Biodivers Conserv (2007) 16:2933–2948
DOI 10.1007/s10531-007-9151-0

ORIGINAL PAPER

Do tree-level monocultures develop following Canadian boreal silviculture? Tree-level diversity tested using a new method

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Received: 22 April 2006 / Accepted: 24 January 2007 / Published online: 7 March 2007

Abstract Concern about forestry practices creating tree-level monoculture plantations exists. Our study investigates tree diversity responses for six early seral boreal forest plantations in Ontario, Canada, representing three conifer species; black spruce (*Picea mariana*), white spruce (*P. glauca*), and jack pine (*Pinus banksiana*), 14 release treatments, and 94 experimental units. Dominance-diversity curves and Simpson's indices of diversity and evenness indicate tree alpha diversity. We propose a new method for assessing diversity, using percentage of theoretical species maximum (%TSM) which is determined by comparing post-disturbance richness (S) with a theoretical species maximum (TSM). Our results support the hypothesis that alternative vegetation release treatments generally do not reduce tree species diversity levels (%TSM) relative to untreated plots. The only %TSM ($P \leq 0.05$) comparison that produced less diversity than in control plots was repeated annual treatments of Vision herbicide at one of the black spruce study sites. Our results generally support the hypothesis that tree monocultures do not develop after vegetation release. Only one out of 94 experimental units developed into a tree layer monoculture (Simpson's reciprocal diversity index = 1). Again this was one of the repeated annual treatments of Vision herbicide at one of the black spruce study sites—a treatment which is atypical of Canadian forest management.

Keywords Biodiversity · Boreal forestry · Conservation · Herbicide alternatives · Plantation · Rank abundance plots · Release treatment · Vegetation management

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Introduction

Concern about artificially established single species forests—monoculture plantations—stems from the widely held belief that increases in species diversity provide stability and maintain ecosystem functioning. Modern forestry in Canada's northern forests often includes clearcutting followed by site preparation, planting, and chemical or motor-manual tending (NRCan 2004). This suite of activities is criticized for reducing biodiversity (May 2005) to the point of creating monocultures (Mosquin et al. 1995) resulting in potentially unstable and unsustainable forest communities. These are serious allegations that, if true, must be addressed in the context of current ecological knowledge and legislation such as the UN World Commission on Environment and Development (Brundtland 1987), the UN Convention on Biological Diversity (Environment Canada 1995; SCBD 2005), the Crown Forest Sustainability Act (Statutes of Ontario 1995), and Ontario's biodiversity strategy (OMNR 2005).

For the purposes of this study, the following qualitative definition of monoculture was adopted, the growing over a large area of a single crop species or of a single variety of a particular species (Allaby 1998). This state can also be defined quantitatively as a community with a Simpson's reciprocal diversity index (SRDI) value of one (Magurran 1988).

The term industrial plantation often used to describe northern Ontario plantations is defined as intensively managed forest stands established to provide material for sale locally or outside the immediate region, by planting and/or seeding in the process of afforestation or reforestation. Individual stands or compartments are usually with even age class and regular spacing and consist of one or two indigenous species (CIFOR 2002). This definition is appropriate to northern Ontario but must be clarified on two points: first, in most cases planted species are indigenous and include black spruce (*Picea mariana*), white spruce (*P. glauca*) and jack pine (*Pinus banksiana*) (Farrar 1995). Second, afforestation (as defined by Sauvageau 1995) is rare in northern Ontario. Almost all northern Ontario regeneration efforts are by definition reforestation (Sauvageau 1995), usually conducted within a few years after harvesting.

Another shortcoming of the definition offered by CIFOR (2002) is the use of the term ''intensive.'' In Canada, this term has come to mean ''Any management action beyond those required by law'' (Dunster and Dunster 1996), and is therefore different than what is implied above. Basic rather than intensive silviculture is typically practiced in Ontario (Bell et al. 2006) and, although usually only one species is planted, mixed species stands often develop (Chen and Popadiouk 2002). According to Dunster and Dunster (1996), basic silviculture is defined as, ''... the practices necessary to establish regeneration of the desired species at specific densities and stocking, free from competing vegetation, and within a certain time limit.''

The boreal and boreal/Great Lakes–St. Lawrence (GLSL) transition forests of Ontario provide an ideal setting for investigating the effects of silviculture on tree α diversity for a number of reasons. First, if Canadian forestry activities were going to create tree-level monocultures, it would most likely occur in northern Ontario where natural origin stands commonly have low tree richness (Thompson 2000). Typical stand compositions in this region include only four to eight tree species (Sims et al. 1989; Taylor et al. 2000). Furthermore, tree-level monocultures can occur even without the influence of forestry activities (Thorpe 1992; Johnson et al. 2003; Thompson et al. 2003). Second, many of the industrial forestry activities linked to creating single species and/or product forests are currently used in the region. For example, for decades regeneration efforts have focused on localized planting of only one native species (usually black spruce, white spruce, or jack pine), followed by herbicide application (NRCan 2004). Third, intensification of forestry activities is anticipated (OFAAB 2002) due to wood supply shortages and land use pressures (OMNR 2004).

Although tree species diversity is just one of many characteristics with debatable applicability in the overall assessment of biodiversity change (see Erdle and Pollard 2002; Betts et al. 2005), trees dominate forest landscapes and hence influence understory vegetation and other biotic communities. Several studies have shown that non-tree-layer monocultures rarely develop following typical forest management activities in northern forests (Esseen et al. 1997; Archibold et al. 2000; Boateng et al. 2000; Reich et al. 2001; Bell and Newmaster 2002; Rees and Juday 2002; Hunt et al. 2003), but few have considered this question exclusively from the viewpoint of tree layer diversity.[1]

In this study, we assess the effects of alternative release treatments on tree species diversity and richness in six industrial plantations in boreal and boreal-Great Lakes transition forests. Our objectives were to (1) quantify early seral tree species diversity and (2) assess whether tree-level monocultures developed ten growing seasons after release treatments at six northern Ontario conifer-planted study sites. Metrics included dominance-diversity curves, Simpson's indices for diversity and evenness, and a new approach, percentage of theoretical species maximum (%TSM), which compares a pre-disturbance theoretical tree species maximum richness (based on reliable, well-documented historical data) to measured post-disturbance tree richness. We test two hypotheses: first, that applying release treatments after clearcutting, site preparation, and planting one conifer species, does not significantly reduce tree species diversity (%TSM) when compared to untreated control plots (i.e., no release treatment applied) and, second, that tree-level monocultures do not develop after typical boreal silviculture that includes release treatments.

Methods

Six study sites (Fig. 1; Table 1) were established between 1990 and 1993 under the Ontario Ministry of Natural Resources (OMNR) Vegetation Management Alternatives Program (VMAP) (Wagner et al. 1995) and are located in four unique North American ecoregions of the Neararctic biogeographical zone (Ricketts et al. 1999). Within 2–4 years of harvest, study sites were individually planted with three different conifer species (white and black spruce and jack pine). After planting, study sites were treated with a range of release treatments including herbicides, cutting and combinations of both applied in different ways (Table 1).

All sites were assessed 10 years after alternative release treatments were applied (Bell et al. 1997; Pitt et al. 2000, 2004; Mallik et al. 2002; FRP 2005). Field data collection methods followed identical procedures, although sampling intensity varied for each study site (Table 1). Randomly laid out sampling transects (10 m by 2 m)

[1] We used the Forest Science database keyword search, searching all fields, limiting search to "Journal Articles" from "09/2005 to 1939" for the basis of this claim. Query 1 = (boreal) = 5,038 records, and Query 2 = (boreal) and ((tree biodiversity) or (tree diversity) or (tree species richness)) = 5 records. Of the articles with keyword "boreal" only 0.1% met our "tree diversity" search criteria (Source: Forest Sciences Database. 1997–2005 WebSPIRS. Ovid Technologies. Version 5.1. Build 20050721).

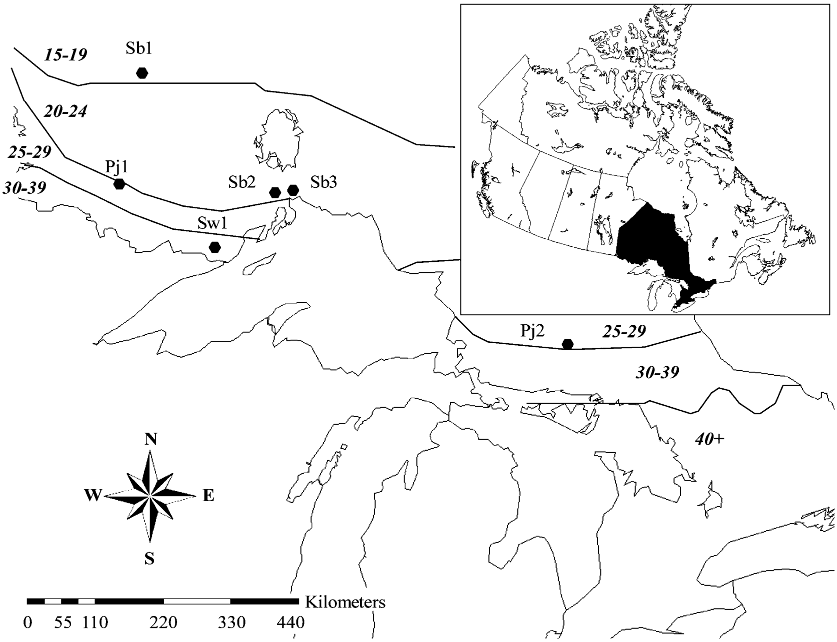

Fig. 1 Location of six early seral post-silviculture disturbance studies in northern Ontario, Canada. For site and treatment descriptions, and study locations refer to Table 1. Values indicate tree species richness (Thompson, 2000)

were established within each block-treatment area (the 94 experimental units). All tree species stems within sampling areas were identified and tallied. The richness and abundance of tree species on the Sb2 and Sb3 sites were likely underestimated because individuals under 130 cm were not recorded and these sites were sampled at the lowest intensity (Table 1). Tree species richness was estimated for conifer and hardwood species separately.

Dominance diversity curves for the trees (Burton et al. 1992) (also referred to as rank abundance plots) were produced for each study site by totaling tree species abundance for each treatment within each study site, across blocks. Relative abundance probabilities were then calculated, ranked, and plotted.

Of the available diversity indices, Simpson's reciprocal diversity index (SRDI) (Eq. 1) was used because of its responsiveness to dominant species (Pielou 1975; Magurran 1988). This was deemed important because expected species richness was low (Sims et al. 1989; Taylor et al. 2000; Thompson 2000). The value of the index ranges from one, denoting a tree-level monoculture, to just above five (Magurran 1988; Lande 1996), possibly the highest value expected for tree diversity in northern Ontario. To illustrate, a northern Ontario site with *very* high tree-level diversity might have eight tree species in the following proportions; 30, 20, 20, 10, 5, 5, 5, 5%. Its SRDI value would be 5.26. If species are equally present, the SRDI value would equal species richness (S).

$$\mathrm{SRDI} = 1/D = 1/\sum p_i^2, \qquad (1)$$

where p_i is the proportional abundance of the ith species.

Table 1 Summary of study site attributes including; tree species planted, study location, terrestrial ecoregion, pre-disturbance vegetation type, year of release treatment, experimental design, sampling intensity, treatment package, and number of experimental units

Study	Species planted	Study name; Location	Terrestrial ecoregion[a]	Pre-disturbance vegetation type[b]	Year of release	Design	Sampling intensity (# of 20 m^2 transects, m^2/exp.unit)	Treatments[e, f]	*n*
Sw1	White spruce	Fallingsnow Ecosystem; 48°08′N, 89°49′W	Western Great Lakes Forest	Block 1 = V-7[c] Trembling aspen—balsam fir /balsam fir shrub; Block 2 = V-5[c] Aspen hardwood; Block 3 = V-28c Jack pine /low shrub	1993	RCBD; 3 blocks	6, 120	1 rep/block BRU, CON, REL, SIL, VIS	15
Pj1	Jack pine	Bending Lake; 48°57′N, 92°02′W	Western Great Lakes Forest	V-17[c] Jack pine mixedwood/shrub rich	1992	RCBD; 4 blocks	6, 120	1 rep/block BRU, CON, CRV, VIS	16
Pj2	Jack pine	Domtar-Espanola; 46°47′N, 82°10′W	Eastern Forest/Boreal Transition	Block 1 = V-18[d] Jack pine/black spruce/blueberry Block 2 = V-17[d] Jack pine/black spruce/feathermoss Block 3 = V-17[d] Jack pine/black spruce/feathermoss	1993	RCBD; 3 blocks	4, 80	1 rep/block BBR, BRR, BRU, CON, CRV, MBV, VIS	21
Sb1	Black spruce	Leether Lake; 50°36′N, 91°45′W	Midwestern Canadian Shield Forests	V-17[c] Jack pine mixedwood/shrub rich	1993	CRD	4, 80	3 reps of 4 treatments BRU, CON, CRV, VIS	12

Table 1 continued

Study	Species planted	Study name; Location	Terrestrial ecoregion[a]	Pre-disturbance vegetation type[b]	Year of release	Design	Sampling intensity (# of 20 m^2 transects, m^2/exp.unit)	Treatments[e, f]	*n*
Sb2	Black spruce	Nipigon-Hele; 48°59′N, 88°33′W	Central Canadian Shield Forests	V-4[c] White birch hardwood and mixedwood	1990	RCBD; 3 blocks	3, 60	1 rep/block CON, CRV, RHV, SGV	12
Sb3	Black spruce	Nipigon-Corrigal; 49°02′N, 88°10′W	Central Canadian Shield Forests	V-14[c] Balsam fir mixedwood	1990	RCBD; 3 blocks	3, 60	1 rep/block BRU, CON, CRV, EZV, RHV, TLR	18

[a] Ricketts et al. (1999)

[b] Based on forest resource inventory (FRI) pre-disturbance stand conditions and unpublished pre-disturbance data

[c] Sims et al. (1989)

[d] Taylor et al. (2000)

[e] For full descriptions of treatments and sites see Bell et al. (1997), Pitt et al. (2000), Mallik et al. (2002), Pitt et al. (2004), FRP (2005)

[f] BBR, basal bark application of Release (triclopyr) herbicide with backpack sprayer; BRR, brushsaw cutting with stump herbicide applicator attachment with release; BRU, brushsaw cutting without herbicide applicator attachment; BRV, brushsaw cutting with stump herbicide applicator attachment with vision (glyphosate) herbicide; CON, untreated control; CRV, repeated annual treatments of vision; EZV, EZ-Ject injection of vision into competition basal stem; MBV, backpack mist blower application of vision; REL, aerial application of release from a Bell 206 helicopter; RHV, reel and hose application of vision; SIL, silvana selective cutting head mounted to a Ford versatile tractor with parallelogram boom; SGV, spot gun application of Velpar L (hexazinone); TLR, thin-line application of release; VIS, aerial application of vision from a Bell 206 helicopter

Measures of evenness indicate how species abundance is distributed within a community. Community can be defined as an assemblage of species inhabiting a common environment and interacting with one another. A community with equally abundant tree species will have a high evenness value (i.e., one), whereas, a community with differences in tree species abundance will have a low evenness value (i.e., approaching zero) (Smith and Wilson 1996). In this study, evenness was calculated using Simpson's evenness index (Eq. 2) because it gives precise and unbiased estimates and meets the requirements of an evenness index (Smith and Wilson 1996; Payne et al. 2005).

$$E_{1/D} = \mathrm{SRDI}/S, \tag{2}$$

where SRDI = reciprocal of Simpson's diversity index and S = species richness.

For a community where all species are equally frequent SRDI and S will be equal, therefore $E_{1/D}$ will equal one (Lande 1996).

In addition to the abovementioned metrics, we propose a new and simple method, percent theoretical species maximum (%TSM), for assessing the effects of silviculture by comparing a theoretical maximum pre-disturbance species count to a measured post-disturbance species count. The theoretical pre-disturbance maximum richness or *reference value* (White and Walker 1997; Terradas et al. 2004) was determined using a checklist approach (Droege et al. 1998). Because natural forests are temporally and spatially variable (White and Walker 1997), reference values must be carefully documented when applied in this manner. Thus, we applied two different theoretical species maximum (TSM) values based on different spatial scales (one local and one regional) to the measured post-disturbance tree richness (total number of tree species per experimental unit) (Boyle 1992).

In the first approach, we classified each site's pre-disturbance vegetation type (Table 1) using the standard forest ecosystem classification system for northern Ontario (Sims et al. 1989; Taylor et al. 2000). We then totaled the number of trees (trees as classified by Farrar 1995) in the V-Type description to obtain the local theoretical species maximum (LTSM) which was then used to calculate percent local theoretical species maximum (%LTSM). The LTSM is based on published, accepted spatial and historical data (White and Walker 1997; Landres et al. 1999) collected from sites with similar characteristics and is therefore directly applicable to our study.

The second approach is based on range data for individual tree species (Farrar 1995) delineated into geographical regional tree richness values (Fig. 1; Thompson 2000). We chose the highest richness value for each delineated region as the regional theoretical species maximum (RTSM) (Table 2) and used this value in our percent regional theoretical species maximum (%RTSM) calculations. For example, tree richness at the Sw1 study site ranged between 30 and 39 (Fig. 1) so the maximum value we used to calculate %RTSM for this site was 39 tree species (Table 2). As for LTSM, RTSM is based on well-founded and reliable data, but at a much broader scale.

The percent local and regional theoretical species maxima (%LTSM, %RTSM) were then calculated by dividing the measured post-disturbance tree richness for each experimental unit by the theoretical species maximum (LTSM and RTSM) for each experimental unit (reference value) and then multiplying by 100 (Eq. 3). This approach provides a method to compare a theoretical maximum to a measured

Table 2 Species richness and diversity measures ten years after conifer release treatments in six northern Ontario conifer plantations

Site	Treatment[a]	# of spp. (S)	# of conifer spp.	# of hard wood spp.	SRDI (1/D)	SEI ($E_{1/D}$)	LTSM[b]	RTSM[c]	%LTSM	%RTSM
Sw1	BRU	3–5	1–3	2	1.67–3.27	0.56–0.65	4–6	39	60–125	8–13
	CON	3–6	1–3	1–3	2.00–2.33	0.39–0.74	4–6	39	75–100	8–15
	REL	6–7	2–4	3–4	2.91–4.80	0.42–0.80	4–6	39	100–175	15–18
	SIL	4–6	2	2	1.92–3.26	0.48–0.65	4–5	39	83–175	10–13
	VIS	3–4	1–2	1–2	1.66–3.10	0.55–0.78	4–6	39	60–75	8–10
Pj1	BRU	2–4	1–2	1–2	1.27–1.78	0.42–0.84	6	29	33–67	7–14
	CON	3–4	1–3	1–3	1.84–3.51	0.52–0.88	6	29	50–67	10–14
	CRV	2–4	2	0–2	1.09–2.89	0.38–0.72	6	29	33–67	7–14
	VIS	2–6	1–4	1–3	1.09–2.50	0.36–0.62	6	29	33–100	7–21
Pj2	BBR	6–7	3–5	2–3	2.62–2.96	0.37–0.44	5–7	29	100–140	21–24
	BRR	6–7	3–5	2–4	1.91–3.99	0.32–0.57	5–7	29	100–140	21–24
	BRU	5–8	4–5	1–3	1.99–4.19	0.40–0.52	5–7	29	100–120	17–28
	CON	4–7	1–4	1–3	1.78–3.25	0.44–0.46	5–7	29	71–140	14–24
	CRV	3–6	2–5	1–2	1.96–3.24	0.46–0.65	5–7	29	60–100	10–21
	MBV	6–11	3–8	3–4	2.91–5.37	0.32–0.52	5–7	29	120–180	21–38
	VIS	5–8	3–5	1–3	1.94–4.74	0.39–0.59	5–7	29	100–114	17–28
Sb1	BRU	3–4	2	1–2	1.70–2.43	0.47–0.81	5	19	60–80	16–21
	CON	3–5	2–3	1	2.28–2.72	0.53–0.76	5	19	60–100	16–26
	CRV	2–3	2	0–1	1.80–2.09	0.60–0.94	5	19	40–60	11–16
	VIS	2–3	2	0–1	1.49–2.24	0.50–0.87	5	19	40–60	11–16
Sb2	CON	2–4	1	1–3	1.41–3.45	0.70–0.86	6	24	33–67	8–17
	CRV	**1**–3	1–2	0–1	**1.00**–1.81	0.60–1.00	6	24	17–50	4–13
	RHV	2–3	2	0–1	1.22–1.98	0.61–0.99	6	24	33–50	8–13
	SGV	2–3	1–2	1	1.60–2.00	0.60–1.00	6	24	33–50	8–13
Sb3	BRV	3–5	1–2	2–3	1.82–2.27	0.36–0.74	8	24	38–63	13–21
	CON	4–5	1–3	2–3	1.72–2.39	0.43–0.52	8	24	50–63	17–21
	CRV	3	2–3	0–1	1.41–2.49	0.47–0.83	8	24	38	13
	EZV	4–6	2–3	2–3	2.51–3.17	0.53–0.63	8	24	50–75	17–25
	RHV	4–5	1–3	2	2.59–4.17	0.65–0.83	8	24	50–63	17–21
	TLR	4	2	2	1.46–2.46	0.37–0.62	8	24	50	17

S, measured tree richness, number of conifer and hardwood species; SRDI, Simpson's reciprocal diversity index; SEI, Simpson's evenness index; LTSM, local theoretical species maximum; RTSM, regional theoretical species maximum; %LTSM, percent local theoretical species maximum; %RTSM, percent regional theoretical species maximum

Results are expressed as ranges rather than averages to demonstrate the variability across blocks within study sites. Possible tree-level monocultures in bold

[a] Refer to Table 1 for treatment codes

[b] Based on values from Sims et al. (1989), Taylor et al. (2000)

[c] Based on maximum values from Thompson (2000)

richness. One caveat, however, applies to this approach: the sampling units within our experimental units are relatively small (60–120 m^2 exp. $unit^{-1}$) relative to very large sampling unit used to determine the LTSM (V-Type), and RTSM (species range data). Although not practically feasible, for precise comparisons sampling intensity should have been 100% for each unit.

$$\%\text{LTSM or }\%\text{RSTM} = (S/\text{TSM}) \times 100, \tag{3}$$

where S = post-disturbance species richness and TSM = either local or regional theoretical species maxima.

Analysis of Variance (ANOVA) was run using the SPSS Ver. 11.0.1 General Linear Model (Univariate) function to compare treatment effects in the underlying experimental designs (RCBD, CRD) for each study site on %LTSM and %RTSM values. Planned comparisons (a priori) related all release treatments to the untreated control because it was assumed that the control would have the greatest relative tree diversity. Diagnostic normal probability plots of model residuals and side-by-side dot plots of residuals were used to verify that the assumptions of normality and homogeneity of variance were met.

An ANOVA was not conducted on Simpson's reciprocal diversity index (SRDI) or Simpson's evenness index (SEI) because diversity indices are based on proportional abundance of species (Magurran 1988) making interpretation difficult (Zar 1984). Furthermore species indices are non-parametric (Magurran 1988) and do not *necessarily* meet ANOVA assumptions. Indices were used to supplement and provide additional insight into the ranked abundance curves and %TSM values.

Results

Tree species sampled across all six study sites included black spruce, white spruce, jack pine, balsam fir (*Abies balsamea*), red pine (*Pinus resinosa*), white pine (*P. strobus*), eastern white cedar (*Thuja occidentalis*), trembling aspen (*Populus tremuloides*), balsam poplar (*P. balsamifera*), largetooth aspen (*P. grandidentata*), white birch (*Betula papyrifera*), sugar maple (*Acer saccharum*), red maple (*A. rubrum*), black ash (*Fraxinus nigra*), and eastern larch (*Larix laricina*).

The slope of dominance-diversity curves can suggest diversity and evenness index values (Fig. 2a–f; Table 2). Curves with steep slopes indicate dominance by one or a few species which usually translates into low richness and low evenness (Burton et al. 1992). For example, the steep slope of the dominance-diversity curve for the Vision herbicide (VIS) treatment at the Pj1 site (Fig. 2b) is confirmed by relatively low diversity and evenness (Table 2; SRDI = 1.09–2.50, SEI = 0.36–0.62). As the slope of the dominance-diversity curve becomes more gradual, dominant species become less conspicuous suggesting high diversity and richness (Burton et al. 1992). For example, the gradual slope for the reel hose application of Vision (RHV) treatment at the Sb3 site (Fig. 2f) indicates relatively high diversity and evenness (Table 2; SRDI = 2.59–4.17, SEI = 0.65–0.83).

Results for all study sites are summarized in Table 2 and are expressed as ranges rather than means to demonstrate variability in response across blocks within study sites, and to show where the possible tree-level monoculture occurred (i.e., the treatment unit with both S and SRDI = 1.00). Of 94 experimental units, one tree-level monoculture was detected in one experimental unit at the Sb2 site in a repeated annual applications of Vision (CRV) treatment.

It was expected that generalizations across studies would be difficult due to the wide variety of site factors (i.e., differences in ecoregions, pre-disturbance vegetation types, site histories, harvesting and site preparation methods and planted species). However one trend was observed: both the VIS and CRV treatments resulted in the

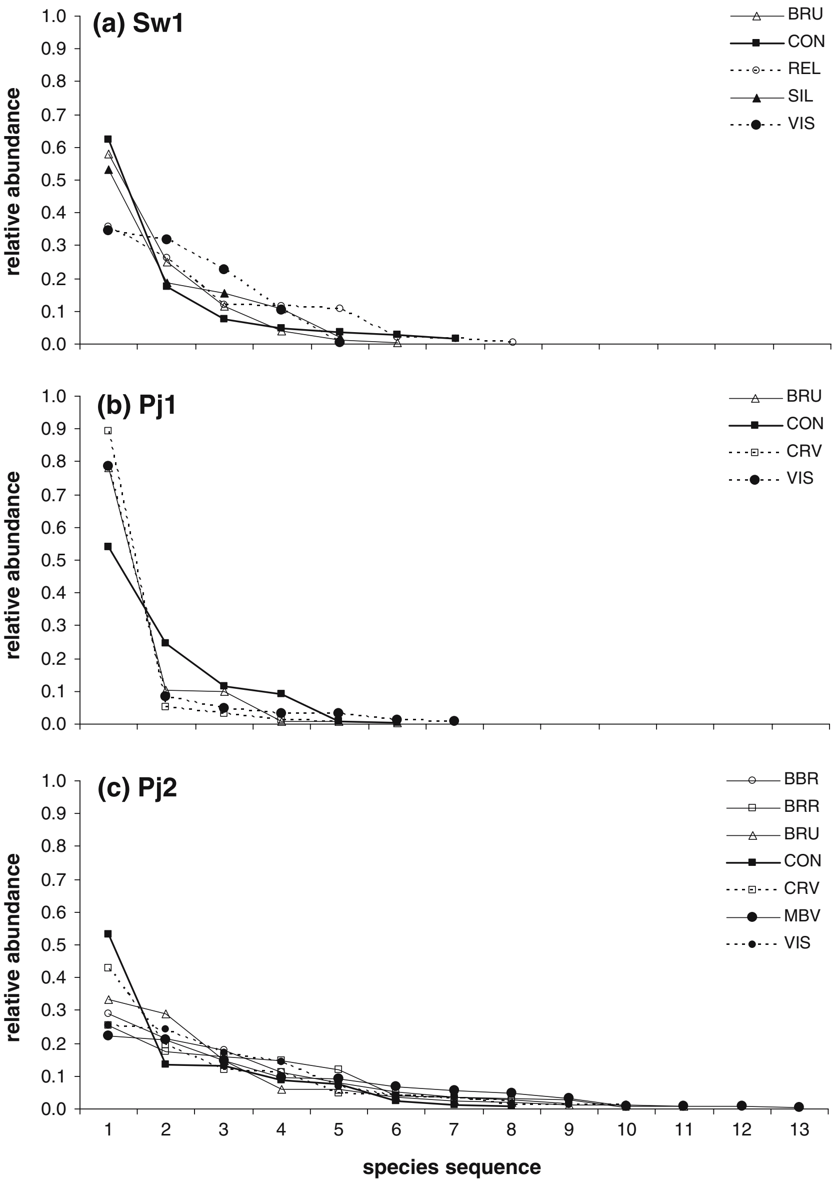

Fig. 2 (**a–c**) Dominance diversity curves for one white spruce (Sw1) and two jack pine (Pj1, Pj2) plantations in northern Ontario, 10 years after conifer release treatments. (**d–f**) Dominance diversity curves for three black spruce plantations (Sb1, Sb2, Sb3) in northern Ontario, 10 years after conifer release treatments. For treatment codes see Table 1

least tree species diversity (lowest %LTSM and %RTSM) at most study sites. Furthermore, VIS and CRV treatments resulted in more conifers than hardwoods at most sites as indicated by the number of conifers and hardwoods in Table 2. Only the Sw1 site had an overall treatment effect ($P \leq 0.05$). Significant a priori comparisons ($P \leq 0.05$) of %LTSM and %RTSM included untreated control (CON)

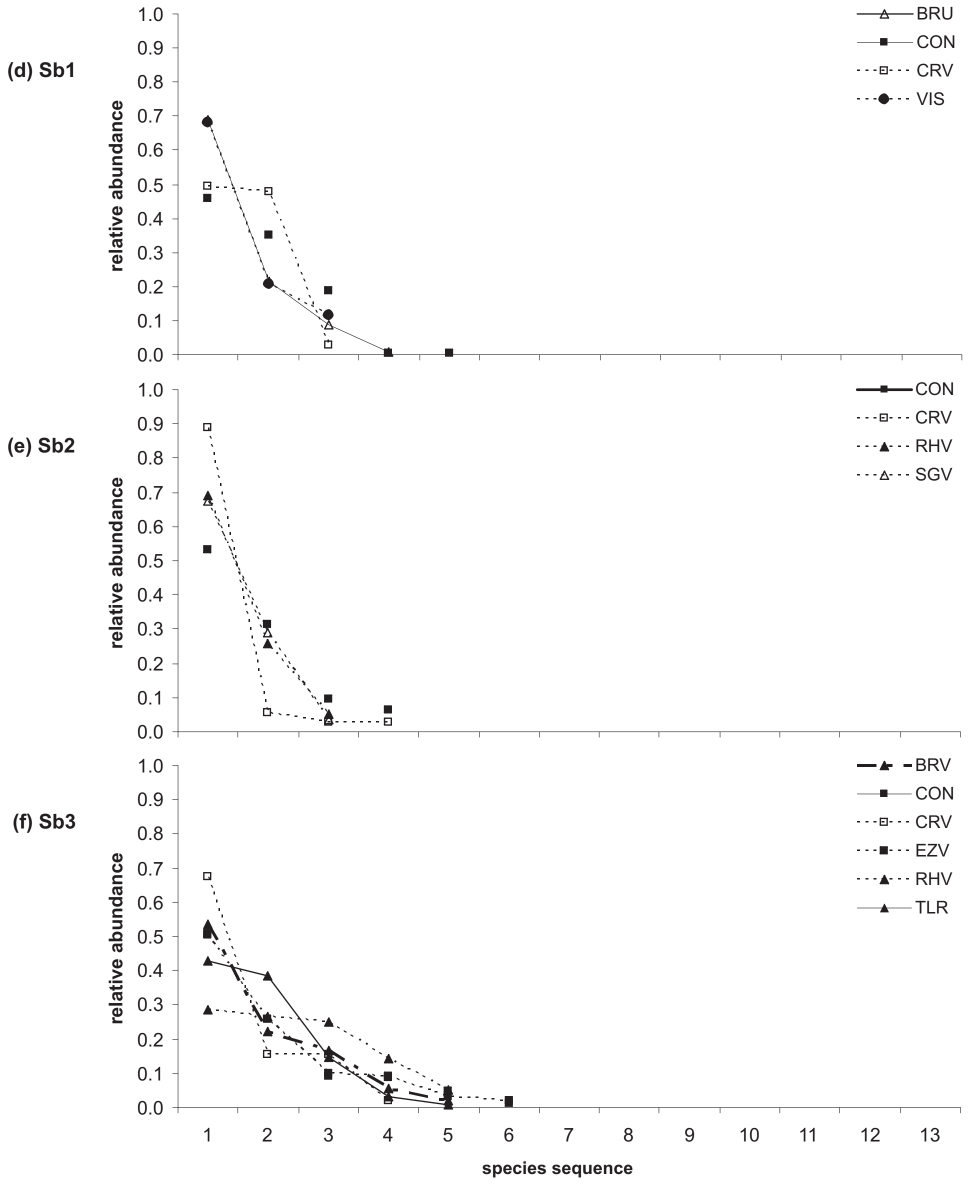

Fig. 2 continued

vs. Release herbicide (REL) at Sw1; CON vs. mist blower application of Vision (MBV) at Pj2; and CON vs. CRV at Sb3 (Table 3).

Discussion

Our findings generally support our first hypothesis that applying alternative vegetation release treatments after clearcutting, site preparation, and planting one

Table 3 ANOVA results (*P*-values) for percent local theoretical species maximum (%LTSM) and percent regional theoretical species maximum (%RTSM) for treatment comparisons 10 years after release treatment in six northern Ontario conifer plantations

Site	Comparison	%LTSM	%RTSM
Sw1	CON vs. BRU	0.55	0.31
	CON vs. REL	**0.02**	**0.01**
	CON vs. SIL	0.86	1.00
	CON vs. VIS	0.28	0.14
	Overall	**0.02**	**0.01**
	SE	0.15	0.02
Pj1	CON vs. BRU	0.13	0.13
	CON vs. CRV	**0.04**	**0.04**
	CON vs. VIS	0.21	0.21
	Overall	0.19	0.19
	SE	0.12	0.12
Pj2	CON vs. BBR	0.29	0.27
	CON vs. BRR	0.39	0.40
	CON vs. BRU	0.72	0.57
	CON vs. CRV	0.47	0.57
	CON vs. MBV	0.10	0.06
	CON vs. VIS	0.96	0.78
	Overall	0.28	0.28
	SE	0.21	0.04
Sb1	CON vs. BRU	0.50	0.50
	CON vs. CRV	0.07	0.07
	CON vs. VIS	0.07	0.07
	Overall	0.16	0.16
	SE	0.09	0.09
Sb2	CON vs. CRV	0.36	0.36
	CON vs. VIS	1.00	1.00
	Overall	0.54	0.54
	SE	0.11	0.11
Sb3	CON vs. BRU	0.55	0.55
	CON vs. CRV	**0.03**	**0.03**
	CON vs. EZV	0.24	0.24
	CON vs. RHV	0.55	0.55
	CON vs. TLR	0.24	0.24
	Overall	0.06	0.06
	SE	0.07	0.07

Values in bold are significant

conifer species, does not significantly reduce tree species diversity levels, although tree-level diversity was significantly reduced in one experimental unit (i.e., CRV at Sb3). However, repeated annual applications of herbicide (CRV) is an intensive, experimental treatment that is not used in operational forest management. Based on the results of this study, treatments typically used in boreal forest management produce similar tree-level diversity as found in the untreated control plots.

Our findings also support our second hypothesis, except in one case. Although one tree-level monoculture occurred at the Sb2 site (CRV), previously published results from this site do not match our findings. At the Sb2 site, 10 years after release, Pitt et al. (2004) report less than 11% competing tree cover on CRV plots, however, no tree-level monocultures were reported. This discrepancy can be explained by differing field methods. Pitt et al. (2004) used 20 crop tree centered sampling sub-plots (4 m^2 each) within each experimental unit, while we used three, 20 m^2 sampling transects within each experimental unit. Furthermore, we counted tree stems while Pitt et al. (2004) used tree cover, which increases the likelihood of sampling more tree species.

The experimental unit deemed a monoculture was also sampled at the lowest intensity (60 m^2 experimental $unit^{-1}$). Larger sampled areas are more likely to provide lists with greater species counts (Magurran 1988; Burton et al. 1992), therefore the likelihood of identifying tree-level monocultures will be higher for smaller sampling areas. Nonetheless, mathematically rare species can be missed even in large samples (Lande 1996). Our results (especially from sites where sampling intensity was relatively low) therefore provide anecdotal evidence which supports Lande (1996).

It is not surprising that the most intensive herbicide treatment (i.e., CRV) produced a tree-level monoculture in one of the experimental units. Pitt et al. (2000) suggested that annual herbicide treatments could produce tree-level monocultures at stand maturity. However, such treatments are very rarely used in Canadian forest operations because application costs are high and single applications tend to be sufficient to accomplish the silvicultural objective of maintaining conifer dominated stands (Dampier et al. 2006). As well, conifer-dominated tree-level monocultures do develop naturally in northern Ontario forests (Rudolph and Laidly 1990; Viereck and Johnston 1990; Thorpe 1992; Johnson et al. 2003; Thompson et al. 2003) and there may be legitimate ecological reasons for establishing such stands within the matrix of a managed forest.

Continuously removing vegetation using annual treatments of Vision might create a tree monoculture when: (1) a single conifer species is present, and (2) little chance for ingress from other tree species seed exists. However, when we accounted for non-tree vegetation classes such as woody shrubs, herbs, grasses, ferns, sedges, mosses, and lichens (unpublished data), no species monocultures were detected in any of the 94 experimental units across northern Ontario. Our results are supported by a recent review (Carnus et al. 2006) and a recent study near Thunder Bay, Ontario (Newmaster et al. 2006). The latter reported reduced plant diversity on afforested, 50-year-old tree plantations on formerly abandoned agricultural lands when compared with natural forest ecosystems dominated by the same tree species but did not report the establishment of monocultures.

A false linkage between reforestation (as opposed to afforestation) and monoculture development appears to exist. For example, Mosquin et al. (1995, 72) state that tree plantations established on *lands marginal for agriculture* (afforestation) tend to be monocultures. Indeed, many of Canada's oldest tree plantations *were* established on former agriculture fields with heavy grass competition; hence, the straight rowed, single species stands common to central and eastern Ontario. Mosquin et al. (1995, 72) implies that the practice of planting single or low numbers of species over large holdings of recently logged public forest lands, coupled with spraying [herbicides] for commercially undesirable species creates monocultures. Our study demonstrates that this practice *does not* generally create tree-level monocultures in the Neararctic biogeographical zone of Canada.

Acknowledgements The collection of data used in this study was sponsored by the Ontario Ministry of Natural Resources (OMNR), Forestry Research Partnership (CEC-FRP), Living Legacy Trust (LLT), Upper Lakes Environmental Research Network (ULERN), and Spray Efficacy Research Group (SERG) members: specifically the Manitoba Department of Natural Resources and Forest Protection Limited (FPL). Financial support for data analysis was provided by the Natural Sciences and Engineering Research Council of Canada (NSERC), Monsanto Canada, Inc., and Forestry Futures Trust Enhanced Forest Productivity Science Program (FFT-EFPS). The projects used in this study were established under OMNR's Vegetation Management Alternatives Program (VMAP). The authors greatly appreciate the efforts of John Winters and Amy Bolduc and the many folks involved in the field assessments. The authors appreciate technical advise provided by R. Maki, Monsanto Canada, Inc. Editorial and technical advice on an earlier draft was provided by P. Comeau (University of Alberta), D. Pitt (NRCan), J. Winters, A. Morneault, and L. Buse (OMNR) and two anonymous reviewers. The lead author also thanks Mark Lesser, University of Wyoming, for stimulating conversations related to this study and for assistance in producing Fig. 1.

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