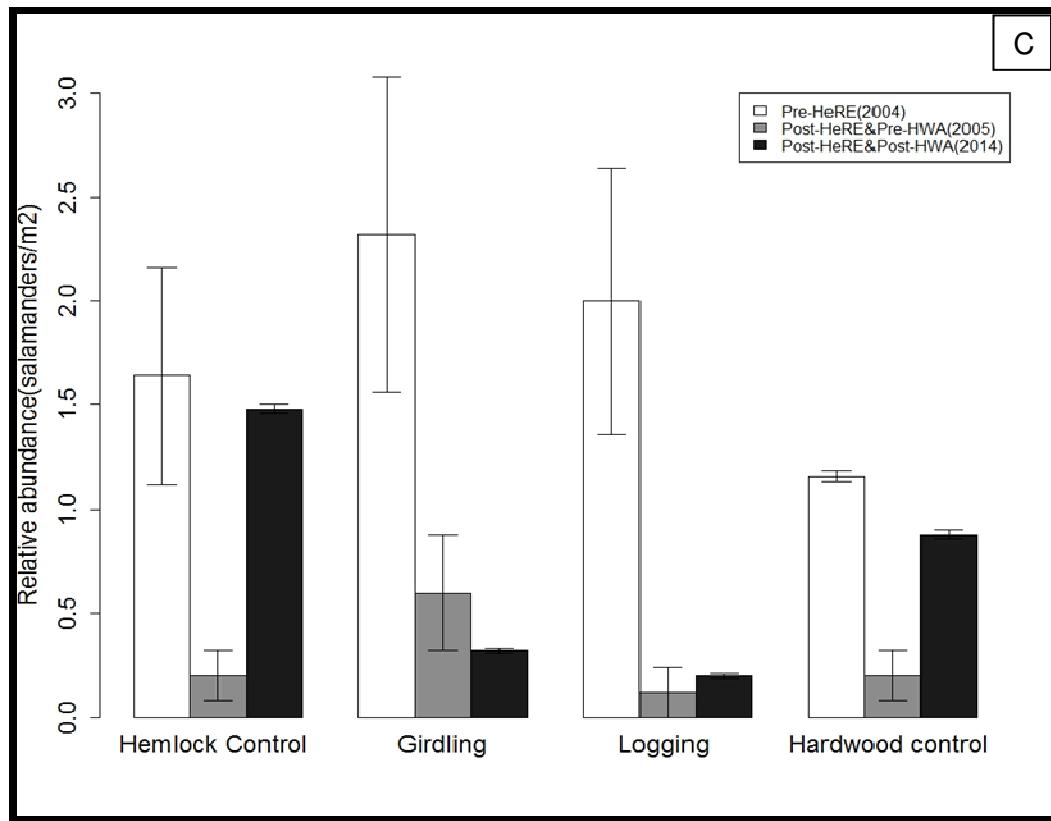


Figure 2.3: Occupancy (A), detection probabilities (B), and relative abundance (C) of *P. cinereus* in Harvard Forest at pre and post to canopy manipulations simulating adelgid outbreak. Error bars represent standard errors of these estimates.

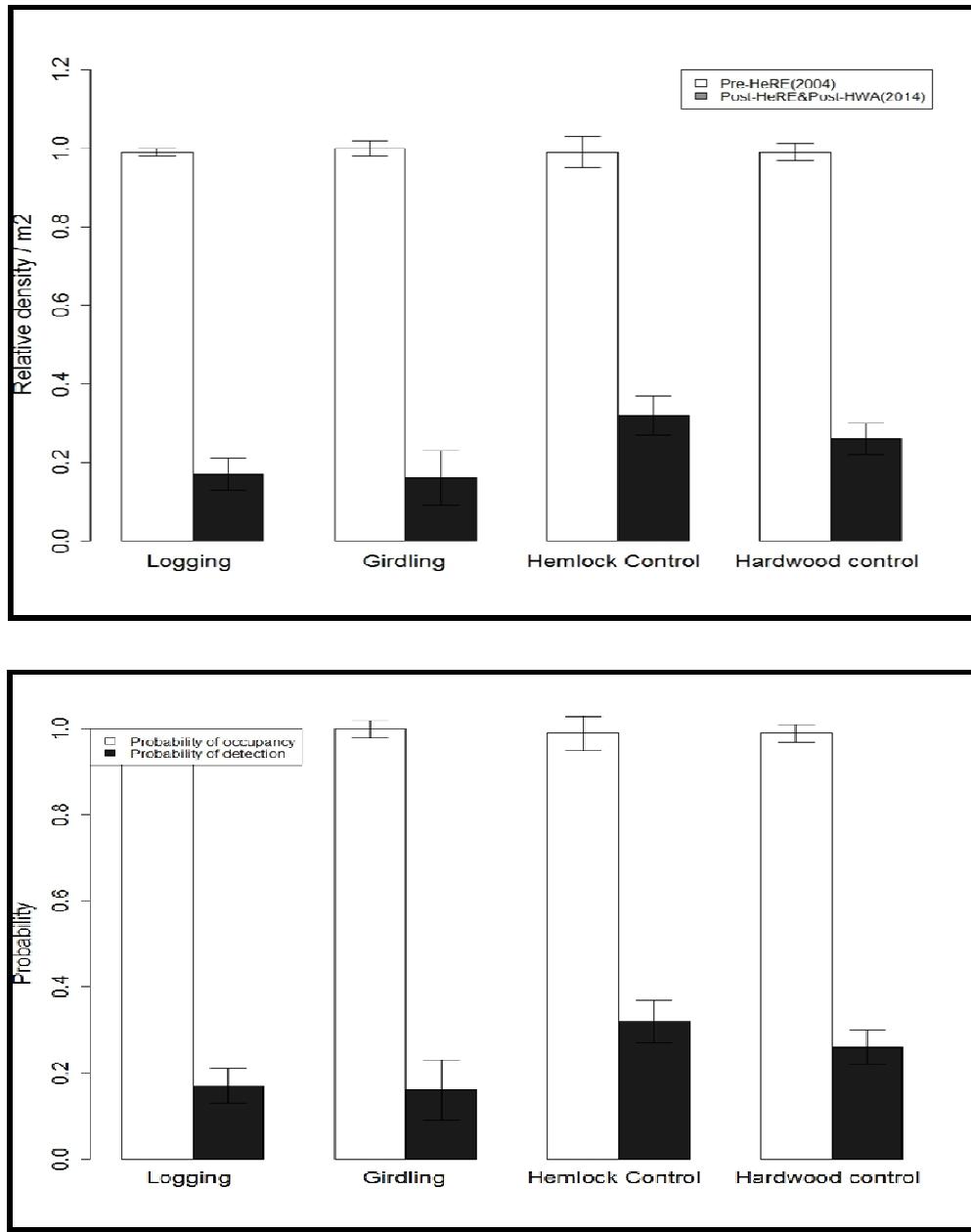


Figure 2.4: Average relative density/ m² of *N. viridescens* (top) in Harvard Forest before and after canopy manipulations simulating adelgid outbreak. The current state of occupancy and detection probability (bottom) of *N. viridescens* in 2014, ten years after canopy manipulations simulating adelgid outbreak in Harvard Forest. Error bars represent standard errors of these estimates.

CHAPTER 3

CALIBRATING ABUNDANCE INDICES WITH POPULATION SIZE ESTIMATORS OF RED BACKED SALAMANDERS (*PLETHODON CINEREUS* (GREEN)) FOR LONG-TERM MONITORING OF POPULATION DYNAMICS AND ENVIRONMENTAL CHANGES IN A NEW ENGLAND FOREST

3.1 Abstract

Herpetologists and conservation biologists frequently use convenient and cost-effective, but less accurate, abundance indices (e.g. number of individuals collected under artificial cover boards or during natural objects surveys) in lieu of more accurate, but costly and destructive, population size estimators to detect and monitor size, state, and trends of amphibian populations. Although there are advantages and disadvantages to each approach, studies, reliable use of abundance indices requires that they be calibrated with accurate population estimators. Such calibrations, however, are rare. The red back salamander, *Plethodon cinereus* (Greene, 1818), is an ecologically useful indicator species of forest dynamics, and accurate calibration of indices of salamander abundance would increase the reliability of abundance indices used in monitoring programs. We associated abundance indices derived from surveys of *P. cinereus* under artificial cover boards and natural objects with a more accurate estimator of their population size in a New England forest. Average densities/m² and capture probabilities of *P. cinereus* under natural objects and cover boards in independent, replicate sites at the Harvard Forest (Petersham, Massachusetts, USA) were similar in stands dominated by *Tsuga canadensis* (L.) Carrière (eastern hemlock) and deciduous hardwood species (predominantly *Quercus*

rubra L. [red oak] and *Acer rubrum* L. [red maple]). Abundance indices from salamander surveys under cover boards and natural objects were correlated with density estimates of *P. cinereus* derived from depletion (removal) surveys. The cover-board abundance index was weakly associated with the population estimator of *P. cinereus*, and it overestimated the estimated true density by a factor of eight. In contrast, the natural-object survey was strongly associated with the estimated true density, but it underestimated the true density by a factor of two. I conclude that when calibrated and used appropriately, abundance indices may provide cost-effective and reliable measures of *P. cinereus* abundance that could be used in conservation assessments and long-term monitoring of forest health at Harvard Forest and elsewhere in the northeastern USA.

Keywords: Abundance index, amphibian monitoring, artificial cover boards, depletion sampling, indicator species, long-term monitoring, *Plethodon cinereus*, population size, regression calibration, removal sampling, salamander, *Tsuga canadensis*.

3.2 Introduction

Amphibians are declining worldwide due to climatic changes, habitat loss and alteration, invasive species, diseases, and environmental pollution (Becker et al., 2007; Dodd, 2010); the number of threatened amphibian species increased nine-fold between 1996 and 2011 (Lanoo, 2005; ICUN, 2011). Amphibians are physiologically sensitive to many local environmental characteristics, so they are thought to be useful indicator species for monitoring local environmental changes (Welsh and Hodgson, 2013; but see Kerby et al., 2010). Indicator species can be used reliably to monitor environmental conditions and to inform conservation programs, but only if the measures used as

indicators reflect the actual abundance and density of the species of interest (Yoccoz et al., 2001).

Two standard methods are used to provide accurate estimates of the size of amphibian populations (Heyer et al., 1994): capture-mark-recapture methods (Seber, 1982; Bailey et al., 2004a) and depletion (removal) methods (Zippin, 1956; Bailey et al. 2004b). Although both of these methods yield reliable estimates of abundance, they are impractical to use when species have very large home ranges, low detection probability, or are cryptic or rare (Royle, 2004). Long-term monitoring programs also may not have sufficient resources to regularly (e.g. annually) repeat intensive mark-recapture or depletion studies. Finally, mark-recapture studies that rely on toe clipping, PIT tags, or skin marking may reduce survival and have been critiqued on ethical grounds (e.g. Clark, 1972; Heyer et al., 1994; Ott and Scott, 1999; Green, 2001; Dodd, 2010; Guimarães et al., 2014), and depletion studies can reduce local population sizes (Hayek, 1994).

Because of these challenges, many herpetologists and conservation biologists who use amphibians, including Plethodontid salamanders, as indicator species use indices of abundance derived from simple counts of individuals under artificial cover boards, random searching of natural objects, pitfall traps, or visual encounter surveys (Heyer et al., 1994; Mathewson, 2009 and 2014; Welsh and Hodgson, 2013). Although abundance indices routinely are assumed to be proportional to absolute measures of abundance, assuming a constant capture probability (i.e. detectability), these indices may not provide accurate estimators of population size. For example, salamanders may be attracted to cover boards or pitfall traps, and random searching or visual encounter surveys may not

provide reliable estimates of detection probability or occupancy, which also are rarely constant (e.g. Krebs, 1999; Pollock et al., 2002). Nonetheless, abundance indices often are easier to obtain than other estimators of population abundance, can be determined for large areas, are less intrusive, minimize harm to individuals, and are cost-effective (Royle, 2004; Pollock et al., 2002).

The trade-off between the need for reliable and cost-effective abundance indices versus labor-intensive but more accurate abundance estimators has led to research that combines both methods using model-based inference (e.g. Smith, 1984; Buckland et al., 2000). Two approaches are used commonly in studies of birds and mammals. *N*-mixture models use Poisson or binomial likelihoods of abundance indices or repeated count data to obtain site-specific estimates of abundance (e.g. Royle, 2004). Alternatively, abundance indices can be calibrated to population estimates obtained from mark-recapture or depletion studies (e.g. Eberhardt and Simmons 1987; Brown et al., 1996). However, neither *N*-mixture models nor direct calibration of abundance indices have been adopted widely by herpetologists, who generally use uncalibrated abundance indices to draw inferences about population sizes and demographic rates, and then use these inferences to determine management applications (e.g. Mazerolle et al., 2007). Here, I calibrate abundance indices derived from transect surveys of counts of salamanders found under cover boards and natural objects with simultaneous estimates of local population sizes of eastern red back salamander (*Plethodon cinereus*) obtained using replicated depletion studies in a New England Forest.

The red back salamander (hereafter *P. cinereus*), is a common woodland amphibian in the family Plethodontidae; the largest salamander family, which contains 240 species (Hairston, 1987; Mathewson 2006; Dodd, 2010). Plethodontid salamanders, including *P. cinereus*, are lungless organisms that respire through their skin (Hairston, 1987). Unlike many other amphibians, *P. cinereus* has no aquatic stage; it is a completely terrestrial species that spends its entire lifetime (3-7 years) in forested areas where it lives in or under moist soils, rotting logs, leaf litter and other natural objects such as rocks. The females lay 3-14 eggs between mid-June and mid-July underneath moist soils and natural objects. The eggs are incubated for 6-9 weeks before they hatch (Petraska, 1998). *Plethodon cinereus* also has a relatively small home range. It averages 13 m² in size, and individuals move about 0.47 m/day, although they may move up to 1 m during rainy days when foraging for prey at the soil surface. Their small home range and limited mobility makes them excellent indicators of changes in local forest ecosystem (Welsh and Droege, 2001; Mathewson, 2009). Burton and Likens (1975) found that the density of *P. cinereus* in New Hampshire was about 0.25 salamanders/m². Their total biomass in the soils at Hubbard Brook equaled that of small mammals at the site and twice that of breeding birds. This abundance makes them important prey to many birds and snakes, as well as a significant predator of many invertebrates and soil fauna such as insects and beetles (Welsh and Hodgson, 2013). Last, *P. cinereus* is quite cryptic. Because it lives underground or beneath natural objects, only 2 – 32% of its population may be present on the soil surface during the warm and moist or rainy nights that are the typical conditions for sampling this species (Burton, 1976; Hairston, 1987).

This study is particularly timely because of the ongoing decline of *Tsuga canadensis*, a foundation tree species in New England forests (Ellison et al., 2005). *Tsuga canadensis* is being killed by a non-native insect, *Adelges tsugae*, which is spreading rapidly throughout the eastern United States (e.g. Orwig et al., 2013). The large spatial scale of *T. canadensis* decline requires rapid, fine-scale studies of the status and trends in populations of species associated with *T. canadensis* stands so that concrete conclusions can be drawn about the likelihood of future persistence or extinction of particular species (Hanski, 1999). For example, the loss of the majority of *T. canadensis* individuals from southern and central New England forests over the next several decades is expected to lead to parallel declines in salamander populations (e.g. Ellison et al., 2005; Mathewson, 2009; Mathewson, 2014). Designing, validating, and implementing a long-term monitoring program for salamanders in these forests requires both accurate base-line estimates of population sizes and methods to rapidly (re)assess populations for many years to come (e.g. Dale and Beyeler, 2001; Bailey et al., 2004b; Welsh and Hodgson, 2013).

3.3 Materials and Methods

My calibration study involved four sequential steps (Figure 3.1):

- 1- Establishment of plots and sampling transects, and emplacement of cover boards (May 2013);

- 2- Simultaneous depletion sampling and surveys of natural cover objects and cover boards (repeated twice in July 2014);
- 3- Estimation of population sizes from depletion sampling;
- 4- Regressions of cover board and natural object survey data on estimated population size of *P. cinereus*.

3.3.1 Study site and locations of calibration plots

This calibration study was done at the Simes Tract (Ellison et al., 2014) within the Harvard Forest Long-term Ecological Research (LTER) site in Petersham, Massachusetts, USA ($42.47^{\circ} - 42.48^{\circ}$ N, $72.22^{\circ} - 72.21^{\circ}$ W; elevation 215 – 300 m a.s.l.; Figure 3.2). All measurements were taken within four stands. Two of these stands were dominated by eastern hemlock (*Tsuga canadensis*), and the other two were composed of mixed deciduous species such as oak (*Quercus*) and maple (*Acer*) species (Figure 3.3). The two hemlock sites were in a moist valley, whereas the two deciduous locations were on a drier ridge \approx 500 m from the valley. Individual stands within a forest type were separated by > 100 m, so all four sites can be considered independent of one another.

Transects for depletion sampling, natural object surveys, and cover boards were established in May 2013. Within each stand, we laid out three, parallel 30×1 -m strip transects, separated from one another by 10 m (Figure 3.2, inset). Along each of the two outer transects, I placed five cover boards ($1 \times 0.25 \times 0.02$ m rough-sawn *T. canadensis* planks), spaced 5 m from one another. Cover boards were placed along the transects in May 2013 as well. To ensure that the lower surface of each cover board was in contact

with the soil surface, leaf litter directly under the cover board was removed before the cover board was laid down. Depletion sampling and natural object surveys were done along all three transects. To minimize effects of the disturbance of establishing the sampling locations on detection of *P. cinereus*, and to allow for appropriate weathering (Hesed, 2012), all sampling was done in July 2014, 14 months after the sites had been selected, transects laid out, and cover boards placed in the field. Following each sampling day, all transects, including natural objects on the forest floor, were left in similar conditions to those seen at the start of the day.

3.3.2 Salamander sampling

Depletion sampling of *P. cinereus*, surveys of these salamanders under natural cover objects, and counts of individual salamanders under cover boards in all four plots occurred during two four-day sessions in July 2014. The first session ran from 14-17 July, and the second from 27-30 July. All sampling was done on the morning of each day between 0700 and 1100 hours.

3.3.3 Depletion sampling

My depletion sampling procedure followed that developed by Hairston (1986), Petraska and Murray (2001), and Bailey et al. (2004b). Every morning during each of the two four-day sampling sessions, I intensively searched for salamanders for \approx 4 hours under dead wood, rocks, and leaf litter in each transect in each plot. All salamanders encountered were removed from the strip transect and placed into $0.7 \times 0.3 \times 0.15$ -m plastic baskets buried 5 m outside of the sampling zones (Figure 3.2, inset). The bottom 10 cm of each basket was filled with dirt and leaf litter to provide wet habitat and food;

small holes were drilled in the bottom of each basket to allow rain water to drain, and baskets were covered with mesh netting to provide shade and protection from predators (Corn, 1994). All salamanders collected remained in these baskets for the entire sampling session (up to 72 hours), and were released thereafter.

3.3.4 Cover-board sampling

I lifted up each cover board, counted the number of *P. cinereus* that I saw under it (Mathewson, 2009; Hesed, 2012), removed the salamanders from under the cover boards, and placed them in the holding baskets.

3.3.5 Abundance estimations and calculation of abundance indices

The three abundance estimates were calculated for each sampling session separately then averaged to a count for temporal variability in these estimates. From the data collected from the depletion surveys, I estimated capture probability and population size of *P. cinereus* in each plot using Zippin's regression method (Zippin, 1956, 1958) as implemented in the Removal Sampling software, version 2.2.2.22 (Seaby and Henderson, 2007). In this method, the total number of individuals captured and removed from the sampling area (i.e. transect) each day was plotted as a function of the cumulative number of captures on previous days in the same transect. The estimated population size for each transect was defined as the point where the regression line intercepted the *x*-axis, and the capture probability as the slope of the regression line. Estimates of population size per m² or per ha were obtained by division (I sampled 30 m² per transect) or multiplication (1 ha = 10,000 m²), respectively.

A transect-level per-cover-board index (salamanders/m²) was estimated as the average of the number of salamanders detected during the first day of each sampling session under all five cover boards in the transect, multiplied by 4 (the area of a single cover board = 0.25 m²). Similarly, a plot-level, natural object survey index (salamanders/m²; excluding the cover boards) was estimated as the total of the number of salamanders captured during the first day of sampling in each transect divided by 30 (the total area of strip transects searched for salamanders was 30× 1 m² = 30 m²). In both cases, I used only the first day of captures to avoid effects of habitat disturbance (from searching) and ongoing removal sampling on the subsequent three days of detection and capture of salamanders in each sampling session.

3.3.6 Calibration of indices

I calibrated the two density indices (from cover boards and natural objects) by regressing them against the estimates of population size derived from the depletion sampling method (Eberhardt, 1982).

3.4 Results

Between both sampling sessions and summed over all three sampling techniques, I captured or detected a total of 101 *P. cinereus* individuals: 53 individuals were captured in the first sampling session and 48 in the second. There was no significant difference between the number of salamanders captured in the hemlock plots (59) or the hardwood plots (42) (Wilcoxon rank sum test: $W = 24$, $P = 0.18$). As is typically found in depletion studies, the total number of captures/day declined continuously in both forest types, and

cumulative captures generally leveled off by the fourth day of sampling during each session (Figure 3.4).

The average population density of *P. cinereus* estimated from the depletion surveys ranged from 0.13 to 0.18 salamanders/m² (1330 to 1816 salamanders/ha), with an overall average of 0.15 salamanders/m² (1550 /ha) (Table 3.1). The average capture probability in the hemlock stands was 0.51, about 15% lower than that in the hardwood stands (0.64). In contrast, the average relative density suggested by cover-board observations was 1.7 individuals/m² in the hemlock stands and 0.7 individuals/m² in the hardwood stand. Last, the estimated density of *P. cinereus* from searches of natural objects within each 30 × 1-m transects was 0.1 and 0.06 individuals/m² in the hemlock and hardwood stands, respectively. Overall, there were no significant differences between forest stand types in any of these estimates (Table 3.1).

Because I found no differences between forest-stand types in salamander density or abundance indices, I pooled the data from the two forest-stand types when I calibrated the two indices using the estimated population density (Figure 3.5). Figure 3.4 reveals that the density of *P. cinereus* was predicted well by the natural-objects survey ($r^2 = 0.65$, $P = 0.001$), but the cover-board index had only a weak and not significant relationship with the estimated population density ($r^2 = 0.30$, $P = 0.158$). The density index obtained from searching under natural objects underestimated the estimated population density of *P. cinereus* whereas the cover board index substantially overestimated the actual density. The population density/m² estimated from cover-board surveys was eight times greater than the estimate of population density based on depletion sampling; whereas the

population density estimated from natural-object surveys was two times lower (Table 3.1 and Figure 3.5).

3.5 Discussion and conclusions

Estimation of the abundance of organisms is at the core of population biology and conservation practice (Krebs, 1999). However, in spite of the importance of accurate estimates of population size, many ecologists and environmental scientists use abundance indices that rarely are calibrated with actual abundance data. We have shown here that, with only modest effort, at least one abundance indices for *P. cinereus* can be calibrated reasonably well, allowing for stronger inferences regarding salamander population size.

Our results represent the first time, to our knowledge, that an abundance index of salamander population size has been calibrated with density estimates in northeastern North America. The abundance estimates we obtained from the removal sampling (1816 salamanders/ha) were 20% lower to those found in hardwood forests at Hubbard Brook, New Hampshire (2243 salamanders/ha; Burton and Likens, 1975). Our finding that *P. cinereus* density was similar in hemlock and hardwood stands mirrored findings of Mathewson (2009) at Harvard Forest. However all of these density estimates are likely to be quite conservative, as Taub (1961) suggested that only 2 – 32% of a local population of *P. cinereus* is available for sampling on the soil surface or within the topsoil during a given period of time.

Although we were successfully able to calibrate the abundance index obtained by natural object surveys with the depletion population size estimator, we could not calibrate the cover board abundance index. Our findings regarding the weaker relationship and

overestimation of cover board index to the salamanders density are not surprising given the apparently attractive nature of the cover boards objects for *P. cinereus* coupled with a very small sample size (N=8) and large variability (CV = 77%). This suggests that density estimates of *P. cinereus* from future natural objects surveys can be used to predict the population size in any monitoring project at least within the study area, but cover board observations are less reliable (Hesed, 2012).

As with many other studies of amphibians, our sample sizes were relatively small. We wanted to reduce the number of depletion sampling plots so as to minimize the number of animals handled and disturbed. Second, calibrating indices with population density estimation using methods such as removal sampling requires that all the different sampling methods be done simultaneously over a large area, a process that is labor (and hence, cost) intensive. Although we were unable to calibrate the cover board index, the results do suggest the magnitude and direction of the difference between the index and actual density estimates (Mazerolle et al., 2007). There is a potential to detect stronger relationship between a cover board index and actual population size estimators if future studies used larger sample sizes.

At the same time, I note that the relationship between density and abundance indices may be non-linear due to detectability issues (see also Pollock et al., 2002). If salamander sampling is part of a long-term monitoring program, I recommend repeated re-calibration of abundance indices at 4–5-year intervals to capture the effects of, for example, changing environments. More intensive mark-recapture surveys should also be done to estimate detection probability, but as with depletion surveys, even mark-recapture

surveys of salamanders are likely to encounter only a fraction of the available population (Taub, 1961; Bailey et al., 2004b)

My results suggest that once it is calibrated, meaningful data on amphibian abundance can be obtained from natural object surveys that take fewer supplies, people, and time than repeating more intensive, invasive, or destructive methods (e.g. capture-mark-recapture surveys, pitfall traps, depletion surveys). Although my data and calibrations are applicable only to the forests I studied in central Massachusetts, the method for calibrating abundance indices is generalizable to any site. I recommend that any abundance index be initially calibrated and routinely re-calibrated, just as one would do with an electronic sensor. Calibrated abundance indices should lead to cost-effective indicators that are straightforward to implement in large-scale conservation programs and broader ecological research (e.g. Noss, 1990; Gitzen et al., 2012, or the U.S. Geological Survey's Amphibian Research and Monitoring Initiative: <http://armi.usgs.gov>).

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Table 3.1: Mean estimates (standard error of the mean) of *P. cinereus* population size (salamanders/m²) based on removal sampling, cover boards, and natural-object searches at the Simes Tract. Tests for significant differences in each estimate were done using the Wilcoxon rank-sum test.

Salamanders/m ²	Forest type		Wilcoxon's W	P
	Hemlock	Hardwood		
Depletion sampling	0.18 (0.03)	0.13 (0.02)	6.5	0.461
Cover-board index	1.7 (0.4)	0.7 (0.17)	0	0.125
Natural-object survey index	0.1 (0.02)	0.06 (0.01)	7	0.562

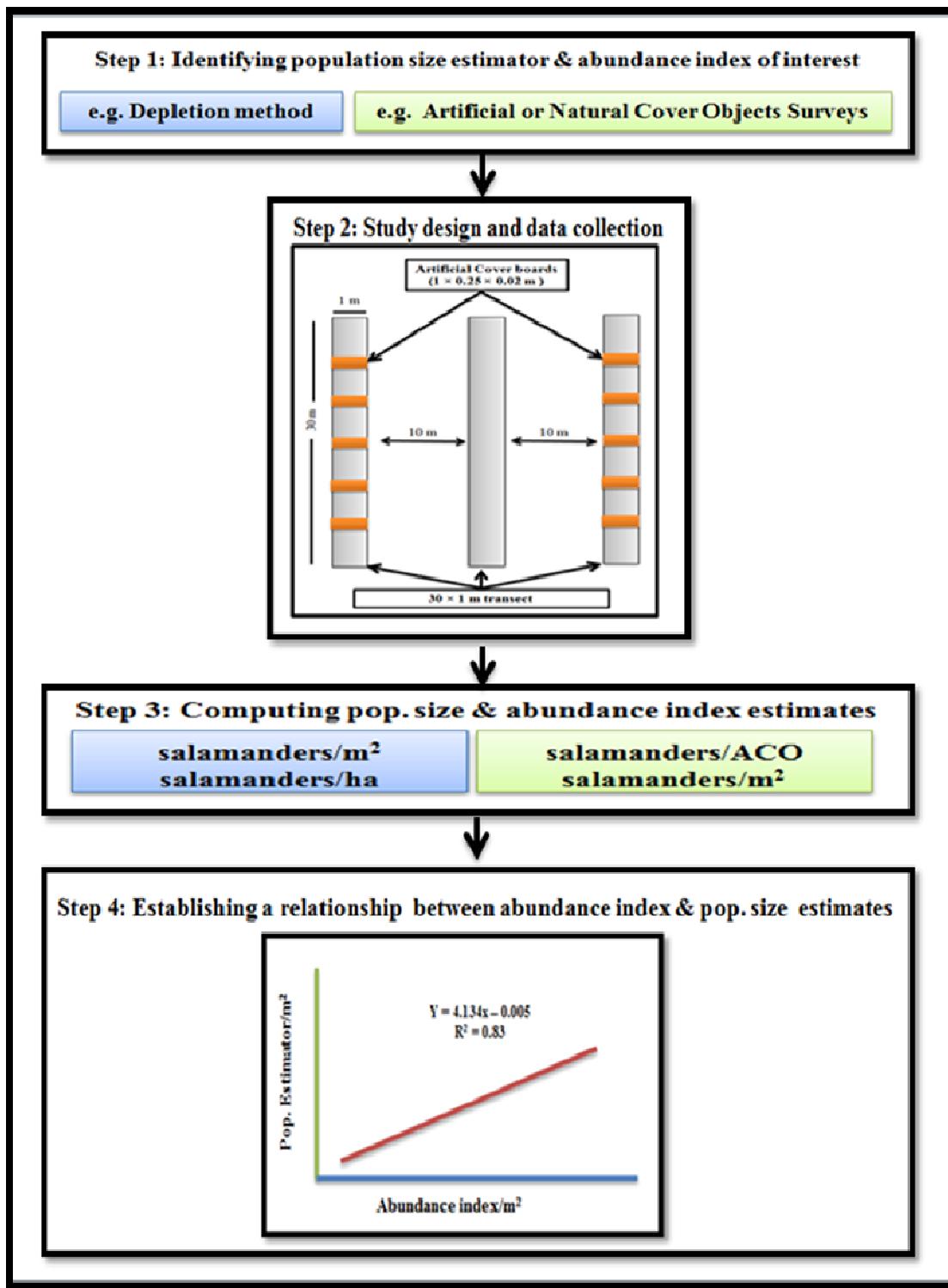


Figure 3.1: Conceptual framework for the methodology of calibrating salamander abundance indices with population size estimators.

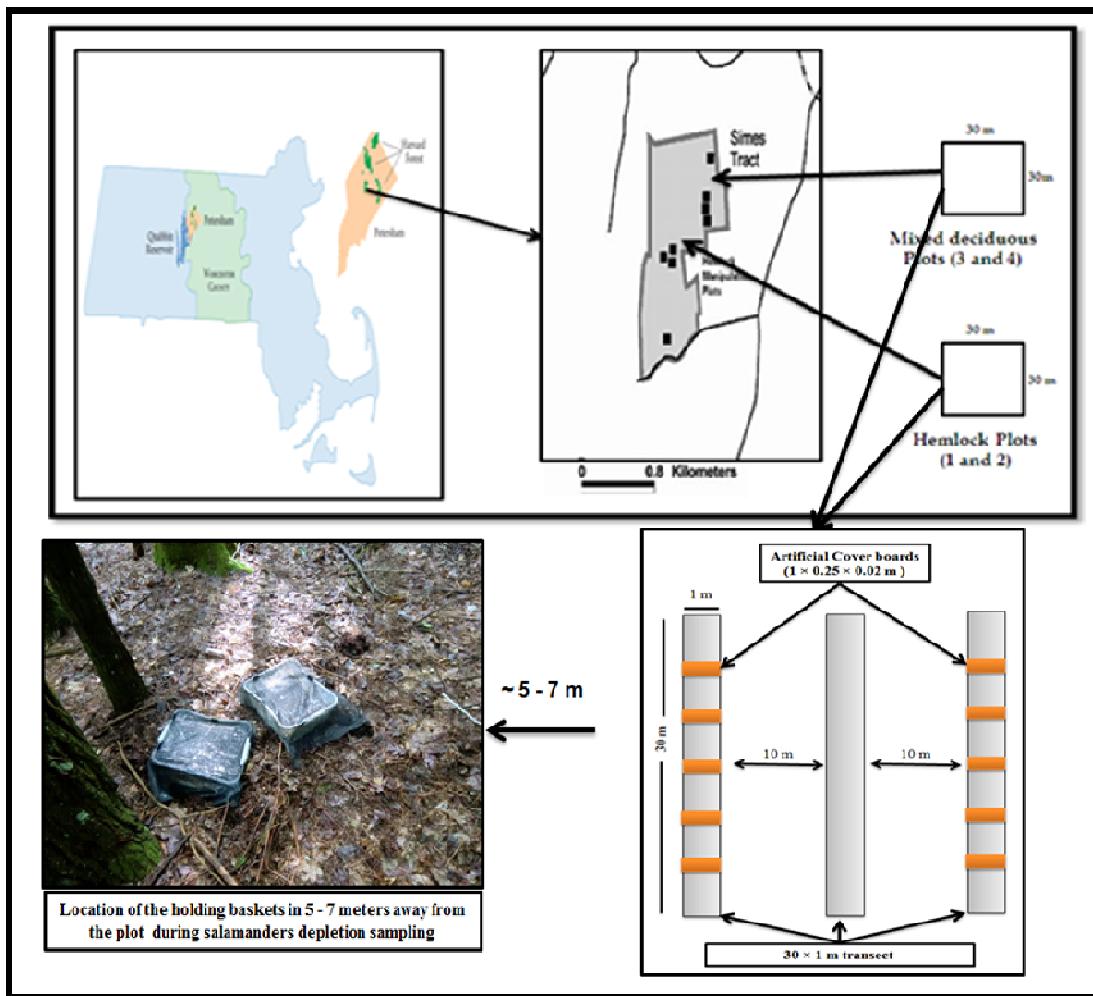


Figure 3.2: Study locations (top) and sampling design (lower inset) showing the layout of the sampling transects, cover boards and holding baskets locations in the Simes Tract at the Harvard Forest, Petersham, Massachusetts.



Figure 3.3: Photographs (June 2014) of the understory of one of the deciduous forest stands (left) and one of the hemlock stands (right) in which calibration plots were established.

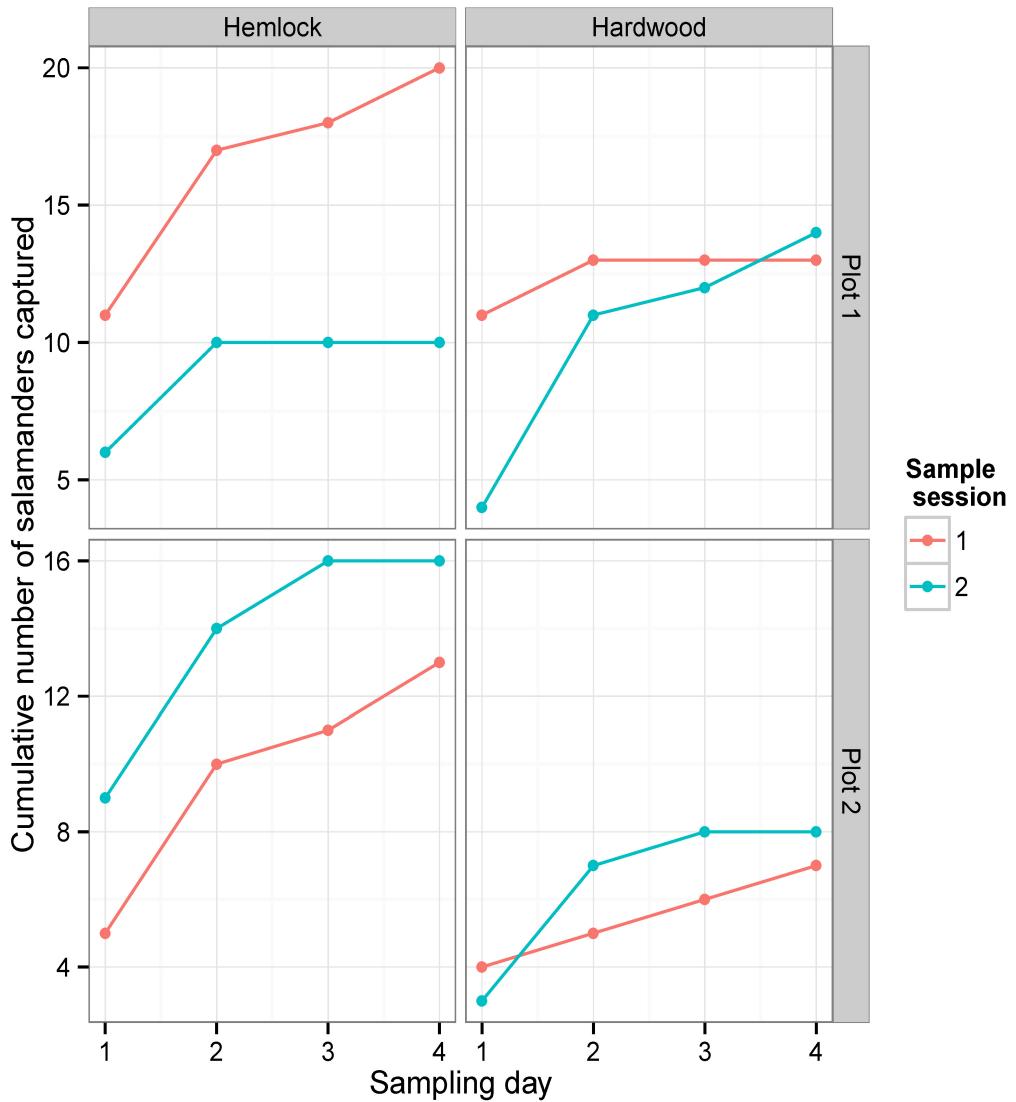
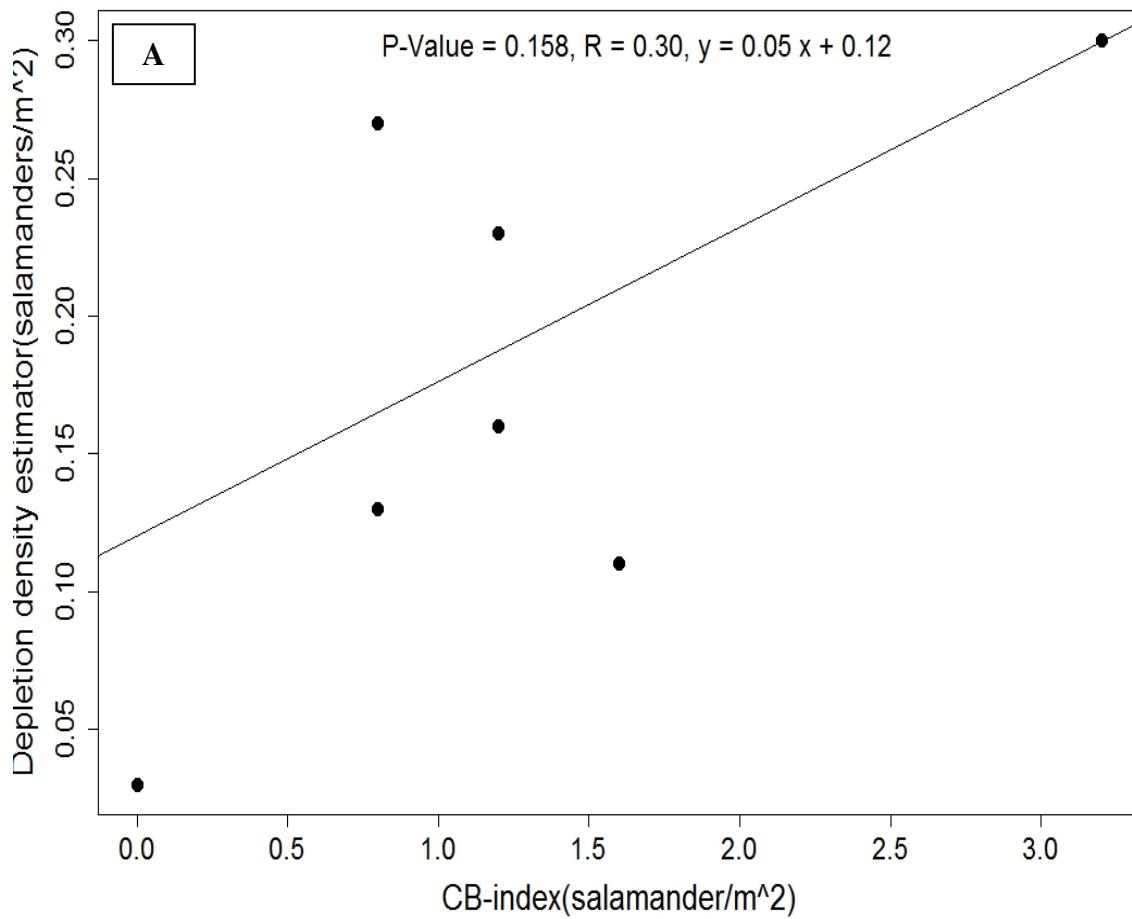


Figure 3.4: Cumulative numbers of salamanders captured during the depletion sampling. Each panel illustrates the cumulative number of salamanders captured in a single plot in either hemlock or the hardwood stands. The data for each 4-day sampling session in each plot × forest type combination are shown in different colors.



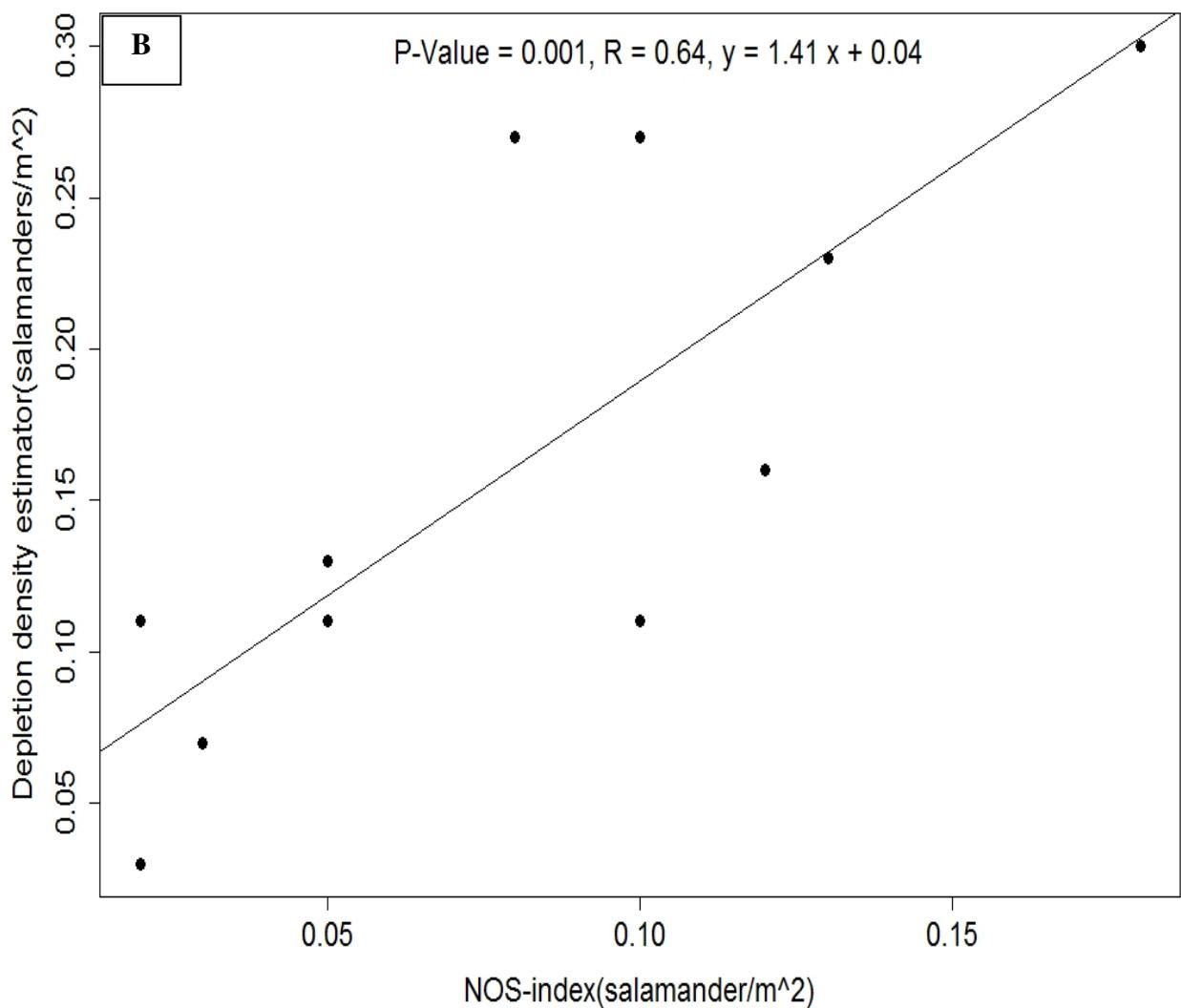


Figure 3.5: Regressions of population estimates (salamanders/m²) based on depletion sampling and abundance indices (salamanders/m²) from cover boards (A) and natural-object surveys (NOS) (B) of *P. cinereus* at the Simes Tract.

CHAPTER 4

THE INDICATOR SPECIES POTENTIAL (ISP) INDEX: A MULTIMETRIC METHOD FOR ASSESSING AND QUANTIFYING THE EFFICACY OF INDICATOR SPECIES IN MONITORING ENVIRONMENTAL CHANGES

4.1 Abstract

Many living organisms, including plants, invertebrates, birds, mammals, fishes, amphibians and reptiles, have been employed effectively as indicator species (IS) for monitoring ecosystem dynamics and environmental changes. Recently, the use of IS has been criticized for lacking robust methods to quantify their effectiveness in achieving monitoring goals. In addition, the Indicator Value index (IndVal), one of the most widely-used techniques for identifying appropriate IS does not account for detection or occupancy probabilities, or consider the association between proposed indicators and monitoring context.

I introduce a multi-metric index that I call Indicator Species Potential (ISP). The ISP quantifies the efficacy of indicator species in classifying sites, monitoring ecological changes, and assessing desired management conditions. The ISP is the average of occupancy probability, specificity, and detection probability, the strength of association between a species and its environment, and the coefficient of variation of its abundance. I suggest that the ISP provides a reasonable way to assess the effectiveness of an indicator species for monitoring environmental change.

Through simulation and an application to real data, I illustrate the use of the ISP. Simulated data on response of small mammals to habitat restoration shows differences

between species chosen by ISP and by IndVal. When applied to salamanders as potential indicators of changes in forests in Massachusetts, ISP suggests that *Plethodon cinerus* is a reasonable IS for ecological change in hemlock stands whereas *Notophthalmus viridescens* is a potential IS in mixed hardwoods. Overall, the ISP shows promise as a method for summarizing ecological and statistical information about potential IS in a single value.

Keywords: Ecological monitoring, ecological changes, ecological indices, terrestrial amphibians, Harvard forest, habitat restoration, eastern hemlock forests.

4.2 Introduction

Many living organisms, including plants, invertebrates, birds, mammals, fishes, amphibians and reptiles, have been employed effectively as indicators for monitoring ecosystems dynamics and environmental changes (Landres et al., 1981; Lindenmayer and Likens, 2011; Siddig et al., *in review EI*). The modern use of indicator species (IS) in environmental management and conservation was first proposed in the early 1900s, when Kokwitz and Marsson (1908) came up with the idea of using some aquatic organisms to assess the levels of pollution in rivers.

According to Carignan and Villard (2002), the first use of the term “indicator species” in the United States dates to 1919, when Hall and Grinnell used it as the basis of their classification of life zones in which plant and animal species are assigned to reflect the structure and composition of a particular location with similar environment in California. Since then, IS have been studied and used widely because of their scientific, economic, and logistical appeal (Borrett et al., 2014; Siddig et al. *in review EI*).

The use of IS also has been criticized because the selection of a particular species to use as an indicator often is based on subjective and arbitrary judgments, or only on a few aspects of the species' life history. Consequently, researchers have sought objective and statistically robust ways to select IS that can be used to monitor ecosystem states reliably and suggest management interventions. Two quantitative methods have been developed to objectively identify IS.

Hill (1979) developed Two-Way Indicator Species Analysis (TWINSPAN). TWINSPAN uses multivariate classification to match relationships between species and site characteristics (Figure 4.1). Hierarchical divisive clustering is used to identify which species best associates with a particular habitat.

Dufrene and Legendre (1997) developed an Indicator Value (IndVal), by which the best indicator species or group of species in the community is selected based on the product of its relative abundance A (also called its specificity) and its relative frequency of occurrence B (also called its fidelity) at the study sites. IndVal has been widely used for ecological monitoring, biodiversity conservation and environmental management because it is easy to construct and has an intuitive interpretation.

The IndVal of species i in group g (of a total of G groups) is calculated as:

$$\text{IndVal}_{ig} = A_{ig}B_{ig}$$

Where A_{ig} is mean abundance of species i in group g divided by the sum of the means of abundance of the same species i across all G groups; and B_{ig} is the relative frequency of occurrence of species i in group g across all sites sampled in group g . Both specificity

and fidelity range from 0 to 1. Finally, the species which has the maximum of all the IndVal_{ig} is taken as the IS for that group of sites.

To demonstrate the calculation of IndVal, consider three rodents species ($i = \{1, 2, 3\}$) being considered by restoration ecologists as IS for monitoring a habitat restoration project. Two restoration treatments – vegetation mowing or burning have been applied in two different locations (hereafter groups $g = \{1, 2\}$ of sites) in the habitat. In each group, ten 20-m² plots were established and the rodents were sampled and the density of the native understory vegetation was measured. The overall goal is to indicate the restoration success (habitat quality) using rodent species as proxy to expected increase in the quality (i.e. composition) and quantity (i.e. densities) of the native understory community in response to these treatments. Table 1 shows simulated data, computation of IndVal, and choice of IndVal for the two restoration treatments.

4.2.1 Limitations of IndVal

According to Noss (1990), Cairns and Pratt (1993), Carignan and Villard (2002), Dale and Beyeler (2001), U.S.EPA (2002), Doren et al. (2009), and Siddig et al. (*in review* EI), an ideal IS for monitoring biodiversity and ecological changes at least should (1) be quickly responsive to the changes in its environment while also strongly reflecting cause-effects relationships in ecosystem changes; (2) have low spatio-temporal variability in abundance; and (3) be easy to detect and sample. IndVal has been criticized because it does not reflect these criteria adequately (Dai et al., 2006; Halme et al., 2009; De Cáceres and Legendre, 2009; De Cáceres et. al., 2010; Podani and Csanyi, 2010; Quinn et al., 2011; Smith and Mather, 2012; Urban et al., 2012).

IndVal does not adequately account for important ecological and statistical information, except for the abundance and physical occurrence of the species in a predefined group of sites (Urban et al., 2012). Thus, in the simulated example (Table 4.1) the locally abundant species 3 in the mown sites is likely to have highest IndVal regardless of its fidelity to the site or its spatial distribution (random, aggregated, and hyper-dispersed). Alternatively, a specialist species with an aggregated distribution, or one which is detectable in only a part of the habitat also could have a maximum IndVal. IndVal also is sensitive to outliers or extreme values. In addition, Urban et al. (2012) and Podani and Csanyi (2010) argued that computation of IndVal from presence-absence data give equal weights to rare and abundant species.

Calculation of IndVal assumes that all individuals are detected perfectly (Quinn et al. 2011, Urban et. al. 2012), an assumption that is rarely true (Mackenzie et al., 2002 and 2003; Royle, 2004). In fact, the fidelity of a species (B_{ig}) is the naïve estimate of occupancy or the proportion of area occupied (POA) by the indicator species within a certain group of sites (Mackenzie et. al., 2002 and 2003; Royle, 2004; Zuur et. al., 2009). Abundance indices used to estimate IndVal may be biased by the expertise of the sampler, sampling methods, timing, and site-specific covariates related to different treatment groups (Quinn et al., 2011; Urban et al., 2012; Siddig et. al., *in review* PeerJ - chapter 3).

Application of IndVal assumes that an IS directly reflects changes in its environment through changes in specificity or fidelity. In the simulated data (i.e. Table 4.1), species 1 and 3 have the highest IndVal in burning and mowing treatments,

respectively, but their abundances are associated more weakly with native understory density (Table 4.7) than the other species (2) at the same location with lower IndVal. This suggests that the highest IndVal may not necessarily reflect the strength of association between an IS and the environment or management activities (Carignan and Villard, 2002; Lindenmayer and Likens, 2011; Urban et al., 2012).

Finally, the form of the site \times species matrix used to calculate IndVal cannot distinguish between abundances that are summaries (e.g. averages) of several temporally replicated samples or those resulting from one-time estimates. To emphasize the consequences of losing the information about the abundance variability of indicator species, species 3 has indicated the mowing sites very well (IndVal = 0.68) and species 1 at second place by 0.5. However, species 1 show less variability relative to species 3 ($CV = 0.43$ and 0.13 , respectively).

These limitations suggest that IS selected based on IndVal alone do not necessarily reflect or predict the relationship between a species and its environment, and could lead to erroneous conclusions in monitoring environmental changes. As a result, there have been a number of recent attempts to improve the IndVal method: see Dai et al. (2006), Halme et al. (2009); De Cáceres and Legendre 2009, De Cáceres et. al. (2010), Podani and Csanyi (2010), Quinn et al (2011), Smitha and Mather (2012), and Urban et al. (2012).

4.2.2 Objectives

Given the limitations of IndVal and motivated by its recent extensions (e.g. De Cáceres and Legendre, 2009; Quinn et al., 2011; Urban et al., 2012), I argue that any

index for choosing an IS needs to be multidimensional in the sense that selected indicators must reflect explicitly not only their specificity and fidelity but also their relationship to the context or environment that they are indicating, have stable and predictable populations, and incorporate information about the ease and availability of sampling (i.e. detectability). Otherwise an IS cannot capture the complexity of the environment nor be effective and meet monitoring goals (Cairns and Pratt, 1993; Carignan and Villard, 2002; Welsh et al., 2008; Lindenmayer and Likens, 2011).

In this chapter, I first present a new framework for selecting and using one or more IS for site classification, monitoring ecological change, and assessing desired management conditions. I introduce the Indicator Species Potential (ISP) index; a multimetric method for quantifying the efficacy of indicator species in ecological monitoring. Second, I compare and contrast my ISP with IndVal using two examples; the simulated rodent data set presented above, and an empirical example from the Harvard Forest Long-term Ecological Research Site (HFR-LTER). For the latter, I applied the ISP to determine which amphibian is best for monitoring response of New England forest ecosystems to hemlock decline caused by an outbreak of an invasive insect. I conclude by identifying limitations with the ISP method and discuss future directions for additional research.

4.3 Methodology

The construction of the ISP is composed of three steps meant to attain and assess the ecological and statistical criteria these best describe the IS. The three steps are: (1) set

the monitoring goals and data preparation; (2) construct the index and assess its statistical properties; (3) apply the ISP to ecological monitoring in a decision framework.

(1) Setting the monitoring goals & data preparation

1. Identify the ecological setting (e.g., forest, watershed, wetland, desert) and spatial extent of the study site (i.e. the scope of inference), then define the monitoring goals, including the clustering factor(s) that the selected indicator(s) species are going to reflect (e.g. habitat type, disturbance level, climate gradients).
2. Identify a set of candidate indicators species based on criteria in Table 4.2. Special emphasis should be given to the suitability of the candidate species to the monitoring context and objectives.
3. Identify the population parameters of interest (e.g. abundance or density) and optimal sampling design (Hirzel and Guisan, 2002; Heywood and DeBacker, 2007; Hooten et al., 2012; Gitzen et al., 2012).
4. Identify the ecological covariates (e.g. vegetation characteristics, climatic factors, soil properties) that link the candidate species to the predefined clustering factor.
5. Sample the abundance of the proposed IS and the relevant ecosystem covariates. Sampling should be conducted in replicated sites and replicated during repeated times separated by short durationsto ensure population closure in the face of any demographic changes due to new births, deaths, emigration and emigration (Gotelli, 2008). This step should yield spatially replicated counts for each species from multiple sampling

occasions, as well as site-specific covariates including the clustering variable, whereas the rows are always representing sites or sampling units in each group (see table 4.3).

(2) Statistical assessment and ISP index construction

Once site data, covariates, and count data, (Table 4.3) have been collected for all candidates IS, the ISP index for each species (i) in group (g) can be computed. The ISP index is an unweighted average of five metrics: occupancy, specificity, detection probability, the strength of the association between species and its group's environmental correlates (covariates), and the spatial and temporal variability (CV) of the abundance for each single indicator species within each group of sites:

$$\text{ISP index for species}_{ig} = 1 / N \sum_{j=1}^5 I_j \quad \dots \quad (1)$$

That is, the ISP of species (i) in group (g) equals the average of five metrics I_1, \dots, I_5 (Table 4.4). Similar to IndVal, ISP ranges between 0.0 – 1.0, with 1 indicating perfect indication and 0.0 no indication.

(3) Decision making and application in management:

The initial ecological or management question was which species to use as an indicator of habitat type, disturbance levels, restoration treatments or other a priori defined characteristic. Based on the ISP values for each species i in group g , I can rank the candidates species from better to worse, where the species with larger ISP values are likely to be better indicators than species with lower ISP values (Table 4.5).

4.4 Test cases and applications

4.4.1 Example 1. Simulated data set

In the rodent example (i.e. Table 4.1), species 1 has an ISP of 0.65 in the burned sites (Table 4.6). This species has the highest levels of specificity, occupancy, and abundance stability (0.50, 1.0 and 0.87 respectively) and the second-highest detection probability (0.90) in the burned sites. In the mown sites, species 2 has an ISP = 0.43 (table 6). The ISP for species 1 in the burned sites is consistent with IndVal. However, in the mown site, the ISP identified species 2 as the best IS, whereas IndVal identified species 3 as the best indicator. Overall, the three rodent species had higher ISP and IndVal scores in the mown sites (Table 4.6).

4.4.2 Example 2. Harvard Forest amphibians:

Amphibians such as Plethodontidae salamanders are ecologically important and reflect changes in their local environment due to their sensitive skins and adaptability to live in wetlands and uplands (Welsh et al., 2008; Mathewson, 2009; Welsh and Hodgson, 2013; Mathewson, 2014; Siddig et al., in prep – chapter 2). Eastern hemlock in the northeastern US are declining as they are infested by the nonnative hemlock woolly adelgid, leading to structural and compositional changes in hemlock-dominated forest stands. These changes – including a shift from hemlock-dominated stands to ones dominated by mixed hardwood (e.g. maple, birch, and beech) already have altered diversity and abundances of many taxa, including amphibians (Ellison, 2005; Orwig et al., 2013; Siddig et al., 2015b).

Using data collected in the Harvard Forest Hemlock Removal Experiment plots in July 2014 (see Chapter 2), I computed the ISP index for two terrestrial amphibians; the

red-backed salamander (*Plethodon cinereus*; PLCI) and the eastern red spotted newt (*Notophthalmus viridescens*; NOVI) occurring in hemlock and mixed hardwood stands at the Harvard Forest (Table 4.7).

The ISP of *P. cinereus* and *N. viridescens* in hemlock stands equaled 0.67 and 0.64 respectively. Despite the near equivalence in ISP for the two species, *P. cinereus* would be preferred because it has a higher population size and specificity in hemlock forests. In contrast *N. viridescens* has a higher ISP (0.46) than does *P. cinereus* (0.28) in mixed hardwood stands.

4.5 Discussion and conclusion

The overall objective of the ISP is to combine multiple metrics to assess and quantify the effectiveness and potential of IS for monitoring changes and suggesting time for management interventions in predefined groups of sites or habitat types. This goal not only is consistent with several recent studies (e.g. Welsh et al., 2008; Podani and Csanyi, 2010; Urban et al., 2012) who critiqued the validity of IndVal and presented an alternative frameworks to compute the index by partially incorporating some of the limitations I mentioned earlier. I think that the ISP has conceptual appeal and good level of performance in assessing the effectiveness of IS.

The ISP is similar to IndVal in that includes specificity and fidelity terms. However, I calculated these terms with *N*-mixture models (Royle, 2004), whereas IndVal calculates them from raw count/ relative abundance. Unlike IndVal, the ISP explicitly

includes terms for the association between the indicators and the environmental context; detection probability; and occupancy. In summary, the ISP:

- 1- Allows identification of the strength of a single species as an indicator in the context of ecological monitoring;
- 2- Is based on the classical repeated counts surveys (i.e. Table 4.4) that provide better estimates of occupancy and detection probability as well as spatio-temporal variation in abundance;
- 3- Explicitly considers the potential influences of ecological covariates on the potential IS;
- 4- Explicitly considers the variability in the abundance of a potential IS;
- 5- Accounts for sampling bias by considering the detection and occupancy probabilities;
- 6- Combines multiple lines of ecological data about IS and their environment.

4.5.1 Potential uses for the ISP index

Generally, IS are used in three categories of ecological monitoring (Cairns and Pratt, 1993). Compliance indicators include those used to detect the success and maintenance of desired states of ecosystems that may be related to management efforts (i.e. restoration). Diagnostic indicators are used to reflect immediate malfunctioning of the environment. Early warning indicators are used to predict expected environmental changes.

The ISP could be used as any of these three types of indicators. It is particularly well suited for situations that involve site classifications, evaluation of management practices on wildlife population dynamics, habitat restoration success, monitoring habitat change. It could be extended to assessment of land resources and agricultural productivity (Pankhurst et al., 2002; Morrison, 2009).

4.5.2 Challenges and ways forward

Despite the intuitive appeal of the ISP, there are several limitations that need to be addressed with future work. The ISP gives equal weight to each of the five metrics and assumes that they are independent (additive) (Table 4.5). However, these assumptions may not be universally true. For example, estimates of the specificity and occupancy (fidelity) depend on detection probability, as may the association between these variables and environmental covariates (Quinn et al., 2011; Urban et al., 2012). Additional investigations of the nature of these relationships in different systems are needed.

The association between an IS and its environment as measured by the correlation coefficient is one metric used in the ISP. However, not all candidate IS respond to the selected ecological covariates in the same way, and not all covariates have the same level of association with a candidate IS. As a result, careful consideration of natural history characteristics will be important when identifying specific covariates, as well as knowing how the correlation changes with detectability (Quinn et al., 2011; Urban et al., 2012).

The ISP needs more data than IndVal because ISP depends on the computation of five separate metrics that together are averaged into the ISP. In particular, estimation of

abundance, occupancy, and detectability using N -mixture models can be complex. At a minimum, computation of these metrics requires two samples repeated in time (within a season, to satisfy the closure assumption), and five sites in any group of sites (Mackenzie et al., 2002 and 2003; Royle, 2004).

Finally, like other methods (IndVal, TWINSPAN), ISP lacks incorporation of other important ecological information, such as body condition or trophic information. As recommended by Welsh et al. (2008) I think that the addition of body condition to the index would improve its suitability for identifying realistic biological responses of the indicators.

Given the increasing trend in the use of indicator species method in response to accelerating environmental challenges (e.g. forest disturbances due to invasive insects), practitioners are limited by the absence of robust methods that allow for quantifying and assessing the potential of the selected indicators. In this study I suggest criteria that could be used to describe the best indicator species that ecologists and conservation biologists could use to guide their decisions in monitoring habitat changes. I also demonstrated the importance of having quantitative, multidimensional, cost-effective measure to evaluate the potential of any selected indicator species. Overall, these findings could provide ecologists and conservation biologists with a protocol to improve the identification of indicator species for ecological monitoring in general as well as promote and encourage the use of multimetrics to better evaluate indicators potential to reflect changes in such complex ecosystems.

4.6 Acknowledgements

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Table 4.1: Hypothetical numerical example showing data and calculation of the IndVal (according to Dufrene and Legendre, 1997) of three rodent species in assessing the success of two vegetation restoration strategies.

Plot number	Burning treatment sites										Mowing treatment sites									
	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20
Species 1	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2
Species 2	0	1	2	0	0	0	1	2	0	0	3	0	0	3	3	1	0	0	3	3
Species 3	0	0	0	10	0	0	0	0	10	0	4	3	2	25	1	3	4	2	15	0
Understory density/m ²	3	3	2	5	2	5	3	2	3	2	3	1	2	3	3	2	1	2	3	3
Species 1																				
A _{1g}	2/4 = 0.5										2/4 = 0.5									
B _{1g}	10/10= 1										10/10= 1									
IndVal _{1g}	0.5										0.5									
Species 2																				
A _{2g}	0.6/2.2 = 0.27										1.6/2.2 = 0.72									
B _{2g}	4/10 = 0.4										6/10 = 0.6									
IndVal _{2g}	0.11										0.43									
Species 3																				
A _{3g}	2/8 = 0.25										6/8 = 0.75									
B _{3g}	2/10=0.2										9/10 = 0.9									
IndVal _{3g}	0.05										0.68									

Table 4.2: The desired ecological, societal and methodological criteria of indicators species for monitoring ecological changes.

Adapted after Cairns and Pratt (1993) and Dale and Beyeler (2001).

Ecological characteristics	Methodological characteristics	Societal characteristics
<ul style="list-style-type: none">• Known life history.• Sensitive and predictive to stresses and environmental changes.• Potential in reflecting the causes of disturbances ahead of time.• Widely applicable to many stressors.• Low variability in space and time.• Ability to integrate and indicate unmeasured information and not redundant though.	<ul style="list-style-type: none">• Logistically convenient to sample with no ethical issues.• Known, straightforward and easy data collection protocols and analysis techniques.• Possibility of sampling in coarse spatial and temporal scales.• Ability to measure the sensitivity or responses by explicit metrics.• Provides cost-effective and timely information for decision making.• Availability of data about historical range of variability of state variables.	<ul style="list-style-type: none">• Socially relevant.• Responses to stresses are easily understood by public.• Responses are scientifically intuitive and interpretable.

Table 4.3: A sample of point-count data for each species resulting from step one.

Sites	Sampling occasion			Site Covariate				Observation Covariate		
	t = 1	t = 2	t = 3	Grouping factor (habitat type)	Soil pH	Understory Veg, density/m ²	DWD/m ²	RH1	RH2	RH3
site 1	2	0	0	Hemlock	4.2	0.7	0.005	78	63	75
site 2	0	0	1	Mixed-deciduous	3.6	2.2	0.003	65	60	70
site 3	0	1	1	Mixed-deciduous	3.7	1.5	0.004	70	62	66
site 4	0	0	0	Hemlock	4.0	0.5	0.005	82	68	69
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Site M	3	1	1	Hemlock	4.4	1.1	0.007	80	70	80

- DWD refers to downed woody debris; RH stands for relative humidity.

Table 4.4: The metrics used in construction of the Indicator Species Potential (ISP) index for the indicator species in monitoring ecological changes.

ISP metrics		Measure	Range of Value
1	Fidelity	Proportion of area occupied (occupancy)	0.0 – 1.0
2	Abundance specificity	Abundance (A_{ig}) as in IndVal	0.0 – 1.0
3	Species-environment associability	correlation coefficient (r^2)	0.0 – 1.0
4	Detection ability	Probability of detection (P)	0.0 – 1.0
5	Species Variability and stability	The complement of the adjusted coefficient of variation (CV) of the abundance of species $_{ig}$. $1 - CV$	0.0 – 1.0
ISP index		The average of the five metrics	0.0 – 1.0

Table 4.5: Qualitative ranking of the Indicator Species Potential (ISP) index

Grade	Explanations	ISP values (0.0 – 1.0)
Excellent	Indicators show highest possible values for all metrics.	0.81 – 1.00
Very good	Indicators show highest possible values in some metrics, but have some relatively low values.	0.61 – 0.80
Good	Indicators show above average values or combinations of some high and low values of the metrics.	0.50 – 0.60
Fair	Indicators show below average values for all metrics.	0.25 – 0.49
Poor	Indicators show very low values for all metrics.	< 0.24

Table 4.6: ISP and IndVal of three simulated rodents considered as indicators of vegetation restoration success.

Treatments	Metrics	Species #		
		1	2	3
Burning sites	Pop. Size	74.90	1.54	2.68
	Specificity	0.50	0.02	0.03
	Occupancy	1.00	0.40	0.20
	Detectability	0.90	0.85	1.00
	1 - CV	0.87	0.62	0.50
	r^2	0.00	0.30	0.20
	naïve IndVal	0.50	0.16	0.05
	robust IndVal	0.50	0.01	0.01
ISP index		0.65	0.43	0.38
Mowing sites	Pop. Size	74.90	74.80	97.00
	Specificity	0.50	0.98	0.97
	Occupancy	1.00	0.67	0.90
	Detectability	0.90	0.66	0.87
	1 - CV	0.84	0.58	0.57
	r^2	0.00	0.84	0.20
	naïve IndVal	0.50	0.44	0.67
	robust IndVal	0.50	0.66	0.88
ISP index		0.64	0.75	0.70

- CV = Coefficient of variation of the abundance of the species in each habitat type and r^2 is the Pearson's correlation coefficient.

Table 4.7: Indicator Species Potential (ISP) index calculations for red-backed salamanders (*Plethodon cinereus*; PLCI) and eastern red spotted newts (*Notophthalmus viridescens*; NOVI) as indicators for monitoring ecological changes in the hemlock and hardwood stands at Harvard Forest.

		Hemlock forests		Hardwood forests	
Species		PLCI	NOVI	PLCI	NOVI
Pop. Size		68.8	36.8	1	10.7
ISP metrics	specificity	0.99	0.77	0.01	0.23
	occupancy	1.00	1.00	0.63	1.00
	detectability	0.46	0.45	0.31	0.43
	1 - CV	0.55	0.45	0.39	0.61
	r^2	0.36	0.52	0.04	0.02
ISP index		0.67	0.64	0.28	0.46
Naive IndVal		0.68	0.47	0.14	0.47
Robust IndVal		0.99	0.77	0.01	0.23

- CV = Coefficient of variation of the abundance of the species in each habitat type and r^2 is the Pearson's correlation coefficient.

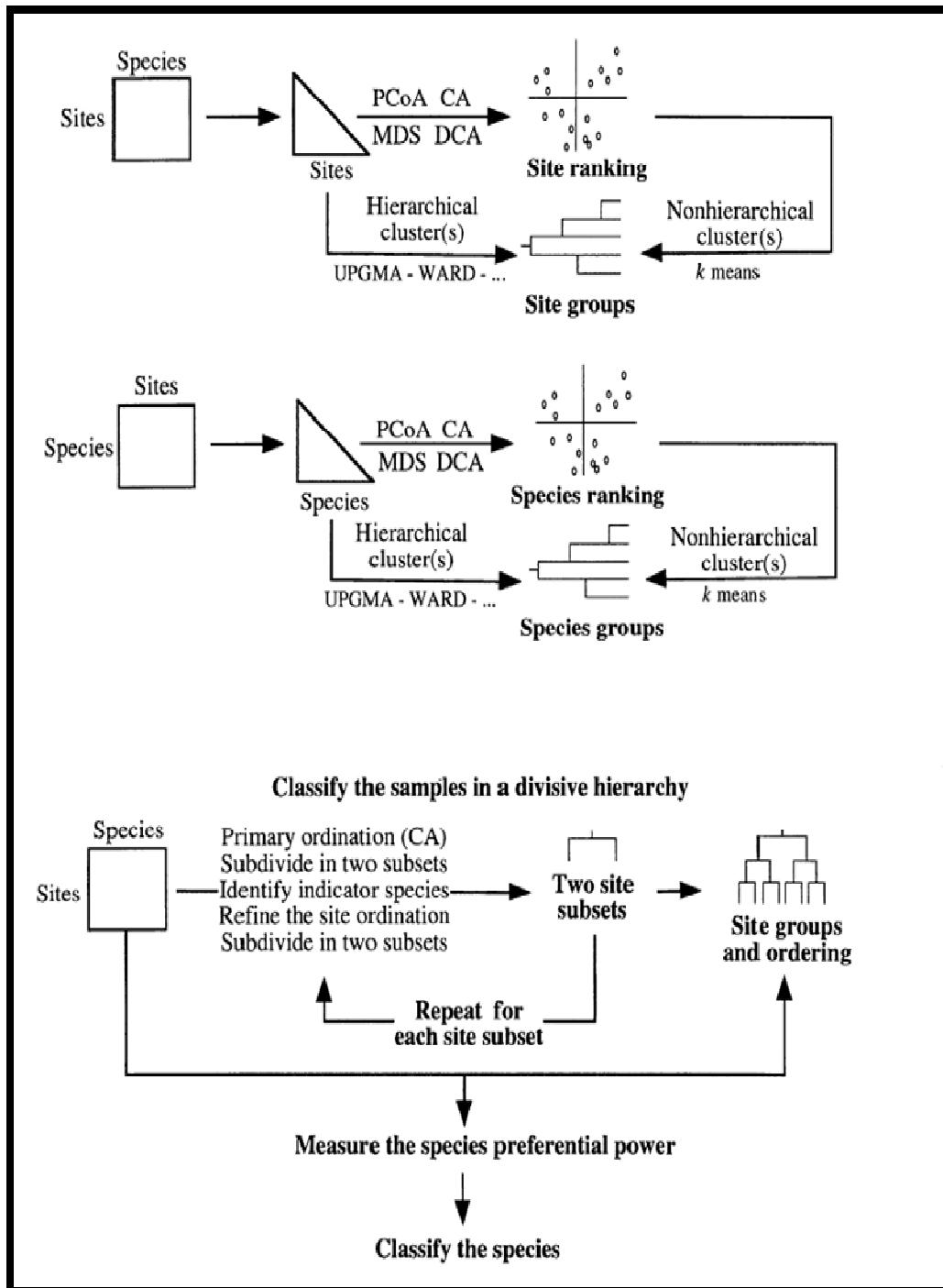


Figure 4.1: The process of identifying indicator species using TWINSPAN, (Hill, 1979) as illustrated in Dufrene and Legendre (1997).

CHAPTER 5

CONCLUSIONS, LIMITATIONS AND FUTURE DIRECTIONS

In this dissertation I aimed to assess the potential and effectiveness of forest amphibians as indicator species in long-term monitoring of ecological changes in New England forests. In the first chapter I introduced the concepts of long-term ecological monitoring and the central role of selecting suitable indicators, and discussed some challenges in applying the indicator species approach in monitoring ecological changes. In the second chapter I assessed the potential impacts of the eastern hemlock decline due to infestations by hemlock wooly adelgid on Red-backed salamanders and eastern red spotted newts. I assessed the impacts of girdling and logging of hemlocks on the abundance, occupancy and detectability of these two focal salamander species. I compared the anticipated future of hemlock forests when it will be replaced by mixed deciduous species (e.g. birch, beech, and maple). My findings and conclusions with this regard are:

- Decline of hemlock as simulated by logging and girdling does not affect the fidelity of these two salamanders, as almost both species had occupancy of 100%. However, canopy manipulations significantly reduced the abundance and detectability of salamanders in manipulated plots relative to hemlock control plots (i.e. healthy hemlock habitat). Although detection was an issue, the high levels of occupancy for both species in the hardwood and hemlock stands suggest that populations of these species will persist in the future.

- Fewer salamanders were found in hardwood stands relative to hemlock stands. At the Harvard Forests, salamanders may decline by as much as 50% as hemlock disappears.
- Difficulty in detection may result in more costly future monitoring programs as sample size and sampling efforts should be increased.
- Overall these three metrics have described the response of salamanders to forest changes.

Despite these conclusions I strongly recommend the following complementary investigations to better understand the impacts of hemlock decline on amphibian populations, and to better inform the conservation and management decisions.

- Continue to monitor the populations of these salamanders to maintain long-term data sets of these indicator species.
- Develop dynamic occupancy models for each salamander species to assess the colonization and extinction probabilities in declining hemlock forests.
- Investigate impacts of hemlock decline on the sex and age structure of red backed salamanders' population and build stage-class models (e.g. Leslie matrix), for this species at Harvard Forest.
- Assess the independent impacts of invasive earthworms on salamander habitat and food availability and how hemlock decline may mediate these impacts in New England forests.

In the third chapter I attempted to calibrate abundance indices derived from transect surveys of cover boards and natural objects with simultaneous estimates of local population sizes of red-backed salamanders (*Plethodon cinereus*) obtained from depletion methods. From this calibration study I concluded the following:

- Samples of salamanders under cover boards overestimate the population size by about eight times whereas natural cover objects survey yield abundance estimates half the size of the estimated population size of red backed salamanders.
- Calibration of at least one abundance indices resulted in a reliable and cost-effective way for long-term monitoring of salamander abundance.
- Modest effort put towards calibration will allow for stronger inferences regarding salamander population size and amphibian population dynamics, and can inform conservation planning.
- Calibration has the potential to be applied usefully in long-term monitoring of other salamanders in the region.
- My approach could be used for other studies, such as calibrating abundance indices of mole salamanders.

Nevertheless, there were limitations to this calibration work. The sample size was small ($n = 8$ transects). The relationship between actual density and abundance indices may be non-linear due to detectability issues, and recalibration should be repeated regularly to capture the effects of, for example, changing environments. Overall

calibration studies may follow the conceptual model that I developed in chapter three (Figure 3.1).

In chapter 4, I introduced the Indicator Species Potential (ISP) index, a multimetric method for quantifying the efficacy of indicator species in ecological monitoring. The index quantifies the strength of the indicator based on the average value of the following metrics:

- 1- Species fidelity
- 2- Specificity
- 3- Detectability
- 4- Species-environment associability
- 5- Abundance variability in space and time.

The overall objective of this new index is to use multiple metrics to assess and quantify the effectiveness and potential of indicator species in indicating predefined groups of sites or habitat types, or management activities. Overall, the index showed great conceptual appeal to assess the potential of indicator species based on multiple criteria, but I was particularly successful in assessing the potential of Red-backed salamanders (*Plethodon cinereus*) and Eastern red spotted newts (*Notophthalmus viridescens*) as indicator species for monitoring ecological changes in the hemlock and hardwood stands at the Harvard Forest. Despite this success, I note limitations in the index that need additional investigation:

- All metrics have equal weight and are assumed to be additive.
- Not all species respond similarly to selected ecological covariates.
- Direct and realistic biological responses (e.g. body condition) are not considered yet in the metric.
- The index requires a lot of data, and its construction involves multiple steps that managers may find difficult to follow.

APPENDIX

METRICS OF INDICATOR SPECIES POTENTIAL INDEX

Figure A.1. Explanations of the standard metrics used to compute the Indicator Species Potential (ISP) index for the indicator species in monitoring ecological changes.

ISP Metrics		Explanation and Rationale
1	Occupancy (O_{ig})	<p>This measures the proportion of area occupied (i.e. occupancy probability) by the candidate indicator species based on hierarchical modeling, particularly Royle's N-mixture models. These are similar to Urban et al. (2012) except they are based on the multinomial probabilities distributions and a maximum likelihood framework (Royle, 2004). According to the findings of Urban et al. (2012), this method has proven to significantly improve the estimates of species occurrence relative to classical estimates of fidelity used in IndVal. The latter is based only on a naïve estimate of relative frequency of occurrence of indicator species.</p>
2	Abundance specificity (A_{ig})	<p>This assesses the weight of the group-specific abundance of the species (i) out of the total abundance of the same species across all sites of G groups.</p> <p>Specificity =</p> $\frac{\text{Lambda}}{\sum \text{average abundance}(\lambda) \text{ across } G \text{ groups}}$ <p>λ is the average sites-specific abundance for each group_g, which is calculated from Poisson distribution as explained in Royle (2004).</p>
3	Species-environment associability (SEA_{ig})	<p>Measures the strength of the association between the average relative abundance and relevant ecological covariates of the species in group (g). This can be assessed by using the correlation coefficient (r^2) derived from simple or multiple regression analysis as in Gotelli and Ellison (2012). The relative abundance of the species as response variable should be summarized from raw counts data (Table 2) by taking the average of the abundance of T sampling occasions in each group (g) then regressing them against the covariate values measured at the same group(g).</p>

4	Detection ability (D)	Given the importance of detection probability in planning sampling and monitoring populations, I argue that the ability to detect indicator species is important not only for estimating occupancy and site-specific abundances but also for determining how easy or difficult it will be to monitor the species. Estimates of detectability for species (i) in group (g) can be obtained along with the occupancy probability by Mackenzie's model (Mackenzie, 2002 and 2003). Detectability ranges between 0.0 – 1.0 , where 0.0 or closer values indicates low detection ability and values equals or close to 1.0 indicates high levels of detectability for the species at a certain group.
5	Species Variability and stability (V_{ig})	Evaluate the spatial and temporal variability in the abundance of the candidate species (i). This value assesses the assumption that an effective indicator species is supposed to have low variability in its abundance through space and time (see Carins and Pratt, 1993; Dale and Beyeler, 2001; Lindenmayer and Likens, 2011). The CV is calculated for each sample (t) separately then averages the CVs of the abundance for all T samples. The CV of the species i in the group g and sample t equals the standard deviation of the abundance of candidate species (i_{gt}) divided by its mean ($SD_{igt}/Average_{igt}$). As CV of a highly variable species can be greater than one we adjusted for that by relativizing the average CV for each species(i) in group (g) out of T sampling occasions by the sum of average CVs of all candidate species in group (g) . This adjusted CV species (i) = CV species (i) / Σ average CVs of all candidate species in group (g)). This way I kept CV values for each species bounded by 0.0 – 1.0 and at same time held the magnitude of differences in variability among species in each group similar to before adjustment. Accordingly I use the complement of the adjusted coefficient of variation (1 - CV) as the metric to assess the variability in the abundance (and population stability during sampling). For more on the importance of considering abundance variability in population monitoring, Gibbs (2000) – table 7-2, page 227 provides estimates of variability in for a variety of plant and animal populations.
ISP index		Describes the overall potential of a certain indicator species in classifying or reflecting the ecological changes in specific group of sites based on its degree of occupancy, detection ability, specificity, strength of environmental associability and abundance variability in time and space. This index ranges between 0.0 – 1.0. Species with greater levels of occupancy (fidelity), greater detection ability, higher abundance

	specificity, stronger associability with local environment, and lower abundance variability in time and space within a certain group are likely to be efficient indicators with ISP index value close or equal to 1.0. In contrast, species with lower levels of occupancy, lower detection ability, lower abundance concentration, weaker associability with local environment and greater abundance variability in time and space in certain group will likely have lower ISP index values ~ 0.0 and therefore be weaker indicators.
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