

TREAD LIGHTLY: AN ASSESSMENT OF RESEARCHER TRAMPLING IMPACTS ON
THE UNDERSTORY VEGETATION OF LAKE WABIKON FOREST DYNAMICS PLOT

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ABSTRACT

Long term and large-scale ecological studies often require intensive sampling and replication. However, the inevitable impacts resulting from intensive researcher activity are often considered negligible and largely ignored in data analysis and interpretation of ecological data. This thesis study examines the impact of researcher trampling activity on the density and diversity of understory fern and woody seedling communities in a northern temperate forest research plot in Northern Wisconsin. I established thirty 1X1 m plots in heavily trampled, moderately trampled, and untrampled locations on and near the Wabikon Lake Forest Dynamics Plot. Fern and woody seedlings were identified to species, and fern species richness and Shannon's diversity scores for woody seedlings analyzed for each plot and across trampling treatments. Fern and woody seedling diversity varied significantly with trampling intensity (ferns: $\chi^2 = 9.772$, $df = 2$, $p = 0.008$; woody seedlings: $\chi^2 = 10.546$, $df = 2$, $p = 0.005$).

Decreases in fern density occurred between control and moderately trampled (MW = 269.000, WW = 734.000, $p = 0.006$) and between control and heavily trampled locations (MW = 283.500, WW = 784.500, $p = 0.012$), however moderately and heavily disturbed locations did not vary significantly in fern density (MW = 405.500, WW = 870.500, $p = 0.485$). Overall seedling density showed no significant variation between trampling treatments, however individual species assessments of *Acer saccharum* and *Fraxinus americana* seedlings indicate a significant decrease between trampling treatments for both

species. *Fraxinus americana* seedling density decreased significantly control and moderately trampled (MW = 306.000, WW = 771.000, $p = 0.031$) and control and heavily trampled plots (MW = 317.500, WW = 782.500, $p = 0.047$), with no significant decrease between moderately and heavily trampled plots (MW = 436.000, WW = 901.000, $p = 0.832$). Conversely, *Acer saccharum* seedlings did not decrease significantly between control and moderately trampled plots (MW = 440.500, WW = 905.500, $p = .874$), but decreased significantly between control and heavily trampled (MW = 310.000, WW = 775.000, $p = 0.008$) and between moderately and heavily trampled locations (MW = 322.500, WW = 787.500, $p = 0.013$).

Significant reductions in density and diversity of understory communities suggest that researcher activity can significantly alter a study ecosystem. This has both ecological and ethical implications, as researcher-induced alterations to understory composition may bias ecologists' understanding of ecosystem dynamics and ecosystem response to environmental change.

INTRODUCTION

This study investigates the potential impact of researcher trampling activity on the forest and understory dynamics of a northern temperate hardwood forest. By assessing the effects of trampling on woody species seedling density and diversity and fern density and diversity, I aim to both draw attention to the ethical dilemmas that arise when a preserved or otherwise pristine location is chosen as a site for ecological study, and to quantify the often-overlooked effect of researchers on their study sites.

Ecological study has long served to advance our understanding of biological communities and their responses to human-related impacts. These studies satisfy the purpose of informing environmental management decisions and conservation strategies. However few researchers, in conducting ecological assessments, experiments, and observations, have studied the extent of their own impact on the locations they are studying (Bezanson et al., 2012). This constitutes a gap in our understanding and execution of ecological field research, and poses considerable scientific and ethical ramifications. As the biological integrity of the world's remaining natural places continues to decline, we must ask the question: are ecologists in danger of contributing to the destruction of the very places they are working to conserve (Bezanson et al., 2012)?

With this consideration in mind, I chose to investigate the impact of researchers conducting a forest tree census on understory vegetation in order to postulate that researcher trampling may potentially alter the long-term dynamics of the forest under study. Some degree of disturbance is inevitable in any ecological study. In order to accurately interpret the results of scientific investigation, it is critical that ecologists understand how and to what degree they are influencing these findings. Researcher impacts on their study sites are likely much more significant than is currently recognized, and may in fact bias research findings through inaccurate representations of ecosystem responses (Comita et al., 2009). I propose that understory trampling resulting from a census of mature hardwood trees has a significant impact on the composition of understory vegetation. This investigation has substantial implications for

ecological protocol for data collection and analysis, for the emerging field of ecological and research ethics, and for long-held perceptions of natural forest ecosystem function and dynamics.

Ecological Research Ethics

The field of environmental ethics attempts to provide moral standing and to generate moral consideration for non-human entities. However, unlike biomedical ethics or business ethics, environmental ethics does not currently constitute an applied framework. Instead, it embodies a theoretical shift in existing moral philosophy (Callicott, 1984) in which moral considerability (Cahen, 1988) is only awarded to overtly sentient beings. The earliest and most conservative attempt at shifting this paradigm, the animal welfare movement, developed with the growth of modern laboratory science and from a direct objection from the public to inhumane treatment of animal study subjects (Farnsworth & Rosovsky, 1993). Today, vertebrate species, complex invertebrate species, such as octopi, and endangered species are protected under ethical codes and legislation, but most plant and invertebrate species and collective entities such as communities, ecosystems, and the biosphere as a whole, are not capable of sentience as it is currently defined, and are thus thought to be undeserving of moral consideration (Parris et al., 2009). In this way, with protection awarded to species easily relatable from the human perspective, existing moral constructs are largely anthropocentric. Unfortunately, plants, ecosystems, and animal species not considered charismatic are typically excluded from protective codes of conduct.

Despite their lack of recognition as such, plants, communities, and ecosystems should be considered moral entities. Our current environmental predicaments, including progressive destruction of the world's forests, anthropogenic local eradication and global extinction of species, and global biological homogenization (Callicott, 1984), are moral issues. There are clearly moral implications associated with the assault of non-human life forms for the purpose of human gain, even if the only wrong recognized is deprivation of future human generations of critical resources and services. We have a moral obligation to maintain the integrity of natural communities and to ensure the healthy functioning of ecosystems, despite their exclusion from existing constructs of moral philosophy.

Because system perturbation is unavoidable in scientific study, ecological research design and conduct raise ethical issues relating to ecologists' commitment to both the public and scientific community, and to the habitats and organisms under study. The merit of this research is not in question. Ecological research has greatly enhanced our understanding of biological life forms and processes, and guided the design and implementation of conservation strategies (Minteer & Collins, 2005b). However, researcher impact is not a central focus of study in the fields of ecology and ecosystem management. Even the simple and seemingly harmless act of observation has reverberating effects on organisms' behavior and the surrounding ecosystem (Fransworth & Rosovsky, 1993). This presents an ethical dilemma; in not considering researcher impact on studied

systems, we fail to fulfill our obligation to either our human or our biological moral commitments.

Currently, there is no field of applied or practical ethics for the purpose of mitigating the specific ethical issues that arise during ecological research (Minteer & Collins, 2005a). Medical researchers have borne the brunt of public backlash associated with the scientific practice of infliction of pain or death on study animals. Alternatively, biologists, although subject to ethical codes surrounding animal welfare, have experienced far fewer direct objections from the public to manipulation of invertebrates, plants, or ecosystems subject to ecological study. At a fundamental level, ecological field research requires modification of organisms, populations, species, or ecosystems in the pursuit of scientific understanding. Research necessarily perturbs the system under scrutiny, potentially in a negative way. For this reason, a subfield of environmental ethics is needed, ecological research ethics, and should be devoted entirely to the design and conduct of ecological field experiments.

The first ethical dilemma that arises from the consideration of researcher impacts concerns the organisms under study. Field research commonly entails some degree of destructive sampling. This could mean removal or transplantation of vegetation, labeling of microsites, collection and sacrifice of specimen, or inadvertent damage to plants or other organisms (Fransworth & Rosovsky, 1993). Additionally, the scale of experimental treatments must be considered in this ethical dilemma, as intensity of research, spatial scope, and time scale of a study can determine whether research activity produces long or

short-term effects on observed sites and organisms. Ecologists are increasingly inclined to initiate long-term or large scale experiments in order to determine the effects of factors such as toxins, climate change, edge effects, and pollutant deposition on existing ecosystems over time (Fransworth & Rosovsky, 1993). Research activity alters these rapidly declining ecosystems, often to the point of requiring tens to hundreds of years of recovery (Gouvenain, 1995). To ensure accurate statistical representations of ecosystem response to these factors, ecologists must ensure a high level of replication and treatment strength in these large-scale studies, which can have long-term cumulative effects on study subjects. Although these studies play a valuable role in informing preservation strategies for larger ecosystems, should this necessarily trump preserving natural condition of a study ecosystem (Minteer & Collins, 2005a)? These ethical questions must be considered in research design and conduct.

A second ethical dilemma concerns not the ecological community, but the scientific community. Failing to study the influence of research activity on the study system, and therefore on any findings or conclusions drawn, could lead to biased and overall inaccurate interpretations of our impact on organisms and their habitats. Examining the influence of anthropogenic impacts such as pollution, deforestation, and climate change on ecosystem dynamics via ecological study without including the potentially significant role of researchers in these assessments means excluding an imperative variable from our perception of ecosystem dynamics. This issue is only magnified in the case of long-term ecological studies, where continual research activity could actually

alter forest dynamics due to compounding effects over a larger time scale (Comita et al., 2009).

Some difficulty exists in first the perception, and then quantification of these impacts. Ecologists may not be able to directly detect their impact or the impact of their treatments on study sites (Fransworth & Rosovsky, 1993). Therefore, it is necessary for ecologists to actively look for researcher impacts in order to discern the long-term influences of experiments and activity on ecological study sites and subjects (Parris et al., 2009). If the purpose of ecological research is to further our understanding of the processes and responses of biological communities, we must consider all factors that provoke these responses in order to inform the most appropriate conservation strategies.

Ecological Effects of Human Trampling

Disturbance is an important ecological factor in many ecosystems, and elicits both positive and negative consequences to ecosystem parameters such as species diversity, recruitment, and species density. A disturbance pattern possessing positive effects is described in the intermediate disturbance hypothesis (Petraitis et al., 1989; Connell, 1978), which predicts increased species diversity occurring at intermediate disturbance frequencies and intensities (Hobbs & Huenneke, 1992). Intermediate levels of disturbance can have a positive effect on species diversity and on the overall resilience of a biological community. Findings from numerous vegetation studies support this prediction, suggesting that disturbance promotes greater species diversity levels in vegetation communities that could not otherwise be achieved due to

competition (Janzen, 1970; Connell, 1978). However, before assuming that intermediate disturbance will positively influence an ecosystem, a number of factors must be considered, including historical context.

Ecological research maintains that total diversity of native species at the landscape level will be greatest when disturbance occurs at its historical frequency and in its historical pattern (Hobbs & Huenneke, 1992). However, if a disturbance regime is introduced that is foreign to a system, such as one imposed by a research crew, it can have negative repercussions for species diversity and vegetation structure, and may increase mortality and vulnerability of a system to invasion by non-native vegetation. For example, when historical disturbance regimes are overly compounded with researcher impact, disturbance can have significant detrimental effects on organisms and biological communities (Hobbs & Huenneke, 1992).

Researcher trampling is one example of an added disturbance that can harm rather than benefit an ecological community. Trampling as a disturbance factor can have both direct and indirect effects on plants. Direct impacts of trampling on vegetation include the mechanical destruction or physical damage of the plant (Stancic et al., 2008), the prevention of germination, or increased mortality (Comita et al., 2009). An indirect impact of trampling on vegetation is soil compaction, which reduces soil aeration and water supply (Stancic et al., 2008). Because certain plant species are better adapted to disturbed conditions, trampling alters competition and recruitment dynamics in affected communities.

These effects, both direct and indirect, culminate in various alterations to vegetation community structure and composition.

Ecological studies of trampling effects have utilized soil assessments to examine indirect impact on vegetation structure. Results show increased soil compaction and lower contents of litter, organic matter, and moisture in trampled locations than in surrounding areas (Kutiel & Zhevelev, 2001). These effects can remain detectable years after closure of an area to human activity (Kuss & Hall, 1991). Soil compaction has been found to be a long-lasting indirect effect of trampling and negatively correlated with species diversity, suggesting that vegetation communities require considerable recovery time after trampling events to return to their initial community structure (Kuss & Hall, 1991).

Direct effects of trampling on vegetation communities are more easily observable than the indirect effects mentioned above. Results of a number of studies investigating the direct effects of trampling on vegetation indicate significant changes in vegetation cover and height, number of species present, species diversity, and seedling survival in areas subject to trampling treatments (Kuss & Hall, 1991; Comita et al., 2009; Kutiel & Zhevelev, 2001). Results also suggest that the more intensively an area is trampled, the more that area's vegetation deviates from untrampled vegetation structure (Hamberg et al., 2010). Time scale further differentiates direct from indirect effects. Short-term disturbance events generally cause direct impacts, mainly in the form of damage to plant tissues, while long-term trampling has both direct effects, such as

reduction in cover, and indirect effects, such as soil compaction and effects on plant roots (Hamberg et al., 2010).

Thus, intensity and frequency of trampling disturbance are major factors in vegetation response. Vegetation cover is found to decrease more slowly in areas of light trampling compared to areas of heavy trampling (Kuss & Hall, 1991). Studies investigating the impact of various levels of trampling intensity confirm that both species diversity and total number of species per trampling treatment progressively declined as trampling frequency increased (Kuss & Hall, 1991). Despite variations in response time to different treatment levels, shifts in species composition and vegetation cover were detected even at low levels of use, suggesting that the balance of natural habitats is easily disrupted by external influences that lead to plant injury and physical change (Kuss & Hall, 1991; Hamberg et al., 2010). Other studies suggest a progression of community response, with vegetation cover and biomass loss occurring rapidly after trampling disturbance, followed by destruction of litter, reduction in soil organic matter, and exposure of the mineral soil to erosion and other physical process. These responses reduce germination and seedling establishment by present species (Kutiel & Zhevelev, 2001).

Although researchers have established the existence of a prominent trend of reduced plant density and diversity with intense trampling, different species exhibit various responses to disturbance, with some showing more resilience than others. Within a given patch, the response of a community to disturbance is determined by the attributes of its component species (Hobbs and

Huenneke, 1992, Comita et al., 2009). Attributes determining resistance, a species' ability to minimize damage, and resilience, a species' ability to regenerate after damage (Cole, 1995) of different plant species vary depending on their morphology, anatomy, reproductive potential, and biomass (Gouvenain, 1995). Kuss and Hall (1991) describe one example of varying responses of species to disturbance. They discovered that the dominance of *F. grandifolia* and *A. saccharum* persisted with little variation throughout all levels of trampling treatment, while other species declined progressively with increased trampling. The differential responses of various species to trampling and disturbance explain why the composition of plant communities subjected to these stresses will shift over time (Gouvenain, 1995). As trampling and disturbance alters ecosystem conditions, native species are replaced with species more tolerant of new soil conditions.

When native-species are reduced and unable to germinate due to trampling disturbance, resultant openings provide opportunities for new, nonnative species to establish and invade the community (Stancic et al., 2008; Hobbs & Huenneke, 1992). Multiple studies confirm this point, finding habitats with weaker anthropogenic impact to have lower proportions of invasives than those with greater anthropogenic impact (Stancic et al., 2008; Hamberg et al., 2010). Trampling potentially aids in the dispersal of species not typical of mature forests into the forest interior (Hamberg et al., 2010). Invasive plants can displace native species, both plants and animals, and can thus not only alter composition, but overall ecosystem function (Hobbs & Huennek, 1992).

Additionally, simulations have confirmed that regularly disturbed forests respond more quickly to climate change and are more susceptible to herbivory than undisturbed forests (Sykes & Prentice, 1996; Cahill et al., 2001).

These findings have significant implications for ecosystem structure and function in light of increased trampling disturbance. Changes in species diversity, number of species, vegetation cover, soil moisture, soil compaction, and species composition resulting from trampling have proven to be significant and frequent consequences of trampling disturbance on vegetation communities. Infrequent or pulse disturbances may only cause short-term changes in seedling and understory dynamics. However, if severe, these effects can accumulate over time, posing serious implications for long-term ecological research sites (Comita et al., 2009).

Research Questions

The central question that motivated my thesis study was whether the impact of research activity on a study site can cause large perturbations in ecosystem function and community structure. Specifically, I questioned whether researcher trampling of understory vegetation during a large-scale census of a forest of mixed hardwood trees had significant impacts on understory composition, and contemplated whether cumulative trampling effects over the long term can alter overall forest dynamics. As mentioned previously, the scale of an ecological study is an important to consider, especially in an assessment of researcher impact on secondary communities. Many ecological studies may fall under a “low” disturbance level classification, as many researchers conscientiously utilize

small research crews, alternate years of access, and design controls that consider human impact (Shane Heschel, personal communication, 2014). However, in light of rapid changes in the global biosphere due to global anthropogenic influences, long-term ecological studies are an increasingly favored approach to understanding the response of natural ecosystems over time. These long-term studies, for example the tree census in question here, may not employ destructive sampling techniques, however researcher presence and activity may become destructive if repeated over large time scales.

In this study, I asked a number of research questions: 1) how does woody seedling density and fern seedling density change between untrampled, moderately trampled, and heavily study locations; 2) how does woody seedling diversity and fern species richness change between untrampled, moderately trampled, and heavily trampled study locations; and 3) do different species of ferns and woody seedlings respond differently to various levels of disturbance.

To answer these research questions, I conducted an assessment of understory vegetation five years after the first complete tree census of the Wabikon Forest Dynamics Plot near Crandon, Wisconsin. I expected vegetation response to trampling to vary by species, and to find an overall trend of reduced vegetation density and diversity with increased trampling intensity. My objective in undertaking this study was to investigate whether ecological studies, in their pursuit of information to advance conservation strategies, may actually threaten conservation efforts due to inadvertent harm on non-study organisms, in this case the understory community.

METHODS

Study Site

The Wabikon Lake Forest Dynamics Plot (45°33'N, 88°48'W) is a 25.5 ha (300X480m) site 10 km east of Crandon, Wisconsin. Established by ecologists at the University of Wisconsin-Green Bay in 2007, the Wabikon plot is part of a global network of forest research plots committed to the study of tropical and temperate forest function and diversity, coordinated by the Smithsonian Tropical Research

Institute (Wang et al., 2013). The glacially formed topography of the site primarily consists of rolling hills, with some steeper outwash features.



Figure 1. Satellite image of Wabikon Lake Forest Dynamics Plot.

The mean annual temperature is about 49.1°C, and the mean annual rainfall is 819 mm (Wang et al., 2013). The Wabikon plot consists mainly of mesic northern hardwoods (Fig 1), with sugar maple (*A. Saccharum*), basswood (*T. Americana*), white ash (*F. Americana*), and ironwood/eastern hop hornbeam (*O. Virginiana*) dominating as the most frequently occurring hardwood species (Wang et. al, 2013). The plot has a relatively well-developed shrub layer and

fairly diverse ground layer due to nutrient rich leaf fall, according to previous studies (mnfi.anr.msu.edu). Common ferns in the understory community include lady fern (*Athyrium filix-femina*), maidenhair fern (*Adiantum pedatum*), and oak fern (*Gymnocarpium dryopteris*). A number of animal species inhabit the area, including gray wolf (*Canis lupus*), black bear (*Urus Americana*), and numerous migrant birds. The Wabikon Lake plot is similar to many other commercially important hardwood forests in the western Great Lakes region, and was subject to commercial logging in the early 1900s. It is since protected, and the understory maintains a status of high quality according to the Smithsonian Tropical Research Institute and the Center for Tropical Forest Science. Five years prior to this study, Wabikon research teams had conducted one census of woody tree species (DBH > 10 cm) on the plot since its 2007 establishment. Thus, this study investigates the impact of one major disturbance event on the understory vegetation community.

Study Design

The 25.5 ha Wabikon Plot is divided into six hundred and thirty 20 X 20 m subplots (Fig. 2), which are marked with wooden posts for ease of census data collection and spatial mapping. Out of convenience, researchers often utilize these posts for navigation throughout the forest



Figure 2. Satellite image of Wabikon Plot, with boundary depicted in red. White lines depict boundaries of 20X20 m subplots. Study site is depicted in yellow.

plot, as they are easily seen across the forest landscape, and mark the four cardinal directions. Thus, most foot traffic throughout the forest plot occurs along the boundaries of these 20X20m subplots. The highest foot traffic occurs during the census years, which are spaced at 5-year intervals. The census crew is likely the most intensive source of trampling disturbance on the plot. Additional research does occur on non-census years, however non-census research teams are typically smaller and less intensive, and are thus thought to cause less significant trampling disturbance relative to the census trampling.



Figure 3. Satellite Image of Wabikon Plot. Yellow rectangle represents on-plot study site. Smaller red rectangle represents control site.

To examine the effects of trampling on the forest dynamics, I designated plots in heavily trampled areas, moderately trampled areas, and untrampled areas. Untrampled plots, or control plots, were placed 100 m south of Wabikon's southern boundary and oriented to match the study site (Fig. 3).

Heavily trampled sites were categorized as the understory areas with the highest likelihood of foot traffic during field research. Thus, I established my thirty 1X1 m heavily trampled plots along the western boundaries of the 20X20 m subplots, at the halfway point between wooden markers.

I placed moderately trampled plots at the center point of each 20X20m subplot, as subplot interiors are subject to considerably lower trampling frequency than the outer boundaries. Trampling of subplot interiors likely

occurs only once every five years, when that subplot is surveyed for the forest census. Any additional trampling in the center of subplots is considered a random occurrence, and the cumulative disturbance of census and random researcher trampling would still be considerably less than on subplot perimeters. It is important to note that intermediately trampled areas, assumed to represent the entire interior of 20 m subplots, are much more representative of the plot as a whole than the heavily trampled areas, which are only approximately 0.5 m wide at most locations.

To test for research impacts on the understory resulting from the 2008 census, I established my untrampled plots approximately 100 m south of Wabikon's southern boundary in order to control for trampling around the immediate perimeter of the Wabikon plot. A potential fault of this study design, with assessment of a single control location, is pseudoreplication. The control, although visually assessed for likeness to on-site sample plots, may not be appropriate, and could bias results toward indicating higher sensitivity of understory vegetation to trampling disturbance. The control area was visually assessed to confirm that control plots and trampled plots possessed similar topography, light availability, orientation, vegetation, and soil type, in order to ensure the most appropriate control was used in light of limited equipment resources. Within the control area, I established six transects 10 m apart and 50 m in length. Sample plots were placed every 10 m along each transect, mirroring the on-plot sampling pattern, for a total of 30 untrampled plots.

Data Collection

Data for this study was collected in July of 2013. I established my 1X1 m quadrats in the center and along the western boundaries of thirty 20X20 m subplots within the 25.5 ha Wabikon plot, and thirty 1X1 m plots south of the Wabikon plot, for a total of 90 sample plots. Within each sample plot, all free standing tree seedlings less than 3 cm in diameter were counted and identified to species. Because ferns dominated the herbaceous groundcover, I used fern species to represent the herbaceous understory overall. Percent fern cover was visually estimated, and ferns identified to species. Data from trampled plots that were within 2 m of an old logging road and were noticeably impacted by this social trail were discarded, and the data supplemented by assessing plots further away from the trail (Comita et al., 2009). Because research activity is concentrated within the boundaries of the Wabikon plot, outside plots serve as an adequate control for assessing the effects of researcher trampling associated with the forest census of the Wabikon Forest Plot.

Data Analysis

To test for researcher trampling impacts on the forest understory, I compared data from my two trampling treatments, heavily and moderately trampled, with the data taken from my control plots adjacent to Wabikon. Using SPSS statistical software, I assessed normality and homogeneity of variance for seedling abundance, seedling diversity, percent fern cover, and fern species richness for each treatment level. All datasets violated normality and were not homogeneous, and therefore non-parametric statistics were employed to determine trampling effect. I utilized the Shannon-Weiner Diversity Index as a

measure for seedling species diversity across treatment types. For ferns, species richness served as my diversity measure. I again conducted non-parametric tests on the diversity datasets in order to determine significance of trampling impact on these parameters.

Because a plant's ability to tolerate and recover from physical damage or disturbance is largely dependent on species (Comita et al., 2009; Hobbs & Huenneke, 1992; Gouvenain, 1995; Kuss and Hall, 1991), I also compared the abundances of two woody species, white ash (*Fraxinus americana*) and sugar maple (*Acer saccharum*), across trampling treatments in order to determine whether the effect of trampling was more pronounced in different species of seedlings. I performed all statistical analyses using the SPSS statistical package 21.1.

In order to represent homogeneity in understory composition, I used Microsoft Excel to construct species area curves for fern and woody seedling species across a cumulative area sampled. To construct these curves, I assigned each of my sample plots a number 1-30. I randomly chose one plot at a time until all plots were accounted for, and recorded the number of additional species contained in that plot not yet encountered in previously drawn plots. If a plot did not contribute a new species, the tally of total species encountered remained the same as the previous plots. Any newly encountered species added to the tally of total number of species encountered within a particular trampling treatment.

RESULTS

Percent cover of ferns was significantly reduced in the areas of moderate and heavy trampling compared to undisturbed locations. The mean percent cover of ferns reduced by half in the disturbed plots, from an average of 33.2%, to 15.0% in the moderately

trampled locations, and to 14.1% in the heavily trampled plots. Because the data were not normally distributed and did not have homogeneous variance, I utilized the Kruskal-Wallis test to determine statistical significance. Percent fern

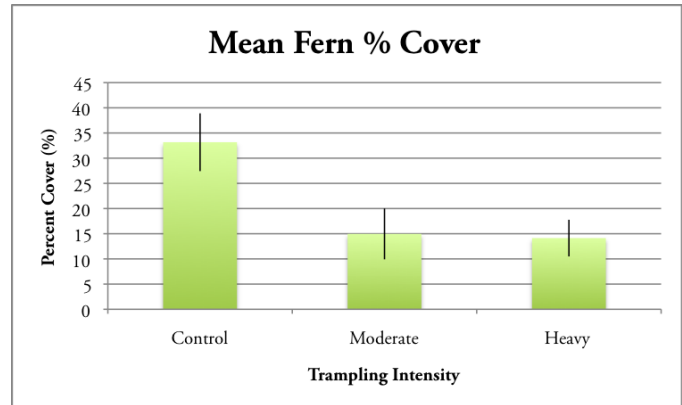
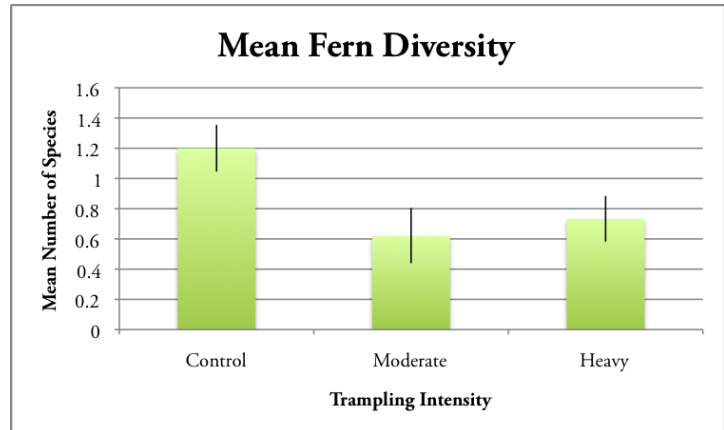


Figure 4. Graph of mean fern percent cover in three trampling treatments. There is a significant decrease in fern density with trampling disturbance.

cover differed significantly between trampling treatments (Chi-square = 9.740, $df = 2$, $p = 0.008$). Further non-parametric tests highlighted the differences between the individual treatments. Differences in percent cover of ferns were statistically significant between untrampled and moderately trampled (Mann Whitney U = 269.000, Wilcoxon W = 734.000, $p = 0.006$) and between untrampled and heavily trampled plots (Mann Whitney U = 283.500, Wilcoxon W = 784.500, $p = 0.012$) (Fig. 4). Percent fern cover in moderately trampled plots did not differ significantly from percent fern cover in heavily trampled plots (Mann-Whitney U = 405.500, Wilcoxon W = 870.500, $p = 0.485$).

Additionally, I analyzed fern species richness as an indicator of trampling impact on diversity. Average fern species richness was greatest in the untrampled plots, at 1.200, and decreased to .633 and .733 in the moderately and heavily trampled plots. Non-parametric tests demonstrate that the differences in species richness



were significant between trampling intensities (Chi square = 9.772, df = 2, p = 0.008). Significant differences in fern species richness existed between untrampled and moderately trampled plots (Mann Whitney U = 259.500, Wilcoxon W = 724.500, p = 0.003), and between untrampled and heavily trampled plots (Mann Whitney U = 310.000, Wilcoxon W = 775.000, p = 0.028) (Fig. 5). There was no significant difference in fern species richness between moderately trampled and heavily trampled sites (Mann Whitney U = 403.000, Wilcoxon W = 868.000, p = 0.442).

The highest number of woody seedlings occurred in the undisturbed locations. Mean number of seedlings per plot decreased with disturbance from 5.500 seedlings in untrampled locations, to 3.467 in moderate and 3.800 in heavily trampled areas. Despite this trend of reduction in number of seedlings with increased trampling, the differences in seedling density were not statistically significant according to a Kruskal-Wallis test (Chi square = 3.664, df = 2, p = 0.160).

After finding no significant differences between overall seedling density and trampling intensity levels, I investigated the effect of trampling intensity on the densities of individual woody species. I focused on the two most abundant species: white ash and sugar maple. White ash (*Fraxinus americana*) seedlings were most abundant in the undisturbed plots, and decreased in abundance in trampled locations. Mean number of white ash seedlings per plot decreased from 4.400 in the untrampled locations, to 2.533 in the moderate, and to 2.500 in the heavily trampled locations. A

Kruskal-Wallis test proved these differences to be marginally statistically significant across trampling treatments (Chi

square = 5.766, df = 2, p = 0.056).

Ash seedling abundances differed significantly between both the

untrampled and the moderately trampled plots (Mann Whitney U = 306.000, Wilcoxon W = 771.000, p = 0.031), and between the untrampled and heavily trampled plots (Mann Whitney U = 317.500, Wilcoxon W = 782.500, p = 0.047) (Fig. 6). Differences between moderately trampled and heavily trampled plots were not statistically significant for white ash seedlings (Mann Whitney U = 436.000, Wilcoxon W = 901.000, p = 0.832).

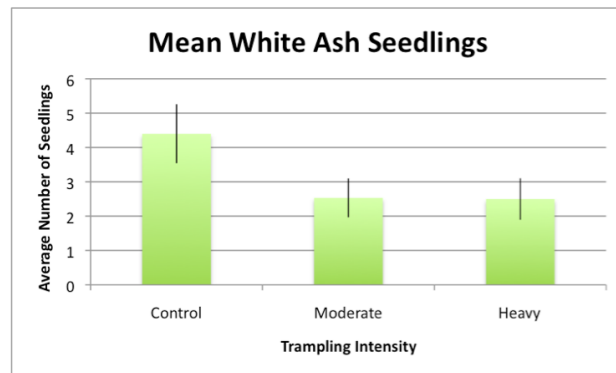


Figure 6. Graph depicting mean white ash seedling density in three trampling treatments. Number of white ash seedlings decreased significantly from control plots. to trampled plots.

I also investigated the effect of trampling intensity on sugar maple (*Acer saccharum*) seedlings, the second most abundant seedling species encountered. Differences in number of seedlings were statistically significant between trampling treatments (Chi square = 7.895, df

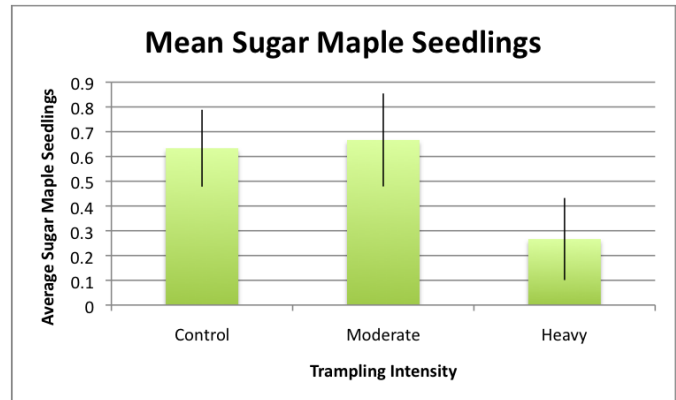
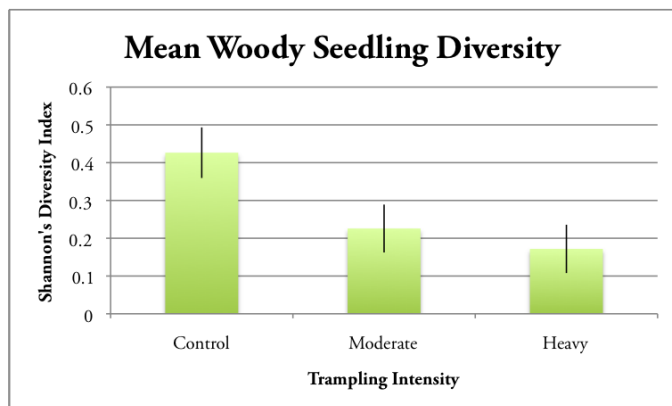


Figure 7. Graph of mean number of sugar maple seedlings in three trampling treatments. Sugar maple density only decreased in heavily trampled locations.

= 2, p = 0.019). Mean numbers of maple seedlings were largely unaffected when subjected to moderate trampling, at 0.6333 in untrampled and 0.6667 in moderately trampled plots, but were reduced by more than half, to 0.2667, when subjected to heavy trampling (Fig. 7).

Abundance of sugar maple seedlings was not significantly different between untrampled and moderately trampled plots (Mann Whitney U = 440.500, Wilcoxon W = 905.500, p=0.874). However, significant differences existed in the abundances of these

seedlings between untrampled and heavily trampled plots (Mann Whitney U = 310.000, Wilcoxon W = 775.000, p=0.008) and between



moderately trampled and

heavily trampled plots (Mann Whitney U = 322.500, Wilcoxon W = 787.500,

Figure 8. Graph of mean woody seedling diversity in three trampling treatments, calculated using Shannon's Diversity Index. Seedling diversity decreased significantly when subject to trampling disturbance.

p=0.013).

In addition to density, I analyzed woody seedling diversity across my three trampling treatments. I used Shannon's Diversity Index to quantify woody seedling diversity by plot. Species diversity was highest in the undisturbed plots, with a mean diversity of 0.426, and decreased with increased trampling intensity to 0.226 in moderately trampled locations and 0.172 in heavily trampled areas. A non-parametric test confirmed that these trends were statistically significant (Chi square = 10.546, df = 2, p = 0.005) between the untrampled and moderately trampled plots (Mann Whitney statistic = 304.500, df = 1, p = 0.021) and between the untrampled and heavily trampled plots (Mann Whitney statistic = 265.000, df = 1, p = 0.003) (Fig. 8). No significant differences existed between the moderately and heavily trampled sites (Mann Whitney statistic = 407.000, df = 1, p = 0.424).

Species area curves for woody seedling species across a cumulative area sampled show the untrampled plots' curve as possessing the largest slope of the three treatments, and thus the greatest degree of homogeneity in understory composition between trample plots (Fig. 9). Both trampled treatments' curves possess smaller slopes, reaching the maximum number of woody seedling species encountered after a

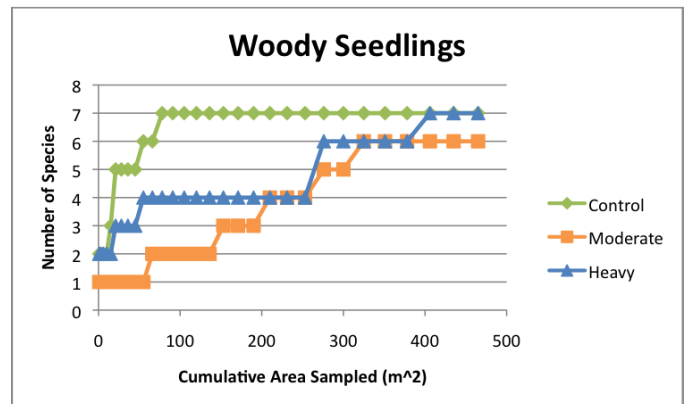


Figure 9. Species area curve of number of woody seedling species across a cumulative area sampled. Green line represents control plots, orange line represents moderate trampled, and blue line represents heavily trampled plots. Slope of line indicates homogeneity of understory composition.

much greater cumulative area sampled, and suggesting a large degree of heterogeneity between plots sampled in disturbed locations. Curves for all treatments level off at between 6 and 7 species, indicating a maximum species richness of 7 woody seedling species in both disturbed and undisturbed sample locations.

Species area curves

for fern species across a cumulative area sampled show moderately trampled plots as possessing the greatest degree of homogeneity in understory composition, while control,

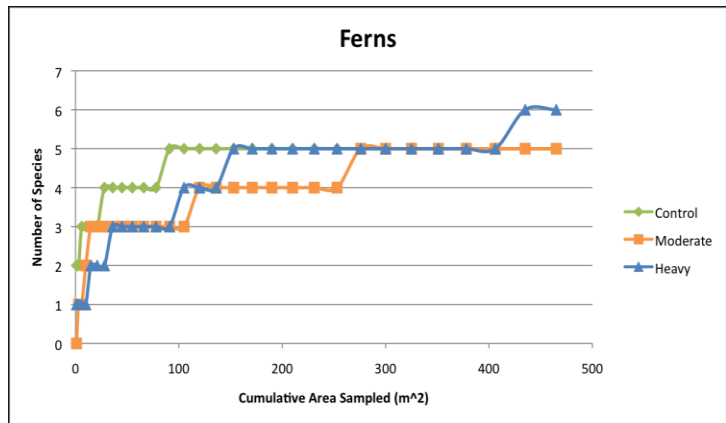


Figure 10. Species area curves for fern species across a cumulative area sampled. Green line represents control plots, orange line represents moderate trampled, and blue line represents heavily trampled plots. Slope of line indicates homogeneity of understory composition.

and then heavily trampled plots increased in heterogeneity of composition between their respective plots (Fig. 10). The maximum number of fern species overall, six, was encountered in the heavily trampled locations, while moderately trampled locations possessed the lowest number of fern species, four, compared to other disturbance levels.

DISCUSSION

My results indicated a significant change in understory composition and structure as a result of researcher impact. Fern density decreased significantly on the Wabikon plot, compared to the control plots just 100 m south of the plot. This decrease was significant between control plots and both trampling

treatments. A number of studies utilizing experimental trampling or assessments of recreational use corroborate these findings. For example, Kutiel and Zhevelev (2000), in their assessment of recreational trampling around picnic sites, found that the more intensively vegetation was trampled, the more percent vegetation cover and vegetation height deviated from untrampled controls. A study by Hamberg et al. (2010) produced similar results, indicating progressive changes in vegetation height and cover with increasing trampling intensity (Hamberg et al., 2010; Kutiel & Zhevelev, 2000). Previous studies support the decreasing trend in vegetation density with trampling disturbance, however unlike the many studies that employed experimental trampling in their study design, my results did not indicate a progression trend in understory changes with increased trampling.

Where previous studies found that trampling progressively changes a system, my results indicate that even slight trampling significantly alters the understory vegetation (Kutiel & Zhevelev, 2000; Kuss & Hall, 1991). Similar to results for fern density data, statistical results for fern species richness data suggest that species richness decreases significantly with trampling treatment, but regardless of intensity. This strays slightly from previous studies in which a clear progression in reduction of variables such as cover, height, species richness, and species diversity with increased trampling was reported (Kutiel & Zhevelev, 2000). Instead of this gradual progression in understory impact, I found that untrampled plots possessed significantly higher fern species richness and density compared to moderately and to heavily trampled plots, but species

richness and density in moderately trampled plots did not differ significantly from heavily trampled plots.

With few exceptions, this appears to be the dominant trend in my findings. Moderate trampling, although of relatively low intensity, caused significant alterations in a number of ecological variables, and these alterations did not change significantly with added trampling beyond moderate levels. In addition to fern cover and fern species richness, both white ash seedling abundance and woody seedling species diversity showed significant declines when subject to moderate trampling, but little change between moderate and heavy trampling intensities.

Although these findings differ slightly from previous reports of gradual changes in understory vegetation with increasing trampling, they do offer important implications, indicating that even low levels of disturbance can significantly influence vegetation community structure. Some studies do confirm significant changes in vegetation even at lower disturbance levels. For example, Kuss and Hall (1991) in their study of ground flora trampling, found that the greatest degree of damage in terms of total cover reduction, reduced number of species, decreased species diversity, and shifts in species composition was brought about by lower trampling intensities. These findings suggest that the balance of natural habitats is easily disrupted by non-historical disturbance or external influences (Kuss & Hall, 1991).

Like the above studies, my findings for changes in within-plot species diversity across trampling treatments are seemingly counter to the intermediate

disturbance hypothesis (Petraitis et al., 1989; Connell, 1978). Native species and community structures are naturally selected for through disturbance regimes. In this way, the continued existence of a particular species or communities typically requires disturbance of some type (Hobbs & Huenneke, 1992). According to the intermediate disturbance hypothesis, I would expect species diversity to be greatest at moderate levels of trampling. However, my results indicate lower species diversity levels for ferns and woody seedlings with both moderate and heavy trampling intensities.

This dissonance in my results despite moderate disturbance levels brings into question the historical disturbance regime of this forest ecosystem. As many studies have recognized, the promotion of increased diversity through the intermediate disturbance hypothesis is contingent upon whether or not the disturbance aligns with historical disturbance frequency and patterns (Hobbs & Huenneke, 1992; Kuss & Hall, 1991). In the case of this study ecosystem, human disturbance is not a new phenomenon. Not only did native peoples inhabit the area, but the plot was logged in the early 1900s, and although it is since protected, plant communities may still be responding to intense management activities from over 50 years ago. From their simulations of forest dynamics, Sykes and Prentice (1996) inferred that their study forest was still adjusting from a shift from an intensive human disturbance regime to protected status, and thus a much less intensive natural disturbance regime. The persistent effects of this intensive disturbance could influence, either amplify or soften, the responses of vegetation communities to more recent disturbance events. Finally,

researcher activity, with new disturbance patterns and frequency for this forest ecosystem, was recently added to its already complex disturbance history. When disturbance novel to an ecosystem is introduced in combination with historical regimes, or when a management strategy is adopted that is inappropriate to the life histories of native plants, it can actually decrease plant species diversity, and create opportunities for invasion by non-native species (Hobbs & Huenneke, 1992).

The recent addition of researcher disturbance to these historical disturbance regimes in 2007 appears to have generated reductions in species diversity due to an added disturbance regime. It is critical to consider the influence of cumulative human land use on an ecosystem, both prior to obtaining protection status, and after protection strategies are implemented, as these cumulative effects may alter local patterns in vegetation (Josefsson et al., 2009). Potential consequences of these trends include homogenization and the formation of new species assemblages through the introduction of invasive species (Josefsson et al., 2009). Future studies should be conducted to assess the presence of invasives at various trampling levels.

Although within-plot species diversity showed a distinct decrease with increased trampling, findings for landscape level diversity do in fact show increased diversity levels with increased disturbance, in fact aligning with the intermediate disturbance hypothesis. Both the woody seedling species and fern species area curves suggest that heterogeneity of the understory increases with trampling disturbance. Both control curves possess the greatest slope, implying

the highest homogeneity in species composition exists in the undisturbed sample areas. The more gradual slopes of the moderate and heavy trampling treatments for both ferns and woody seedlings suggest that the plots in the disturbed locations were much more heterogeneous from one plot to the next in terms of species composition. Within-plot species diversity, as we know, declined with increased trampling. However, these species area curves provide an important consideration to our overall picture of trampling effect, that between plot diversity increases with trampling disturbance. This aligns with previous findings that all forms of secondary disturbance are shown to increase forest floor heterogeneity (Hart & Chen, 2006; Connell, 1978). Although this at first appears to be a positive outcome of disturbance, a further look at the species area curve suggests more concerning management implications.

Each of the curves in the two species area curve graphs level off at similar points along the y-axis. These asymptotes represent the maximum number of fern or woody seedling species encountered in this assessment, or the maximum species richness. The point at which each curve levels off along the x-axis is the amount of survey area required before all species were encountered. Thus, theoretically for undisturbed locations, we would need to conserve an area of approximately 80 m² of land in order to conserve all seven of the woody seedling species encountered. To protect these same seven species in the heavily disturbed location, we would need to conserve approximately 400 m², or five times as much land. A similar story exists for fern species. To conserve the maximum number of fern species in an undisturbed location, an area of

approximately 90 m² is required. To conserve the maximum number of fern species in a heavily disturbed area, the conservation area would need to be expanded to approximately 420 m². Although trampling appears to increase forest floor heterogeneity, these management implications, as well as the increased risk of invasion brought about by heavy trampling, must also be considered in our understanding of researcher impact.

Further investigation of seedling abundance across trampling treatments revealed that different plant species possess varying responses to disturbance. My results indicate that white ash seedlings were highly sensitive to trampling, and their declined significantly from the control to the trampled plots. On the other hand, sugar maple seedlings did not exhibit the same sensitivity to moderate researcher trampling as most other study parameters did. Results indicate that maple seedlings did not change significantly between control and moderately trampled plots, and in fact increased slightly. When subject to heavy trampling however, sugar maple seedlings declined significantly compared to undisturbed plots. These findings suggest that, unlike white ash seedlings, sugar maple seedlings are not sensitive to moderate levels of trampling intensity, but are highly sensitive when subject to heavier trampling.

The differential responses of vegetation to trampling are most commonly attributed to differences in morphological features of plant species. For example, Hobbs and Huenneke (1992) found that the species present in impact areas did possess morphological features, such as stem flexibility (Dumitrascu et al., 2010), plant height, leaf size, and root length (Kuht & Reintam, 1999; Sun & Liddle,

1993), that improved their survival in trampled areas. Similar to my findings, Kuss and Hall (1991) observed high recovery rates and resiliency in sugar maple seedlings. Sugar maple was one of few species assessed that showed no change in frequency of occurrence between disturbed and undisturbed plots. My results, supported by previous studies, indicate that response and recovery vary according to species and plant-growth characteristics. An additional assessment immediately following trampling disturbance would reveal whether differences between sugar maple response and white ash response to trampling are more closely linked with trampling resistance of the particular species, or if necessary recovery time is significantly shorter for sugar maple seedlings than for white ash seedlings.

Although trends seem to vary according to vegetation species, fern and woody seedling densities and diversity decreased significantly when subject to heavy trampling. The fact that these trends were significant five years after the first Wabikon census has notable implications. A single disturbance event caused significant alterations of the understory that were detectable five years post-disturbance. A study by Kuss & Hall (1991) suggests that these long-lasting effects may derive from increased soil compaction and its associated consequences. Kuss and Hall (1991) found that soil compaction was negatively correlated with species diversity. Thus, direct effects of trampling on community composition and structure are linked to indirect effects, such as soil compaction and changes in soil moisture. A study by Gouvenain (1995) investigating trampling impact in the Doubtful Lake area, confirms this inference. Gouvenain's

(1995) findings indicate that changes in soil characteristics such as soil water content and soil surface temperature resulting from past trampling impacts can have long-term effects on the successional development of plant communities.

In a study assessing the impact of multiple five year censuses on understory vegetation in the tropics, Comita et al. (2009) suggest that trampling impacts during intense pulses of research activity (repeated censuses) can cause short-term changes in seedling dynamics for certain plants. The researchers further imply that these alterations in seedling dynamics could accumulate over time and cause a shift in the seedling layer (Comita et al., 2009). A number of vegetation studies indicate that, depending on the severity of the trampling impacts, recovery of a plant community can take from a few years to several hundred years (Gouvenain, 1995; Joseffson et al., 2009). Because impacts of the first census are still detectable five years later, and that censuses are conducted every five years, impacts of researcher activity on the Wabikon plot have considerable potential to compound over time. Thus, significant researcher-induced alterations to the understory community are highly likely, and have serious implications for future interpretations of forest dynamics and ecosystem response.

The magnitude of these implications lies in the importance of the understory to the forest ecosystem overall. Although the understory and canopy communities are commonly assessed as separate entities, the composition and structure of one drives the composition and structure of the other. However, forest management and the ecological studies informing it have tended to focus

on the overstory community without addressing the equally consequential role of the understory as an ecosystem driver (Hart & Chen, 2006). Evidence is emerging that understory vegetation is a major driver of forest condition, both in the short term, effecting seedling establishment, and in the long term, by driving soil processes that cycle nutrients and regulate decomposition (Nilsson & Wardle, 2005) and by affecting canopy succession and ecosystem productivity (Hart & Chen, 2006; Liang & Seagle, 2002). In addition to its role as driver of numerous ecosystem processes, the forest understory also serves to modify the abiotic and biotic environment of the forest floor, and thus to influence the recruitment of tree seedlings (George & Bazzaz, 1999). Understory plants, such as ferns, reduce the light levels, increase litter layer, and reduce soil-surface temperatures below their canopies, thus acting as a barrier for seedlings reaching the soil, and for seedlings emerging from below the litter mat (George & Bazzaz, 1999). In this way, the understory has the potential to influence the density and species composition of the seedling bank, and thus future forest tree species composition and structure. The prominent role of the understory in both ecosystem processes and community structure suggests that alterations to the understory composition will have serious long-term consequences for overstory communities and for forest dynamics (Liang & Seagle, 2002). Finally, little is known about how major global change drivers such as climate change and chemical deposition may influence composition of understory vegetation, as most studies investigating ecosystem response focus on overstory communities (Nilsson & Wardle, 2005). The forest census that motivated this investigation is

one such study in which the understory was not considered. If we hope to understand the influence of these change drivers on forest ecosystems, changes to understory communities, major drivers of overstory composition and function, cannot be ignored.

CONCLUSIONS

Researcher trampling effect on understory vegetation is an often overlooked but unavoidable occurrence in most field research programs (Cahill et al., 2001). My findings, supported by previous research, indicate that researcher trampling significantly alters understory vegetation community structure. In long-term assessments of forest dynamics, repeated visitations to study sites are required to obtain more precise representations of forest demographics. However in the effort to make these representations more precise, frequent visitations may alter future demographic parameters, forest dynamics, biodiversity, ecological processes, and nutrient availability (Cahill et al., 2001; Josefsson et al., 2009). Based on my own findings and on the magnitude of these potential consequences, I contend that researchers working in forest ecosystems must consider their impact to secondary communities as a more central concern in ecological study.

Ecological research is both needed and valued in the effort to understand anthropogenic impact on natural ecosystems. While some manipulation and observation is necessary in order to answer scientific questions, we can no longer disregard the disturbance, deliberate or inadvertent, of organisms in nature (Fransworth & Rosovsky, 1993). The long-standing assumption that field

researchers are harmless observers must be abandoned as an unsound practice, as the act of conducting an experiment can alter experimental results (Cahill et al., 2001). Ecologists must understand the extent of their impact on a study system in order to understand their results.

Ecologists must also understand the extent of their impact in order to acknowledge the extent of harm inflicted on study ecosystems, pristine or otherwise. As this investigation demonstrates, scientists are impacting their study ecosystems to a greater extent than one may think. A model of practical research ethics must be developed in order to minimize and account for the impact of researchers on their study sites. Examples of conduct requirements outlined in a future model could include incorporating ethical considerations, conservation, and restoration efforts into research design, necessitating a plan of minimum impact for sampling and study execution, and requiring impact mitigation strategies in grant and funding proposals (Parris et al., 2009). If an area is to be set aside for research purposes, an ecological research ethics model could require the protection of a similarly sized area in the same ecosystem solely for conservation purposes to offset the consequences of researcher disturbance to these study ecosystems. Only when an ecological research ethics model is applied to research practice, and all disturbances and communities considered in ecosystem assessments can we begin to truly understand the responses of ecological communities to environmental change.

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