Forest Ecology and Management 406 (2017) 19–27

Contents lists available at ScienceDirect

Forest Ecology and Management

journal homepage: www.elsevier.com/locate/foreco

Salvaging has minimal impacts on vegetation regeneration 10 years after severe windthrow

Anthony R. Taylor[a,b,*], David A. MacLean[b], Donnie McPhee[a], Evan Dracup[a], Kevin Keys[c]

[a] Natural Resources Canada, Canadian Forest Service – Atlantic Forestry Centre, 1350 Regent Street, PO Box 4000, Fredericton, New Brunswick E3B 5P7, Canada[1]
[b] Faculty of Forestry and Environmental Management, University of New Brunswick, 28 Dineen Drive, Fredericton, NB E3B 5A3, Canada
[c] Nova Scotia Department of Natural Resources, 15 Arlington Place, Suite 7, Truro, Nova Scotia B2N 0G9, Canada

ARTICLE INFO

Keywords:
Regeneration
Succession
Diversity
Salvage
Disturbance

ABSTRACT

Salvage harvesting is an important means of recovering wood fiber after disturbance, but remains controversial because it removes many unique biological legacies produced by natural disturbance. In this study, we assessed the effects of salvaging on the recovery of young forests approximately 10 years after severe windthrow in eastern Canada. Results showed that salvaging significantly reduced the abundance of residual overstorey trees from an average of 7.8 to 1.1 m^2/ha and downed woody debris from 197 to 46 m^3/ha, and altered forest soil attributes. However, we did not detect as clear an impact on regenerating vegetation. Although tree regeneration diversity was greater in salvaged stands (7.3 versus 5.6 species), the diversity and occurrence of all other nontree plant species did not significantly differ between treatments. Interestingly, mean tree seedling height was significantly higher in salvaged stands (1.5 versus 0.9 m), but saplings were taller in nonsalvaged stands (3.9 versus 3.2 m), largely due to the presence of advanced regeneration. Overall, salvaging had minimal effects on regenerating vegetation 10 years after windthrow and resulted in potential benefits, including increased mineralization of the forest floor, enhanced growth of seedlings, and improved access to conduct silviculture.

1. Introduction

Wind is an important driver of forest dynamics along Canada's eastern, coastal forests (Seymour et al., 2002; Neily et al., 2008; Bouchard et al., 2009). High annual precipitation limits the occurrence of wildfire, permitting the development of old forests in which gap-forming disturbances, such as wind, play a vital role (Loo and Ives, 2003; Bouchard et al., 2008). Wind also interacts with periodic severe spruce budworm (*Choristoneura fumiferana* Clem.) outbreaks; Taylor and MacLean (2009) showed that 11–15 years after defoliation ceased, postoutbreak stands were more vulnerable to wind-related mortality, which peaked at 11 m^3/ha/year. More severe, stand-replacing wind disturbances are also common, including strong gales and hurricanes (Johnson, 1986). However, although the influence of wind on Canada's coastal forests is well recognized, knowledge of how forests recover following wind disturbance is less known. This is of concern as climate change is expected to alter the frequency and severity of strong wind storms in this region and area of wind-disturbed forest (Overpeck et al., 1990; Knutson et al., 2010).

On 29 September 2003, Hurricane Juan made landfall over Nova Scotia as a Category 2 hurricane with sustained winds of 158 km/h, and gusts of up to 185 km/h (Fogarty, 2004). As Hurricane Juan moved northward, through central Nova Scotia, it damaged over 600,000 ha of forest (Fig. 1A). Although a large effort was made to salvage as much wood fiber as possible, many areas were left not salvaged due to lack of timely resources (e.g., limited harvesters and high road-building costs) and public controversy over whether areas should be left to regenerate naturally. Indeed, there remains much controversy among conservation biologists and foresters over whether to salvage or not, largely due to a lack of studies that directly compare recovery and regeneration of salvaged versus nonsalvaged forests (Lindenmayer and Noss, 2006).

Opposition to salvaging stems largely from the fact that salvage harvesting removes many of the unique biological legacies (e.g., organically derived structures and patterns) produced by natural disturbance, altering the structure of the forest (Lindenmayer and Noss, 2006). For instance, following wind disturbance, forest stands become "windthrown", whereby trees are overturned due to stem breakage and uprooting. Depending on the strength and direction of wind, forest composition, and site conditions, this can produce a unique entanglement of overturned and standing trees (Mitchell, 2013). The

* Corresponding author at: Natural Resources Canada, Canadian Forest Service – Atlantic Forestry Centre, 1350 Regent Street, PO Box 4000, Fredericton, New Brunswick E3B 5P7, Canada.
E-mail address: anthony.taylor@canada.ca (A.R. Taylor).
[1] Current affiliation.

http://dx.doi.org/10.1016/j.foreco.2017.09.061
Received 11 August 2017; Received in revised form 13 September 2017; Accepted 24 September 2017
Available online 07 October 2017

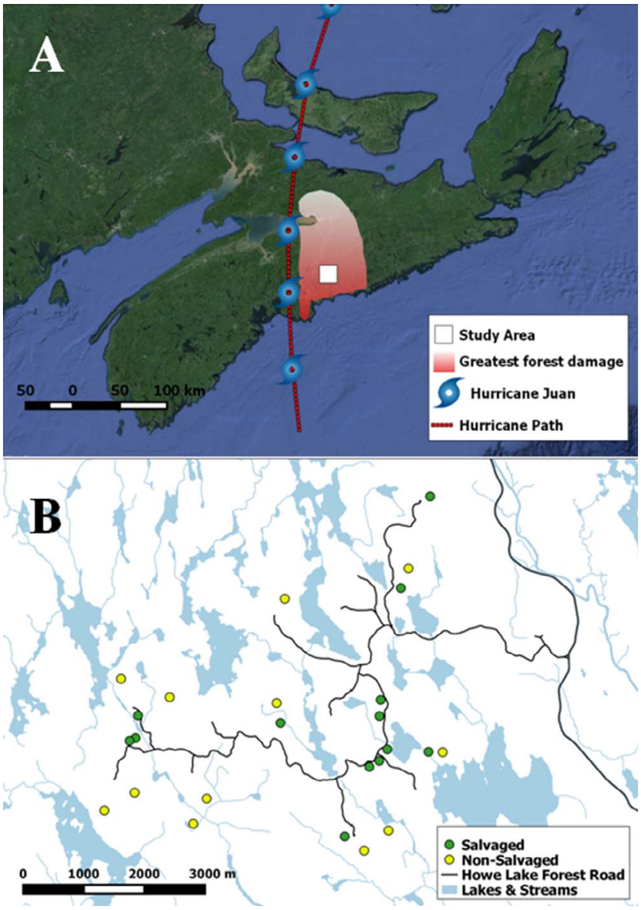

Fig. 1. Study area in central Nova Scotia, Canada. Maps showing (A) the path of Hurricane Juan and area of greatest forest damage relative to our study area; (B) the Howe Lake Road study area and sample plots of nonsalvaged and salvaged sites.

resulting physical environment is structurally complex, generally consisting of a more open forest canopy, an abundance of leaning and lying dead trees, substantial accumulation of forest floor woody debris, and the exposure of mineral soil caused by tree uprooting.

Downed trees provide habitat for a myriad of microbial, insect, plant, and animal communities (Bouget and Duelli, 2004; Jonsson et al., 2005; Brassard and Chen, 2006; Dittrich et al., 2014), including important regeneration substrate for late-succession tree species, e.g., *Picea glauca* and *Thuja occidentalis* (Simard et al., 1998). Unlike many other disturbance agents (e.g., wildfire and insect outbreaks), wind-uprooted trees expose and overturn volumes of mineral soil, creating pit–mound complexes that can persist for hundreds of years (Mitchell, 2013). This process can help counter soil podsolization (Kramer et al., 2004), which is common in eastern Canada, and introduce microtopographic and microclimatic heterogeneity within stands. Mounds are generally warmer and drier than pits and adjacent undisturbed soil, which promotes understorey biodiversity (Beatty, 1984; Peterson and Leach, 2008). Alternatively, salvage harvesting removes most deadwood and residual live trees, altering postdisturbance habitat structure and plant propagule availability, thus affecting forest regeneration and succession (Rumbaitis del Rio, 2006; Lain et al., 2008; Man et al., 2013; Schafer et al., 2014). During salvaging, wind-uprooted stumps are pushed over, reducing the abundance of pit-mound formations (Waldron et al., 2013). Furthermore, the activity of harvesting equipment during salvage operations (which is generally more intensive than standard harvesting practices) can cause soil compaction and mixing, altering the physical and chemical properties of forest soils (Rumbaitis del Rio, 2006; Lang et al., 2009; Hume et al., 2017).

Nonetheless, although the impacts of salvaging are apparent immediately following treatment, how long these impacts persist as forests recover is not clear (Lindenmayer and Noss, 2006; Mitchell, 2013). Whereas some studies have shown little to no detectable differences in community structure 25 years following salvaging (Lang et al., 2009), some differences in physical structure may persist for decades (e.g., abundance of standing dead trees). In this study, we aimed to directly compare the effects of salvaging versus not salvaging on the structure of young, recovering forests approximately 10 years after severe stand-replacing wind disturbance in the Acadian Forest Region of eastern Canada. Specifically, we measured the structure of the residual overstorey and downed woody debris complex, the physical and chemical properties of the forest soil, and the composition, growth, and diversity of regenerating vegetation. We hypothesized that plant community composition, growth, and diversity would significantly differ between salvaged versus nonsalvaged forest because salvaging removes most residual live trees and deadwood, impacts understorey vegetation, affects plant propagule availability, and potentially affects the physical and chemical properties of the soil.

2. Materials and methods

2.1. Study area

Our study area was located approximately 50 km east of Halifax, Nova Scotia, Canada, between 44°85′N to 44°80′N and 63°20′W to 63°30′W at 50–100 m elevation (Fig. 1). This area is part of the Acadian Forest Region (Rowe, 1972) and Eastern Granite Uplands Ecodistrict (Neily et al., 2005). It is characterized by cool summers and mild winters, with mean annual temperature and precipitation of 6.6 °C and 1396 mm, respectively, and an average annual frost-free period of 163 days (Environment Canada, 2016). The terrain of our study area is underlain with granite bedrock and was largely shaped by the retreat of the Laurentide Ice Sheet approximately 10,000 years ago. Steep cliffs, rocky ridges, and granite outcrops are common and are dissected by many long narrow lakes and streams. Soils on our sample stands were derived from coarse textured, stony glacial tills high in granite (Gibraltar series; MacDougall et al., 1963) and were predominately classed as Orthic Humo-Ferric or Orthic Ferro-Humic Podzols (Soil Classification Working Group, 1998).

In our study area, red spruce (*Picea rubens* Sarg.), black spruce (*Picea mariana* Mill.), balsam fir (*Abies balsamea* (L.) Mill.), red maple (*Acer rubrum* L.), eastern hemlock (*Tsuga canadensis* (L.) Carr.), and eastern white pine (*Pinus strobus* L.) form predominant tree associations, with red spruce dominating on the better drained soils, and black spruce dominating in poorer drained areas (Loucks, 1962; Neily et al., 2013). Sugar maple (*Acer saccharum* Marsh.), striped maple (*Acer pensylvanicum* L.), yellow birch (*Betula alleghaniensis* Britt.), white birch (*Betula papyrifera* Marsh.), white ash (*Fraxinus americana* L.), and beech (*Fagus grandifolia* Ehrh.) can also be found. Common shrub and herb species found across our study stands include black huckleberry (*Gaylussacia baccata*), red raspberry (*Rubus idaeus*), sheep laurel (*Kalmia angustafolia*), low-sweet blueberry (*Vaccinium angustifolium*), bunch berry (*Cornus canadensis*), mayflower (*Eigaea repens*), winter green (*Gaultheria hispida*), star flower (*Trientalis borealis*), wild lily-of-the-valley (*Maianthemum canadense*), and gold thread (*Coptis trifolia*).

Strong wind storms (e.g., tropical cyclones) and fire are the two dominant natural stand-replacing disturbances in our study area. The return interval of stand-replacing fire has been estimated to be > 1000 years since human suppression of wildfire began in the 20th century (Wein and Moore, 1979). The return interval of stand-replacing wind storms is less known, but a review of historical reports on the frequency and extent of strong wind storms over the past 300 years in Nova Scotia indicates it is likely no greater than 500 years—possibly as short as 200 years along Nova Scotia's eastern shore (Dwyer, 1979; Johnson, 1986; Seymour et al., 2002). Rather, gap-forming disturbances (≈10–1000 m^2) are the most common form of natural disturbance driving forest dynamics in our study area, primarily caused by wind, pathogens, and insect herbivory, with average return intervals of 50–200 years (Seymour et al., 2002; Neily et al., 2008).

Fig. 2. Photographs from our study sites showing (A) a salvaged forest approximately 10 years after disturbance, (B) a windthrown forest (non-salvaged), (C) the condition of a mature, intact softwood forest before Hurricane Juan, and D) a typical soil profile.

2.2. Sampling design

To examine the effects of salvaging on forest recovery following stand-replacing windthrow disturbance, we conducted a survey of salvaged and nonsalvaged windthrow sites during the summers of 2013 and 2014, approximately 10 years after Hurricane Juan. Our study sites were located on the Howe Lake Logging Road, operated by Taylor Lumber Company Limited (Fig. 1B), within 50 km east of the storm's eye (Fig. 1A), which was one of the forest areas most severely damaged by the storm. Although effort was made to salvage as much wood as possible, much damaged forest was not salvaged due to limited harvesting resources and was left to recover naturally. We used stratified random sampling to select wind-damaged forest sites that were either salvaged or not salvaged (Fig. 2A, B, respectively). Because Taylor Lumber Company Limited only constructed one main logging road in this area, most salvaged sites are closer to the main logging road than nonsalvaged forests. This was simply due to haul distance restrictions and not due to preferential site characteristics or wood quality. Indeed, we carried out extensive surveys using pre- and post-Hurricane Juan aerial photos and multiple on-site visits and evaluated all sites using Nova Scotia's forest ecosystem classification guide (Neily et al., 2013) to ensure selected sites were as similar as possible in terms of site quality, pre-disturbance tree species composition, age and level of windthrow damage.

All sites selected for our analysis sustained major wind damage, with at least 80% of all trees overturned. All salvaged sites that were selected were harvested by Taylor Lumber within 18 months following the storm, between the fall of 2003 and the spring of 2005. All harvesting took place between April and November each year and was completed by the same harvesting contractor using a single grip harvester, cable skidder, and forwarder. Because of the difficulty in harvesting downed trees, little attempt was made to protect advanced regeneration, and harvested areas were cleared of most merchantable wood.

To minimize the confounding effects of different site conditions, we only sampled sites on mesic, flat to mid-slope positions, with no slope exceeding 20%. Because Hurricane Juan originated from the south, with strong southeasterly winds, south-facing forest slopes were the most damaged. Consequently, all our sample sites were located on south-facing slopes on well-drained, coarse textured soils at least 30 cm thick, which is the prevailing soil type across the area. Although granite outcrops and small pocket wetlands with thick organic layers are scattered throughout the study area, these sites were avoided. Nonetheless, we had to remove two sampled plots from our analysis (sites S10a and NS5) because, upon closer inspection of their soil profiles, they did not meet our site type criteria.

The predisturbance forests in the study area were predominantly mature, dominated by red and black spruce, hemlock, white pine, and balsam fir (Figs. 2C and 3) (Taylor et al., 2007). Therefore, we only selected forest sites that were mature (> 80 years old) and conifer dominated (> 60% softwood species) before disturbance. For site selection, predisturbance composition was estimated by interpreting aerial photographs that were taken several months before Hurricane Juan, and local forest inventory records. Following plot sampling, we cross-referenced predisturbance composition estimated from the aerial photos with ground measurements of residual standing trees, harvested stumps, and downed trees. Only one measured plot (site S7) did not meet our softwood-dominated criteria and was removed from our analysis. A comparison of the predisturbance conditions of the salvaged and nonsalvaged sample plots (Fig. 3 and Table 1) demonstrated the similarity of site conditions before Hurricane Juan. Additionally, to minimize the impact of spatial autocorrelation between sample plots, we avoided sampling in close proximity to one another. This was achieved by selecting replicate plots from different patches of disturbed forest area resulting in a minimum distance between plots of 250 m. Overall, we selected and analyzed 11 salvaged and 11 nonsalvaged plots (Table 1). All patches were at least 1 ha in size and visually homogeneous in structure and tree species composition based on aerial photos and field reconnaissance.

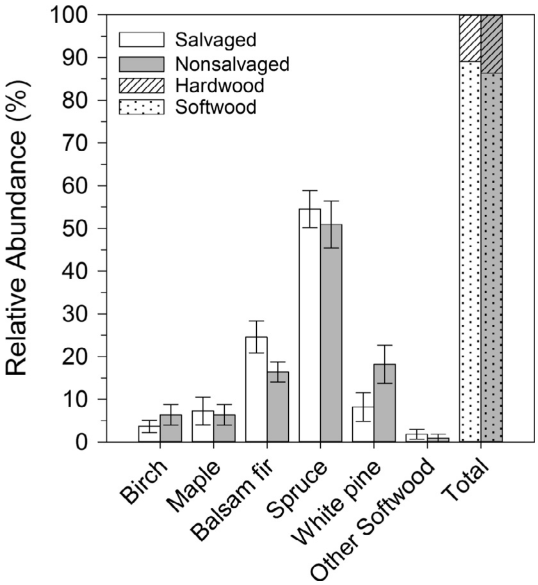

Fig. 3. Mean relative abundance (% basal area) of overstorey tree species composition before Hurricane Juan, along with total softwood and hardwood content for nonsalvaged and salvaged plots. Because of difficulties in determining closely related species from data used to reconstruct predisturbance composition, some species have been grouped. Birch includes white and yellow birch, Maple includes red and sugar maple, and Spruce includes red, black and white spruce.

2.3. Plot measurements

2.3.1. Plot establishment

At each salvaged and nonsalvaged site we located and established one 400 m^2 sample plot at least 50 m away from treatment boundaries and roads to avoid edge effects. The center of each plot was marked with an aluminum post, and its position recorded with a Garmin® GPS.

2.3.2. Residual overstorey trees, snags, and stumps

Within each plot, all standing trees with a diameter at breast height (DBH; 1.3 m above the root collar) ≥10 cm were identified by species, measured for DBH and height, recorded by status (i.e., live, dead, leaning, broken), and tagged with a tree number. This included all snags and broken-off stems > 1.3 m in height with an elevational angle ≥45° (as determined using a Suunto® Clinometer). Heights were measured using a TruPulse™ 200 laser range finder.

All stumps that originated from either harvesting or stem breakage ≥10 cm in diameter were measured for root collar diameter and identified by species. In the nonsalvaged plots, all blown-over tree stems (elevational angle < 45°) with DBH ≥10 cm were identified by species, measured for root collar diameter, DBH, height (i.e., length from root collar to tree tip), recorded by status (i.e., live or dead), and tagged with a tree number. Any dead stems or stumps that could not be identified by species were identified as either softwood or hardwood or unknown. Diameter at breast height of all stumps was estimated by using locally developed, species-specific stem diameter ratios calculated from root collar and DBH measurements taken from 168 downed trees

Table 1
Comparison of vegetation attributes of nonsalvaged (NS) and salvaged (S) plots before and after Hurricane Juan.

Plot	Before Hurricane			After Hurricane										
	% SW	% CC	Ht (m)	RSBA (m^2/ha)	DWD		Regen density (stems/ha)		Regen height (m)		Richness		Evenness	
					Vol (m^3/ha)	% Cover	Seedling	Sapling	Seedling	Sapling	Regen	Veg	Regen	Veg
NS1	80	75	15.6	6.3	167	13	15,300	2000	1.0	3.3	6	21	0.5	0.3
NS2	80	80	14.6	7.4	148	8	22,700	1100	0.6	4.5	7	13	0.3	0.4
NS3	80	75	16.2	8.5	312	15	34,800	4800	1.0	4.7	6	14	0.6	0.3
NS4	80	70	16.2	9.7	165	14	19,000	1800	1.0	4.1	4	16	0.6	0.3
NS6	90	70	18.3	8.4	286	15	17,400	2000	0.7	4.3	5	17	0.8	0.4
NS7	100	75	16.3	8.3	143	13	49,600	2300	1.3	3.2	7	18	0.4	0.3
NS8	100	65	14.3	5.4	215	24	10,400	1100	0.9	4.1	5	14	0.4	0.4
NS9	80	65	16.2	9.7	154	18	12,600	1300	0.8	4.0	5	23	0.6	0.3
NS10	80	60	15.8	3.9	88	9	12,000	2000	0.7	2.9	5	22	0.3	0.2
NS11	80	65	17.8	12.3	295	16	25,400	1200	1.0	3.8	7	24	0.3	0.4
NS12	100	60	15.7	5.6	189	8	7800	1000	0.9	3.8	5	22	0.3	0.3
Mean	*86.4*	*69.1*	*16.1*	*7.8*	*197*	*14*	*20636.4*	*1872.7*	*0.9*	*3.9*	*5.6*	*19*	*0.5*	*0.3*
SE	*2.8*	*2.0*	*0.4*	*0.7*	*22*	*1*	*3713.4*	*323.9*	*0.1*	*0.2*	*0.3*	*1.2*	*0.1*	*< 0.0*
S1	90	65	16.0	2.1	46	5	65,000	1100	1.2	2.1	8	18	0.4	0.3
S2	100	60	15.0	1.0	6	3	10,200	1300	1.5	4.1	5	25	0.4	0.4
S3	90	90	14.5	0.0	31	2	34,000	2700	1.5	3.1	7	24	0.3	0.3
S4	80	75	16.0	0.0	49	10	10,100	5400	1.7	3.6	8	21	0.5	0.2
S5	100	85	15.0	0.6	23	4	18,400	10,600	2.0	3.1	7	23	0.5	0.3
S6	90	80	15.6	0.6	43	1	31,400	7400	1.7	3.4	10	13	0.5	0.5
S8	70	75	15.8	0.0	39	5	50,100	5300	1.4	3.3	9	21	0.2	0.1
S9	80	70	16.8	5.0	74	2	16,700	2700	1.2	3.4	7	21	0.6	0.3
S10	90	80	15.3	0.0	32	4	13,300	1300	1.4	3.4	7	20	0.5	0.2
S11	100	75	15.8	3.1	102	7	16,700	1500	1.3	2.6	6	14	0.5	0.3
S12	90	60	14.6	0.0	56	6	14,500	1200	1.4	3.6	6	18	0.3	0.4
Mean	*89.1*	*74.1*	*15.5*	*1.1*	*46*	*4*	*25490.9*	*3681.8*	*1.5*	*3.2*	*7.3*	*20*	*0.4*	*0.3*
SE	*2.8*	*2.9*	*0.2*	*0.5*	*8*	*1*	*5409.5*	*944.4*	*0.1*	*0.2*	*0.4*	*1.2*	*< 0.0*	*< 0.0*
p-value	*0.50*	*0.18*	*0.16*	***< 0.01***	***< 0.01***	***< 0.01***	*0.47*	***0.09***	***< 0.01***	***0.01***	***0.01***	*0.45*	*0.57*	*0.48*

Abbreviations : SW, softwood; CC, crown closure; Ht, height; RSBA, residual standing basal area of live and dead trees; DWD, downed woody debris; Vol, volume; Regen, tree regeneration (seedlings and saplings); Veg, all nontree vegetation (bryophytes, ferns, herbs and shrubs). Statistics are italicized, and statistically significant ($p < 0.05$) and marginal ($p < 0.1$) effects are shown in bold font.

(a minimum of five downed trees per species was used to calculate ratios).

Measured standing live and dead trees were used to estimate post-disturbance residual overstorey basal area (m^2/ha) for each sample plot. These data were then combined with the measurements of stumps and downed trees to reconstruct predisturbance overstorey basal area and species relative abundance (i.e., % basal area) in all plots. These estimates were then cross-referenced with our previous photo-interpretation estimates of overstorey composition to ensure consistency.

2.3.3. Tree regeneration

Within each 400 m^2 main plot, four square 25 m^2 regeneration subplots were systematically located in the center of each of the NE, SE, SW, NW quadrants of the main plot, in which all live seedlings and samplings were identified by species and counted. All stems < 2 cm DBH were counted as seedlings and recorded into one of five height classes, i.e., 0–50 cm, 51–100 cm, 1–2 m, 2–3 m, and 3–4 m. All stems ≥ 2 cm DBH were counted as saplings and measured for DBH and height and marked and numbered with a wire tag. In cases of vegetative clumps of live stems originating from a common root system, stump, or log, only live stems that originated from a main stem with their point of origin below DBH were recorded.

2.3.4. Shrubs and herbaceous plants

At the NE and SW corner of each regeneration subplot, we established 1 m^2 square shrub and herb subplots, for a total of eight 1 m^2 subplots for each main plot. All shrub and herbaceous plants within subplots were identified by species (except for some bryophytes, which were identified to the genus level) for presence and had their percent coverage estimated within the subplot using the Braun-Blanquet scale (nine cover classes; Rodwell, 2006) for visual estimates of cover. Furthermore, we also recorded the percent coverage of any obstructions to vegetation growth in each 1 m^2 subplot, including the presence of boulders and downed tree stems.

2.3.5. Downed woody debris

Downed woody debris (DWD) was defined as all dead wood lying or standing (with a elevational angle < 45°) with a mid-length diameter ≥ 2.0 cm and was measured using the line intercept method (Husch et al., 2003). Two 30-m transects were established in each sample plot, oriented N to S and E to W, crossing, and with their midpoints located at plot center (i.e., spoked design). The diameter of each piece of DWD (≥ 2.0 cm), where it intercepted the transect line, was recorded along the length of each transect. Downed woody debris volume on an area basis was calculated according to Husch et al. (2003):

$$DWD = \frac{\pi^2 \sum d^2}{8L} \tag{1}$$

where d is the piece intercept diameter, and L is the length of the transect. Although the potential for a downed log to be counted more than once is possible when using a "spoked" transect design (particularly where the two transects cross at plot center, i.e., Van Deusen and Gove (2011)), we analyzed each 30-m transect independently as well combined and found the same overall result and thus used the combined transect results.

2.3.6. Forest floor and mineral soils

Soil pits were excavated at a representative location within each main plot based on assessment of topography, vegetation, and surface stoniness (Fig. 2D). Due to high stoniness and difficulty in digging, only one pit was assessed per plot. When possible, pits were excavated to the bottom of the rooting zone (maximum 60 cm depth). Mineral soil profiles and textures were described and field classified using the Canadian System of Soil Classification (Soil Classification Working Group, 1998), whereas humus forms were classified using the system described by Green et al. (1993). Some field calls for Bf and Bhf horizons were later adjusted after lab determination of organic C.

Forest floor bulk density (combined F and H horizons) was sampled at each soil pit location using a 20 cm × 20 cm square frame with the average of four depth measurements taken to estimate sample volume. Live moss, lichens, and fresh litter were removed before measuring the depth of the forest floor layer. Mineral soil bulk density volumes were estimated after excavation using glass beads and a volumetric cylinder. When thick enough, mineral samples were collected by horizon, but horizons were sometimes combined to get sufficient volume (target 1000 cm^3). In all cases, the horizon(s) sampled were noted for later interpretation. Separate soil samples from each horizon (except F and H horizons, which were combined) were also collected for chemical analysis. All chemical samples were placed in a cooler while in the field and later refrigerated until processed in the laboratory.

Bulk density was determined by drying samples at 105 °C until a constant weight was achieved (typically 48 h) and correcting for coarse fragment (> 2 mm) volume. Soil chemical samples were air dried, sieved (2 mm mesh), and ground in a Wiley mill prior to analysis. Chemical analyses, including pH; total carbon (C) and organic matter content; total nitrogen (N); available phosphorous (P); exchangeable potassium (K), calcium (Ca), and magnesium (Mg); and exchangeable acidity were conducted by the Laboratory for Forest Soils and Environmental Quality, University of New Brunswick. pH was measured using 0.01 $CaCl_2$ solution (ratio 1:1) and a bench top meter. Total C and N were measured using a LECO induction furnace, with organic matter estimated from total C by multiplying by 1.72. Exchangeable K, Ca, and Mg were determined through 1 M NH_4OAc extraction (adjusted to pH 7) followed by AAS analysis. Available P was determined through colorimetric analysis (Technicon AutoAnalyzer) after extraction with 0.5 M $NaHCO_3$ solution. Exchangeable acidity was determined by titration after extraction with 1 M KCl.

2.4. Data analysis

All vegetation and soils data collected were aggregated and summarized for each of the salvaged and nonsalvaged plots at the per ha level. Predisturbance overstorey species composition, crown closure and height were tested for similarity using two-sample t-tests.

To test the effects of salvaging versus not salvaging on post-disturbance forest attributes we compared: (1) overall and species-specific residual standing basal area of live and dead trees; (2) volume and percent cover of DWD; (3) overall and species-specific tree regeneration density, height, and diversity; (4) diversity of nontree vegetation; and (5) forest floor and mineral soil physical and chemical properties. The mean and standard error of each variable were calculated for the salvaged and nonsalvaged plots and tested for statistically significant differences using two-sample t-tests. We also tested for differences in the presence–absence of all nontree plant species measured on the salvaged and nonsalvaged plots using Fisher's Exact Test for count data. All data analyses were performed using the R Statistical Environment, version 3.2.3.

For plant diversity measures, species richness (i.e., the number of plant species in a community) was calculated from the number of bryophyte, fern, herb, shrub, and tree species counted in each main plot; and species evenness (i.e., the closeness in the relative abundance of different species in a community) was calculated using Simpson's dominance index (the inverse of Simpson's evenness index). Simpson's index ranges from 0 (infinite diversity) to 1 (monoculture) (Smith and Smith, 2012).

For soils, only mean data from the same humus forms (hemimor and humimor) and mineral horizons (Ae, Bf, Bhf) were compared. Where found, data from partially cemented horizons (Bfcj and Bhfcj) were included for chemistry comparisons, but not for bulk density. Where there were two Bf or Bhf horizons found in the same profile, only data from the top-most horizon were included in the statistical analyses.

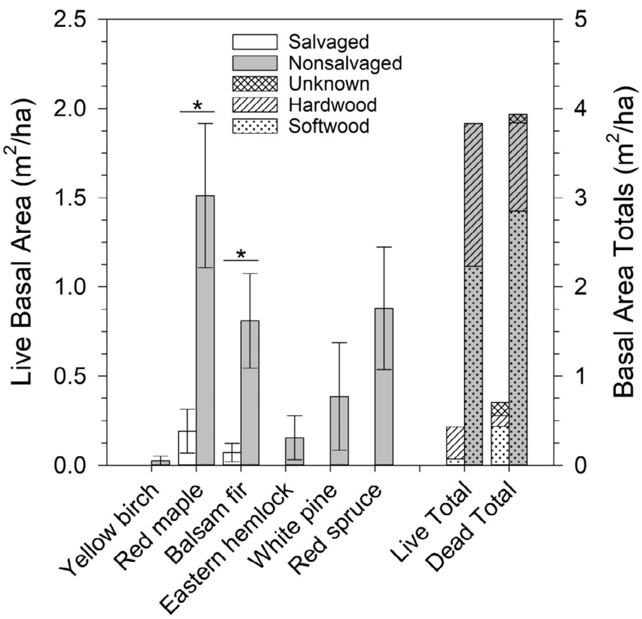

Fig. 4. Mean basal area (m^2/ha, ± 1 standard error) of residual live overstorey trees (≥10 cm DBH) by species, for nonsalvaged and salvaged plots, 10 years after Hurricane Juan. Total live and dead standing basal area are also shown and divided into softwood, hardwood, and unknown species categories. Significant differences ($p < 0.05$) between nonsalvaged and salvaged plots are indicated by *.

3. Results

The mean basal area of residual standing live and dead trees (Table 1) was significantly ($p < 0.01$) higher in the nonsalvaged plots at 7.8 m^2/ha compared with only 1.1 m^2/ha in the salvaged plots. Standing residual stems were composed evenly of live and dead trees with approximately 3.8 m^2/ha of the residual overstorey remaining alive in the nonsalvaged plots, whereas only 0.4 m^2/ha remained alive in the salvaged plots (Fig. 4). Similarly, the mean volume and percent ground coverage of DWD was also significantly ($p < 0.01$) higher in the nonsalvaged plots at 197 m^3/ha and 14%, respectively, versus the salvaged plots, which had 46 m^3/ha and 4% coverage (Table 1).

Most treatment differences related to soil were confined to the forest floor, where salvaged plots had significantly or marginally higher bulk density ($p < 0.01$), pH ($p = 0.07$), and exchangeable Ca ($p = 0.07$) compared with nonsalvaged plots, along with significantly or marginally lower exchangeable Mg ($p = 0.04$), thickness ($p = 0.14$), and exchangeable acidity ($p = 0.11$) (Tables 2 and 3). For mineral soils, the salvaged plots also had marginally higher exchangeable Ca ($p = 0.05$) and Mg ($p = 0.06$) in Ae horizons, as well as a significantly higher Bf horizon bulk density compared with the nonsalvaged plots ($p = 0.04$). There were no significant differences or trends between treatment plots with respect to organic matter content, total N, or available P (Table 3).

The mean density of seedlings was higher in salvaged than nonsalvaged plots at 25,490 stems/ha versus 20,640 stems/ha, but this difference was not significant ($p = 0.47$, Table 1). Sapling density was also marginally ($p = 0.09$) higher in the salvaged than in nonsalvaged plots at 3680 versus 1870 stems/ha. Although no significant differences in seedling and sapling density were detected between individual tree species (Fig. 5), red spruce comprised the largest proportion of seedlings for both nonsalvaged and salvaged plots (Fig. 5a), and red maple comprised the largest proportion of saplings for both nonsalvaged and salvaged plots (Fig. 5b).

Mean seedling height was, overall, 0.6 m taller in the salvaged plots ($p < 0.01$, Table 1) with all species being, on average, taller in the salvaged plots; red spruce and red maple were significantly ($p < 0.05$) taller in the salvaged plots (Fig. 6a). In contrast, saplings were 0.7 m taller in nonsalvaged plots ($p = 0.01$; Table 1), with balsam fir being the tallest overall sapling species, but especially in the nonsalvaged plots ($p < 0.05$; Fig. 6b).

Species richness of tree regeneration was significantly higher in the salvaged plots, which averaged 1.7 more tree species present than in nonsalvaged plots (Table 1). Evenness of tree regeneration diversity did not differ significantly between salvaged and nonsalvaged plots, with both treatments showing moderate values (0.4 versus 0.5), suggesting that a few species made up most tree regeneration. Prominence of red maple and red spruce likely contributed to this small imbalance in evenness (Fig. 5).

No significant differences were detected in mean species richness or evenness between salvaged and nonsalvaged plots for all nontree vegetation, including bryophytes, ferns, herbs, and shrubs (Table 1). Overall, 72 different nontree plant species occurred across all salvaged and nonsalvaged plots. Fisher's Exact Test for count data found that salvaging had no significant effect on the occurrence (or presence) of nontree vegetation.

Table 2
Comparison of forest floor and mineral soil horizon bulk density (Db) and thickness in nonsalvaged (NS) and salvaged (S) plots. Standard errors of means are shown in parentheses, *p* values are in italics, and statistically significant ($p < 0.05$) and marginal ($p < 0.1$) effects are shown in bold font.

Treatment	Horizon	n	Db (Mg/m^3)	Thickness (cm)
S	FF	10	0.10 (0.01)	13.2 (1.6)
NS	FF	10	0.07 (0.00)	17.2 (2.1)
p-value			***< 0.01***	*0.14*
S	Ae	3	1.05 (0.08)	–
NS	Ae	3	1.09 (0.07)	–
p-value			*0.703*	–
S	Bf	6	0.95 (0.11)	–
NS	Bf	7	0.63 (0.04)	–
p-value			***0.04***	–

Note: n = number of plots used in analysis. FF = forest floor. No mean thickness values are given for mineral soils as these would not be impacted by treatment. No separate Db values for Bhf horizons are available as these horizons were usually too thin for individual sampling.

4. Discussion

4.1. The effect of salvaging on the residual overstorey, deadwood, and soils

As expected, the abundance of residual overstorey trees and DWD differed significantly between salvaged and nonsalvaged plots, primarily because salvage harvesting removed most large live and dead trees. However, less expected was the impact that salvaging had on forest soil attributes 10 years after treatment.

Forest floor horizons are naturally thick and acidic in the softwood-dominated vegetation type assessed in our study area (Neily et al., 2013). Salvaging-related increases in light and moisture availability likely promoted increased forest floor mineralization, which, in turn, could have led to the small increases in pH and total exchangeable bases (and slight drop in exchangeable acidity) found in salvaged versus nonsalvaged plots (Hume et al., 2017). Forest floor horizons, however, were still very acidic in all plots regardless of treatment (mean pH 3.45–3.67). Increased mineralization could also be expected to reduce thickness and increase forest floor bulk density because H horizons are typically denser than F horizons. Other factors that may have contributed to reduced thickness and increased density in salvaged plots include snow press and water movement, along with machine traffic as noted by Lang et al. (2009). However, contrary to similar studies that have directly compared salvaged and nonsalvaged windthrow sites (Rumbaitis del Rio, 2006; Lang et al., 2009), we did not detect any significant effect of salvaging on forest floor or mineral soil N concentrations.

Table 3
Comparison of mean forest floor and mineral soil horizon chemical properties in nonsalvaged (NS) and salvaged (S) plots. Standard errors of means are shown in parentheses, *p* values are in italics, and statistically significant ($p < 0.05$) and marginal ($p < 0.1$) effects are shown in bold font.

Treatment	Horizon	n	OM (%)	pH	TotN (%)	Avail P (ppm)	xK (cmol/kg)	xCa (cmol/kg)	xMg (cmol/kg)	xAC (cmol/kg)
S	FF	11	73.0 (2.8)	3.67 (0.10)	1.29 (0.06)	95.8 (12.0)	1.33 (0.11)	8.86 (1.40)	4.36 (0.56)	6.30 (0.62)
NS	FF	9	76.3 (0.7)	3.45 (0.05)	1.15 (0.06)	76.8 (14.0)	1.42 (0.13)	5.24 (1.20)	5.91 (0.40)	8.02 (0.80)
p-value			*0.27*	***0.07***	*0.142*	*0.315*	*0.614*	***0.07***	***0.04***	*0.11*
S	Ae	10	1.04 (0.13)	4.00 (0.11)	0.09 (0.01)	3.44 (0.64)	0.03 (0.01)	0.31 (0.08)	0.09 (0.02)	2.23 (0.31)
NS	Ae	11	0.94 (0.09)	3.95 (0.07)	0.08 (0.01)	2.58 (0.53)	0.03 (0.00)	0.12 (0.02)	0.05 (0.01)	2.25 (0.27)
p-value			*0.54*	*0.69*	*0.32*	*0.32*	*0.72*	***0.05***	***0.06***	*0.55*
S	Bhf	5	12.71 (1.10)	4.55 (0.17)	0.34 (0.03)	16.30 (11.00)	0.07 (0.01)	0.15 (0.08)	0.11 (0.03)	5.08 (1.10)
NS	Bhf	6	12.20 (0.87)	4.57 (0.11)	0.34 (0.02)	16.50 (11.00)	0.09 (0.01)	0.03 (0.01)	0.07 (0.01)	4.21 (1.40)
p-value			*0.73*	*0.91*	*0.92*	*0.99*	*0.17*	*0.22*	*0.44*	*0.65*
S	Bf	8	3.93 (0.67)	4.87 (0.07)	0.16 (0.02)	2.53 (1.10)	0.04 (0.01)	0.35 (0.11)	0.04 (0.01)	2.02 (0.46)
NS	Bf	7	5.53 (0.86)	4.66 (0.12)	0.21 (0.02)	18.50 (9.50)	0.05 (0.01)	0.15 (0.10)	0.05 (0.01)	3.51 (1.10)
p-value			*0.17*	*0.17*	*0.12*	*0.15*	*0.21*	*0.19*	*0.50*	*0.25*

Abbreviations: n, number of plots used in analysis; FF, forest floor; OM, organic matter; TotN, total nitrogen; Avail P, available phosphorus; xK, exchangeable potassium; xCa, exchangeable calcium; xMg, exchangeable magnesium; xAC, exchangeable acidity.

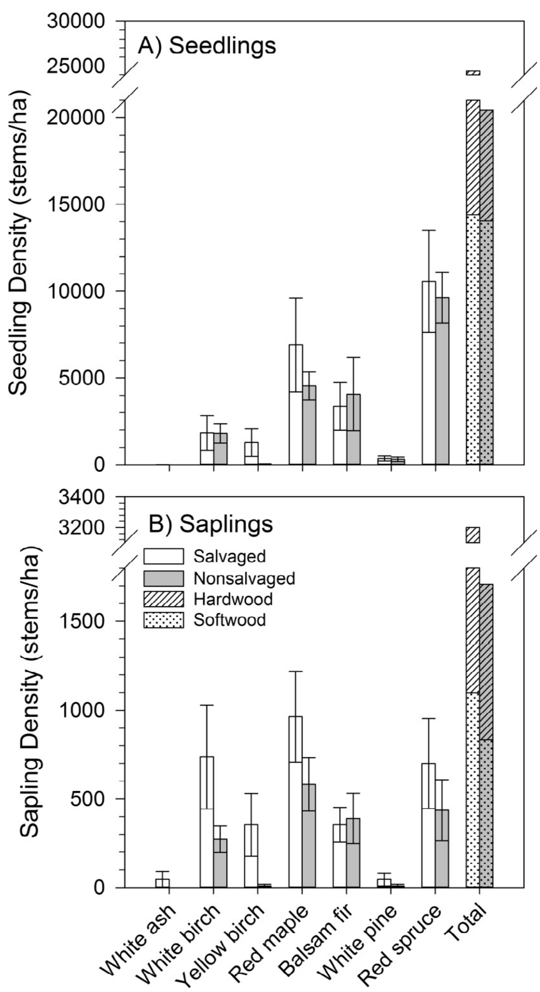

Fig. 5. Mean density (stems/ha, ± 1 standard error) of (A) seedlings (< 2 cm DBH), and B) saplings (≥ 2 cm and < 10 cm DBH) by species, for nonsalvaged and salvaged plots, 10 years after Hurricane Juan.

Mean Ca and Mg concentrations were greater in the salvaged plot Ae horizons, however, concentrations were more variable compared with nonsalvaged plots, and may have been due to variation in posttreatment decay of the coarse roots, and/or by leaching inputs from increased mineralization (Hume et al., 2017). Lack of detected differences in deeper mineral soil horizons agrees with previous reports that the effect of harvesting on soil properties diminishes with soil depth (Johnson and Curtis, 2001; Chen and Shrestha, 2012).

Thick forest floors and high sand and coarse fragment content in the Ae horizons probably mitigated possible impacts of machine traffic on surface mineral soil compaction—as noted by almost identical bulk density values between treatments (1.05 Mg/m^3 salvaged versus 1.09 Mg/m^3 nonsalvaged). This supports the recognized resilience of this soil type to compaction damage (Neily et al., 2013) and suggests that the higher mean bulk density found in salvaged plot Bf horizons (0.95 Mg/m^3 salvaged versus 0.63 Mg/m^3 nonsalvaged) was likely due to differences in organic matter content (3.93% salvaged versus 5.53% nonsalvaged) and posttreatment settling rather than machine traffic.

4.2. The effect of salvaging on vegetation regeneration

Salvaging had lasting impacts on the overstorey and deadwood, but we did not detect a clear impact on regenerating vegetation. This was in conflict with our original hypothesis that plant community structure would differ significantly between salvaged and nonsalvaged stands, but is consistent with previous reports (e.g., Peterson and Leach, 2008).

Contrary to studies that reported higher seedling density and abundance of shade-tolerant seedlings in nonsalvaged, windthrown forests (Lain et al., 2008; Man et al., 2013; Waldron et al., 2014), overall and species-specific seedling density did not differ significantly between our salvaged and nonsalvaged plots, indicating salvage had little impact on seedling abundance and composition, as similarly reported by Peterson and Leach (2008). However, further analysis using multivariate techniques (e.g., nonmetric multidimensional scaling, Schafer et al., 2014) may be warranted to more closely examine differences in community composition. Nevertheless, seedlings were found to be significantly taller in salvaged than in nonsalvaged plots and mean height of all individual tree species was taller, particularly for red spruce, a locally important softwood lumber species, and red maple. This suggests salvaging may have enhanced recruitment and early growth of both deciduous and coniferous seedlings by removing residual overstorey and DWD cover, permitting more growing space and improving germination substrate through soil perturbation.

Sapling development in nonsalvaged and salvaged plots directly supported the idea that windthrow "accelerates succession" by releasing shade-tolerant advanced regeneration, versus salvaging, which "delays succession" by removing advanced regeneration and promoting the recruitment of early succession species (Spurr, 1956; Rich et al., 2007). In salvaged plots, overall density of saplings were marginally higher, but that of broadleaf white birch and red maple was significantly higher, with most broadleaf saplings originating from harvested stump sprouts, similar to previous reports (Lain

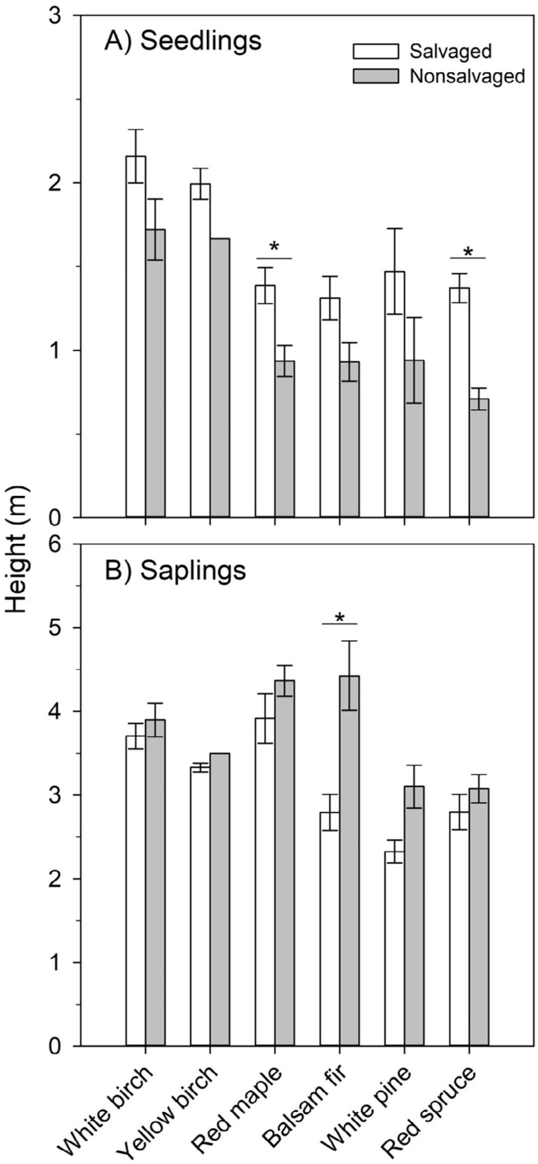

Fig. 6. Mean height (m, ± 1 standard error) of (A) seedlings (< 2 cm DBH, measured in five height classes up to 4 m), and (B) saplings (≥ 2 cm and < 10 cm DBH) by species, for nonsalvaged and salvaged plots, 10 years after Hurricane Juan. Significant differences ($p < 0.05$) between nonsalvaged and salvage plots are indicated by *.

et al., 2008; Lang et al., 2009; Man et al., 2013; Schafer et al., 2014). Nonsalvaged plots had a slightly higher density of balsam fir saplings that were significantly taller than all other saplings (mean height > 4 m). Balsam fir, a very shade-tolerant species, is well known to form dense understoreys in mature softwood forests in our study region (e.g., Spence and MacLean, 2012) and succeeds following windthrow or spruce budworm outbreaks (Baskerville, 1975; Spence and MacLean, 2012). However, as seedlings were 8–10 times more dense than saplings, it seems likely that within 10–20 years, species composition would likely converge between salvaged and nonsalvaged forests, as observed by Lang et al. (2009).

Although salvage harvesting has been shown to negatively impact plant species diversity (Rumbaitis del Rio, 2006; Waldron et al., 2014), we found that salvaged plots contained higher tree regeneration species richness. Overall, 72 nontree plant species were recorded, with no significant differences in species richness or evenness between salvaged and nonsalvaged plots, as similarly reported by Schafer et al., (2014). There were also no significant differences in the presence–absence or percent cover of nontree plant species between the salvaged and nonsalvaged plots, in contrast to a reduction in bryophyte cover following salvage harvesting found by Rumbaitis del Rio (2006) and Man et al. (2013). Following 10 years of recovery, the salvaged and nonsalvaged plots may have begun to converge (Lain et al., 2008; Peterson and Leach, 2008). Furthermore, the small size of disturbed areas in our study, generally < 3 ha, and close proximity to intact, mature forest may have permitted seed dispersal and rapid recovery of plant species, but without early assessments immediately following disturbance, it is difficult to confirm early community developmental patterns. Although we did not detect any significant difference in the presence–abundance of nontree plant species, some species did exhibit trends: red raspberry, velvet-leaved blueberry (*Vaccinium myrtilloides*), bunch berry, sheep laurel, and mountain holly (*Ilex mucronata*) all had mean percent cover values an order of magnitude higher in salvaged plots, consistent with their ruderal nature and preference for more open and/or disturbed areas (Dickinson et al., 2004; Boland, 2012; Neily et al., 2013). In contrast, Schreber's moss (*Pleurozium schreberi*), hay-scented fern (*Dennstaedtia punctilobula*), star flower, and creeping snow berry (*Gaultheria hispida*) were an order of magnitude more abundant in nonsalvaged plots, perhaps indicating higher sensitivity to more open and disturbed conditions created by salvaging (Dickinson et al., 2004; Boland, 2012; Neily et al., 2013).

4.3. Management implications

Windthrow plays a major role in the natural disturbance regime of eastern, coastal forests in Canada, and salvage harvesting is an important means of recovering wood fiber losses after disturbances. Our results suggest that salvage harvesting had minimal effects on regenerating vegetation 10 years after windthrow and salvage. Although salvaged plots had substantially less residual overstorey large trees and DWD (but not less than a typical clearcut harvest in the study area), which could impact habitat and diversity of animal communities (Brassard and Chen, 2006; Lain et al., 2008), this had little impact on the structure of regenerating vegetation. Salvage operations had minimal impacts on soil properties and may have even resulted in some short-term benefits through increased mineralization of thick forest floor horizons. From a nutrient sustainability perspective, salvage harvesting on the site type investigated can be considered similar to conventional clearcut harvesting in that the main concern is total nutrient removal over time (Keys et al., 2016), and harvest removals should not exceed cumulative inputs from soil weathering and atmospheric deposition (especially for Ca and N). Therefore, it may be important to retain some residual coarse woody debris after salvage harvesting to avoid potential future nutrient deficits.

From an operational point of view, salvage harvesting permits forest managers to have continued access to disturbed forest to conduct silviculture. If windthrow areas are left unsalvaged after massive blowdown, such as Hurricane Juan, they are impassable due to the remaining entanglement of fallen trees and turned-up stumps, inhibiting forest surveys and planting or tending treatments. Salvage harvesting enhanced recruitment and growth of seedlings, including abundance and growth of red spruce, while providing a useful means to recover valuable wood fiber. However, further study is warranted to examine how physical changes to forest structure caused by salvaging may influence other forest attributes, including habitat and carbon stocks.

Acknowledgments

We thank Jenna Doaks, Allison Dykstra, Andrew MacLean, Breton McPhee, Edward Wilson, and Dave Arseneau for assistance with fieldwork. We thank Taylor Lumber Co. Ltd. and the Nova Scotia Department of Natural Resources for road access and information. We also thank Caroline Simpson and the anonymous reviewers, who provided useful comments on an earlier version of this manuscript. This study was funded by Natural Resources Canada and by a NSERC grant to DAM.

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