ENVIRONMENTAL AND ECONOMIC RESEARCH AND DEVELOPMENT PROGRAM

Impacts of Harvesting Forest Residues for Bioenergy on Nutrient Cycling and Community Assemblages in Northern Hardwood Forests, Wisconsin

Final Report
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Executive Summary

Background

The most readily available source of woody biomass to the logger is through whole-tree harvesting that removes what has been traditionally left as slash (i.e., fine woody debris - FWD). This material has potential to be used as energy feedstock. However, a critical element of managing for biodiversity is maintaining woody debris on the forest floor. Woody biomass is important for nutrient cycling, providing seed beds, and creating habitat structure for wildlife. Researchers recognize the link between biodiversity and ecosystem functioning, but this relationship is not well understood. A change in species may have cascading effects across trophic levels, and cause shifts in the size, distribution, and vertical zonation of vegetation over large areas. Our goal was to investigate the impact of FWD removal on nutrient availability and above and belowground community assemblages on rich soils under regenerating northern hardwood stands in Wisconsin. Land managers are concerned with removing FWD in this system because of the existing lack of large woody debris and structural diversity (e.g., understory shrubs). In addition, exploiting slash for bioenergy purposes will compete with ot her ecological services Wisconsin forests provide.

Research Objective

We manipulated the amount of fine woody debris removed after timber harvest within the Chequamegon-Nicolet National Forest to compare soil carbon:nitrogen availability and community change [i.e., the abundance and diversity of plants (forbs, shrub, tree regeneration), beetles (Coleoptera), and amphibian assemblages) across 4 forest residue removal treatments: (1) 0% tipwood removed (100% retention; current practice), (2) 65% tipwood removed (intermediate retention; amount based on Minnesota biomass harvesting guidelines), (3) 100% tipwood removed (tipwood from every tree harvested removed from site; some tipwood remained on site due to incidental breakage during skidding), and (4) no-cut control

Methods

Fine woody debris removal was applied winter 2009-2010 across nine treatment blocks within a second-growth northern hardwood forest. We used a randomized complete block experimental design; each FWD removal treatment was randomly assigned to a >8 ha section of each block. Treatment sites ranged from 8.5 – 17.4 ha. Within each treatment site, we established a 100 x 100 m (10,000 m^2) plot near the center to ensure sufficient spacing and independence of replicates, and to minimize edge effects from the surrounding forest and other treatment areas. Within each plot, four transects spaced 33.3 m apart were run the length of the plot. We used time constrained searches along each transect to sample the amphibian assemblage. Pitfall traps spaced 25 m along each transect were used to sample the beetle assemblage (Coleoptera). Plant community and fine woody debris amounts were measured in 1 m^2 quadrats placed every 10 m along each transect. Soil samples were collected at 25 m intervals along each transect to a depth of > 20 cm. Measurements were taken May – August, 2009 (pre-treatment), 2010 91-year post-harvest), and 2011 (2-years post-harvest)

Results and Conclusions

In general, there were few short-term qualitative changes in species composition at the plot-level across fine woody debris removal treatments for all species groups. While forb and fern species

richness did not change, fewer shrub species were found immediately post-harvest compared to pre-harvest and 2-years post-harvest. Tree regeneration was similar across treatments. Changes in the plant community appeared to be the result of the harvest rather than changes in FWD amounts.

There were changes in abundance across treatments for amphibians and beetles. For amphibian species, we found more individuals in the 100% removal treatment, but by 2-years post-harvest, 2 of the 4 species showed no differences among treatments (American toad and spring peepers). These results may be due to our collecting method because retained slash piles were difficult to search, or an interaction with the functional changes associated with uneven-aged harvest practices (i.e., more direct sunlight to forest floor for open canopy over slash piles). Abundances for all beetles (Coleoptera) were reduced at the plot-level, and by 2-years post-harvest, abundances were reduced by half for the more abundant species. The reduced numbers of almost all beetles, and no qualitative changes in species composition is consistent with trends found in other soil arthropod studies investigating slash removal.

For soils, levels of carbon, nitrogen, pH, and C:N ratio did not consistently decrease when greater volumes of FWD were removed; changes in pre- and post-harvest levels did not occur in the upper mineral soil layers but were significantly more prominent in the deeper depths. While not investigated in the current study, one explanation for the differences and changes in the lower soil layers may be impacts from earthworm invasions that are known to disrupt C, N, and C:N levels and ultimately the balance of forest floor ecosystems.

This study represents the short-term responses of multiple trophic levels to forest residue removal. Future studies will be required to determine the long-term impacts of removing this material from nutrient rich northern hardwood second-growth forests. These studies will be able to build on our current baseline information to evaluate long term impacts of biomass removal.

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CHAPTER 1

Background, Objectives, and Study Description

Deahn M. Donner

Background

The demand for using woody biomass as an energy feedstock and alternative to fossil fuels is growing. The most readily available source of woody biomass to the logger is through whole-tree harvesting that removes what has been traditionally left as slash (i.e., forest residue or fine woody debris – FWD). This strategy decreases inputs of fine woody debris (material < 4 inches in diameter typically stems and branches) into the system. However, a critical element of managing for biodiversity is maintaining woody debris on the forest floor. Woody biomass is important for nutrient cycling, providing seed beds, and creating habitat structure for wildlife (Maser et al. 1979, Harmon et al. 1986). Researchers recognize the link between biodiversity and ecosystem functioning, but this relationship is not well understood. A change in species may have cascading effects across trophic levels, and cause shifts in the size, distribution, and vertical zonation of vegetation over large areas. Exploiting slash for bioenergy purposes, however, will compete with other ecological services Wisconsin forests provide.

The environmental impacts of harvesting woody biomass to our forest's established biodiversity and ecosystem health requirements will have to be addressed by land managers (e.g., through the NEPA process). Current site-level guidelines emphasize retaining large diameter coarse woody debris (CWD) based on many studies documenting the important role it plays in managing biodiversity and contributing to nutrient cycling. However, little information exists to help guide land managers on appropriate levels of FWD retention for biodiversity and nutrient cycling concerns. There is an urgent need to develop sustainable woody biomass harvesting practices within the context of conventional timber harvesting guidelines that already outline how to protect water, soil, and biodiversity. Minnesota's established harvesting guidelines recommend retaining 1/3 of FWD on-site. Just recently, Wisconsin's established harvesting guidelines recommend retaining even less of the FWD on a site (only 10%;) However, these numbers are based on few studies that have shown variable responses depending on the system, and may not be applicable to all systems. Research is needed to better understand the impacts of whole-tree harvesting, and the corresponding reduction of FWD inputs on biodiversity within uneven-aged silvicultural treatments.

Our goal is to investigate the impact of FWD removal on nutrient availability and above and belowground community assemblages on rich soils under regenerating northern hardwood stands in Wisconsin. Federal land managers are concerned with removing woody biomass in this system because of the disproportionate number of sensitive plant species associated with the rich soils (e.g., American ginseng, goblin fern), and how the lack of FWD recruitment and continued availability may influence existing communities. Woody biomass harvesting has increased in frequency in the last few years, but such efforts have occurred primarily in conifer-dominated situations. The majority of studies on residue removal have been done outside the Lake States

contributing to a considerable knowledge gap on the impacts of FWD removal in Wisconsin forests.

Our major objectives are to compare species diversity and abundance within and across several trophic levels to determine if there is a threshold amount of FWD that can be removed without significantly affecting biodiversity and nutrient availability (i.e., soil carbon and nitrogen levels). A better understanding of how structure affects biodiversity requires integrating diversity within and across trophic levels. Slash influences microclimate and soil conditions that are important to species composition and abundance of smaller organisms, vascular plants, and tree regeneration. Changing the nutrient levels available to plants, especially nitrogen, and increasing soil temperatures could result in flora and fauna community assemblage changes. A shift in species may have cascading effects across trophic levels, and cause shifts in the size, distribution, and vertical zonation of vegetation over large areas, especially if slash removal is done on a large scale throughout Wisconsin's forests. Results from this study will provide base-line environmental information on the short-term response of flora and fauna to the removal of FWD; this information does not currently exist for systems in the Lake States.

Site Description

This study was conducted on approximately 365 ha of northern hardwood forest on the Lakewood/Laona District, Chequamegon-Nicolet National Forest (Fig. 1.1). The approximately 365 ha study site is primarily an even-aged, second-growth mature forest with an overstory composed mostly of sugar maple (*Acer saccarum*) and American basswood (*Tilia americana*) with scattered yellow birch (*Betula alleghaniensis*), ash (*Fraxinus* spp.) and eastern hemlock (*Tsuga canadensis*). Butternut (*Juglans cinerea*) is scattered in the northern sites. Poletimber and small sawtimber predominated (i.e., 8 – 12 inches in diameter) with few larger trees, and small canopy gaps existed pre-harvest. Stands were last harvested in 1998. For this study, stocking rates were reduced to 80-85 ft^2, and most timber was taken from the small sawtimber size.

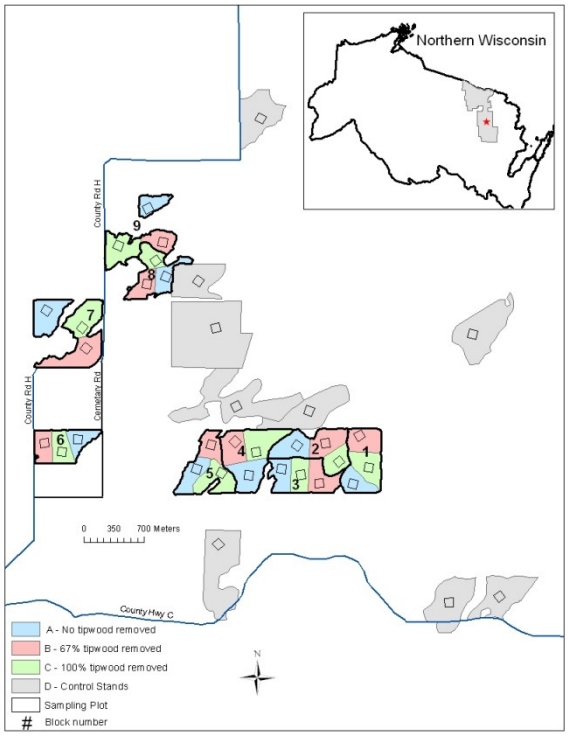


Figure 1.1. Study sites in northern hardwood forests, Lakewood/Laona Ranger District, Chequamegon-Nicolet National Forest, northern Wisconsin.

Experimental Design

Woody biomass harvesting guidelines and techniques must be practical for foresters and loggers to implement in the field. When this study began, the Chequamegon-Nicolet National Forest (CNNF) and the Wisconsin Department of Natural Resources (WDNR) did not have established woody biomass harvesting guidelines. We used existing Minnesota guidelines to establish biomass removal treatment levels. To achieve the recommended 1/3 FWD retention on a site, Minnesota guidelines recommend retaining 20% of the FWD by leaving the tops and limbs (i.e., tipwood) from one "average sized" tree out of every five trees harvested, and the remaining 10 – 15% received from incidental breakage during skidding. Based on these guidelines, we evaluated the response of soil carbon and nitrogen, and biotic species (i.e., forbs, tree regeneration, arthropods, and salamanders/frogs) abundance and diversity to four FWD removal treatments:

A. 0% *tipwood removed; 100% retention* (current practice; loggers retained all tipwood from harvested trees on site and was scattered consistent with standard operating practices).

B. *65% tipwood removed* (*intermediate retention*; equates to 1/3 tipwood retained on site after harvest based on Minnesota removal guidelines; four out of every five trees were removed (20% retained on forest floor) plus incidental breakage (13-15%) and remaining tipwood was scattered consistent with standard operating practices).

C. *100% tipwood removed; 0% retention* (tipwood from every tree designated for harvest was removed from the site; some tipwood remained on site due to incidental breakage during skidding)

D. Control (no cut)

Nine blocks were located within a northern hardwood forest scheduled to be harvested (Fig. 1.1). The location of the blocks within the planned harvest area was determined in consultation with district foresters and planners to avoid sensitive ecological and archeological sites, and to ensure adequate access points for the logging operations. We used a randomized complete block experimental design; each FWD removal treatment was randomly assigned to a >8 ha section of each block. Treatment sites ranged from 8.5 – 17.4 ha. Nine no-cut treatment sites were established as close as possible to the harvested blocks (Fig. 1.1). Controls were northern hardwood stands with approximately the same basal area, stand history, soil types, and Kotar habitat types.

Within each treatment site, we established a 100 x 100 m (10,000 m^2) plot near the center to ensure sufficient spacing and independence of replicates, and to minimize edge effects from the surrounding forest and other treatment areas. Within each plot, four transects spaced 33.3 m apart were run the length of the plot (Fig. 1.2). Before going into the field, we used a geographic information system to place 4 points over the treatment area; we then randomly selected one of the 4 points to establish the starting point for establishing transects. This point was transect 1, distance 0. After locating the point in the field using GPS, we then spun a compass while not looking at it to determine transect direction.

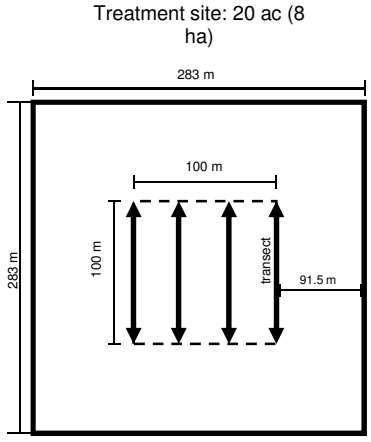


Figure 1.2. Illustration of four 100 m transects placed within 100 x 100 m plots located near the center of treatment sites (> 8 ha) creating a 10,000 m^2 sampling plot.

Using larger block sizes ensured that the proposed harvest and treatments were economically feasible for loggers, and it emulated the size of an area that woody biomass harvesting will likely take place. In addition, plot size was sufficient to incorporate the within plot variability of FWD caused by the unequal spacing between individual trees.

Soil, herpetofauna (i.e., amphibians), insect, herbaceous plants and tree regeneration sampling was conducted along each transect (Fig. 1.3; see Methods sections under each chapter for more detail) during May-August, 2009 (pre-treatment), 2010 (1-year post-treatment), and 2011 (2-years post-treatment).

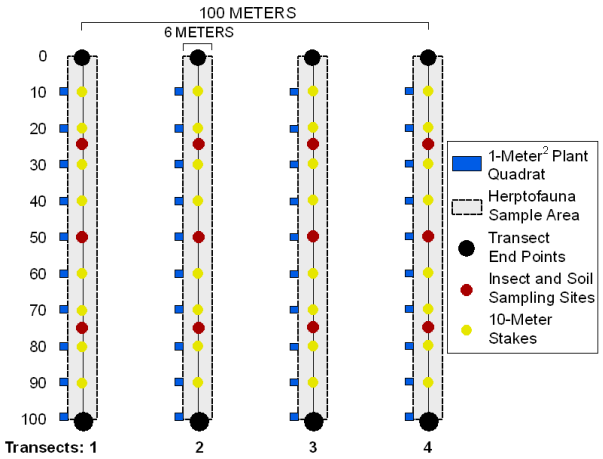


Figure 1.3. Illustration of soil and biotic sampling design along transects within treatment plots. See Methods section listed under each chapter for detailed description of sampling techniques.

Treatment Details

Forest woody biomass (i.e., FWD or tipwood) was removed during a winter-only selection harvest, November 2009 – March 2010 using individual tree selection and whole-tree harvesting to promote uneven-aged conditions. All applicable Forest Plan Standards and Guidelines, Wisconsin State Best Management Practices (BMP), and Timber Sale contract provisions were implemented during harvest activities. For blocks 1- 6, trees were harvested using a processor that cut-to-length merchantable timber and separated the tipwood into small piles throughout the treatment site (i.e., tipwood was piled from 5 trees and then required tipwood removal treatment levels were skidded to landings with forwarders). For blocks 7-9, designated trees for harvest were hand cut with chainsaws and the whole tree was skidded to landings where fowarders sorted roundwood and piled tipwood at the landings. Woody biomass at landings was later chipped and hauled to nearby mills (Fig. 1.4). By removing forest woody biomass in a manner that resembled techniques used by logging operations, our results have direct and immediate application to land managers and policy makers.

Figure 1.4. Woody biomass or tipwood stacked at a landing that was later chipped on site and hauled to a local mill.

Quality Assurance

To determine if FWD treatments were applied appropriately, we sampled percent cover of FWD within 1 m^2 quadrats spaced every 10 m along each transect for a total of 40 quadrats per plot (see chapter 3 Methods for a detailed description). Fine woody debris was woody material < 4 inches in diameter. We found FWD cover retained on the forest floor to be representative of the treatment removal levels 1-year post-harvest (Fig. 1.5). All treatment levels increased in FWD cover with treatment A (0% removal treatment in Fig. 1.5) having the largest increase and greatest amount as expected; FWD cover remained the same across years for the control stands.

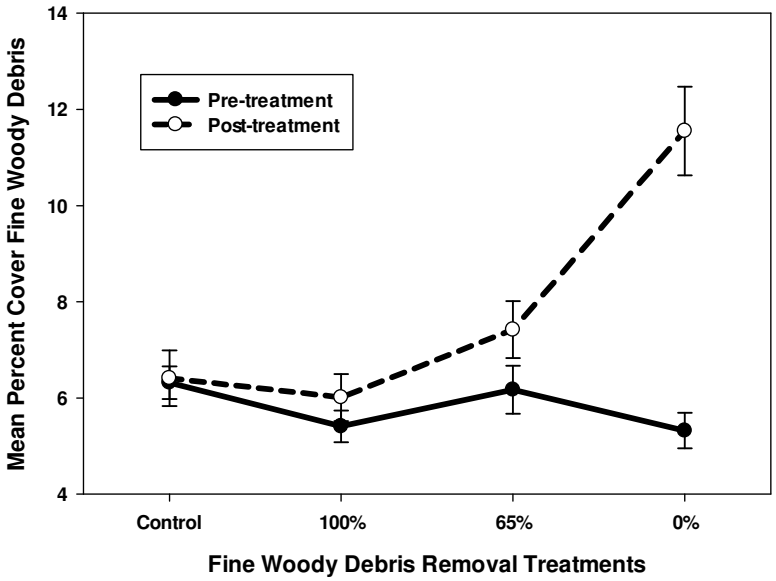


Figure 1.5. Mean percent cover of fine woody debris pre- and 1-year post-harvest.

Literature Cited

Harmon ME, JF Franklin, FJ Swanson, P Sollins, SV Gregory, JD Lattin, NH Anderson, SP Cline, NG Aumen, JR Sedell, GW Lienkaemper, K Cromack Jr., and KW Cummins. 1986. Ecology of coarse woody debris in temperate ecosystems. Advances in Ecological Research. 1986. Department of Forest Science. Oregon State University. Corvallis, Oregon 97331, USA. Vol 15, pp. 133-302.

Maser, C, RG Anderson, K Cromack, Jr., JT Williams, and RE Martin. 1979. Dead and down woody material. Pages 78 – 103 In Thomas JW (editor), Wildlife habitats in managed forests: the Blue Mountains of Oregon and Washington. USDA Forest Service, Agriculture Handbook No. 553, Washington, D.C.

CHAPTER 2

Amphibian assemblage response to forest residue removal treatments

Deahn M. Donner and Christine A. Ribic

Background

Amphibians are the most abundant vertebrate group in many forest ecosystems and play a key role in forest nutrient cycling by regulating populations of soil invertebrates that breakdown organic material (Burton and Likens 1975, deMaynadier and Hunter 1995). However, amphibians have relatively poor dispersal capabilities compared to other vertebrates (Sinsch 1990), so local disturbances to habitat can affect amphibian abundances. Little information exists on the impacts of uneven-aged forestry practices on amphibian populations during their terrestrial life-stage, and even less is known on the impacts of removing woody material during harvest. Large woody debris is an important structural habitat feature to amphibians providing mating sites, nesting cover, feeding, and thermoregulation (Greenberg 2001). However, it is not well known how amounts of fine woody debris influence amphibian populations. Given the importance of woody material to amphibian populations, we hypothesized that anuran and Plethodon salamander numbers would decline on treatment sites with less fine woody debris due to microclimate temperature and moisture changes.

Methods

Amphibian sampling

We conducted visual encounter surveys (VES) (i.e., time-constrained searches) where individuals actively searched for animals during a set amount of time (Crump and Scott 1994). This technique is used most often to determine species richness because it is more likely to encounter rare species that are difficult to trap. It is also an effective method to estimate relative abundances in large forest areas of uniform habitats where visibility is good. We standardized our search time to a cumulative 180 person-minutes, and further subdivided this time to 45 person-minutes per transect to ensure the full plot was searched. All animals observed within 3 m of each side of the 100 m transects were recorded (Fig. 1.3) including the distance along the transect it was captured. Captured animals were measured (snout-to-vent and total length), and microhabitat data such as substrate type and location (e.g., under leaf litter) were recorded. Individuals were marked using the visible implant fluorescent elastomer (VIE) tagging system to record recaptures. Because of behavioral differences among species, each plot was sampled twice (May-June and August) each year.

Analyses

We modeled species abundance (i.e., the counts) as a function of site-specific environmental data and observation processes (detection probability). We used ‘unmarked’ R package (Fiske and Chandler 2011) to fit the models. We used open N-mixture models using the function pcount for our repeated count data (i.e., 2 sampling periods per year) to compare number of individuals across treatments for each year. Relative abundance combined total number of individuals captured from both survey periods. Being ectotherms, amphibian activity is influenced by climate conditions that can influence detectability, so we used maximum temperature and total

precipitation recorded from the nearest weather station Laona 6 SW on the sampling date.

Because control sites were not within block boundaries encompassing the 3 biomass removal treatments, we compared pre-treatment (2009) abundance data between the controls sites and a randomly selected treatment site within each block to determine if our treatment sites were representative of northern hardwood forests in the broader landscape. We found total amphibian abundance in the controls was significantly greater compared to the removal treatments ($p < 0.01$), so subsequent analyses compared relative abundances only among forest biomass removal treatments (i.e., A,B,C). The higher abundances of amphibians in the control stands that we found are similar to abundance patterns found after even-aged silvicultural practices, but little information exists on abundance patterns after uneven-aged silvicultural treatments (DeMaynadier and Hunter 1995).

Results and Conclusions

Eight amphibian species were captured across the 3 years (Table 3.1). Of the 4 frog and toad species, the wood frog (*Rana sylvatica*) was the most abundant species followed by American toad (*Bufo americanus*), spring peeper (*Pseudoacris crucifer*), and gray treefrog (*Hyla versicolor*). Of the 4 salamander species, the red-backed salamander (*Plethodon cinereus*) was the most abundant species captured followed by blue-spotted salamander (*Ambystoma laterale*), spotted salamander (*Ambystoma maculatum*), and central newt (*Notophthalmus virdescens*). These species represent the species pool expected in this upland, forested system. These findings are similar to other studies that have found abundant wood frog numbers in upland, wooded habitats. The low numbers of blue-spotted and spotted salamanders reflect the species' behavior of burrowing in soils and duff after migrating to the uplands from ephemeral breeding ponds (Oldfield and Moriarty 1994) making it difficult to encounter the salamanders during sampling.

Table 3.1. Number of individuals captured using visual encounter surveys across fine woody debris removal treatments and years (2009 – pretreatment, 2010- 1-year post-harvest, 2011 – 2-years post-harvest)

Species	n	Year	Treatment A 0% removal	Treatment B 65% removal	Treatment C 100 % removal	Total Individuals
American toad	590	2009	35	25	24	84
		2010	72	50	129	251
		2011	70	96	89	255
Gray treefrog	11	2009	0	0	0	0
		2010	0	0	0	0
		2011	3	4	4	11
Spring peeper	590	2009	63	117	48	228
		2010	14	46	88	148
		2011	64	76	74	214
Wood frog	1537	2009	134	107	90	331
		2010	40	69	73	182
		2011	312	301	411	1024
Red back salamander	181	2009	21	31	32	84
		2010	5	4	17	26

Species	n	Year	Treatment A	Treatment B	Treatment C	Total
		2011	6	38	27	71
Blue-spotted salamander	8	2009	0	2	2	4
		2010	0	0	0	0
		2011	0	2	2	4
Spotted salamander	80	2009	3	0	1	4
		2010	0	0	0	1
		2011	0	0	0	0
Central newt	1	2009	0	0	0	0
		2010	0	0	0	0
		2011	0	1	0	1

We captured more individuals 2-years post-harvest (n=895) compared to 1-year post-harvest (n=413) and pre-treatment (n=597). The low number of individuals caught in 2010 (1-year post-harvest) compared to 2009 (pre-) and 2011 (2-year post-harvest) may be due to drought conditions that year. As temperatures increased, our models showed detectability of red-backed salamanders decreased, but precipitation did not significantly influence detectability of this species. For the anurans and American toad, however, as temperatures and precipitation increased, detectability (or abundances) increased as well.

We found no significant difference in total number of amphibians captured among removal treatments in 2009 (pre-treatment), and similar results for red-backed salamanders, American toads, and wood frogs. Spring peepers, however, were more abundant in treatment B (intermediate biomass removal treatment; $p < 0.01$) during the pre-treatment year.

One-year post-harvest, we captured significantly more individuals in the 100% removal treatment (C) than the intermediate (B) and 0% removal treatments (A, $p < 0.01$). These abundance patterns were found for wood frogs and spring peepers. However, findings were different for the other species. For red-backed salamanders, significantly more individuals were found in only the 100% removal treatment ($p < 0.01$). Results for American toads was variable with significantly more individuals being captured in 100% FWD removal ($p < 0.02$) and significantly less individuals in the intermediate FWD removal treatment ($p < 0.02$) than the 0% removal treatment.

By 2-years post-harvest, significantly more individuals were captured in 100% tipwood removal treatment than the intermediate and 0% removal treatments ($p < 0.01$), this was also found for wood frogs. Wood frogs were most likely driving these results as this species represented 51% of total captures. Red-backed salamanders were significantly more abundant in 100% and intermediate levels of FWD removal treatments ($p < 0.01$). There was no significant differences in American toad and spring peeper captures among treatments.

Our findings did not support our initial hypothesis of a positive relationship between amphibian abundance and FWD retention on the forest floor. These patterns may be due to detectability issues related to our sampling method. Often, residue piles from the logging operation were left on the forest floor. These potentially large piles were difficult to search under for amphibians, so we may have under-estimated the number of individuals in the 100% FWD retention (0% removal) treatments. However, these piles may attract potential predators of amphibians (e.g., small mammals and snakes) resulting in decreased numbers of amphibians

within these treatments. Another explanation may be that FWD is too small to create favorable microclimate conditions similar to coarse woody debris, and differences in abundance may be related to the amount of coarse woody debris present prior to harvesting activity and CWD left on the forest floor after the current harvest. Many of the tops and limbs from harvested large trees were greater than 4 inches in diameter (pers. observation).

An alternative explanation may be the changes in microclimate conditions created after harvesting activities. Small canopy gaps are created from individual tree removal allowing more direct sunlight to reach the forest floor increasing microclimate temperatures and reducing moisture thereby increasing desiccation potential (Greenberg 2001). Additionally, residue piles were placed often in small openings between trees with open canopies and the lack of herbaceous plant cover creating warmer microhabitats. Heatwole (1962) found significantly cooler and moister conditions in the interior of shaded logs than unshaded logs where no salamanders were found. After 2-years post-harvest, the over-story canopy began to close through natural succession and herbaceous plants began growing around and among the residue piles creating greater shade and less difference among treatments in terms of microclimate conditions. These short-term, natural succession changes may explain spring peeper and American toad abundances becoming similar among FWD removal treatments after 2 years.

Literature Cited

Burton, TM and GE Likens. 1975. Energy flow and nutrient cycling in salamander populations in the Hubbard Brook Experimental forest, New Hampshire. Ecology 56:1068-1080.

Crump, ML and NJ Scott. 1994. Visual Encounter Surveys. Pages 84-92 *In* Heyer, WR, MA Donnelly, RW McDiarmid, LC Hayek, and MS Foster (editors), Measuring and monitoring biological diversity: standard methods for amphibians. Smithsonian Institution Press, Washington, 364 p.

DeMaynadier, PG and ML Hunger, Jr. 1995. The relationship between forest management and amphibian ecology: a review of the North American literature. Environmental Review 3:230-261.

Fiske, IJ and RB Chandler. 2011. unmarked: An R package for fitting hierarchical models of wildlife occurrence and abundance. Journal of Statistical Software 43:1-23.

Greenberg, CH. 2001. Response of reptile and amphibian communities to canopy gaps created by wind disturbance in the southern Appalachians. Forest Ecology and Management 148:135-144.

Heatwole H. 1962. Environmental factors influencing local distribution and activity of the salamander, Plethodon cinereus. Ecology 43:460-472.

Oldfield B and JJ Moriarty. 1994. Amphibians and reptiles native to Minnesota. The University of Minnesota Press. 237 p.

CHAPTER 3

Tree regeneration and herbaceous plant assemblage response to forest residue removal treatments

Deahn M. Donner, Christine A. Ribic, and Matthew StPierre

Background

Harvest forest residue represents a major source of downed woody debris in uneven-aged managed (i.e., single tree selection) hardwood forests, and helps to maintain levels comparable to unmanaged stands (Angers et al. 2005, Vanderwel et al. 2010). The importance of this material to many ecosystem functions including plant diversity is well documented, especially larger material (Harmon et al. , Miller et al. 2002), but less is known about the role fine woody material may have on maintaining ground-layer plants and tree regeneration. Ground-layer plants are highly sensitive to environmental conditions (Miller et al. 2002, Kern et al. 2006), and removing forest residues may impact species composition (i.e., diversity). Additionally, whole-tree harvesting rather than cut-to-length harvesting is employed when removing residue meaning small woody material is removed from the site that may allow more direct solar radiation to the forest floor. Increased sunlight may affect recruitment responses from the forest floor seed bank, and loss of insulating woody material may impact mortality of shade-tolerant tree seedlings (McInnis and Roberts 1994). The ground-layer plant community was found to shift from species representative of old-growth conditions to weedy and early successional species in an uneven-age managed second-growth northern hardwood stands in Wisconsin and Michigan (Scheller and Mladenoff 2002). Greater regeneration densities of shade-intolerant and mid-tolerant hardwoods were found after full-tree harvests of mixed stands (McInnis and Roberts 1994). Thus, we hypothesized that amounts of early successional forbs and regenerating tree species will differ across FWD removal treatments.

Methods

Vegetation sampling

Vegetation was measured within 1 m^2 quadrats placed at 10 m intervals along each 100 m transect (40 points per treatment plot; Fig. 1.3). The lower left corner of a quadrat was placed 1 m out perpendicularly from the main transect. Quadrat sampling has been found to record more species than other vegetation cover methods. The number of seedlings (<1 inch DBH) per tree species was counted within each quadrat. Cover of ground-layer vegetation (herbaceous and shrubs <1.3 m in height and < 1.0DBH) was estimated using 7 cover classes (0.1-1%, 1-5%, 6-15%, 16-25%, 26-50%, 51-75%, and 75-100%). Midpoints of cover classes were used in percent cover calculations similar to Kern et al. 2006 (e.g., fine woody debris percent cover; Fig. 1.5). Sampling was conducted May-June to detect and estimate spring ephemerals, and again in July-August for summer flowering species.

Analyses

For species richness, we included all species recorded at the plot level regardless of sampling period. We compared plot-level shrub, forb, and fern species richness across treatments and years using generalized linear models with a poisson link. Similar to amphibians, we compared pre-treatment species richness between control sites and randomly-selected treatment sites within

each block to determine if treatment sites were representative of the broader northern hardwood forest. We found plot-level forb and fern species richness was significantly greater in the treatment sites than control plots ($p < 0.05$); subsequent analyses were conducted comparing across treatment sites (i.e., A, B, C).

Frequency estimates were the number of quadrats occupied per plot by the species. There were 360 quadrats per treatment. We used results from the May-June survey for spring ephemerals. We used nonmetric multidimensional scaling (NMS) to compare composition of ground-layer plant communities (shrubs, herbaceous) among treatments (VEGAN R package for community ecology; Dixon 2003). For each ordination, species that occurred in < 20% of the plots were deleted.

Results and Conclusions

A total of 189 species were recorded including 20 trees, 21 shrubs, 22 fern and fern allies, 105 forbs, 17 grasses, and 13 sedges (Tables 3.1-3.5). The most frequent shrub species were elderberry, prickly gooseberry, and red raspberry. The most frequent forb species were sweet cicely, Virginia waterleaf, moss, trout lily, and wild lily-of-the-valley, and the most frequent ferns were Lady, silvery glade, and maidenhair.

Forb and fern species richness was not significantly different among treatments and across years. Due to the difficulty in identifying grasses and sedges to species, we did not compare species richness for these plant groups. Shrub species richness was significantly lower 1-year post-harvest compared to pre- and 2-year post-harvest ($p=0.04$). Only 7 species were found immediately post-harvest compared to 13-14 for pre- and 2-years post-harvest (Table 3.2). Dogwood species were less common post-harvest, while more *Ribes* and *Rubus* species were found 2-years post-harvest compared to pre-treatment; however these species were not recorded immediately after harvest as well (Table 3.2). Only alternate-leaved dogwood, beaked hazelnut, prickly gooseberry, red raspberry, and elderberry were found across all three years.

Sugar maple was the most common seedling found across treatments and years, but numbers declined post-harvest (Table 3.1). The second most common seedling recorded was white ash, but seedling numbers were similar to pre-harvest numbers 2-years following harvest. The number of seedlings for basswood and ironwood (mid-tolerant species) increased 2-years post-harvest across all treatments, while yellow birch and red maple numbers declined. American elm declined, but slippery elm increased 2-years post-harvest. This discrepancy may be identification errors; considering Ulmus spp. in general, regeneration remained similar across years and treatments. The similarity of tree regeneration across treatments suggests changes to regeneration are a result of the harvest rather than changes in FWD amounts. However, the number of seedlings was small for some species so it is unknown if our results are representative of the impacts FWD removal may have on regeneration of some of these species.

Table 3.1. Number of seedlings recorded by tree species during summer sampling period. Number represents the total number of seedlings counted from 360 1 m^2 quadrats per treatment each year.

				Treatment			
CODE	Common name	Scientific name	Year	A	B	C	Total
ABBA	Balsam fir	*Abies balsamea*	2009	1	1	0	2
			2010	1	1	0	2
			2011	2	0	0	2
ACRUR	Red maple	*Acer rubrum*	2009	7	3	9	19
			2010	0	1	1	2
			2011	1	2	3	6
ACSAS	Sugar maple	*Acer saccharum*	2009	183	198	219	600
			2010	117	132	127	376
			2011	138	148	159	445
BEAL2	Yellow birch	*Betula alleghaniensis*	2009	6	7	1	14
			2010	0	1	0	1
			2011	2	1	1	4
BEPA	Paper birch	*Betula papyrifera*	2009	0	0	0	0
			2010	2	0	0	2
			2011	0	0	0	2
CACAV	American hornbeam	*Carpinus caroliniana*	2009	2	2	0	4
			2010	0	0	0	0
			2011	0	0	0	0
CACO15	Bitternut hickory	*Carya cordiformis*	2009	3	2	4	9
			2010	0	3	2	5
			2011	5	3	2	10
FRAM2	White ash	*Fraxinus Americana*	2009	65	86	65	216
			2010	53	56	46	155
			2011	56	83	66	205
FRPE	Green ash	*Fraxinus pensylvanica*	2009	0	1	0	1
			2010	0	0	0	0
			2011	0	0	0	0
JUCI	Butternut	*Juglans cinerea*	2009	0	0	2	2
			2010	1	0	0	1
			2011	1	0	0	1
OSVI	Ironwood	*Ostrya virginiana*	2009	19	16	13	48
			2010	19	8	8	35
			2011	26	39	34	99
POGR4	Big-tooth aspen	*Populus grandidentata*	2009	0	0	0	0
			2010	0	1	0	1
			2011	0	1	0	1
POTR5	Quaking aspen	*Populus tremuloides*	2009	0	0	0	0
			2010	0	0	0	0

			2011	0	4	1	5
PRSE2	Black cherry	*Prunus serotina*	2009	37	16	16	69
			2010	31	13	19	63
			2011	31	16	18	65
QUAL	White oak	*Quercus alba*	2009	0	0	0	0
			2010	0	0	0	0
			2011	0	1	0	1
QURU	Northern red oak	*Quercus rubra*	2009	1	1	2	4
			2010	0	0	3	3
			2011	0	1	2	3
TIAMA	Basswood	*Tilia Americana*	2009	2	5	9	16
			2010	1	2	5	8
			2011	8	12	11	31
TSCA	Eastern hemlock	*Tsuga Canadensis*	2009	1	1	0	2
			2010	0	0	0	0
			2011	0	0	0	0
ULAM	American elm	*Ulmus Americana*	2009	7	14	15	36
			2010	1	0	1	2
			2011	0	1	2	3
ULRU	Slippery elm	*Ulmus rubra*	2009	0	0	0	0
			2010	4	9	14	27
			2011	1	8	5	14

Table 3.2. Shrub species[a] identified during spring or summer sampling periods.

Code	Common Name	Scientific Name	2009	2010	2011
ACSP2	Mountain maple	*Acer spicatum*	X		
CORNU	Dogwood	*Cornus spp.*	X		
COAL2	Alternate leaved-dogwood	*Cornus alternifolia*	X	X	X
CORU	Roundleaf dogwood	*Cornus rugosa*	X		
COAM3	American hazelnut	*Corylus americana*	X		X
COCO6	Beaked hazelnut	*Corylus cornuta*	X	X	X
DILO	Bush honeysuckle	*Diervilla lonicera*	X	X	
DIPA9	Leatherwood	*Dirca palustris*	X	X	X
RUST	Rubus strigosus	*Grayleaf red raspberry*	X		
LONIC	Honeysuckle	*Lonicera*			X
LOCA7	Northern fly honeysuckle	*Lonicera canadensis*	X		X
PAQU2	Virginia creeper	*Parthenocissus quinquefolia*	X		
PRPE2	Pin cherry	*Prunus pensylvanica*			X
RICY	Prickly gooseberry	*Ribes cynosbati*	X	X	X
RIHI	Hairy-stem gooseberry	*Ribes hirtellum*	X		X
RILA	Bristly/spiny swamp-currant	*Ribes lacustre*			X
RUAL	Allegheny blackberry	*Rubus alleghaniensis*			X
RUIDS2	Red raspberry	*Rubus ideaus*	X	X	X
SACA12	Elderberry	*Sambucus canadensis*	X	X	X
SACAR3	Red elderberry	*Sambucus racemosa spp.*	X		X
VIBUR	Viburnum	*Viburnum*			X

[a] Taxa were referenced in the USDA Plants Database (USDA, 2005).

Table 3.3. Fern and fern ally species[a] identified during spring or summer sampling period.

Code	Common Name	Scientific Name	2009	2010	2011
ADPE	Maidenhair fern	*Adiantum pedantum*	X	X	X
ATFIA	Lady fern	*Athyrium filix-femina*	X	X	X
BOLAA2	Lance-leaved grape fern	*Botrychium lanceolatum*	X	X	
BOMA2	Daisy-leaf grape fern	*Botrychium matricarifolium*			X
BOSI	Little moonwort	*Botrychium simplex var. simplex*	X	X	
BOTRY	Grapefern	*Botrychium spp.*			X
BOVI	Rattlesnake fern	*Botrychium virginianum*	X	X	X
DEAC4	Silvery glade fern	*Deparia acrostichoides*	X	X	X
DICOX0	Northern running pine	*Diphasiastrum complanatum*			X
DRCAXX	Toothed wood fern	*Dryopteris carthusiana*	X	X	X
DRCR4	Crested wood fern	*Dryoptersis cristata*	X	X	
DRGO	Goldie's wood fern	*Dryopteris goldiana*			X
DRIN5	Fancy wood fern	*Dryoptersis intermedia*	X	X	X
EQAR	Field horsetail	*Equisetum arvense*			X
EQHY	Scouringrush horsetail	*Equisetum hyemale*	X	X	
EQSY	Woodland horsetail	*Equisetum sylvaticum*	X		
EQUIS	Horsetail	*Equisetum spp.*		X	X
GYDR	Oak fern	*Gymnocarpium dryopteris*	X	X	X
MAST	Ostrich fern	*Matteuccia struthiopteris*	X	X	X
ONSE	Sensitive fern	*Onoclea sensibilis*	X	X	X
OSCL2	Interrupted fern	*Osmunda claytoniana*	X	X	X
PHCO24	Northern beech fern	*Phegopteris connectilis*	X	X	X
PTAQL	Bracken fern	*Pteridium aquilinum*	X	X	X

[a] Taxa were referenced in the USDA Plants Database (USDA, 2005).

Table 3.4. Grass and sedge species[a] identified during spring or summer sampling periods.

Code	Common Name	Scientific Name	2009	2010	2011
Grasses					
AGGI2	Red top	*Agrostis gigantea*		X	
AGHY	Tickle grass	*Agrostis hyemalis*	X		
BRER2	Bearded shorthusk	*Brachyelytrum erectum*	X	X	X
BRCI2	Fringed brome	*Bromus ciliatus*		X	
BRPU6	Hairy woodland brome	*Bromus pubescens*	X	X	
CACA4	Blue joint grass	*Calamagrostis candensis*			X
CINNA	Woodreed	*Cinna*	X		
ELHY	Bottlebrush grass	*Elymus patula*	X	X	X
FESU3	Nodding fescue	*Festuca subverticillata*	X	X	X
GLST	Fowl mannagrass	*Glyceria striata*			X
MEIF	Milium grass	*Milium effusum*			X
ORAS	Rice-cut grass	*Oryzopsis asperifolia*	X	X	X
PHAR3	Reed-canary grass	*Phalaris arundinacea*	X	X	X
PIRA5	Blackseed ricegrass	*Piptatherum racemosum*			X
POAL3	Bluegrass	*Poa alsodes*	X	X	X
POPR	Kentucky blue grass	*Poa pratensis*	X	X	
SCPU	False melic	*Schizachne purpurascens*	X	X	X
GRASS	Grass unknown		X	X	X
Sedges					
CAAR3	Drooping woodland sedge	*Carex arctata*			X
CAAS2	Walking sedge	*Carex assiniboinensis*	X		X
CACE	Oval-leaf sedge	*Carex cephalophora*	X		
CARO22	Curly-styled wood sedge	*Carex convoluta*			X
CACRC2	Fringed sedge	*Carex crinita*	X		X
CADE9	Dewey sedge	*Carex deweyana*	X	X	X
CAGR2	Graceful sedge	*Carex gracillima*	X	X	X
CAINX2	Greater bladder sedge	*Carex intumescens*	X	X	X
CALE11	Few-nerved wood sedge	*Carex leptonervia*		X	X
CAPE11	Peck's sedge	*Carex peckii*		X	
CAPE6	Pennsylvania sedge	*Carex pensylvanica*	X	X	X
CAPL4	Plantain-leaf sedge	*Carex plantaginea*	X	X	X
CAREX	Sedge	*Carex spp.*	X	X	X
CASP7	Sprengel's sedge	*Carex sprengelii*			X

[a] Taxa were referenced in the USDA Plants Database (USDA, 2005).

Table 3.5. Forb species[a] identified during spring or summer sampling periods.

Code	Common Name	Scientific Name	2009	2010	2011	Code	Common Name	Scientific Name	2009	2010	2011
ACPA	White baneberry	*Actaea pachypoda*	X	X	X	MACA4	Wild-lily-of-the-valley	*Maianthemum canadense*	X	X	X
ACRU2	Red baneberry	*Actaea rubra*	X		X	MARAR	False Solomon's seal	*Maianthemum racemosum*	X	X	X
ACTAE	Baneberry	*Actaea sp.*	X	X	X	MAST4	Starry false lily-of-the-valley	*Maianthemum stellatum*		X	
AMBR2	Hog-peanut	*Amphicarpaea bracteata*	X	X	X	MEAL	White sweet clover	*Melilotus alba*			X
ANQU	Wood anemone	*Anemone quinquefolia*	X	X	X	MIDI3	Two-leaf miterwort	*Mitella diphylla*	X	X	X
APAN2	Spreading dogbane	*Apocynum androsaemifolium*			X	MOUN3	Indian pipe	*Monatropa uniflora*			
ARGL	Tower rockcress	*Arabis glabra*			X	MOSS	Moss	*Moss*	X	X	X
ARNU2	Wild sarsaparilla	*Aralia nudicaulis*	X	X	X	MYAQ	Giantchickweed	*Myosoton aquaticum*			X
ARRA	Spikenard	*Aralia racemosa*	X	X	X	OSCL	Sweet cicely	*Osmorhiza claytonii*	X	X	X
ARCTI	Burdock	*Arctium*			X	PATR4	Dwarf ginseng	*Panax trifolius*			X
ARTR	Jack-in-the-pulpit	*Arisaema triphyllum*	X	X	X	PARTH3	Creeper	*Parthenocissus Planch.*	X	X	X
ASCA	Wild ginger	*Asarum canadense*	X	X	X	PHLE5	Lop-seed	*Phryma leptostachya*	X	X	X
ASEX	Poke milkweed	*Asclepias exaltata*		X	X	POBI2	Greater Solomon's seal	*Polygonatum biflorum*	X		
ASMA2	Big-leaf aster	*Aster macrophyllus*	X	X	X	POPU4	Hairy Solomon's seal	*Polygonatum pubescens*	X	X	X
ASTER	Aster	*Aster sp.*	X	X	X	PRAL2	White rattlesnake root	*Prenanthes alba*	X		
BUUM	Flowering rush	*Butomus umbellatus*	X			PRVU	Self-heal	*Prunella vulgaris*			X
CACO26[b]	Five-parted anemone	*Cardamine concatenata*	X	X	X	RAAB	Small Flowered crowsfoot	*Ranunculus abortivus*		X	
CADI10[b]	Broad-leaved cardamine	*Cardamine diphylla*	X	X	X	RARE2	Hooked buttercup	*Ranunculus recurvatus*			
CATH2	Blue cohash	*Caulophyllum thalictroides*	X	X	X	RARH	Labrador buttercup	*Ranunculus rhomboideus*	X		
CHUM	Pipsissewa	*Chimaphilla umbellata*		X		SAGR	Grassy arrowhead	*Sagittaria graminea*	X		
CILU	Enchanter's night-shade	*Circaea quadrisulcata*	X	X	X	SARAR3	Red elderberry	*Sambucus racemosa*			X
CIPA6	Marsh thistle	*Cirsium palustre*		X		SACA13	Bloodroot	*Sanguinaria canadensis*	X	X	X
CLVI3[b]	Spring beauty	*Claytonia viginica*	X	X	X						
CLVI5	Devil's darning needles	*Clematis virginiana*		X		SAMA2	Black snakeroot	*Sanicula marilandica*	X	X	X
CLBO3	Bluebead lily	*Clintonia borealis*	X	X	X	SAOD	Clustered blacksnakeroot	*Sanicula odorata*	X		
COCA13	Bunchberry	*Cornus canadensis*				SCLA2	Blue skullcap	*Scutellaria lateriflora*			X
CRCA9	Honewort	*Cryptotaenia canadensis*			X	SOLID	Goldenrod	*Solidago*	X		X
DICA[b]	Squirrel corn	*Dicentra canadensis*	X	X	X						
DICU[b]	Dutchman's breeches	*Dicentra cucullaria*		X		SOAL6	Canada goldenrod	*Solidago altissima*	X		X
DOUMU	Parasol whitetop	*Doellingeria umbellata*			X	SOFL2	Zigzag goldenrod	*Solidago flexicaulis*	X	X	X
EPRE2	Trailing arbutus	*Epigaea repens*				SORU2	Wrinkleleaf goldenrod	*Solidago rugosa*			
ERAM5[b]	Trout lily	*Erythronium americanum*	X	X	X						
ERAN	Eastern daisy fleabane	*Erigeron annuus*			X	STRO4	Rosy twisted stalk	*Streptopus roseus*	X	X	X
ERCH9	Wormseed wallflower	*Erysimum cheiranthoides*		X		SYLA6	White panicle aster	*Symphyotrichum lanceolatum*	X	X	
FRVE	Woodland strawberry	*Fragaria vesca*	X	X		SYMPH4	Aster	*Symphyotrichum nees*			X
FRVI	Strawberry	*Fragaria virginianum*	X	X		SYUR	Arrow-leaved aster	*Symphyotrichum urophyllum*	X	X	X
FUNGI	Fungi	*Fungi*	X	X	X	TAOF	Dandelion	*Taraxacum officinale*	X	X	X
GATE2	Brittlestem hempnettle	*Galeopsis tetrahit*	X	X	X	THDI	Early meadow-rue	*Thalictrum dioicum*	X	X	X
GALA3	Bedstraw	*Galium lanceolatum*			X	TRBO2	Starflower	*Trientalis borealis*	X	X	X

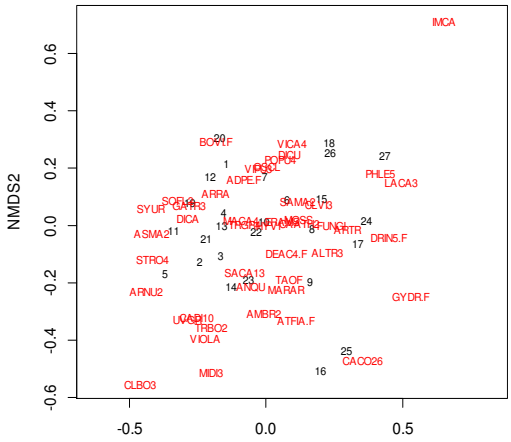
Code	Common Name	Scientific Name	2009	2010	2011	Code	Common Name	Scientific Name	2009	2010	2011
GATR3	Sweet-scented bedstraw	*Galium triflorum*	**X**	**X**	**X**	TRIFO	Clover	*Trifolium*			**X**
GAPR2	Wintergreen	*Gaultheria procumbens*			**X**	TRCE[b]	Nodding trillium	*Trillium cernuum*	**X**	**X**	**X**
GEAL3	Yellow avens	*Geum aleppicum*		**X**		TRGR4[b]	Large trillium	*Trillium grandiflorum*	**X**	**X**	**X**
GECA7	White avens	*Geum canadense*			**X**	UVGR	Bellwort	*Uvularia grandiflora*	**X**	**X**	**X**
HENOA	Sharplobe hepatica	*Hepatica nobilis*	**X**	**X**	**X**	UVSE	Sessileleaf bellwort	*Uvularia sessilifolia*	**X**	**X**	
HIERA	Hawkweed	*Hieracium*			**X**	VETH	Common mullein	*Verbascum thapsus*		**X**	**X**
HIAU	Orange hawkweed	*Hieracium aurantiacum*	**X**		**X**	VERON	Speedwell	*Veronica*			**X**
HICA10	Yellow hawkweed	*Hieracium caespitosum*		**X**	**X**	VEOF2	Common gypsyweed	*Veronica officinalis*	**X**	**X**	
HYVI[b]	Virginia waterleaf	*Hydrophyllum virginianum*	**X**	**X**	**X**	VIRA	Downy arrowwood	*Viburnum rafinesqueanum*			**X**
IMCA	Jewelweed	*Impatiens capensis*	**X**	**X**	**X**	VICA4[b]	Canada violet	*Viola canadensis*	**X**	**X**	**X**
LACA	Canada lettuce	*Lactuca canadensis*			**X**	VIPU4[b]	Yellow violet	*Viola pubescens*	**X**	**X**	**X**
LACA3	Canadian wood nettle	*Laportea canadensis*	**X**	**X**	**X**	VISO[b]	Woody blue violet	*Viola sororia*			**X**
LIOF	European gromwell	*Lithospermum officinale*			**X**	WAFR	Barren strawberry	*Waldsteinia fragariodes*			**X**
LYCO	Rose campion	*Lychnis coronaria*		X							

[\a] Taxa were referenced in the USDA Plants Database (USDA, 2005).

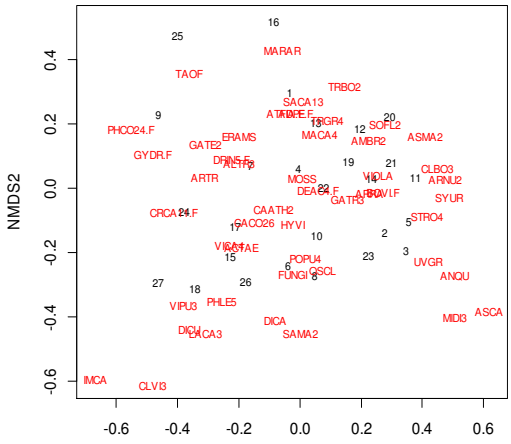
[b] spring ephemeral

We did not find a change in community composition of forbs and ferns across treatments and years (i.e., treatment sites were well distributed and interspersed along NMD axes; Fig. 3.1). Several individual plots, however, changed positions along one or more of the NMD axes after harvest. For example, site 16, which was treatment B (intermediate FWD removal) recorded more false solomon seal and oak fern post-harvest. Our results are comparable to Kern et al. (2006) who also did not find significant changes to community composition in ground-layer plant species between even-age and uneven-age silvicultural treatment in Wisconsin northern hardwood forests. Brosofske et al. (2001) found that higher order factors such as overstory type influenced understory vegetation richness more than site variables such as canopy cover, soil pH and forest floor characteristics. Because our FWD removal treatments were applied in northern hardwood forest stands with similar silvicultural practice treatments (i.e., similar thinning), factors such as canopy cover and pH may still be acting at a higher level than FWD in ground-layer plant species diversity.

2009



2010



2011

Figure 3.1. NMS ordination of ground-layer forb and fern plant communities for FWD removal treatments within a northern hardwood forest. Numbers 1-9 represent treatment A sites (0% removal), numbers 10-18 represent treatment B sites (65% removal), and numbers 19-27 represent treatment C sites (100% removal). Refer to Tables 3.3 and 3.5 for plant species code (red).

Literature Cited

Angers, VA, C Messier, M Beaudet, and A. Leduc. 2005. Comparing composition and structure in old-growth and harvested (selection and diameter-limit cuts) northern hardwood stands in Quebec. Forest Ecology and Management 217:275-293.

Brosofske KD, J Chen, and TR Crow. 2001. Understory vegetation and site factors: implications for a managed Wisconsin landscape. Forest Ecology and Management 146:75-87.

Dixon, P. 2003. VEGAN, a package of R funcitons for community ecology. Journal of Vegetation Science 14:927-930.

Kern, CC, BJ Palik, and TF Strong. 2006. Ground-layer plant community responses to even-age and uneven-aged silvicultural treatments in Wisconsin northern hardwood forests. Forest Ecology and Management 230:162-170.

Harmon ME, JF Franklin, FJ Swanson, P Sollins, SV Gregory, JD Lattin, NH Anderson, SP Cline, NG Aumen, JR Sedell, GW Lienkaemper, K Cromack Jr., and KW Cummins. 1986. Ecology of coarse woody debris in temperate ecosystems. Advances in Ecological Research. 1986. Department of Forest Science. Oregon State University. Corvallis, Oregon 97331, USA. Vol 15, pp. 133-302.

McInnis BG and MR Roberts. 1994. The effects of full-tree and tree-length harvests on natural regeneration. Northern Journal of Applied Forestry 11:131-137.

Miller TF, DJ Mladenoff, and MK Clayton. 2002. Old-growth northern hardwood forests: spatial autocorrelation and patterns of understory vegetation

Scheller, RM, and DJ Mladenoff. 2002. Understory species patterns and diversity in old-growth and managed northern hardwood forests. Ecological Applications 12:1329-1343.

USDA, NRCS, 2005. The PLANTS Database, version 3.5. National Plant Data Center, Baton Rouge, LA USA.

Vanderwel MC, HC Thorpe, JL Shuter, JP Caspersen, and SC Thomas. 2008. Contrasting downed woody debris dynamics in managed and unmanaged northern hardwood stands. Canadian Journal of Forest Research 38:2850-2861.

CHAPTER 4

Beetle (Coleoptera) assemblage response to forest residue removal treatments

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Background

Downed and decaying wood is important to biodiversity of beetles (Siitonen 2001). Invertebrate species dependent on dead wood are considered saproxylic organisms (Speight 1989). Removing woody biomass from systems is considered the most severe threat to saproxylic insect species (Grove 2002). A reduced saproxylic beetle fauna is of concern because of their importance to decomposition and nutrient cycling (Speight 1989). From a forest management perspective, providing a diversity of substrates will maintain beetle diversity (Siitonen 2001, McGeoch et al. 2007). Saproxylic insect assemblages are known to differ between different substrate types such as logs, low and high stumps due to different feeding guilds (Gibb et al. 2006, Hjältén et al. 2010). Removing substrates such as logging residue may affect microhabitat complexity and indirectly impact beetle populations (Gunnarsson et al 2004). We hypothesized that treatment sites with less FWD will have reduced species richness due to less woody structure, and also have lower numbers of early successional beetle species (i.e., cambium consumers) and more fungivores and predators given the availability of older coarse woody debris available.

Methods

Insect Sampling

To sample the beetle fauna, pitfall traps were installed at 25, 50, and 75 m along each transect totaling 12 traps per plot (Fig. 1.3). Pitfall traps were 12-oz. plastic cup dug into the soil to soil surface level. Funnels were placed on top of the plastic cups and smaller collection cups inside the plastic cups were filled with a 50:50 mix of water and propylene glycol, which acts as a killing agent and preservative (per D. Coyle, *personal communication*). Traps were run 3 days per month, May – August each year. In total, there were 108 trap nights per treatment giving 324 trap nights per sampling period and 1,296 trap nights per year. Between sampling dates, traps were closed by turning the plastic cup upside-down in the hole. All insect samples were collected, transferred, and stored in 70% EtOH for sorting in the laboratory.

Analyses

Because some traps were disturbed by bears, we used only 6 samples per plot to standardize trapping effort. Samples from transects 1 and 3 were used initially, then samples from transect 2 25m progressing to 75 m were used until reaching 6 samples. We modeled plot-level richness (i.e., number of Families present) as a function of site-specific environmental data and observation processes (detection probability) similar to the amphibian analyses (see Methods chapter 2). We used 'unmarked' R package (Fiske and Chandler 2011) to fit the models. We used open N-mixture models using the function pcount for our repeated count data (i.e., 4 sampling periods per year) to compare number of Families represented across treatments for each year. Additionally, we compared relative abundance of the 4 most common beetle families to determine if abundances were different among treatments We were especially interested in weevil number as this group of species are plant eaters and their numbers may impact the plant

community. We used maximum temperature and total precipitation recorded for the day prior to collecting samples and closing traps.

Table 4.1. Coleoptera families identified during pitfall trapping, May – August, 2009 – 2011. Treatment A is 0% harvest residue removed, B is intermediate residue removed, and C is 100% harvest residue removed.

Family	Common description	n	Year	Treatment			
				A	B	C	Total
Curculionidae (superfamily)	weevil	816	2009	110	100	121	331
			2010	103	93	121	317
			2011	66	50	52	168
Staphylinidae	rove beetle	679	2009	141	123	141	405
			2010	77	68	63	208
			2011	27	21	18	66
Elaterida	click beetle	80	2009	14	11	7	32
			2010	9	10	8	27
			2011	6	8	7	21
Lampyridae	firefly	17	2009	3	4	4	11
			2010	4	1	1	6
			2011	0	0	0	0
Cerambycidae	longhorned beetle	8	2009	2	2	2	6
			2010	0	0	0	0
			2011	0	1	1	2
Coccinellidae	ladybug	7	2009	3	1	1	5
			2010	0	1	1	2
			2011	0	0	0	0
Histeridae	hister beetle	17	2009	3	5	4	12
			2010	2	0	1	3
			2011	0	1	1	2
Scolytidae	bark beetle	139	2009	43	55	41	139
			2010	0	0	0	0
			2011	0	0	0	0
Meloidae	blister beetle	12	2009	6	2	1	9
			2010	1	1	0	2
			2011	0	1	0	1
Silphidae	carrion beetle	64	2009	15	10	5	30
			2010	11	9	9	29
			2011	3	0	2	5
Scarabaeidae	scarab beetle	71	2009	20	14	23	57
			2010	1	4	2	7
			2011	3	1	3	7
Nitidulidae	sap beetle	3	2009	2	1	0	3

			2010	0	0	0	0
			2011	0	0	0	0
Calligrapha	leaf beetle	3	2009	2	1	0	3
			2010	0	0	0	0
			2011	0	0	0	0
Buprestinae	wood boring beetles	3	2009	2	1	0	3
			2010	0	0	0	0
			2011	0	0	0	0
Carabidae	ground beetles	925	2009	139	103	135	377
			2010	139	126	116	381
			2011	63	55	49	167
Cicindelidae	tiger beetles	3	2009	0	0	0	0
			2010	2	0	0	2
			2011	1	0	0	1
Contharidae	soldier beetles	4	2009	0	0	0	0
			2010	1	1	1	3
			2011	0	1	0	1
Bostrichoidea	spider beetles	3	2009	0	0	0	0
			2010	1	1	1	3
			2011	0	0	0	0
			2011	0	0	1	1

Results and Conclusions

In total, 2,854 beetles belonging to 18 families were collected (Table 4.1). The most abundant beetles were ground beetles (Family Carabidae) followed by weevils (Superfamily Curculionidae), rove beetles (Family Staphylinidae), and bark beetles (Family Scolytidae). Abundances of ground beetles were similar across treatments, but declined to half their abundance 2-years post-harvest (Table 4.1); ground beetles were likely the most abundant species because of their ease in being collected with pitfall traps. This abundance pattern was also seen in weevil numbers. Rove beetle abundances declined significantly 1-year post-harvest. Rove beetles occur in diverse habitats often under objects on ground, are proficient fliers, and feed on other insects. Bark beetles were not collected post-harvest by pitfall traps, and many of the other beetle species that feed on wood such as wood boring and long-horned beetles were not collected in great numbers.

Although we did not find a treatment effect for the most abundant species, the greatly reduced numbers in all beetle families indicates this group of species is affected by harvesting activities in this system for at least 2 years post-harvest. Niemela et al. (1993) found many boreal ground beetle assemblages (Coleopteran: Carabidae) recovered 1-2 years following clear-cutting in a mature lodgepole pine-white spruce forest; in our hardwood system, recovery appears to take longer. Our results are opposite from (Wermelinger et al. 2002) who found Coleoptera numbers greatly increased after an influx of breeding substrate from windthrow in an alpine

spruce forest. Both of these systems, however, were conifer systems.

We found no significant differences in plot-level Family richness across treatments and years. These findings did not support our hypotheses, however, care must be used when interpreting our results. Gunnarsson et al. (2004) found higher number of species on sites where slash was left, and a positive relationship between numbers of individuals per trap and slash removal. However, richness and abundance were measured at the trap-level, where our response was measured at the plot-level making comparisons difficult. Additionally, we only identified beetles taxonomically to the family level. Moreover, comparison of beetle diversity at the plot-level scale may not be appropriate given the smaller ranges of these species and only 6 samples per plot. Saproxylic insect assemblage compositions are known to differ between different substrate types, such as logs and high stumps due to different feeding guilds (Gibb et al. 2006, Hjältén et al. 2010), so the location of our traps to different substrates may influence beetles caught. Instead, a fine-scale analysis may be more appropriate (e.g., quadrat-scale) where beetle assemblages and numbers are counted closer to actual FWD residuals to find differences at the community level.

Overall, the reduced numbers of almost all beetle species, and no qualitative changes in species composition in our study is consistent with trends found in other soil arthropod studies investigating slash removal (Bird and Chatarpaul 1986, Bengtson et al. 1997).

Literature Cited

Bird GA and L Chatarpaul. 1986. Effect of whole-tree and conventional forest harvest on soil microarthropods. Canadian Journal of Zoology 64:1986-1993.

Bengtsson J, T Persson, and H Lundkvist. 1997. Long-term effects of logging residue addition and removal on macroarthrops and enchytraeids. Journal of Applied Ecology 34: 1014-1022.

Gibb H, RB Pettersson, J Hjältén, J Hilszczanski, JP Ball, T Johansson, O Artlegrim, and K Danell. 2006. Conservation-oriented forestry and early successional saproxylic beetles: responses of functional groups to manipulated dead wood substrates. Biological Conservation 129:437-450.

Grove SJ. 2002. Saproxylic insect ecology and the sustainable management of forests. Annual Review of Ecology and Systematics 33:1-23.

Hjältén J, F Stenbacka and J. Andersson. 2010. Saproxylic beetle assemblages on low stumps, high stumps and logs: Implications for environmental effect of stump harvesting. Forest Ecology and Management 260:1149-1155.

McGeoch MA, M Schroeder, B Ekbom, S. Larsson. 2007. Saproxylic beetle diversity in a managed boreal forest: important of stand characteristics and forestry conservation measures. Diversity Distribution 13:418-429.

Niemelä J, D Langor, and JR Spruce. 1993. Effects of clear-cut harvesting on boreal ground-beetle assemblages (Coleoptera:Carabidae) in Western Canada. Conservation Biology &:551-561.

Siitonen J. 2001. Forest management, coarse woody debris and saproxylic organisms: Fennoscandian boreal forest as an example. Ecological Bulletin 49:11-41.

Speight MCD. 1989. Saproxylic invertebrates and their conservation. Nature and Environment Series 42. Council of Europe, Strasbourg. 79 pp.

Wermelinger B, P Duelli, and MK Obrist. 2002. Dynamics of saproxylic beetles (Coleoptera) in windthrow areas in alpine spruce forests. Forest Snow and Landscape Research 77:133-148.

CHAPTER 5

Soil carbon and nitrogen response to forest residue removal treatments

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Background

Forests comprise ~35% of the 1,584 million acres within the lower 48 contiguous states (Lubowski et al. 2006), resulting in forest biomass constituting ~30% of the total biomass that can be produced in the United States (Smith et al. 2009). Therefore, adequate woody feedstock availability is necessary for environmental and economic sustainability. Woody feedstock production is vital for achieving our national goal of 16 billion gallons of cellulosic ethanol by 2022 (U.S. Energy Independence and Security Act of 2007), as well as supplying local combined heat and power operations. In addition to biofuels and bioenergy markets, forest-grown biomass is a major contributor to the North American woody bioproducts industry.

As described in Chapter 1, fine woody debris (FWD) that has been traditionally left as slash following harvesting operations is a substantial source of biomass for biofuels, bioenergy, and bioproducts, provided that residue removal is ecologically sustainable. This is especially true for forest soils, where such residues contribute large pools of carbon (C) and nitrogen (N) that support long-term ecosystem productivity (Powers et al. 2005). In addition to pools, changes in soil chemistry and fertility following harvesting are also important and have been met with varying results that trend in the direction of having minimal effects of harvesting on soil C and N (Homann et al. 2008, Hoover 2003, Johnson and Curtis 2001). For example, Hendrickson et al. (1989) studied changes in soil pH and N three years after harvesting a northern mixed forest in Ontario, Canada, and they reported no differences in N among uncut stands and those subjected to conventional harvesting or whole tree harvesting prescriptions across four soil layers. For pH, significant increases were observed to 10 cm depth for both harvesting treatments relative to the control (Hendrickson et al. 1989). Likewise, Strong (1997) reported minimal differences among four thinning treatments and an uncut control for soil C in four depths down to 40 cm in a northern hardwood ecosystem in northern Wisconsin. Since Strong (1997), there is limited information on how such interventions impact soil carbon and nitrogen levels in the northern hardwood forests of the Great Lakes region, and published literature on the impacts of FWD removal on soil C and N levels and associated changes are altogether lacking.

To address this need, our overarching objective was to compare soil pH, C, and N levels (including C:N ratios) before and two years after thinning treatments commenced (see Chapter 1 for an overall description of treatments). Our specific objectives were to: 1) test for differences among years and treatments for all four parameters within five different soil layers (litter/humus; 0 to 5 cm, 5 to 10 cm, 10 to 20 cm, >20 cm mineral soil), and 2) test for differences among treatments for the relative changes in the parameter levels pre- and post-thinning, within the layers. Our overall hypotheses were that levels would decrease on sites with less woody residue, and that changes would be most evident in the upper soil layers.

Methods

Collection

During August 2009 and 2011, litter/humus and mineral soil samples were collected at the 25-, 50-, and 75-m points on each of the four transects per plot (i.e., 432 soil samples per year). On

occasions with rocks, roots, and other impediments, samples were taken on the north edge of the original sample point. Litter/humus was collected by centering a 15-cm diameter polyvinylchloride (PVC) ring on each point and bagging the surface litter (e.g., undecayed leaves, twigs, etc.) down to the beginning of the upper mineral soil layer. One mineral soil sample (3.8 cm diameter) to a 30 cm depth was then collected from each sample point using a stainless steel soil core sampler with a plastic liner (AMS Inc., American Falls, ID, USA). The sampler was removed from the ground, disassembled, and the liner removed. The lower end was capped immediately, 'spacer' blocks of wood were added to the top to minimize soil decompaction during transport, and the top was capped. After collection, samples were held at ambient temperature and returned within 9 hours to the U.S. Forest Service, Institute for Applied Ecosystem Studies in Rhinelander, WI, USA, where they were stored at 5 °C until processing.

Processing

The liners were taken out of cold storage, caps removed, and soil depths demarcated using an indelible marker on the outside of each liner. Specifically, the top of the mineral soil layer (i.e., 0 cm depth) was marked first and used as a reference point for the 5, 10, and 20 cm depths. This defined the respective depths of 0 to 5 cm, 5 to 10 cm, 10 to 20 cm, and greater than 20 cm from which the analyses described below were conducted. Using locally fabricated extraction tools, the soils within each depth were then carefully removed and placed in paper bags (2009) or plastic cups (2011) and air-dried before being oven-dried at 55 °C until constant mass, which was at most 24 hours. Samples were weighed immediately upon removing from the drying oven and placed in capped plastic cups for final processing, which consisted of sieving to pass through a 2-mm screen (to remove rocks), and then grinding through a 0.5 mm screen using a Cyclotec 1093 grinder (FOSS Analytical A/S, Eden Prairie, MN, USA). All soils were returned to the plastic cups for capped storage until analytical analyses were performed.

Analytical Analyses

Soils were analyzed for pH using a Fisher Scientific Accumet Model No. XL50 pH meter with a combination reference-glass electrode (Fisher AccuCap combination pH electrode; Fisher Scientific, Waltham, MA, USA), as well as carbon (C) and nitrogen (N) content using a Flash EA1112 N-C analyzer with a model MAS 200 autosampler (Thermo Electron, via CE Elantech, Inc., Lakewood, NJ, USA). The ratio of C to N (C:N) was calculated by dividing percent C by percent N in each sample. Change in all four parameters was determined by subtracting 2011 from 2009 values (pH_{Δ}, C_{Δ}, N_{Δ}, $C{:}N_{\Delta}$).

Data Analyses

All data analyses were conducted within the five soil depths sampled (i.e., we did not test for differences among soil depths). With one exception, year (2009 = pretreatment; 2011 = post harvesting) and residue removal treatments (0% removed; 67% removed; 100% removed) were tested via analyses of variance (ANOVAs) according to the experimental design described in Chapter 1. Unlike the analyses of parameters described in Chapters 2 to 4, the exception for soils is that the control treatment (i.e., no harvesting interventions) was included in the analyses. Individual ANOVAs were conducted to test for differences among the four treatments for all soil variables in 2009, resulting in a lack of discernible trends that could justify non-inclusion of the control treatment.

The following linear additive model was used in the analyses of pH, C, N, and C:N ratio, by soil depth:

$Y_{ijk} = \mu + Y_i + B_j + YB_{ij} + T_k + YT_{ik} + BT_{jk} + YBT_{ijk}$ (Error)

where: Y_{ijk} = response variable to be analyzed, μ = overall mean, B_i = main effect of i^{th} year, B_j = main effect of j^{th} block, YB_{ij} = effect of interaction between i^{th} year and j^{th} block, T_k = main effect of k^{th} treatment, YT_{ik} = effect of interaction between i^{th} year and k^{th} treatment, BT_{jk} = effect of interaction between j^{th} block and k^{th} treatment, and YBT_{ijk} = effect of interaction among i^{th} year, j^{th} block, and k^{th} treatment.

Interactions including the random block effect were universally significant at $P < 0.25$ and, therefore, not pooled. Fisher's protected least significant difference (LSD) was used to compare all means, which were considered different at probability values of $P < 0.05$.

Similarly, the following linear additive model was used in the analyses of pH_Δ, C_Δ, N_Δ, $C:N_\Delta$, by soil depth:
$Y_{ijk} = \mu + B_i + T_j + BT_{ij}$ (Error)
where model definitions and mean comparisons were identical to those for the original parameters.

Results and Conclusions

Overall, data from the current study corroborated commonly-reported results in that thinning of our northern hardwood compartments had a minimal impact on soil pH, C, and N (Homann et al. 2008, Hoover 2003, Johnson and Curtis 2001). Both of our hypotheses were disproven; that is, levels of all parameters did not consistently decrease when greater volumes of FWD were removed, and changes in pre- and post-harvest levels did not occur in the upper mineral soil layers but were significantly more prominent in the deeper depths tested. Despite a difference in the duration between harvesting interventions and soil sampling, these results were similar to those of Strong (1997), who tested soil C responses to four different harvesting regimes at soil depths down to 40 cm. He reported only one significant difference among treatments, whereby soil C under a diameter-limit cut was lower than all other treatments at 3 to 10 cm depth (Strong 1997).

Differences among years, treatments, and their interactions, within soil depths, were highly variable for all parameters, resulting in an overall lack of generalized trends (Table 5.1). The

Table 5.1. Probability values from analyses of variance (ANOVAs) testing the effects of year (2009 = pretreatment; 2011 = post harvesting) and residue removal treatments plus the control (0% removed; 67% removed; 100% removed; no harvesting) on soil pH, C, N, and C:N. Analyses were conducted by soil depth. Values in bold mean there was a significant difference in means.

		Mineral Soil Depth (cm)			
Source	Litter/Humus	0 to 5	5 to 10	10 to 20	>20
	---------- pH ----------				
Year	**<0.0001**	**0.0011**	**0.0409**	ns	**0.0159**
Treatment	ns	**0.0001**	**0.0007**	ns	ns
Year × Treatment	ns	ns	ns	ns	**0.0292**
	---------- C ----------				
Year	**0.0002**	**0.0342**	ns	ns	**0.0477**
Treatment	ns	ns	ns	ns	ns
Year × Treatment	ns	ns	ns	ns	ns
	---------- N ----------				
Year	**<0.0001**	ns	ns	**0.0064**	**0.0235**
Treatment	ns	ns	ns	ns	ns
Year × Treatment	ns	ns	ns	ns	**<0.0001**
	---------- C:N ----------				
Year	ns	**0.0015**	**0.0077**	ns	ns
Treatment	ns	ns	ns	ns	ns
Year × Treatment	ns	ns	**0.0233**	**0.0011**	**<0.0001**

ns = not significant at $\alpha = 0.05$.

majority of differences occurred for the year main effect, where pH, C, and N were all different for the litter/humus and >20 cm depths, and all four parameters exhibited differences within the intermediate layers. Differences between years were most prominent for pH, where levels increased from pre- to post-harvest in the upper soil layers but decreased from 5 cm downward (Table 5.2). Interestingly, there were no differences in the 10 to 20 cm depth. The percent C and N in soils consistently decreased for all depths, despite not being significant in the middle soil layers. In contrast, C:N did not differ in the litter/humus or >20 cm depth but did significantly decrease from 0 to 10 cm (Table 2).

Table 5.2. Mean (± one standard error) values for soil pH, C (%), N (%), and C:N across years (2009 = pretreatment; 2011 = post harvesting). Analyses were conducted by soil depth. Mean values in bold and with different letters within a depth are different according to Fisher's protected least significant difference ($\alpha = 0.05$) signifying a difference between years.

		Mineral Soil Depth (cm)			
Year	Litter/Humus	0 to 5	5 to 10	10 to 20	>20
	---------- pH ----------				
2009	**5.24 ± 0.02 b**	**4.85 ± 0.02 b**	**4.83 ± 0.02 a**	4.58 ± 0.02	**4.62 ± 0.02 a**
2011	**5.49 ± 0.02 a**	**5.00 ± 0.02 a**	**4.76 ± 0.02 b**	4.57 ± 0.02	**4.54 ± 0.02 b**
	---------- C (%) ----------				
2009	**33.89 ± 0.34 a**	**8.70 ± 0.27 a**	3.23 ± 0.12	1.41 ± 0.05	**1.13 ± 0.04 a**
2011	**30.94 ± 0.41 b**	**7.79 ± 0.26 b**	2.96 ± 0.10	1.30 ± 0.03	**0.91 ± 0.03 b**
	---------- N (%) ----------				
2009	**1.50 ± 0.02 a**	0.62 ± 0.02	0.27 ± 0.01	**0.15 ± 0.00 a**	**0.12 ± 0.00 a**
2011	**1.38 ± 0.02 b**	0.59 ± 0.02	0.26 ± 0.01	**0.13 ± 0.00 b**	**0.10 ± 0.00 b**
	---------- C:N ----------				
2009	23.41 ± 0.23	**13.67 ± 0.10 a**	**11.48 ± 0.11 a**	8.95 ± 0.10	9.35 ± 0.16
2011	23.32 ± 0.26	**12.85 ± 0.11 b**	**10.97 ± 0.11 b**	9.69 ± 0.09	8.86 ± 0.13

Treatment main effects only differed on two occasions and both were for pH: 1) 0 to 5 cm depth ($P = 0.0001$), and 2) 5 to 10 cm depth ($P = 0.0007$). In both layers pH significantly increased with less biomass removal, yet the pH levels between 0% removal and 67% removal were not different in either case (0 to 5 cm: 4.94 ± 0.03 and 4.87 ± 0.03, respectively; 5 to 10 cm: 4.78 ± 0.03 and 4.76 ± 0.03, respectively). In contrast, the interaction between year and treatment for the >20 cm depth resulted in significant declines in pH from pre- to post-harvest for the 0% removal and control treatments (Figure 5.1), while similar differences for the 67% and 100% removal treatments were non-existent. Despite the discrepancy in baseline pH values, however,

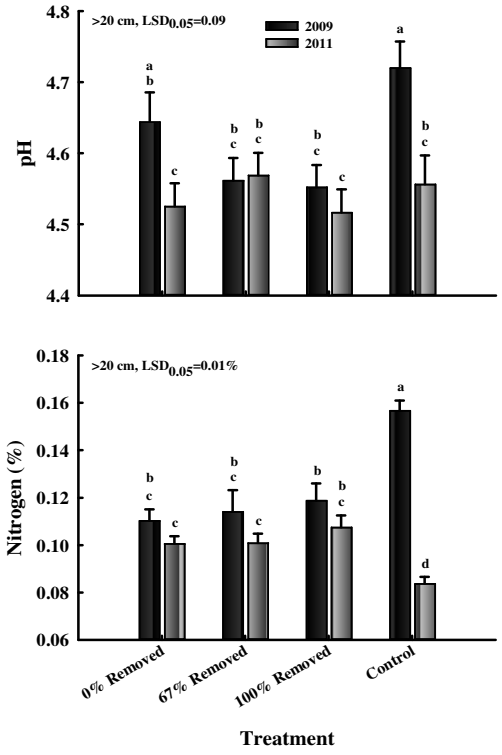

Figure 1. Soil pH and nitrogen content (%) at >20 cm depth in 2009 (pretreatment) and 2011(post harvesting) for all residue removal treatments plus the control (no harvesting). Means with different letters are different at α = 0.05.

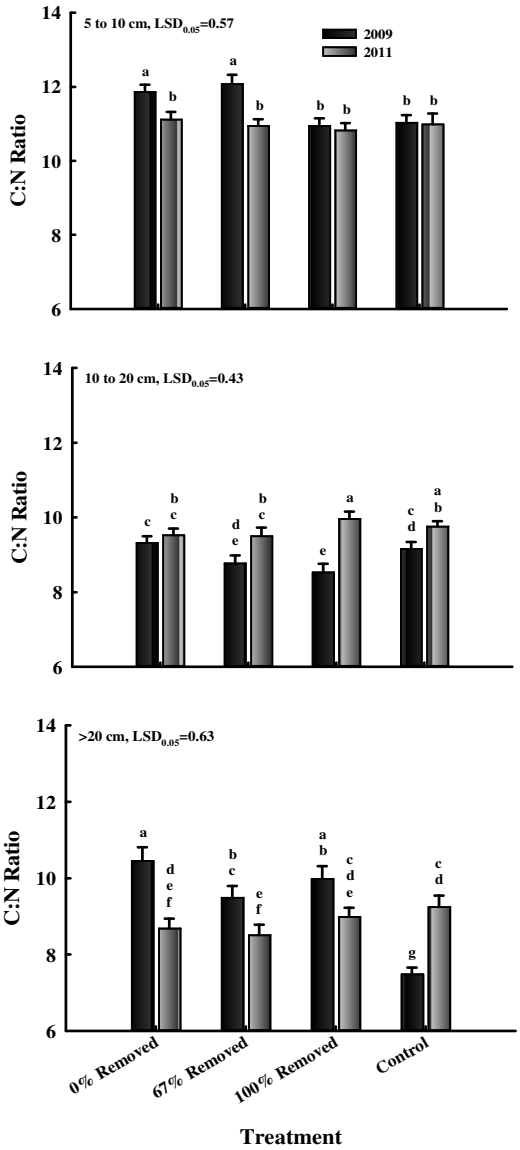

Figure 2. Soil carbon:nitrogen ratio at three soil depths in 2009 (pretreatment) and 2011(post harvesting) for all residue removal treatments plus the control (no harvesting). Means with different letters are different at α = 0.05.

ending values in 2011 were the same. This was not the case for N at the >20 cm depth (Figure 5.1), whereby the pre- and post-harvest N levels were different in the control treatment but not the harvesting treatments, which also exhibited similar magnitudes of decreases in soil N. Furthermore, the interaction between year and treatment was also significant for C:N at 5 to 10, 10 to 20, and >20 cm soil depths (Table 5.1). Differences in the 5 to 10 cm layer existed for the 0% and 67% removal treatments, with C:N decreasing in 2011. In contrast, C:N increased for all treatments except for the 0% removal for the 10 to 20 cm depth, but this trend was then further reversed for soils below 20 cm for all harvesting treatments. That is, the C:N significantly decreased following logging operations, which is in line with the data presented in Table 5.2. Changes among treatments within soil depths were non-existent for litter/humus and the 0 to 5 cm layer, but prevalent for the remaining three mineral depths for pH_{Δ} and $C:N_{\Delta}$ (Table 5.3). While not significant, an observational trend existed for pH_{Δ} such that removing any amount of harvest residue resulted in decreased pH for soils below 10 cm (Figure 5.3). For all but the 5 to 10 cm depth, the change in pH was significantly greater for the control than the 67% and 100% removal treatments, and the 0% removal was not different than any of the other treatments. The pH_{Δ} for the control was also nearly twice as much as for all harvesting treatments, which requires further investigation. Furthermore, $C:N_{\Delta}$ was highly variable among the lower three mineral soil depths (Figure 5.4). For the 5 to 10 cm layer, all changes were positive and were not linked to harvesting intensity, as illustrated by the lack of differences among the 0%, 100%, and control treatments. The $C:N_{\Delta}$ was greatest for the 67% removal treatment, which was also not different than the thinning without residue removal. In contrast, changes for the 10 to 20 cm depth were all negative and the greatest change occurred following 100% removal of harvesting residue. The $C:N_{\Delta}$ was also similar and positive for all harvesting interventions beyond 20 cm soil depth, while that for the control was negative, which tracks the significant decrease in N that is illustrated in Figure 5.1. Lastly, C_{Δ} and N_{Δ} for the >20 cm depth exhibited similar trends, whereby the change in each was significantly greater for the control than any of the harvesting treatments, which were the same.

In summary, our results corroborated results of others such as Hendrickson et al. (1989) who tested pH and N levels in forest floor (i.e., our litter/humus), and three mineral soil layers (0 to 5, 5 to 10, 10 to 20 cm) three years after harvesting a northern mixed forest in Ontario, Canada, and reported a lack of impact from harvesting on N and a slight increase in pH. While not tested in the current study, one explanation for the differences and changes in the lower soil layers may be the impacts from invasions of earthworms which are known to disrupt C, N, and C:N levels and ultimately the balance of forest floor ecosystems (Bohlen et al. 2004a, 2004b). A more detailed evaluation of the current data is underway to assess such hypotheses, in addition to abiotic causal factors leading to differences in levels of and changes in soil parameters that are necessary for fertility and long-term C storage.

Table 5.3. Probability values from analyses of variance (ANOVAs) testing the effects of residue removal treatments plus the control (0% removed; 67% removed; 100% removed; no harvesting) on changes in soil pH, C, N, and C:N from 2009 (pre-harvest) to 2011 (post-harvest). Analyses were conducted by soil depth.

		Mineral Soil Depth (cm)			
Source	Litter/Humus	0 to 5	5 to 10	10 to 20	>20
	---------- pH_{Δ} ----------				
Treatment	ns	ns	**0.0136**	**0.0284**	**0.0054**
	---------- C_{Δ} ----------				
Treatment	ns	ns	ns	ns	**0.0352**
	---------- N_{Δ} ----------				
Treatment	ns	ns	ns	ns	**<0.0001**
	---------- $C:N_{\Delta}$ ----------				
Treatment	ns	ns	**0.0174**	**0.0013**	**<0.0001**

ns = not significant at $\alpha = 0.05$.

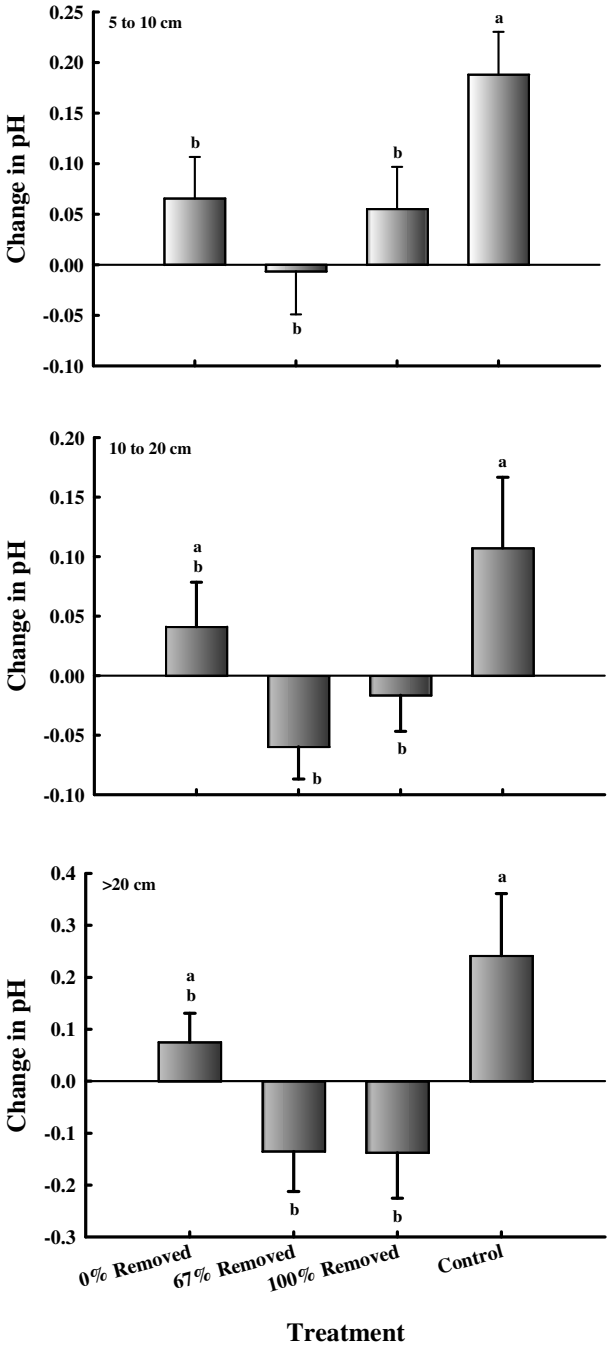

Figure 3. Changes in soil pH from 2009 (pretreatment) to 2011(post harvesting) for all residue removal treatments plus the control (no harvesting), at three depths. Means with different letters are different at α = 0.05.

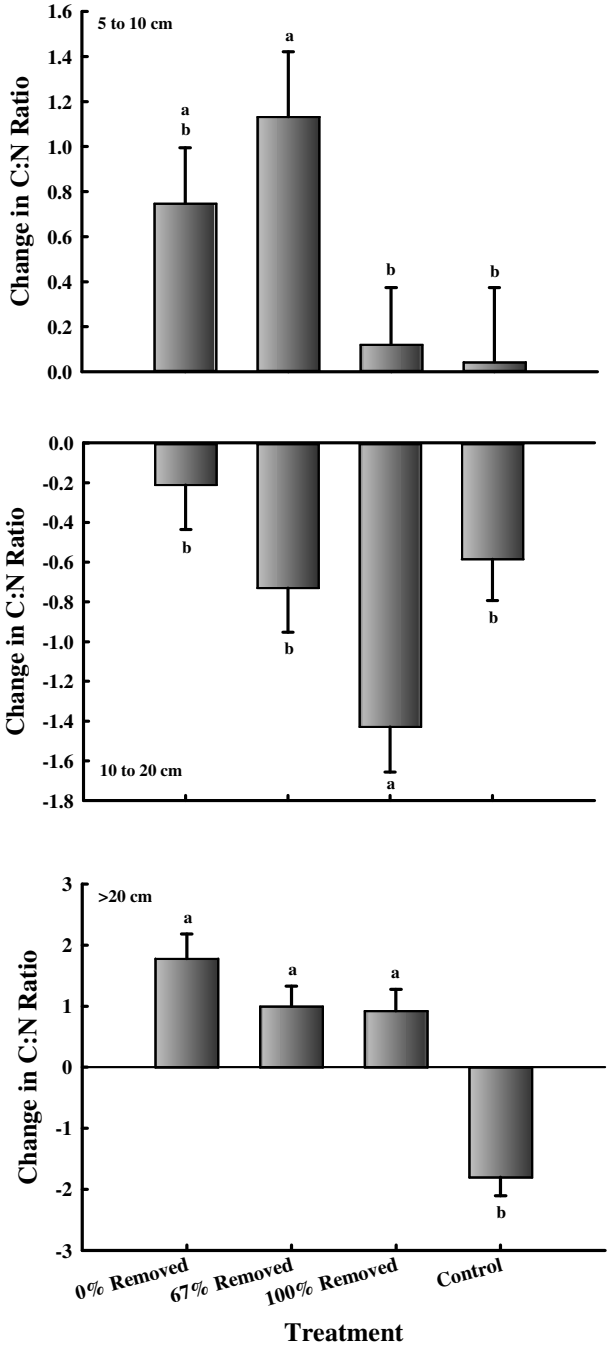

Figure 4. Changes in soil carbon:nitrogen ratio from 2009 (pretreatment) to 2011(post harvesting) for all residue removal treatments plus the control (no harvesting), at three depths. Means with different letters are different at α = 0.05.

Literature Cited

Bohlen PJ, Groffman PM, Fahey TJ, Fisk MC, Suárez E, Pelletier DM, Fahey RT. 2004. Ecosystem consequences of exotic earthworm invasion on north temperate forests. Ecosystems 7:1-12.

Bohlen PJ, Pelletier DM, Groffman PM, Fahey TJ, Fisk MC. 2004. Influence of earthworm invasion on redistribution and retention of soil carbon and nitrogen in northern temperate forests. Ecosystems 7:13-27.

Hendrickson OQ, Chatarpaul L, Burgess D. 1989. Nutrient cycling following whole-tree and conventional harvest in northern mixed forest. Canadian Journal of Forest Research 19:725-735.

Homann PS, Bormann BT, Boyle JR, Darbyshire RL, Bigley R. 2008. Soil C and N minimum detectable changes and treatment differences in a multi-treatment forest experiment. Forest Ecology and Management 255:1724-1734.

Hoover CM. 2003. Soil carbon sequestration and forest management: challenges and opportunities, Chapter 14. In: The potential of U.S. forest soils to sequester carbon and mitigate the greenhouse effect. Edited by: Kimble JM, Heath LS, Birdsey RA, Lal R. CRC Press, Boca Raton, FL. pgs 211-238.

Johnson DW, Curtis PS. 2001. Effects of forest management on soil C and N storage: meta analysis. Forest Ecology and Management 140:227-238.

Lubowski RN, Vesterby M, Bucholtz S, Baez A, Roberts MJ. 2006. Major uses of land in the United States, 2002. Economic Information Bulletin No. 14. Washington, DC: U.S. Department of Agriculture, Economic Research Service. 47 p.

Powers RF, Scott DA, Sanchez FG, Voldseth RA, Page-Dumroese D, Elioff JD, Stone DM. 2005. The North American long-term soil productivity experiment: findings from the first decade of research. Forest Ecology and Management 220:31-50.

Smith W, Miles PD, Perry CH, Pugh SA. 2009. Forest resources of the United States, 2007. General Technical Report WO-GTR-78. Washington, DC: U.S. Department of Agriculture, Forest Service, Washington Office. 336 p.

Strong TF. 1997. Harvesting intensity influences the carbon distribution in a northern hardwood ecosystem. Research Paper NC-RP-329. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station. 7 p.

Chapter 6

Synthesis and future direction of research

Deahn M. Donner

We stress that our results are short-term responses to forest residue removal. Future studies will be required to determine the long term impacts of removing this material from nutrient rich northern hardwood second-growth forests. These studies will be able to build on our current baseline information to evaluate long term impacts of biomass removal. Our permanent plots will be maintained annually, so we can re-sample treatment sites 5-, 10- and 15- years post-harvest. Of great interest is the impact of removing this material during the second 15-year thinning rotation for these systems.

Overall, we found the greatest changes in the amphibian assemblage (i.e., highest trophic level for this study) to forest residue removal treatments. There may be several reasons for these results. We focused on the plot scale for this report but this scale may not have been the best scale for insect and plant assemblage species. These species may be responding to their environment at a finer-scale; for example, finer-scale substrate differences may be more important to insects and plants. These individual factors may explain why some species changed in presence and abundance across years and others did not. However, if the question is large-scale impacts of forest residue on regional diversity then plot-level responses may be most appropriate. The scale of an investigation will impact the number of species found (Godfray and Lawton 2001), and the size of 'local' scale compared to 'regional' scale will depend on the species being investigated so it is difficult to compare results across studies. Recent advances in statistical approaches will allow us to begin linking trophic interactions given the scaling issues that are associated with comparing species groups.

Literature Cited

Godfray HC and JH Lawton. 2001. Scale and species numbers. Trends in Ecology and Evolution. 16:400-404.

Appendix A. Complete list of insects identified during pitfall trapping, May-August, 2009 – 2011.

Species	2009	2010	2011	Species	2009	2010	2011
Order: Aranae				**Order: Lepidoptera**			
Spiders	X	X	X	Moths	X	X	X
Ticks	X	X		Butterflies	X		
Order: Acarina				Moth/ Butterfly Family Other	X		X
Beetle-mite	X	X	X	Moth/Butterfly Larvae	X	X	X
Acarina family unknown	X	X	X	Geometer Moth (F: Geometridae)	X		X
Velvet mite	X	X	X	Geometridae Larvae	X		
Order Coleoptera				Lepidoptera Unknown	X		
Weevils (F: Curculionidae)	X	X	X	**Order: Orthoptera**			
Rove Beetle (F: Staphylinidae)	X	X	X	Grasshoppers (F: Acrididae)	X	X	X
Click Beetle (F: Elateridae)	X	X	X	Crickets (F: Gryllidae)	X	X	X
Fireflies (F: Lampyridae)	X	X		Camel Crickets (F: Gryllotalpidae)	X	X	X
Longhorned (F: Cerambycidae)	X		X	Orthoptera Unknown	X	X	
Ladybugs (F: Coccinellidae)	X	X		**Order: Hemiptera**			
Hister Beetle (F: Histeridae)	X	X	X	Leaf bugs (F: Miridae)	X	X	X
Bark Beetle (F: Scoytidae)	X			Stink bugs (F: Pentatomidae)	X		X
Blister Beetle (F: Meloidae)	X	X	X	Hemiptera Other	X	X	X
Carrion Beetle (F: Silphidae)	X	X	X	Lace bugs (F: Tingidae)			X
Scarab Beetle (F: Scarabaeidae)	X	X	X	**Order Hompotera**			
Sap Beetle (F: Nitidulidae)	X			Suborder Homoptera Other	X	X	X
Leaf Beetle (F: Calligrapha)	X			Aphids (F: Aphidoidea)	X	X	X
Wood Boring Beetle (F: Buprestidae)	X			**Order Neuroptera**			
Soldier Beetle (F: Contharidae)			X	Lacewings	X		
Beetles Other	X	X	X	Neuroptera Other	X		
Larvae	X		X	Lacewing Larvae	X	X	X
Ground Beetle (F: Carabidae)	X	X	X	**Other**			
Tiger Beetle (F: Cicindelidae)		X	X	Scorpionfly (F: Panorpidae)	X		X
Scydmaenidae			X	Thrips (Order: Thysanoptera)	X	X	X
Soldier Beetle (F: Contharidae)		X		Harvestmen (F: Opiliones)	X	X	X
Spider Beetle (F: Bostrichidae)		X		Scale insect (F: Coccoidea)	X		
Order: Hymenoptera				Snails	X	X	X
Ants (F: Formicidae)	X	X	X	Slugs	X	X	X
Ichneumonid Wasps (F: Ichneumonidae)	X	X	X	Worms	X	X	X
Chalcid (F: Chalcidoidea)	X	X	X	Centipede	X	X	X
Hymenoptera Family Other	X	X	X	Millipede	X	X	X
Hymenoptera Unknown	X	X	X	Pillbugs	X	X	X
Hymenoptera Larvae	X	X	X	Psocoptera	X	X	X
Order: Diptera				Caterpliller	X		
Crane Flies (Nematocera)	X	X	X	Unknown Larva	X	X	X
Mosquitoes (Nematocera)	X	X	X	Pseudoscorpiones	X	X	
Nematocera Other	X	X	X	Bag Worm	X		
Brachycera/ Cyclorrhapha	X	X	X	Isoptera	X		X
Brachycera Other	X	X		Neuroptera			X
Diptera Other	X	X	X	Mecoptera	X	X	X
Diptera Larvae	X	X	X	Nabidae	X		
Order: Collembola				Aradidae		X	
Family Isotomidae	X	X	X				
Family Sminthuridae	X	X	X				

Appendix. B. List of acronyms.

Acronym	Definition
C	Carbon
CNNF	Chequamegon-Nicolet National Forest
CWD	Coarse woody debris (> 4 inch diameter)
DBH	Diameter breast height
FWD	Fine woody debris (< 4 inch diameter)
N	Nitrogen
NEPA	National Environmental Protection Act
NMS	Nonmetric multidimensional scaling
VES	Visual encounter survey
VIE	Visible implant elastomer
WDNR	Wisconsin Department of Natural Resources