Forest dynamics in mesic northern hardwoods following windthrow and salvage logging

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Forest dynamics in mesic northern hardwoods following windthrow and salvage logging

by

Katharyn Dianne Derr

A thesis submitted to the graduate faculty
in partial fulfillment of the requirements for the degree of

MASTER OF SCIENCE

Major: Forestry (Forest Biology)

Program of Study Committee:
Lisa A. Schulte, Major Professor
Glenn Guntenspergen
Lee Burras
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Iowa State University
Ames, Iowa
2006

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CHAPTER 1. GENERAL INTRODUCTION

The typical response of forest managers following a catastrophic natural disturbance (e.g., wind, fire) event is to salvage the dying or downed trees to recover some of the economic value. The knowledge regarding the effects of this practice on forest regeneration and biodiversity is limited, however, and often conflicts with the general prescriptions that managers seek (Greenberg et al. 1995, Radeloff et al. 2000, Morissette et al. 2002, Lindenmayer et al. 2004, Donato et al. 2006a, Lindenmayer 2006).

Many studies currently discuss the impacts, positive and negative (though mostly negative), of salvage logging on forest community recovery (Greenberg et al. 1995, Radeloff et al. 2000, van Nieuwstadt et al. 2001, Morissette et al. 2002, Lindenmayer et al. 2004, Donato et al. 2006a,b, Lindenmayer 2006). Natural disturbances, such as wildfires, blowdowns, and other natural events, are seen as destructive catastrophes rather than a natural occurrence (Lindenmayer et al. 2004, Lindenmayer 2006, Lindenmayer and Noss 2006a). The resulting early successional forests are very biologically diverse intact ecosystems (Cooper-Ellis et al. 1999, Lindenmayer et al. 2004, Lindenmayer and Noss 2006a, b) which many people, including ecologists, perceive as degraded and feel the need to “clean up” the forest to aid recovery (van Nieuwstadt et al. 2001, Lindenmayer et al. 2004, Lindenmayer and Noss 2006a, b). Salvage harvesting can destabilize ecosystem benefits gained from the natural disturbance (Cooper-Ellis et al. 1999, Lindenmayer et al. 2004, Donato 2006a, Lindenmayer and Noss 2006b). Most decisions are made rapidly after the disturbance and result in relaxed regulations to aid recovery of “lost” timber (van Nieuwstadt et al. 2001, Morissette et al. 2002, Lindenmayer et al. 2004, and Lindenmayer 2006), but long-term implications of salvage harvesting need to be considered (Cooper-Ellis 1999,
Timing of the salvage harvest, intensity, and size of the harvest are all important factors affecting the ability of the forest to recover (Lindenmayer et al. 2004, Lindenmayer 2006).

In an effort to inform this debate, here I study the long-term effects of salvage logging following catastrophic windthrow in a forest in northern Wisconsin. This work takes place in and around the Flambeau River Hemlock-Hardwood Scientific Area. The Scientific Area is located within the Flambeau State Forest and was not salvaged logged after the blowdown because of its scientific area status, while nearby areas were. Prior to the 1977 blow down event, the Scientific Area and adjacent areas were dominated by *Tsuga canadensis* (L.) Carr., while *Betula alleghaniensis* (Britt.) and *Acer saccharum* (Marsh.) were present in the sub-canopy (Dunn et al. 1983). *Pinus strobus* (L.), *Ulmus Americana* (L.), and *Tilia americana* were also present, but not as common as the above species (Dunn et al. 1983). My work expands upon the effort begun by Dunn et al. (1983). In 1979, Dunn et al. (1983) established a vegetation sampling network in the Flambeau River Hemlock-Hardwood Scientific Area and documented the immediate changes in forest composition and structure associated with this catastrophic wind event through a pre- and post-disturbance comparison. I employ the long-term data set collected on this vegetation sampling network and adjacent salvage logged forest beginning in 1979, and resampled in 1990 and 2004.

**Background on the Flambeau River Hemlock-Hardwood Scientific Area**

The 150 year history of Wisconsin’s northern mesic forests has been one of utilization (Curtis 1959). These forests underwent highgrading for white pine in the late 1800’s- early 1900’s, followed by the harvesting of *Tsuga canadensis* bark for tannins,
highgrading for *Betula alleghaniensis* around Word War I, and finally clear cutting became
the method of harvesting due to forest fire fears, tax laws, and economics around 1930
(Curtis 1959). The large reduction in downed logs negatively affected regeneration of *Betula
alleghaniensis* and *Tsuga canadensis* (Curtis 1959). With this situation in mind, the
Flambeau River Hemlock-Hardwood Scientific Area was set aside as a reserve area in the
Flambeau River State Forest (Curtis 1959). Prior to blowdown, this was one of the last large
tracts of old-growth northern hardwood forest left in the Lake States region (Dunn et al.
1983). The vegetation plots established by Dunn et al. (1983) in and around this area have
since been used to record forest recovery from windthrow.

THESIS ORGANIZATION

This thesis is composed of three chapters. This first chapter contains some
background information that sets the stage for my thesis research. Chapter 2 is written as a
paper for subsequent publication in a scientific journal and comprises the main body of my
thesis. This chapter examines the effects of windthrow and salvage logging on forest
recovery and tip-up structure and composition. This study was conducted within and nearby
the Flambeau River Hemlock-Hardwood Scientific Area. Chapter 3 contains the general
conclusions of my thesis research.

LITERATURE CITED

Cooper-Ellis, S., D.R. Foster, F. Carlton, and A. Lezberg. 1999. Forest response to


CHAPTER 2. FOREST DYNAMICS IN MESIC NORTHERN HARDWOODS
FOLLOWING WINDTHROW AND SALVAGE LOGGING

A paper to be submitted to Conservation Biology
Authors: Katharyn D. Derr, Lisa A. Schulte, Glenn R. Guntenspergen

ABSTRACT

Post-disturbance, or salvage, logging is a highly debated topic within natural resource science and management. Although initial plant and wildlife response to salvage logging is known to be variable depending on the species considered, understanding of its impacts on long-term forest recovery is largely absent. We examine a 25-year response of forest overstory and understory conditions to a 1977 severe wind disturbance event on the Flambeau River State Forest in Wisconsin, USA, some of which was salvage logged. We also examine the effects of salvage logging on forest microtopography and soil characteristics.

Within these mesic mixed conifer and broad-leaved deciduous forests, we document a slower rate of recovery following salvage logging. Tree biomass is twice as high within the unsalvaged area (P= 0.03), while sapling and herb layers are more developed within the salvaged area ((P<0.01) and (P=0.02), respectively). Tree species composition, however, is similar between salvaged and unsalvaged sites today (Tilia americana, Ulmus americana, and Acer saccharum dominant tree and sapling species across disturbance types). Microtopography of the forest floor is more pronounced in the unsalvaged area, with higher mounds (unsalvaged 0.61 m; salvaged 0.34 m; P<0.01) and deeper pits (unsalvaged -0.24 m; salvaged -0.13 m ; P<0.01). Soil composition also differs by disturbance type. Bulk density
organic carbon ($P<0.01$) and organic nitrogen ($P<0.01$) are higher in the salvaged area; pH showed no difference ($P=0.26$).

Both disturbance type and microtopography have a significant effect on the richness and cover of herbaceous species. Species richness was greater within the salvaged area ($P<0.01$), but this does not indicate higher site quality: *Solidago* spp. and *Maianthemum canadense*, indicators of disturbance, were more prevalent in the salvaged area ($P<0.01$). Species richness also differed by tip-up position ($P<0.01$); the top position had the highest average number of species (12.8 species/m$^2$) while the pit had the lowest average number of species (6.8 species/m$^2$). Herbaceous cover was greater on average on mound tops (15.2%), in comparison to the pit position (6.3%), for wind-dispersed ($P<0.01$), shade intolerant ($P<0.01$), and shade tolerant species ($P<0.01$). Some species showed strong relations to the tip-up positions. Moss and *Athyrium filix-femina* highly favored the tip-up top position while *Hydorphyllum virginianum* showed a preference for pit. *Solidago* spp. cover was generally high over all positions showing no strong relation.

This work can aid management decisions after severe wind disturbance in terms of setting goals for retaining or encouraging desired tree species and understory biodiversity, and for reducing impacts on soil resources. We found that salvage logging slows overstory recovery and alters soil structure and composition. Although overall herbaceous richness and cover were higher within salvaged areas, species that benefit from the high light condition and bare soil associated with disturbance tended to be dominant. To minimize the impacts of salvage logging, general disturbance response guides that are attentive to biological legacies should be developed; these guides should specify the timing, location, and intensity of salvage harvests.
**INTRODUCTION**

Salvage logging after a major natural forest disturbance is an ardently debated topic within natural resource science and management communities (Donato et al. 2006a,b, Newton et al. 2006, Baird 2006, and DellaSala et al. 2006). Research supports differing positions depending on the plant or wildlife species considered (Greenberg et al. 1995, Radeloff et al. 2000, Morissette et al. 2002, Hutto 2006, Donato et al. 2006a, Lindenmayer 2006). For example, Greenberg et al. (1995), in comparing bird species assemblages, suggest a neutral response. In comparing a clearcut logged area to an area that was burned by a high-intensity wildfire and subsequently salvage logged, Greenberg et al. (1995) found bird assemblages to be related more to habitat structure than the type of disturbance. Radeloff et al. (2000) found that salvage logging was effective in recreating open habitat in areas historically in pine barrens vegetation. The number of leks for Sharp-tailed Grouse (*Tympanuchus phasianellus* L.), a species of concern within the area Radeloff et al. (2000) were working, increased following the salvage logging practice. Elliot et al. (2002) showed plant recovery response to be high for both early and late successional species after salvage logging, while other studies indicate slow recovery of the herbaceous community (Duffy and Meier 1992, van Nieuwstadt et al. 2001, Donato et al. 2006a,b, Lindenmayer 2006). Some evidence suggests that salvage logging affects forest recovery (Cooper-Ellis et al. 1999, Foster and Orwig 2006) and the spatial patterning of forest understory plants, but with
unknown consequences to the future functioning of the entire plant community (Stuart et al. 1993, Scheller and Mladenoff 2002, Lindenmayer et al. 2004, Lindenmayer 2006, Stokstad 2006). Reeves et al. (2006) suggest that riparian communities should be protected post fire with the same pre-fire restrictions until more research is conducted, since so little is known about the effects of salvage logging on this system. The post-disturbance timing and intensity of the salvage harvest can also affect plant community resilience (Cooper-Ellis 1999, Starr 2000, van Nieuwstadt et al. 2001, Lindenmayer et al. 2004, Lindenmayer and Ough 2006). An ecosystem well adapted for recovery after a natural disturbance may be maladapted to multiple disturbances in rapid succession (Starr 2000, van Nieuwstadt et al. 2001, Lindenmayer et al. 2004, Lindenmayer 2006). van Nieuwstadt et al. (2001) found that rapidly repeated disturbances reduced potential tree sprouting, tree sprout survival rates, and lowered the density of vital seed trees. van Nieuwstadt et al. (2001) also noted the largely reduced seedbank after initial germination. The second disturbance damages the initial germination leaving a diminished seedbank for the second recovery attempt (van Nieuwstadt et al. 2001). Most decisions are made rapidly after the disturbance and result in relaxed regulations to aid recovery of “lost” timber (van Nieuwstadt et al. 2001, Morissette et al. 2002, Lindenmayer et al. 2004, Lindenmayer 2006, Schmiegelow et al. 2006), but long-term implications of salvage harvesting need to be considered (Cooper-Ellis 1999, Lindenmayer 2006). Timing of the salvage harvest, intensity, and size of the harvest are all important factors affecting the ability of the forest to recover (Lindenmayer et al. 2004, Lindenmayer 2006, Schmiegelow et al. 2006).

Another important factor in post-disturbance forest recovery is the effect of salvage logging on forest microtopography. Microtopographic features add spatial heterogeneity to
the forest floor and related microsite conditions can be important for the regeneration of some tree and understory species (Beatty 1984, Carlton and Bazzaz 1998, Palmer et al. 2000, Ulanova 2000). Noteworthy microtopographic features following wind disturbance are tip-up. A tip-up is a mound of earth and associated pit resulting from the raising of a quantity of soil when a tree tips over (Curtis 1959, Dunn et al. 1983, Burns and Honkala 1990, Peterson 2000a, Ulanova 2000). As the enmeshing roots decay, the soil slumps to form a hillock at the base of the trunk while the area from which the soil came persists as a depression or pit (Curtis 1959, Beatty 1984). This pit-and-mound microtopography increases the structural complexity of the forest floor by exposing organic and mineral soil, disrupting soil horizons, altering soil development, creating gradients of soil water content, small-scale temperature differences (Beatty 1984, Peterson 2000a, Ulanova 2000), and may remain long after the treefall event that created them (up to 500 years) (Tyrrell and Crow 1994). Though recognized as adding spatial complexity, the effect of tip-ups vegetation re-establishment has not been closely examined. Furthermore, the effect of salvage logging on tip-up microsites and vegetation recovery are not presently known. In a recent review of the salvage logging literature, however, Lindenmayer and Noss (2006b) observed that most studies assessing the effects of salvage logging are substantially limited. Many lack adequate controls—either sites where no natural disturbance occurred or sites where natural disturbance occurred, but salvage logging was prohibited (Lindenmayer and Noss 2006a,b).

Prior to Euro-American settlement, windthrow, rather than fire, were the major form of disturbance for the higher latitude hemlock-hardwood forest in Wisconsin (Curtis 1959, Goder 1961, Schulte and Mladenoff 2005, Webster et al. 2005). High severity winds are a natural phenomenon that can cause broad-scale catastrophic disturbances in forested systems.
(Palmer et al. 2000, Peterson 2000a,b, Ulanova 2000, Elliott et al. 2004) and usually have a longer rotation period (>1000-year interval) than lower severity ones (White and Mladenoff 1994, Frelich 1995), thus creating a patchwork of disturbance severities on the ground (Dunn et al. 1983). These high severity wind events typically set back succession by removing late successional dominants and enrich diversity through the creation of higher light environments (Palmer et al. 2000, Webb and Scanga 2001). Some forest types, such as boreal spruce forests (Ulanova 2000), hardwood-white pine-hemlock forests (Boose et al. 1994), and hemlock-hardwood forest (Curtis 1959, Dunn et al. 1983), rely on windthrow disturbances for regeneration. The removal of overstory dominants changes the canopy dynamics, patch light availability, and the spatial heterogeneity in soil resources through tip-up mound creation (Goder 1961, Palmer et al. 2000, Webb and Scanga 2001). The patch light dynamics created by windthrow disturbance help maintain the moderately shade tolerant species within the forest system (Curtis 1959, Frelich 2002, Marx 2005, Schulte and Maldenoff 2005, Webster and Lorimer 2005). Woods (2000) suggest that species such as Betula alleghaniensis (Britt.) and Tilia Americana (L.) need the larger gap dynamics created by windthrow disturbances to remain an important secondary species within the hemlock-hardwood forests.

Catastrophic windthrows change the structure of the forest canopy through the removal of dominant trees, and also soil and moisture conditions, tree recruitment patterns, and understory vegetation growth related to microtopography conditions (Curtis 1959, Dunn et al. 1983, Palmer et al. 2000, Peterson 2000a, Ulanova 2000). Forest recovery after catastrophic windthrow is less commonly studied than the effects of damage itself (Peterson 2000a). Understanding the effects of catastrophic wind events on forest systems is important
not only for informing current forest management practices, but also because the frequency of such storms may increase in the future (Everham and Brokaw 1996, Peterson 2000b, Frelich 2002, Lindenmayer 2006). Peterson (2000b) suggests that warmer air masses over middle latitudes in North America, associated with global climate change, combined with typical vertical dynamic movement of convection currents creates the potential for more downburst producing cells in the future. Another factor that may contribute to an increased incidence of windthrow in the future is the further fragmentation in forested areas with housing development (Ward et al. 2005).

Wind damage is not only related to wind intensity, but also to the vegetative composition and its resistance to wind damage (Everham and Brokaw 1996), which in turn is dependent on stand age, tree type, stem size, wood strength, crown form, root formation, and soil characteristics (Carlton and Bazzaz 1998). Old growth stands are typically more prone to damage than younger stands due to the size and age of the trees (Everham and Brokaw 1996, Carlton and Bazzaz 1998, Ward et al. 2005). Conifers are more susceptible to wind damage than hardwoods, and larger stem sizes also increase chances for windthrow (Everham and Brokaw 1996, Ward et al. 2005). The fuller crowns of conifers have a higher degree of wind drag making them more prone to wind damage (Everham and Brokaw 1996, Carlton and Bazzaz 1998). Differences between species in the wood strength and crown shape affect susceptibility. The bending force required for hardwoods is greater than twice that for *Pinus strobus* (L.) (Everham and Brokaw 1996). Species that have a more lateral, shallower root system are more susceptible to wind damage than species with larger, deeper root systems (Everham and Brokaw 1996). Shallow or impermeable soil, high water table,
and soil texture (i.e. high clay content) can affect root systems by restricting the growing conditions that are needed for expansion.

Here we examine the combined effects of windthrow and salvage logging on hemlock-hardwood forests of the northern U.S. Great Lakes region. Our study overcomes one of the limitations noted by Lindenmayer (2006) in that salvaged and unsalvaged areas in our study area, the Flambeau River Hemlock-Hardwood Scientific Area, are proximal to one another and were both struck by a natural catastrophic windthrow event. In the context of most salvage logging studies, this work is furthermore long-term, spanning 25 years. In 1977, a severe wind event composed of 25 separate downbursts with winds that reached 253 km/h (Fujita 1978) removed most of the overstory on 344,000 ha of forest in the study region (Dunn et al. 1983). Dunn et al. (1983) documented the immediate changes in forest composition and structure associated with this catastrophic wind event through a pre- and post-disturbance comparison. Our work expands upon the effort begun by Dunn et al. (1983), employing the long-term data set collected on the Flambeau River Hemlock-Hardwood Scientific Area and adjacent salvage logged forest beginning in 1979, and resampled in 1990 and 2004. Based on review of literature addressing the effects of salvage logging, we formed the following hypotheses:

1. salvage logging lengthens the overstory recovery period and, thus, also alters understory community composition;
2. salvage logging alters tip-up structure and soil composition; and
3. the composition and structure of tip-ups affect forest community composition
(Figure 1).
We use a 25-year time series of vegetation data (1979, 1990, 2004) collected on and nearby the Flambeau River Hemlock-Hardwood Scientific Area to evaluate Hypothesis 1. We expect that the mechanism underlying the lengthening of recovery period within salvage logged areas is its negative effect on advanced regeneration at the time of harvest. van Nieuwstadt et al. (2001), Donato et al. (2006a), and Lindenmayer (2006) have shown this to be the case within their study areas.

Hypothesis 2 examines structural and compositional differences in tip-up microtopography between the Flambeau River Hemlock-Hardwood Scientific Area and the adjacent salvage logged area. We expect that, at the time of harvest, logging equipment disturbs and compacts soil associated with tip-ups. Over time, this could affect other soil attributes (Starr 2000, Lindenmayer 2006, Lindenmayer and Noss 2006b). We use a data set, which includes both soil and vegetation data, collected on and nearby the Flambeau River Hemlock-Hardwood Scientific Area in 2004 to assess this hypothesis.

In Hypothesis 3, we test the relationship between forest community composition and tip-up structure and composition. Beatty (1984) studied the effects of microrelief (tip-up mounds) on soil properties and vegetation in a maple-beech forest in eastern New York and found microtopography created a mosaic of soil properties resulting, in non-hemlock areas, in patchy distributions of most understory species. Beatty (1984) also noted the pit offered more favorable growing conditions (except extended periods of soil saturation in the spring and thick litter), but the mound generally had greater species richness and cover. Here we make use of the same data set as that used in Hypothesis 2.
MATERIALS AND METHODS

Study Area

The study area is located within the Flambeau State Forest in Sawyer County, WI, at 90° 45’ W, 45° 44’ N (Figure 2). The Flambeau State Forest is located within the Laurentian Mixed Forest Providence (Keys et al. 1995), which has a humid-continental climate characterized by mild summers and cold winters. Mean monthly temperature in July is 18.7°C and -11.5°C in January based on a 30 year mean at Winter, WI (1971-2000; Wisconsin State Climatology Center 2006). The extreme low for January temperature was -43.0°C in 1927, and the extreme high for July was 41.0°C in 1936. Mean annual precipitation is around 84 cm with most precipitation occurring in the summer months (34.7 cm) (Wisconsin State Climatology Center 2006).

The land that the Flambeau State Forest is located on has a complex glacial history. It is on the Chippewa Lobe, which was formed about 15,000 to 18,000 years ago by a glacier that advanced from the northeast (Dott and Attig 2004). The Chippewa Lobe is composed of extensive outwash plains, till plains, recessional moraines, esker fields, and drumlin fields distributed in a complex spatial pattern (Dott and Attig 2004). During glacial advances and wasting, as well as into the early Holocene, loess and eolian sands were deposited across this area (often in an intermixed fashion); their combined current thickness ranges from a few centimeters to one meter or more. Within the Flambeau Forest Scientific Area, the key surficial geologic strata appear to be loess, outwash, and eolian sands (NRCS 2006). All of these strata are inherently acidic due to the absence of carbonate rocks within the glacial drift.
Four soil series are mapped within the Flambeau Forest Scientific Area: Antigo, Pence, Sconsin, and Wormet, with Sconsin being the predominant soil within the salvaged area and the Pence being the predominate soil within the unsalvaged area. All four series formed under forest, which tended to further acidify these soils (Hole 1976). These soil series are nearly identical in the top 20 cm or so and are naturally acidic with textures that are sandy or silty—the distinction between these being unusually small because of the intermixed conditions of loess and sands that were blown across this landscape around 10,000 years ago (Lee Burras, November 07, 2006, personal communication). This similarity in texture was qualitatively verified by hand analysis in the laboratory (data not presented). The four series differ primarily in parent material and secondarily due to pedogenic processes (Lee Burras, November 07, 2006, personal communication). The major and most consistent difference in these four soils series occurs below a depth of about 50 cm. It could be speculated that perched water tables, dense strata, and other such conditions would likely limit the effectiveness of root anchoring by trees; however, such an analysis is beyond this thesis. More specifically, the Sconsin (coarse-loamy, mixed, superactive, frigid Oxaquic Glossudalf) is a moderately well drained soil that formed from loess over outwash or till (NRCS 2006). It abruptly transitions to dense conditions at a depth of 50 cm or so. That means a seasonally perched water table as well as other root restrictive conditions are present at about 50 cm; however, above that depth the soil is well suited for tree root growth. The Pence (sandy, isotic, frigid Typic Haploothod) is an excessively drained soil that formed from eolian sands, sandy outwash, and local alluvium (NRCS 2006). This suggests that drought potential is higher, but otherwise this soil is as suitable for tree roots as the Sconsin. In terms of our analysis, all four of these soil series can logically be grouped together in terms of
understanding shallow soil properties and plant growth (e.g., soil organic carbon content, soil organic nitrogen content, bulk density to 15 cm depth) (Lee Burras, November 07, 2006, personal communication).

Study plots are located within and around the Flambeau River Hemlock-Hardwood Scientific Area (Figure 2). Our assumption, vegetation cover was similar prior to the blowdown event, is based upon unpublished data associated with Dunn et al. (1983), personal communications, and analysis of aerial photographs taken prior to the blowdown event. The Scientific Area was about 360 acres surrounded by old growth forest known as the “Big Block”; a 3,000 acre tract of virgin northern hardwood and hemlock forest (Glenn Guntenspergen, November 26, 2006, personal communication). Severe fires in the 1930’s missed the “Big Block” and there were small blowdowns in the “Big Block” in 1949 and 1951 with some salvage logging (Glenn Guntenspergen, November 26, 2006, personal communication). The Scientific Area was established in the least damaged area of the Big Block but disturbance in the rest of the Big Block was minimal. The Big Block was considered “virgin timber” in various Wisconsin DNR reports and an evaluation done by Dr. Philip Whitford for a Natural Landmark evaluation (Glenn Guntenspergen, November 26, 2006, personal communication). Prior to the 1977 blow down event, the Scientific Area and adjacent areas were dominated by Tsuga canadensis (L.) Carr., while Betula alleghaniensis and Acer saccharum (Marsh.) were present in the sub-canopy (Dunn et al. 1983). Pinus strobus, American elm Ulmus Americana (L.), and Tilia americana were also present, but not as common as the above species (Dunn et al. 1983). Tree ages ranged from 250 to 400 years and the mean basal area within the Scientific Area was estimated to be about 41 m$^2$/ha.
Although not allowed within the Scientific Area, adjacent windthrown forest was salvaged logged within two years of the wind event.

**Field methods**

In 1979, Dunn et al. (1983) established a sampling network in the Flambeau River Hemlock-Hardwood Scientific Area, which because of its scientific area status was not salvaged logged after the blowdown. Hereafter, we refer to this area and sampling plots falling within it as ‘unsalvaged’. Twenty-eight plots were established in this area, 15 in a north-south transect and 13 in an east-west transect (Figure 2). Although the data were not published by Dunn et al. (1983), sampling plots were also established in nearby (<0.5 km) areas that were salvage logged. Two transects were located to the north of the unsalvaged area in an area that had been salvage logged during the winter. Another two transects were located in a summer salvage area to the northeast. Each of these transects is comprised of eight plots, for a total of 32 salvage plots (Figure 2). Sampling in both salvage and unsalvaged areas was conducted at three time periods: 1979, 1990, and 2004.

Each sampling plot is composed of a 0.025 ha sampling area (10 m x 25 m), spaced 30m apart from each other along the transects (Figure 4). All live trees and tip-ups falling within the 0.025 ha sampling area were recorded. Saplings, shrubs, herbs and tree seedlings were inventoried in subplots nested within the overall sampling plot. Dimensions of the subplots were 0.0125 ha (5m x 25m), 0.0025 ha (2.5m x 10m), and 0.00025 ha (2.5m x 1m) for saplings, shrubs, and herbs/seedlings, respectively. Within the 0.025 ha tree plot, all stems ≥10 cm dbh (diameter at breast height) were recorded to species; dbh was also measured and tallied (Dunn et al. 1983). All saplings (stems 2.5-10cm dbh and >3m tall) and
shrubs (stems <2.5 cm dbh and 1-3 m tall) falling within their respective subplots were counted and recorded to species. Herb and seedlings (<1 m tall) were tallied as percent cover by species. Protocol for capturing stems measuring >2.5 cm dbh and < 3 m tall were absent from the Dunn et al. (1983); thus, we established a new category, called large-woody seedlings, to fill this gap. All large-woody seedlings falling within the sapling subplot were recorded.

In addition to resampling vegetation data, we intensively sampled tip-ups within the unsalvaged and summer salvage logged areas in 2004. Using the plots established by Dunn et al. (1983), we randomly selected and sampled four tip-up mounds within each plot. At each tip-up, we measured mound height and pit depth using a stadia rod, level, and a string; three measurements were taken for mound height and pit depth and averaged to obtain mean values per tip-up. Vegetation associated with the tip-ups was measured as percent cover in a 25-cm by 50-cm quadrat at five positions: (1) mound top, (2) mound slope, (3) pit, (4) opposite slope, and (5) a 'control' position located 2 m away from the mound on the side of the downed tree (Figure 5). We also gathered soil samples to a depth of 15 cm (a 188.4 cm³ sample) at each of these five positions. These soil samples were stored in an air tight container and kept cool (5°C) until they could be later assessed for pH, bulk density, texture, color/value rank, organic carbon content, and organic nitrogen content using standard protocol in a soil analysis lab. Percent weight for organic carbon and organic nitrogen was calculated using the following formula: (weight of organic carbon or nitrogen / 2g soil sample weight)*100.
Data analysis

Hypothesis 1.—Vegetation data from all three sampling periods were combined to determine the effect of salvage logging on forest regeneration and understory community composition. We employed graphical analysis and a general linear model (GLM) within SAS (SAS Institute 1999) to understand vegetation changes over time and test for differences in recovery between salvaged and unsalvaged sites. Because soils as delineated by the USDA Natural Resources Conservation Service were not equally represented among study salvaged and unsalvaged sampling plots (USDA NRCS 2006a; Figure 3), we first ran a conservative GLM model accounting for soil affects on vegetation variables. We found that soil was not significant for any vegetation variable within this initial model and, thus, subsequently removed it from further models associated with Hypothesis 1. Within the GLM model, the disturbance (salvaged/unsalvaged) effect was analyzed by blocking through time and testing for linear contrast. Disturbance and time were class variables within our model. Disturbance was furthermore treated as a fixed effect and sampling plot identification number was treated as a random effect. Using this model, we tested for differences in total tree basal area, sapling density, shrub density, and herbaceous species richness.

Hypothesis 2.—We used topographical and soil attribute data to evaluate the effects of salvage logging on the overall composition and structure of tip-ups. Variables evaluated using a mixed linear models within SAS (SAS Institute 1999) included tip-up structure (mound height and pit depth), soil bulk density, soil pH, and percent organic carbon and percent organic nitrogen from the total weight (g) of the soil sample. Again, because soils were not equally represented among study salvaged and unsalvaged plots (Figure 3), we used four levels of statistical modeling to understand the effects of salvage logging on tip-up
structure and composition (Table 1). First, we tested for the effect of the different soils on
the attribute data, ignoring the effect of disturbance, to develop a baseline understanding of
the effect of the different soil types on soil characteristics. Soil was treated as a fixed effect
within this mixed linear model, and this test was conducted on control positions (i.e., no
microtopography) only. Secondly, we evaluated the effect of disturbance alone on the
Sconsin (soil code 648B) soil (Table 1), as it was represented at several sites within both
salvaged and unsalvaged areas (Figure 3). Disturbance was treated as fixed effect within the
mixed linear model, and we evaluated its effect on both tip-up structure and soil composition.
Because of differences in mound structure between salvaged and unsalvaged areas for the
Sconsin soil series, we performed a second test on this type, which also evaluated the
combined effects of disturbance and tip-up position (Model III in Table 1); disturbance and
position were treated as fixed effects within the mixed linear model. Our fourth model
(Table 1) evaluated our original hypothesis of salvaged logging effect on the overall
composition and structure of tip-ups. Within this mixed model, soil and disturbance were
treated as fixed effects while position, position by soil, and position by disturbance were
treated as random effects. Position by soil and position by disturbance showed no significant
differences in an initial model and were thus removed from subsequent models. We used the
combined results of these tests to understand the effect of salvage logging on tip-ups.

Because of left skewdness and high levels of variability within the soil data, we
originally used a log transformation to obtain a normal distribution among soil variables.
Preliminary analysis showed that transforming had no effect on the model outcomes; hence,
we report the results from our original, untransformed data.
Hypothesis 3.—We assessed the relationship between tip-up composition and structure and forest community composition. We first used cluster analysis to understand the relationships between forest community composition and tip-up topography, making no *a priori* assumptions regarding species response to disturbance (unsalvaged/salvaged) or tip-up position. Because our data set is large (855 samples), we followed a two phase process, in which preliminary clustering (PROC FASTCLUS; SAS Institute Inc. 1990) was used to reduce the dimensionality of the data and hierarchical agglomerative clustering was used to show the relationships among groups. Data included in the cluster procedure included a trimmed species data set: species included in the analysis were represented at ≥8% of all sampling sites. We removed rare species occurrences because of a large number of zero values, which obfuscated patterns in the data. The 8% cutoff was chosen according to a natural break in the data. Eight species that met the 8% cutoff and included ‘moss’ (all mosses combined into one category), *Athyrium filix-femina* ((L.) Roth), *Poa* (L.) spp., *Laportea canadensis* ((L.) Weddell), *Hydorphyllum virginianum* (L.), *Rubus allegheniensis* (Porter), *Solidago* spp. (L.), and *Gymnocarpium dryopteris* ((L.) Newman). We also included ‘bare soil’ in the analysis, representing sampling positions that had no herbaceous cover (22% of total samples).

The FASTCLUS procedure within SAS finds inherent groups within data sets and then assigns samples to clusters using an iterative procedure and a minimum distance rule (SAS Institute Inc. 1999). To understand the patterns within our data, we ran FASTCLUS multiple times using a range of starting cluster seeds (10 to 100 initial cluster seeds in intervals of 10). We determined which cluster level to proceed with by grouping the number of output clusters (clusters remaining after completion of the iterative procedure) and related
R² values (measure of variation explained by the clustering procedure). R² values generally increased with the number of initial cluster seeds to about 0.82 at 80 cluster seeds, and then leveled off at around 0.75 (Figure 6). Although the 80 cluster seeds had the highest R² value (R² = 0.82), we chose to use output from the run with 50 initial cluster seeds (R² = 0.72) for the second phase of classification because we wanted to explain most of the variance with as few clusters as possible. The 50 initial cluster seed run met these criteria (Figure 6).

The second phase of clustering used Ward’s minimum-variance method (Ward 1963) to create a dendrogram through hierarchical agglomerative clustering. The dendrogram allowed us to examine the relationships between output clusters from the FASTCLUS procedure (Figure 6). Final clusters were named according to the following guidelines:

(1) groups in which the first ranking species represented ≥25% of the herbaceous cover and had at least two times the cover as the second ranking species were named according to the first ranking species and given a ‘high’ designator (e.g., ATFE high);

(2) groups in which the first ranking species that did not meet the ≥25% herbaceous cover cutoff but had two times the cover as the second ranking species were named according to the first ranking species and given a ‘low’ designator (e.g., ATFE low);

and,

(3) groups where species were mixed were defined by the first and second ranking species (e.g., ATFE-POA).

The bare soil cluster was an exception as there were no dominant species, and it was named according to this characteristic. In analyzing the results by cluster, we down-weighted the proportions of samples from the unsalvaged area to account for the disparity in sample sizes between salvaged and unsalvaged areas.
The second phase of analysis regarding Hypothesis 3 used mixed linear modeling within SAS (SAS Institute Inc. 1999) to test for statistical differences in forest community composition by tip-up characteristics (i.e., tip-up position, percent organic carbon, percent organic nitrogen). Because of high levels of absence for most species tallied, we grouped species together by life trait for this analysis. Although we categorized all species according to their primary form of seed dispersal (i.e., wind, deposited near plant, or animal), life term (i.e., perennial or annual), growth habit (i.e., forb/herb, grass, moss, shrub, or vine), and level of shade tolerance (i.e., shade intolerant, shade mid-tolerant, shade mostly tolerant, shade tolerant), we only tested for differences within the following subset because they showed variation among the observed species: (1) wind-dispersed, (2) shade intolerant, and (3) shade tolerant. Many of the species ranged in level of shade tolerance (USDA NRCS 2006b); therefore, the shade mid-tolerant species were combined with the shade intolerant species and the shade mostly tolerant species were combined with the shade tolerant species. We also tested for differences in the presence and abundance of indicator species, including *Trillium grandiflorum* ((Michx.) Salisb.), *Adiantum pedatum* (L.), *Solidago* (L.) species, and *Maianthemum canadense* (Desf.). *T. grandiflorum* is common in rich woods (Duncan and Duncan 1999) especially on moist shaded slopes (Ladd 2001). *T. grandiflorum* is the most common trillium species found in Wisconsin hardwood forests and does not respond well to disturbance such as logging (Rooney 2000). *A. pedatum* is common in rich, moist, well-drained woods and places with deep hummus (Wharton 1971) indicating a more intact canopy and little disturbance. *Solidago* species are common in moist to dry open places (Duncan and Duncan 1999) and are abundant in disturbed areas (Ladd 2001). *M. canadense* is abundant in all types of woods and prefers more open woods, but can persist in the deep
shade under conifers (Ladd 2001). *M. canadense* is an indicator of a northern mesic forest regime (Epstein et al. 2002).

Within the mixed model, disturbance and soil were treated as fixed effects while position, position by soil, and position by disturbance were treated as random effects. Position by disturbance showed no significant difference in an initial model and was removed from subsequent models. Metrics tested included species richness, percent cover for *Trillium grandiflorum, Adiantum pedatum, Solidago, Maianthemum canadense*, percent cover of wind-dispersed species combined, percent cover of shade intolerant species combined, and percent cover of shade tolerant species combined. Percent cover by life trait group was derived by summing the percent cover of corresponding species for each position.

**RESULTS**

*Salvage logging affect on overstory and understory forest community regeneration*

We found significant differences between the unsalvaged and salvaged areas in the overstory and understory regeneration rates and through time (Figures 7-11). Tree basal area (m$^2$/ha) was significantly greater in the unsalvaged area than the salvaged ($F=4.95, P=0.03$) and also carried significance through time ($F=71.26, P<0.01$) (Figure 7a). Mean tree basal area was 1.1 times higher in the unsalvaged than the salvaged area in 2004. *Acer saccharum* and *Tilia americana* were the dominant species in the unsalvaged area while *Betula alleghaniensis* and *Ulmus americana* were dominant in the salvaged area (Figure 8). In the unsalvaged area, *Acer saccharum* and *Tilia americana* increased fairly steadily through time. *Ulmus americana* appeared to be a dominant species in 1979, but only one large tree was
recorded. Its basal area sharply declined in 1990 (tree lost from 1979), and then recovered over time with the 2004 tree basal area double that of 1990 (Figure 8).

*Betula alleghaniensis* was more successful in the salvaged area, continually increasing in average tree basal area over time (~1.60 m$^2$/ha per collection) (Figure 8). *Betula alleghaniensis* in the unsalvaged area started with a high average basal area (6.36 m$^2$/ha) in the 1979 sample, but then sharply declined (0.97 m$^2$/ha) by 1990 (Figure 8). It is likely that live trees damaged by the windthrow event were counted in the 1979 collection, but died before the 1990 collection, accounting for this decline. *Betula alleghaniensis* basal area increased (2.45 m$^2$/ha) in 2004, but was still well below average basal areas for *Acer saccharum* (5.42 m$^2$/ha), *Tilia americana* (4.48 m$^2$/ha), and *Ulmus americana* (4.41 m$^2$/ha) (Figure 8). Tree densities for *Betula alleghaniensis* largely mimic those of basal area; however, high densities within the unsalvaged area during 2004 sampling period suggest that this species may be a strong component of the future forest (Table 2).

Though dominant prior to the windthrow event, *Tsuga canadensis* has been steadily declining in tree basal area over time and was not recorded in the 2004 sampling period (Figure 8). Although it appeared in 1990 that hemlock had an increased in basal area for the salvaged area (Figure 8), the tree numbers had actually declined (Table 2).

Overall, saplings were more abundant in the salvaged than the unsalvaged area ($F$=19.66, $P<0.01$), but showed no statistical significance through time ($F$= 2.72, $P=0.1$) (Figure 7b). Sapling density doubled in the unsalvaged area from 1979 to 1990 (~13,400 saplings/ha) and then declined (~6100 saplings/ha) in 2004 (Figure 7b). Density quadrupled in the salvaged area from 1979 to 1990 (~17,300 saplings/ha) and then declined (~10,700 saplings/ha) in 2004 (Figure 7b). *Betula alleghaniensis* saplings followed a pattern similar to
that of *Betula alleghaniensis* trees. *Betula alleghaniensis* steadily increased in the salvaged area while decreasing overall in the unsalvaged area (Figure 9). *Acer saccharum* decreased while *Tilia americana* increased in the salvaged area (Figure 9). *Salix* species and *Prunus pensylvanica* (L. f.) were commonly observed in the salvaged area in 1990 (Table 3), but show large overall temporal shifts, likely in response to changes in canopy cover. *Prunus pensylvanica*, a shade intolerant species, decreased while *Salix* species, *Ulmus thomasii* (Sarg.), and *Ulmus americana* increased in density (Table 3).

Advanced canopy development in the unsalvaged area likely affected the sapling density and composition over time. *Acer rubrum* (L.), *Tilia americana*, *Prunus pensylvanica*, and *Acer saccharum* were most prevalent from 1979 to 1990, but *Prunus pensylvanica* density was greatly reduced as canopy cover increased (Table 2). The increase in canopy drove the sapling species composition as more shade tolerant saplings (*Acer saccharum*, *Fraxinus americana* (L.), *Ulmus rubra* (Muhl.), and *Ulmus thomasii*) replaced less shade tolerant saplings (Table 3). These shade tolerant saplings created a sub-canopy and likely filled remaining canopy gaps.

Shrub density (shrubs/ha) in the salvaged area was slightly greater than the unsalvaged area ($F=4.21$, $P=0.04$), and increased through time ($F=11.18$, $P<0.01$) (Figure 7c). *Acer saccharum* was commonly found within the shrub layer of both disturbed areas, but steadily increased in the salvaged area while declining in the unsalvaged area (Figure 10). *Sambucus* species, a shade intolerant species, was more common in the salvaged than in the unsalvaged area in 1979 (Figure 10). *Sambucus* declined in both areas in 1990 and was absent in 2004. Some shade mid-tolerant species, such as *Rubus* species and *Rubus allegheniensis*, had much higher densities in 2004 in both salvaged and unsalvaged areas in
comparison to previous years (Figure 10). *Hamamelis virginiana* (L.), a mid-tolerant species, was observed in the unsalvaged, but not the salvaged, area (Figure 10). Densities for the ten most frequent shrub species are listed in Table 4.

Herbaceous species richness was significantly higher in the salvaged area, though not strongly so \((F=4.02, P=0.05)\), and showed no statistical significance through time \((F=2.80, P=0.09)\). Herbaceous species cover was also significantly greater in the salvaged area \((F=5.39, P=0.02)\), though this pattern decreased over time \((F=12.49, P<0.01)\) (Figure 7d). *Betula alleghaniensis* and *Viola* (L.) started with high percent cover frequencies, but decreased through time (Figure 11). Vegetative reproducing species, such as *Maianthemum canadense* (Desf.), took longer to recover in the salvaged area, and shows a consistent decline within the unsalvaged area (Figure 11). By 2004, the percent frequency for *Solidago* spp., generally shade intolerant species, was 2.8 times higher in the salvaged than the unsalvaged area (Figure 11). Densities for the ten most frequent herbaceous species are listed in Table 5.

**Salvage logging affect on the overall structure and composition of tip-up mounds**

We found a significant difference in tip-up structure between the disturbance areas. The mean mound height and pit depth for tip-ups in the unsalvaged area was greater than that in the salvaged area (height: \(F=114.50, P<0.01\); depth: \(F=25.10, P<0.01\)). The mean difference in tip-up mound height between the two areas was 0.27 m (Figure 12). In the unsalvaged area, tip-up mound height averaged 0.62 m and pit depth averaged -0.24 m. In the salvaged, tip-up mound height averaged 0.34 m and pit depth averaged -0.13 m.
In analyzing tip-up structure between the soils, we found a significant difference for pit depth \( (F=9.74, P<0.01) \), but not for tip-up mound height \( (F=1.47, P=0.20) \). The Sconsin (648B) soil averaged 0.61 for tip-up mound height in the unsalvaged area and 0.36 in the salvaged area. Pit depth on Sconsin soils averaged -0.20 m for unsalvaged and -0.14 m for salvaged.

In evaluating our first model (Table 1), we found significant differences between all soils for bulk density \( (F=5.5, P=0.01) \) and percent weight of organic nitrogen \( (F=2.73, P=0.05) \), but not pH \( (F=2.34, P=0.08) \) or percent weight of organic carbon \( (F=0.92, P=0.4) \). These results suggest that the soils our sampling sites are located on are different and, because soils are not equally represented across sampling sites, that the effect of soil must be accounted for in any conclusion regarding the effect of disturbance, especially with regard to interpretations involving bulk density and nitrogen.

Our second model, which evaluated the effect of disturbance on just the Sconsin soil series, removed the soil effect. We tested the control position and found percent weight of organic nitrogen to be higher in the salvaged area \( (F=5.39, P=0.03) \), but bulk density \( (F=0.01, P=0.93) \), pH \( (F=0.29, P=0.59) \), and percent weight of organic carbon \( (F=2.19, P=0.14) \) showed no statistical difference. These results suggest that salvage logging has at least some effect on soil nitrogen dynamics. Because of differences in mound structure between salvaged and unsalvaged areas for the Sconsin soil series, we performed a second test on this type, which also evaluated the combined effects of disturbance and tip-up position (Model III in Table 1). The disturbance effect found bulk density \( (F=8.21, P=0.01) \), percent weight of organic carbon \( (F=28.69, P<0.01) \), and percent weight of organic nitrogen \( (F=54.98, P<0.01) \) to be higher in the salvaged, but pH \( (F=1.28, P=0.26) \) showed no
statistical difference. In evaluating the tip-up position effect, we found significant
differences in bulk density \((F=6.66, P<0.01)\), percent weight of organic carbon \((F=2.12, P=0.08)\), and percent weight of organic nitrogen \((F=7.31, P<0.01)\), but pH showed no
statistical difference. These results suggest that salvage logging may be affecting soil bulk
density, soil carbon, and soil nitrogen dynamics.

In evaluating both soil and disturbance effects across all tip-up positions (Model IV in
Table 1), we found significant differences in all three effects for percent weight of organic
nitrogen \((\text{soil } F=8.95, P<0.01; \text{disturbance } F=71.67, P<0.01; \text{and position } F=2.70, P=0.03)\). The average percent weight of organic nitrogen was 0.18% higher for salvaged versus
unsalvaged areas (Figure 13b). Bulk density \((\text{soil } F=19.60, P<0.01 \text{ and disturbance } F=7.18, P=0.01)\) and percent weight of organic carbon \((\text{soil } F=4.11, P=0.01 \text{ and disturbance } F=29.50, P<0.01)\) varied by both soil and disturbance, but not for position (bulk density
position \(F=1.66, P=0.16, \text{Figure 13d}; \text{percent weight of organic carbon position } F=1.75, P=0.14, \text{Figure 13a})\). The average percent weight of organic carbon was 1.9% higher for
salvaged versus unsalvaged areas (Figure 14a). pH varied by soil only \((F=6.76, P=0.01)\);
disturbance \((F=1.25, P=0.26)\) and position \((F=0.52, P=0.72)\) showed no difference (Figure
14c). These results match our Sconsin soil test results showing that salvage logging at least
has an effect on soil organic nitrogen and may also effect bulk density and soil organic
carbon, although our lack of replication across salvaged and unsalvaged areas for all soils
precludes strong conclusions.
Composition and structure affect on forest community composition

Cluster analysis revealed distinct patterns by disturbance and tip-up position (Table 6 and Figure 14). Ward’s minimum variance method identified moss, *Solidago*, *Athyrium filix-femina*, and *Hydorphyllum virginianum* as single species clusters explaining most of the variance (Figure 7). Moss and *Hydorphyllum virginianum* were represented in higher proportions in the unsalvaged area while *Solidago* and *Athyrium filix-femina* had higher proportions in the salvaged area (Table 7). *Hydorphyllum virginianum* followed the same pattern within each area (unsalvaged/salvaged), but also showing a preference for pit and opposite side tip-up positions (Table 7). The top tip-up position was highly favored by moss in the unsalvaged area while *Athyrium filix-femina* favored the top position in the salvaged area (Table 7). Although *Solidago* cover was generally high over all positions in the salvaged area, the control, mound side, and opposite side positions had larger proportions than did top and pit positions (Table 7).

There was a higher occurrence of more bare ground on tip-ups within the unsalvaged than the salvaged area (Table 7). But we found the same distribution of bare ground samples by positions regardless of disturbance type: the top position had the lowest occurrence and the pit had the highest occurrence of bare ground, respectively (Table 7).

The results of our test for differences by tip-up position and composition showed significant differences in species richness by disturbance ($F=24.28$, $P<0.01$), position ($F=22.57$, $P<0.01$), and soil ($F=7.53$, $P<0.01$). Species richness was higher in the salvaged area averaging 2.27 more species per collection than in the unsalvaged area. Both areas followed similar patterns for the species richness by tip-up position. The top position had the highest average number of species (~3.20) while the pit had the lowest average number of
species (~1.7); however, high levels of variation exist among our samples (Figure 15).

Sconsin soil had the highest average number of species (2.57) and Antigo had the lowest average number of species (1.70).

Significant differences by tip-up structure were found within all the three life trait groups analyzed. Percent cover of wind-dispersed species was significantly higher within salvaged \((F=5.32, P=0.02)\) areas and on the top of tip-ups \((F=26.87, P<0.01)\); no significant difference was exhibited by soil \((F=1.30, P=0.27)\). The top position average percent cover was 13.8% higher than the pit position in the unsalvaged and 9.0% higher in the salvaged (Figure 16a). The percent cover of shade intolerant species showed significant differences by disturbance \((F=34.85, P<0.01)\), position \((F=11.10, P<0.01)\), and soil \((F=8.42, P<0.01)\). The percent cover for the salvaged area was 16.1% greater than the percent cover for the unsalvaged area (Figure 16b). The average percent cover at the top position was higher than average percent cover for the pit position in both areas (Figure 16b). The percent cover of shade tolerant species did not differ by disturbance type \((F=0.95, P=0.33)\), but both the position \((F=10.37, P<0.01)\) and soil \((F=3.18, P=0.02)\) effects showed differences. The average percent cover at the top position was 10.8% higher than that of the pit position in the unsalvaged area; and the corresponding number was 5.8% higher in the salvaged area (Figure 16c).

We found significantly higher levels of *Solidago* species within the salvaged area \((F=22.00, P<0.01)\), complementing what we found with our cluster analysis, and by soil \((F=3.16, P=0.02)\). The average percent cover for *Solidago* species was 5.4 times higher for the salvaged versus unsalvaged area, a difference that is significant despite the large variation within the data (Figure 17c). *Solidago* species had no statistical significance for position \((F=
0.25, \( P=0.90 \)) or position by disturbance (\( F=0.17, P=1.00 \)).  *Maianthemum canadense* had significant difference for position (\( F=8.61, P<0.01 \)), soil (\( F=5.14, P=0.01 \)), and position by soil (\( F=2.88, P=0.01 \)). Top position average percent cover of *Maianthemum canadense* was 10% higher than the pit position in the unsalvaged and 16.7% higher in the salvaged forest (Figure 17d). No statistical difference was found by disturbance (\( F=2.89, P=0.09 \)). We did not find statistical differences for either *Trillium grandiflorum* or *Adiantum pedatum* (Figure 17a, b).

**DISCUSSION**

*Forest recovery following windthrow and salvage logging*

The rapid establishment of canopy cover appears to be a key factor in recovery differences seen between unsalvaged and salvage logged forest. Canopy cover establishes more quickly in unsalvaged forest due to the lack of disturbance to remaining biological legacies (e.g., downed logs, tip-up mounds, re-leafing damaged canopies). Our work supports that of others in showing the salvage logging, by removing downed canopies and disturbing to root masses, sets back the successional development of naturally disturbed forest (Cooper-Ellis et al. 1999, van Nieuwstadt 2001, Lindenmayer 2006).

Despite the fact that *Betula alleghaniensis* is a prolific seeder (Burns and Honkala 1990, Marx 2005) and seedlings were highly abundant shortly following the windthrow event (Dunn et al. 1983, unpublished data) (Table 5), we found that *Tilia americana* was more successful in re-establishing on our site, likely because of even mature trees to stump sprout and because it can better compete with the shade tolerant sugar maple (Burns and Honkala
The extensive root system that *T. americana* has allows its sprouts to expend energy on growing vertically; in comparison, *Acer saccharum* sprouts are using energy to expand their root systems (Burns and Honkala 1990). *B. alleghaniensis* is not as shade tolerant as *A. saccharum* (Burns and Honkala 1990) and, therefore, we expect that the higher basal area it shows today in the salvage logged forest, where the canopy is less well developed, is due to differences in light availability.

*Tsuga canadensis* has not recovered as did the hardwoods despite dominance prior to the windthrow event (Dunn et al. 1983). A few *T. canadensis* trees and saplings are recorded in the 1979, fewer still in 1990 and *T. canadensis* is absent from the 2004 collection (Figure 8 and 9). Several factors influencing *T. canadensis* decline in this forest system include salvage logging, competition and rapid regeneration of other tree species (hardwood competition), increase in deer browse, and climate shift (Mladenoff and Stearns 1983, Frelich and Lorimer, 1985, Peterson 2000b, Frelich 2002).

The shifts in sapling density across disturbance types are influenced by three main factors: shifts between sampling categories (i.e., sapling and tree) due to growth, sapling mortality, and fluctuations among species. As some saplings grow large enough to be classified as trees, they affect the other two factors. Shade intolerant species, including *Populus tremuloides* (Michx.) and *Prunus pensylvanica* are reduced or disappear while the shade tolerant species remain (Table 3).

Witch hazel is a shade mid-tolerant species that should be present in both areas, but has only been recorded in the unsalvaged area. Witch hazel mainly propagates by seed (Smith 2006), but its roots are sensitive to disturbance (Evans 2006). The changing environmental conditions associated with windthrow followed in close succession by salvage
logging disturbance may have been too substantial for witch hazel to survive in the salvaged areas. Witch hazel will take more time to recover in the salvaged area as the seeds are slowly dispersed between the areas.

Although herbaceous cover was similar between the unsalvaged and salvaged areas directly following the disturbances, we found higher overall herbaceous cover within the salvage logged forest today (Figure 7d). The richness of herbaceous species was also slightly higher in the salvaged forest, although individual herb response is highly variable over time (Table 5). We expect these stronger responses in the salvaged area to be due to lower initial canopy cover (Figure 7a) and, although not directly measured by this study, the likely larger amount of exposed mineral soil, providing optimal locations for the establishment of some species. The salvaged area has more shade intolerant species such as Polygonum cilinode, Taraxacum officinale, and Lactuca biennis and species that are are associated with disturbance such as Solidago (Table 7) (Ladd 2001, USDA NRCS 2006b). The unsalvaged has more conservative species such as Gymnocarpium dryopteris, Moss species, and Lycopod (Wharton 1971), and has quicker regeneration of shade tolerant shrubs such as Ribes species (Table 5, Figure 11) (USDA NRCS 2006b).

Salvage logging effects on the structure and composition of tip-ups

We found salvage logging had an effect on the structure and composition of tip-up mounds (Figures 12 and 13). We found significantly higher tip-up mound heights and deeper pits were deeper in unsalvaged forest. Tip-up pits fill with sediment and debris over time through erosional processes (Beatty 1984, Ulanova 2000); salvage logging practices tend to
speed the process by further disturbing the soil and increasing the level of erosion. Although we can not draw strong conclusions regarding soil variability due to the lack of replication across salvaged and unsalvaged areas for all soils, our analysis does suggest that salvage logging does have some impact on soil composition. Salvage logging appears to be having an effect on soil nitrogen dynamics. Bulk density and soil carbon had statistical significance for soil and disturbance suggesting that salvage logging may have different effects on different soils. Betchta et al. (2006) found that soils with higher clay content have a higher rate of compaction from the logging machinery, thus affecting the bulk density. Recent studies have shown that salvage logging does affect soil nutrient composition and soil compaction (Beschta et al. 2006, Johnson et al. 2005), however, these studies focused on post fire salvage logging where fire can also alter the composition of the soil. Further research needs to be conducted for salvage logging following windthrow to determine how salvaged logging effects soil composition in absence of fire effects.

Vegetation response to forest microtopography

Our analysis of the effect of microtopography on plant communities shows that plants strongly respond to tip-up structure and exposure of bare mineral soil agreeing with findings from previous tip-up studies (Beatty 1984, Carlton and Bazzaz 1998, Palmer et al. 2000, Ulanova 2000). The pit position appears to be the poorest position for plant growth; bare soil (no plant cover) was most frequently recorded in pit positions (Beatty 1984, Carlton and Bazzaz 1998, Ulanova 2000) (Table 7). _H. virginianum_ seems to be an exception; the highest cover levels for this species were recorded within tip-up pits and in the opposite slope position (Table 7). Generally, the top position has the highest plant cover, even higher than
the controls (Beatty 1984, Carlton and Bazzaz 1998, Palmer et al. 2000, Ulanova 2000) (Figure 15). Moss, *Poa* spp., and *G. dryopteris* are especially associated with tip-up mound tops (Table 7). An interesting relation was noted for position and disturbance between moss and *Athyrium filix-femina*. The top position was important for both species, however disturbance affected which species was more successful. *Athyrium filix-femina* had greater cover on the top position in the salvaged area as it could handle more sun light. Moss established more quickly in the undisturbed area due to a faster recovering canopy (Table 7). *Solidago* has a strong positive response to salvage logging disturbance in both the cluster analysis and indicator species analysis (Table 7, Figure 17c). *Maianthemum canadense* responded to tip-up position consistently having more cover for the mound top position than the pit position (Figure 17d). *Trillium grandiflorum* and *Adiantum pedatum* did not have any difference between disturbance areas, which was unexpected as both species prefer fewer disturbances and are fairly shade tolerant (Wharton 1971, Rooney 2000). This suggests that salvaged logged areas that now have a well-developed canopy are now similar to the preferred habitat of these shade tolerant species.

The results of our statistical modeling also suggest that soils have an effect on vegetation response to forest microtopography. However, response relationships between plant composition and soil characteristics make it difficult to draw strong conclusions about the direct effect of soil composition on the plant cover. Further research needs to be conducted to better understand the relationship between these factors.
CONCLUSIONS

In discussing previous research on forest response to salvage logging, Lindenmayer (2006) notes the importance of having a naturally-disturbed, unsalvaged area as a baseline from which to evaluate the effects of salvage logging. Our research is powerful in that we are able to make such a comparison: our wind-disturbed, unsalvaged study area is located in close proximity to our salvage logged study area. Our study is also valuable in that we record a 25-year forest response to the combined effects of wind and salvage logging disturbance.

We found that salvage logging generally affects the rate of forest canopy recovery and has a long-term effect on vegetation patterns in the forest understory. Our research also shows that salvage logging also affects forest microtopography by altering the structure of tip-ups and possibly by altering soil composition, especially soil nitrogen. Finding biological legacies (tip-up formations) in the salvaged area 25 years post-harvest may suggest that a fairly low impact salvage harvest operation was conducted. Our results may be conservative compared to salvage logging operations as a whole, and especially high impact salvage operations.

Although 25 years is a short time in the recovery of an overall forest—many of the dominant tree species within our study area can live for of 250 to 300 years on average (USDA NRCS 2006b)—it is established within ecology that the removal or alteration of biotic and structural legacies can effect forest ecosystems over long time periods (Cooper-Ellis et al. 1999, Lindenmayer et al. 2004, Lindenmayer 2006). The differences in biotic and structural legacies that we record after 25-years are, thus, likely to last long into the future.
While there is no generic prescription for salvage logging that can be applied to all forest ecosystems (Lindenmayer and Noss 2006b), our research suggest that salvage logging alters the natural recovery response of this forest system further disturbing soil composition and understory vegetative community. Shade tolerant understory plants that can not compete with faster growing shade intolerant plants depend on quick canopy recovery to survive. In areas where severe natural disturbances, such as wind or fire, are frequent and salvage logging is a common response, forest managers should develop general disturbance response plans to guide the timing, location, and intensity of salvage harvests with attentiveness to minimizing its impacts (Lindenmayer et al. 2004, Lindenmayer 2006, Lindenmayer and Noss 2006b). Long-term research studying the effects of salvage logging effects on forest recovery and soil composition is also needed to inform forest management practices.

ACKNOWLEDGMENTS

Funding for this project was provided by the U.S. Federal McIntire-Stennis Funds, Iowa State University, and the U.S. Geological Survey. We respectively thank Dr. Lee Burras and Dr. Petrutza Caragea for advice on soils and statistical analysis. Thomas Anderson assisted with field work, Mary Jo Burkgren assisted with soil sample preparation, Zach Ankrom provided interpretations of soil texture, and Todd Hanson provided GIS assistance.

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CHAPTER 3. GENERAL CONCLUSIONS

I found that salvage logging affects the forest canopy recovery rate following a disturbance. I also found salvage logging affects the soil composition and understory communities. Although it appeared that important indicator species for mesic northern hardwoods, such as Trillium species, did not differ between the two disturbance areas, I noticed an interesting pattern between the long term and tip-up vegetation data sets. Trillium species appear to have even coverage between the two disturbance areas in my tip-up vegetation data (Figure 17a); however, the long term data set shows Trillium to be more consistently collected in the unsalvaged area suggesting that Trillium persists better with less disturbance and salvage logging may affect the distribution of mesic northern hardwood understory species.

Disappearance of eastern hemlock

The composition of a forest before and after a major disturbance influences the recovery of that system (Dunn et al. 1983). The Flambeau River Hemlock-Hardwoods Scientific Area appears to be shifting from Hemlock-Hardwoods to a Northern Hardwoods forest type. Acer saccharum, Tilia americana, Ulmus americana, Betula alleghaniensis, and Tsuga canadensis were all present in the canopy prior to the catastrophic wind storm (Dunn et al. 1983). After the disturbance, however, T. canadensis has not recovered as did the hardwoods. A few hemlock trees and saplings are recorded in the 1979, fewer still in 1990 and hemlock is absent from the 2004 collection (Figure 9 and 10). Several factors influencing T. canadensis decline in this forest system include competition and rapid
regeneration of other tree species (hardwood competition), increase in deer browse, and climate shift.

After a major disturbance in a Hemlock-Hardwood forest system, hardwoods typically replace the *T. canadensis* (Frelich and Graumlich 1994, Marx 2005). *T. canadensis* have a narrow set of conditions that must be met for germination and also have a high seedling mortality rate (Burns and Honkala 1990, Marx 2005). Most hardwoods, on the other hand, are able to establish more quickly and have better seedling success. Sugar maple’s rapid growth and abundant reproduction allow for quick establishment in the sapling layer (Dunn et al. 1983, Burns and Honkala 1990, Marx 2005). Another disadvantage for pines is the lack of ability to re-sprout or support a canopy once uprooted (Burns and Honkala 1990, Cooper-Ellis et al. 1999, Marx 2005). Hardwoods having the ability to re-sprout are more likely to replace fallen pines (Cooper-Ellis et al. 1999, Marx 2005). Re-leafing downed canopies and vegetative sprouting are important factors in the recovery of a low canopy (Cooper-Ellis et al. 1999).

White-tailed deer (*Odocoileus virginianus* Zimmermann) have increased in abundance in the Lake States over the last fifty years (Frelich and Lorimer 1985, Cornett et al. 2000, Frelich 2002). Deer can affect many plant species acting as a “keystone” herbivore altering Midwestern forest composition (Rooney et al. 2000). Plants that are susceptible to deer browse may fade from the forest composition over time (Frelich and Lorimer 1985, Cornett et al. 2000). In the winter, deer seek protection from winter winds and predators in area known as deer yards (Frelich and Lorimer 1985, Cornett 2000, Frelich 2002). Conifer stands are typical locations for deer yards as the conifers help reduce the wind and offer green browse (Frelich and Lorimer 1985, Frelich 2002). *T. canadensis* seedlings and
saplings are a “highly preferred” species by white-tailed deer for browsing (Frelich and Lorimer 1985, Frelich 2002). *T. canadensis* are not able to re-sprout once browsed as are the hardwood species like sugar maple (Rogers 1978, Frelich and Lorimer 1985, Frelich 2002). The browsing pressure by deer alters the sapling composition to favor hardwood regeneration while reducing the likelihood of *T. canadensis* recruitment success (Rogers 1978, Frelich and Lorimer 1985, Rooney et al. 2000, Frelich 2002).

Climatic shifts may have mixed effects on the retention of *T. canadensis* in the area. The Lake States region is one of the most active weather zones in the northern hemisphere (Frelich and Lorimer 1991, Frelich 2002). Outbreaks of severe weather occur in this region due to the polar jet stream lying just to the north and the subtropical jet stream lying just to the south during the summer months (Frelich and Lorimer 1991, Frelich 2002). If mid-latitude climates warm as predicted, an increase in down burst frequency and intensity is possible (Peterson 2000b). This could potentially increase the hardwood dynamic by reducing *T. canadensis* canopy cover and further fragmenting *T. canadensis* seed sources. Temperature and soil moisture shifts associated with climate change may also affect *T. canadensis* success. *T. canadensis* respond favorable to cooler climates and moist soil conditions (Burns and Honkala 1990, Rooney et al. 2000, Parshall 2002, Marx 2006). If soil moisture increases with rising temperatures *T. canadensis* may persist in the region (Parshall 2002). However, a decrease in soil moisture will make the region less habitable for *T. canadensis* (Parshall 2002).
Implications for future research

This study has shown that salvage logging is affecting this forest system. Research and future data collections are needed to track the long term implications of the salvage harvest. If I were to do this study again, I would change collection methods for herbaceous data. I would not only count the number of different species, but also try to estimate number present for each species and give an approximation of actual percent cover instead of using the current cover code. This would help to give a more accurate view of species cover and count.

For the tip-up study, I would establish new research plots for the different soils to attain more equal sampling between the two areas for better comparisons of salvage logging effect on soil composition and vegetation response. Concerning data analysis, I would first test tip-up vegetation data using a two phase process. The first phase would be preliminary clustering to reduce the dimensionality of the data and hierarchical agglomerative clustering to show the relationships among groups. The second phase would use Ward’s minimum-variance method (Ward 1963) to create a dendrogram through hierarchical agglomerative clustering. This would speed the understanding of what relationships exist and help inform further data analysis decisions.

LITERATURE CITED


Figure 1. Relationship between study components and hypotheses ($H_1$, $H_2$, and $H_3$).
Figure 2. Location of the Scientific Study Area within the Flambeau River Hemlock-Hardwood State Forest in Wisconsin. Inset: Location of the Flambeau State Forest with in North America.
Figure 3. Soils by sampling plot locations within the Flambeau study area.
Figure 4. Nested plot sampling design used by Dunn et al. (1983).
Figure 5. Tip-up positions used in vegetation and soil sampling.
Figure 6. Results from FASTCLUS procedure relating number of output cluster with $R^2$ values.
Figure 7. Comparisons between unsalvaged and salvaged areas over time: (A) average tree basal area (m²/ha), (B) sapling density (sa/ha), (C) shrub density (sh/ha), and (D) average percent herbaceous cover for 10 most abundant species.
Figure 8. Temporal comparison of tree basal area for key species between unsalvaged and salvaged areas.
Figure 9. Temporal comparison of sapling density for key species between unsalvaged and salvaged areas.
Figure 10. Temporal comparison of shrub density for key species between unsalvaged and salvaged areas.
Figure 11. Temporal comparison of herb occurrence for key species between unsalvaged and salvaged areas.
Figure 12. Comparison of average tip-up mound height (m) and pit depth (m) between unsalvaged and salvaged areas.
Figure 13. Comparison between salvaged and unsalvaged of soil composition characteristics by tip-up position: (A) percent total organic carbon, (B) percent total organic nitrogen, (C) average pH, and (D) average bulk density (g/cm$^3$). Percent values for organic carbon and organic nitrogen were derived from the percent of the weight of the 2g sample.
Figure 14. Dendrogram produced through hierarchical agglomerative clustering on tip-up vegetation cover data. See Table 6 for species frequencies within clusters.
Figure 15. Comparison between the unsalvaged and salvaged areas of average species richness by tip-up position.
Figure 16. Comparison between unsalvaged and salvaged areas of average percent cover within life trait groups by tip-up position: (A) wind-dispersed seeds, (B) shade intolerant species, and (C) shade tolerant species.
Figure 17. Comparison between unsalvaged and salvaged areas of average percent cover of indicator species by tip-up position: (A) *Trillium* species, (B) *Adiantum pedatum*, (C) *Solidago* species, and (D) *Maianthemum canadense*. 
Table 1. Components of the mixed linear models used to evaluate the effect of salvage logging on tip-up mound composition and structure.

<table>
<thead>
<tr>
<th>Model</th>
<th>Sites evaluated</th>
<th>Positions evaluated</th>
<th>Fixed effects</th>
</tr>
</thead>
<tbody>
<tr>
<td>I: Evaluation of soil only</td>
<td>All</td>
<td>Control only</td>
<td>Soil</td>
</tr>
<tr>
<td>II: Evaluation of disturbance</td>
<td>Sites on Sconsin (648B) soil only</td>
<td>Control only</td>
<td>Disturbance</td>
</tr>
<tr>
<td>III: Evaluation of disturbance and position</td>
<td>Sites on Sconsin (648B) soil only</td>
<td>All</td>
<td>Disturbance, position</td>
</tr>
<tr>
<td>IV: Evaluation of soil, disturbance, and position</td>
<td>All</td>
<td>All</td>
<td>Soil, disturbance, position, position<em>soil, position</em>disturbance</td>
</tr>
</tbody>
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Table 2. Tree density (D; trees/ha) and average tree basal area (BA; m²/ha) of common tree species for unsalvaged and salvaged areas.

<table>
<thead>
<tr>
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<tbody>
<tr>
<td></td>
<td>D</td>
<td>BA</td>
<td>D</td>
<td>BA</td>
<td>D</td>
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<tr>
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<td>120</td>
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<tr>
<td>Total</td>
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<td>7.81</td>
<td>4000</td>
<td>26.77</td>
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Table 3. Mean density (saplings/ha) of most common sapling species for unsalvaged and salvaged areas.

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<th>Species</th>
<th>1979</th>
<th>1990</th>
<th>2004</th>
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<td>Unsavaged</td>
<td>Salvaged</td>
<td>Unsavaged</td>
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<td><strong>Acer rubrum</strong></td>
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<td>1600</td>
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<td><strong>Acer saccharum</strong></td>
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<td><strong>Fraxinus nigra</strong></td>
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<td><strong>Ulmus thomasii</strong></td>
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<tr>
<td><strong>Total</strong></td>
<td>2204</td>
<td>1360</td>
<td>6426</td>
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Table 4. Mean density (shrubs/ha) of ten most frequent shrub species for unsalvaged and salvaged areas.

<table>
<thead>
<tr>
<th>Species</th>
<th>1979</th>
<th></th>
<th>1990</th>
<th></th>
<th>2004</th>
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<td>--------------</td>
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</tr>
<tr>
<td><em>Acer saccharum</em></td>
<td>1754</td>
<td>400</td>
<td>633</td>
<td>560</td>
<td>489</td>
<td>700</td>
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<tr>
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<td>400</td>
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<td>0</td>
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<td>0</td>
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<tr>
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<td>0</td>
<td>733</td>
<td>800</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td><em>Ribes spp.</em></td>
<td>1600</td>
<td>0</td>
<td>0</td>
<td>400</td>
<td>2867</td>
<td>2727</td>
</tr>
<tr>
<td><em>Rubus spp.</em></td>
<td>3350</td>
<td>1200</td>
<td>1600</td>
<td>1200</td>
<td>5900</td>
<td>8575</td>
</tr>
<tr>
<td><em>Sambucus spp.</em></td>
<td>2076</td>
<td>2600</td>
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<td>800</td>
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<td>0</td>
</tr>
<tr>
<td><em>Tilia americana</em></td>
<td>909</td>
<td>533</td>
<td>450</td>
<td>400</td>
<td>1867</td>
<td>3200</td>
</tr>
<tr>
<td><em>Ulmus americana</em></td>
<td>7600</td>
<td>1200</td>
<td>400</td>
<td>1600</td>
<td>533</td>
<td>1050</td>
</tr>
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<td><strong>Total</strong></td>
<td>22889</td>
<td>6333</td>
<td>19045</td>
<td>7560</td>
<td>18593</td>
<td>16252</td>
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Table 5. Ten most frequent herbaceous species and their percent frequencies for unsalvaged and salvaged areas.

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</thead>
<tbody>
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<td><em>Acer saccharum</em></td>
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<td>0</td>
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<td>19</td>
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<td>9</td>
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<td>0</td>
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<tr>
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<td>13</td>
<td>32</td>
<td>6</td>
<td>14</td>
<td>31</td>
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<td>16</td>
<td>25</td>
<td>13</td>
<td>21</td>
<td>23</td>
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<td>6</td>
<td>0</td>
<td>0</td>
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<td>31</td>
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<tr>
<td><em>Ribes spp.</em></td>
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<td>3</td>
<td>32</td>
<td>9</td>
<td>25</td>
<td>15</td>
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<tr>
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<td>53</td>
<td>93</td>
<td>44</td>
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<td>54</td>
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<td>16</td>
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</table>
Table 6. Species frequency of occurrence within output clusters. Cluster names are based on these data (see text for naming convention).

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<th>Cluster 1</th>
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<th>Cluster 3</th>
<th>Cluster 4</th>
<th>Cluster 5</th>
<th>Cluster 6</th>
<th>Cluster 7</th>
<th>Cluster 8</th>
<th>Cluster 9</th>
<th>Cluster 10</th>
<th>Cluster 11</th>
<th>Cluster 12</th>
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<tbody>
<tr>
<td>Moss sp.</td>
<td>2.09</td>
<td>4.72</td>
<td>0.00</td>
<td>2.63</td>
<td>0.00</td>
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<td>0.45</td>
<td>1.02</td>
<td>0.80</td>
<td>0.68</td>
<td>1.97</td>
<td>37.50</td>
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<td>Athyrium filix-femina</td>
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<td>0.14</td>
<td>0.36</td>
<td>0.00</td>
<td>0.63</td>
<td>0.88</td>
<td>28.64</td>
<td>0.58</td>
<td>7.40</td>
<td>1.70</td>
<td>0.76</td>
<td>1.25</td>
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<tr>
<td>Poa sp.</td>
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<td>10.63</td>
<td>1.79</td>
<td>1.84</td>
<td>0.94</td>
<td>1.49</td>
<td>0.76</td>
<td>0.59</td>
<td>6.20</td>
<td>1.36</td>
<td>1.44</td>
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<td>0.56</td>
<td>0.71</td>
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<td>1.88</td>
<td>1.55</td>
<td>1.21</td>
<td>0.45</td>
<td>0.60</td>
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<td>0.76</td>
<td>0.83</td>
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<td>0.28</td>
<td>1.07</td>
<td>0.26</td>
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<td>7.50</td>
<td>1.82</td>
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<td>0.60</td>
<td>1.02</td>
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<td>Solidago sp.</td>
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<td>0.00</td>
<td>0.76</td>
<td>0.00</td>
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<td>Gymnocarpium dryopteris</td>
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<td>0.00</td>
<td>16.84</td>
<td>0.00</td>
<td>0.20</td>
<td>0.00</td>
<td>0.45</td>
<td>0.20</td>
<td>0.23</td>
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<td>0.56</td>
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<tr>
<td>Bare Soil</td>
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<td>0.00</td>
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<td>0.00</td>
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</tbody>
</table>

<table>
<thead>
<tr>
<th>Name</th>
<th>ATFE low</th>
<th>POA high</th>
<th>SOLI high</th>
<th>GYDR high</th>
<th>HYVI high</th>
<th>HYVI low</th>
<th>ATFE high</th>
<th>Bare soil</th>
<th>ATFE-POA</th>
<th>LACA high</th>
<th>RUAL high</th>
<th>MOSS high</th>
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</table>
Table 7. Species sample cell proportions by position for the (a) unsalvaged and (b) salvaged areas.

<table>
<thead>
<tr>
<th>Position</th>
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<th>POA high</th>
<th>SOLID high</th>
<th>GYDR high</th>
<th>HYVI high</th>
<th>HYVI low</th>
<th>ATFE high</th>
<th>Bare soil</th>
<th>ATFE-POA</th>
<th>LACA high</th>
<th>RUAL high</th>
<th>MOSS high</th>
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<tbody>
<tr>
<td>a) Unsalvaged</td>
<td></td>
<td></td>
<td></td>
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<td></td>
<td></td>
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<td></td>
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<tr>
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<td>7.15</td>
<td>13.75</td>
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</tr>
<tr>
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<td>18.05</td>
<td>5.37</td>
<td>2.21</td>
<td>4.77</td>
<td>8.56</td>
<td>5.80</td>
<td>3.70</td>
<td>13.16</td>
<td>60.36</td>
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<tr>
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<td>11.08</td>
<td>13.50</td>
<td>4.80</td>
<td>9.02</td>
<td>5.37</td>
<td>11.07</td>
<td>7.15</td>
<td>12.17</td>
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<td>0.00</td>
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<td>4.77</td>
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<tr>
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<td>21.47</td>
<td>16.61</td>
<td>14.30</td>
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<td>5.80</td>
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<tr>
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<tr>
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<td>6.89</td>
<td>8.19</td>
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<td>3.64</td>
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<tr>
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<td>21.97</td>
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<td>18.59</td>
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Appendix A. Map of the distribution of the soils for the Flambeau delineated by USDA NRCS.
Appendix B. Table containing the life trait data gathered for all tip-up species.

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<th>Life term</th>
<th>Dispersal (main)</th>
<th>Growth habit</th>
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<td>FB</td>
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<td>d</td>
<td>GR</td>
<td>Y &amp; introduced</td>
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<td>d</td>
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<td>d</td>
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## Appendix B. Continued

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Appendix C. Figure comparing average species richness for the top, pit, and control tip-up positions between the unsalvaged and salvaged areas.
Appendix D. Examples of SAS code used when analyzing hypotheses one, two, and three.

Hypothesis One:

Proc glm data=hypothesis one data;
class Salvaged;
model Vegetation variable= Salvaged Time;
lsmeans Salvaged;
estimate 'Salvaged vs. Unsalvaged' Salvaged 1 -1;
title 'Variable Output';
run;

Hypothesis Two:

Model I:

Proc mixed method=type3;
class Soil_Type;
model Soil variable= Soil_Type;
title 'Variable output';
run;

Model II & III:

Proc mixed method=type3;
class Salvaged;
model Soil variable= Salvaged;
title 'Variable output';
run;

Model IV:

Proc mixed method=type3;
class ST Salvaged Position;
model Soil variable= ST Salvaged Position Salvaged*Position Position*ST ;
title 'Variable output';
run;

Hypothesis Three:

Proc mixed method=type3;
class Salvaged Position SoilType;
model Vegetation variable= Salvaged Position SoilType SoilType*Position Position*Salvaged;
title 'Vegetation output';
run;