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Exotic plant responses to large-scale commercial logging in ponderosa pine forests of the Black Hills National Forest, South Dakota, USA

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ABSTRACT

Current and long-term historic commercial logging in the Black Hills National Forest provided a cost-effective opportunity to investigate exotic plant responses to a wide range of thinning intensities on a large scale. We asked two questions. First, after over 125 years of post-settlement commercial logging, what role does the pre-harvest exotic plant community play in post-harvest responses to logging. Second, how does a broad range of harvest intensities influence exotic plant responses on a large spatial scale. We averaged exotic species relative cover and relative richness to create a synthetic pre-harvest Exotic Vegetation Index (EVI) that was used as one of 15 predictor variables. The response variable was the difference between the pre-harvest and post-harvest EVI. We evaluated plots in ten widely scattered timber sales at pre-harvest and 1-, 2-, 3-, and 7-years post-harvest. In individual plots where the pre-harvest EVI was greater than 20 %, post-harvest exotic species responses were highly variable that, when averaged, the difference in EVI between pre-harvest and 7-years post-harvest was zero. In plots where the pre-harvest EVI was $\leq$ 20 %, the difference in EVI between pre-harvest and 7-years post-harvest was significantly and positively correlated with harvest intensity ($r = 0.65$, $p < 0.001$). Plots with the highest levels of harvest intensity showed the greatest increases above pre-harvest values, which included several noxious weed species. In contrast, plots with the lowest levels of harvest intensity showed low levels of exotic increases with some plots exhibiting reductions in EVI below pre-harvest levels at 7-years post-harvest.

1. Introduction

Ponderosa pine (*Pinus ponderosa*) forests are a major forest type found throughout the western United States, southern Canada, and northern Mexico, which represents a considerable gradient in latitude, longitude, and elevation (Graham and Jain, 2005). Included in this range are the isolated ponderosa pine forests in the Black Hills of southwestern South Dakota and the nearby Bear Lodge Mountains in northeastern Wyoming (Shepperd and Battaglia, 2002). Although geographically widespread, ponderosa pine forests throughout their range evolved under similar environments that largely centered around a distinctive fire regime (Moore et al., 1999). A common theme among pre- Euro-American settlement ponderosa pine forests is that recurrent low-intensity surface fires maintained open forests characterized by large trees at low density with an abundant herbaceous understory (Cooper, 1960; Covington and Moore, 1994a; Brown and Cook, 2006; Brown and Sieg, 1996, 1999). Following Euro-American settlement, decades of fire exclusion, selective logging, and livestock grazing have dramatically altered many forests (Cooper, 1960). Post-settlement changes in ponderosa pine forest structure and function are largely associated with high density, small diameter trees, reduced understory cover, insect outbreaks, and high intensity wild fires (Covington and Moore, 1994a; 1994b).

Lowering the risk of wildfire and insect outbreaks by mechanically reducing ponderosa pine tree densities is often used as a prelude to prescribe fire and as part of long-term plans to restore pre-settlement open canopy forest structure and function (Covington et al., 1997; Moore et al., 1999; Allen et al., 2002; Fulé et al., 2002; Moore et al., 2006; Roccaforte et al., 2010). However, disturbances associated with many mechanical thinning efforts increases the risk of exotic species

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expansion (Schwilk et al., 2009; Abella and Springer, 2015) where invasive species can have a major and long-lasting impact on the ecology and management of forest resources (Chornesky et al., 2005). A single invasive plant species, such as cheatgrass (*Bromus tectorum*), can drastically alter the trajectory of ponderosa pine forest restoration efforts (McGlone et al., 2009, 2012). Even the larger number of exotic species considered benign and not invasive ("weak invaders" sensu Ortega and Pearson, 2005) could become environmentally damaging with shifts in one or more elements of global change.

A meta-analysis by Willms et al. (2017) on understory responses to fuel reduction treatments in fire-prone forest ecosystems found that significant increases in exotic species richness were a consistent theme following relatively small-scale thinning treatments in several mixed conifer forests in the western United States (also see Sutherland and Nelson, 2010). However, broadly applying reported exotic plant responses to thinning treatments beyond individual small-scale studies is often complicated by a host of interrelated factors that vary among forest types, thinning intensities, geographic regions, and duration of the studies (Duguid and Ashton, 2013; Abella and Springer, 2015). Managers of ponderosa pine forests considering mechanical thinning treatments to improve forest health could benefit from large-scale studies that include a wide range of thinning intensities. Commercial logging can provide managers with economically feasible operational-scale opportunities to reduce tree densities in forests that have significantly departed from pre-settlement conditions (Schwilk et al., 2009). Thinning through logging can also be part of long-term restoration plans if they are based on established ecological guidelines for restoring pre-settlement forest structure and function in a way that promotes natural resilience and sustainability (Allen et al., 2002). This long-term goal of reestablishing some semblance of the pre-settlement evolutionary environment requires a landscape approach (Moore et al., 1999) that would benefit from large-scale studies that include a wide range of thinning intensities. However, such studies are largely absent (Abella and Springer, 2015), ostensibly because of the financial cost, which commercial logging could subsidize.

In 2007, we initiated a large-scale study in the Black Hills National Forest (BHNF), South Dakota and Wyoming, designed to evaluate the short- and long-term responses of the understory plant community to commercial timber harvest. The ponderosa pine forest of the Black Hills is an intensively managed forest with a long history of timber harvest that dates back to the late 19th century when large areas experienced unregulated clear cutting to support the burgeoning mining industry (Freeman, 2015). In 1899, the first timber sale in the United States, sanctioned under the new Forest Reserve system established in 1897, took place in what is now the Black Hills National Forest (Freeman, 2015). By the mid-1970s, nearly all of the land in the suitable base of the Black Hills had been logged at least once with many sites receiving multiple harvests (Alexander and Edminster, 1981; Boldt and Van Deusen, 1974; Shepperd and Battaglia, 2002). From 1997 to 2011, an average of 154,543 hundred cubic feet/year of sawtimber was sold, which increased to 193,107 hundred cubic feet/year from 2012 to 2017 (reported by Graham et al., 2021 based on Black Hills National Forest records).

Like many forest landscapes with a long post-settlement history of fire suppression intermixed with logging, livestock grazing, insect outbreaks and wildfires, exotic plants are a common component of the understory vegetation throughout much of the Black Hills (Larson and Johnson, 2007). Tree canopy cover is a significant predictor of understory biomass production in the Black Hills with the total amount of vegetation produced dependent upon local topo-edaphic characteristics (Bennett et al., 1987). While biomass was not separated based on species origin (native or exotic) in this study (Bennett et al., 1987), several landscape-scale studies have reported positive correlations between native and exotic species richness and abundance (Stohlgren et al., 1999; Gilbert and Lechowicz, 2005; Fornwalt et al., 2003; Abella et al., 2012). Because of the negative relationship between the overstory and understory reported for the Black Hills (Bennett et al., 1987), it is reasonable to expect thinning treatments that include a gradient of decreasing overstory would result in a gradient of increasing understory vegetation (Uresk and Severson, 1989, 1998; Wienk et al., 2004) that would include both native and exotic species.

Our research addresses the broad and specific responses of exotic vegetation, as a functional group relative to their native counterparts, to forest-wide commercial timber harvests that occurred in the BHNF from 2007 to 2011. Because of the large geographic and temporal scale of the project, not all pre-harvest plots were established in the same calendar year. To facilitate making cross-plot comparisons over different time periods within a broad landscape, we used relative values of richness and cover rather than absolute values in our analyses. Guo and Symstad (2007) and Guo et al. (2015) described how relative exotic species richness and abundance (cover or biomass) may be used in combination as a measure of the degree of invasion while also providing a mechanism for making comparisons across plant communities and across time periods. In our study, we averaged relative exotic richness and relative cover to create a single synthetic Exotic Vegetation Index (EVI). Although species richness and abundance (cover or biomass) contribute differently to observed patterns in community composition (Bernard-Verdier and Hulme, 2015) and to the invasion process following silvicultural practices (Sutherland and Nelson, 2010), giving equal weight to relative richness and foliar cover allowed us to combine pre-harvest exotic composition and post-harvest colonization and expansion into single values.

We used the BHNF as a large-scale laboratory to address two questions relevant to pre-harvest exotic composition and post-harvest responses to thinning through commercial logging. First, what role does the level of pre-harvest infestation of exotic vegetation play in post-harvest responses to large-scale thinning? The pre-harvest exotic plant community is likely an important reference point in evaluating exotic species responses to a range of thinning intensities through commercial logging. For example, in evaluating restoration efforts for pine-oak forests in Arizona, Fulé et al. (2005) used pre-existing differences in understory plant cover among plots as a covariate in evaluating understory responses to thinning and burning treatments. They found that the understory responses varied more with pre-treatment community characteristics and climate than by the restoration treatments. However, while the importance of the pre-treatment understory community to post-treatment responses is generally recognized, especially if an aggressive exotic plant species is part of the pre-treatment understory (McGlone et al., 2009, 2012), few studies are available that incorporated pre-treatment conditions in evaluating post-treatment responses to

forest restoration experiments (Abella and Springer, 2015). Our second question asks what is the relationship between relatively broad disturbance intensities associated with commercial logging and post-treatment responses of exotic species? While the influence of disturbance intensity on exotic responses is widely recognized, it is likely resource-prohibitive in relatively small plot studies to test the importance that a wide range of thinning intensities on a large spatial scale may have on post-thinning responses.

Decision makers and managers recognize that numerous small-scale studies show that significant increases in exotic herbaceous vegetation following thinning treatments in ponderosa pine forests are possible, if not likely. While these studies contribute substantially to our understanding of understory responses to thinning, their relatively small spatial and temporal scales and limited thinning intensities may restrict their applicability to the broader scales necessary for long-term forest-level planning. Management planning can benefit from an increased understanding of the potential role that the pre-harvest exotic plant community may play in post-harvest responses (question 1), and by additional insight into how a broad range of harvest intensities may influence the spatial and temporal responses of exotic plants (question 2).

2. Methods

2.1. Study area

The Black Hills are often characterized as a forested island dominated by ponderosa pine and surrounded by a sea of northern mixed-grass prairie (Brown and Sieg, 1999; Shepperd and Battaglia, 2002). The area has a semi-arid continental climate with cold winters and warm summers. Normal annual precipitation (1961–90) for Black Hills is 485 mm while the long-term (1931–98) annual average is 472 mm with most of the precipitation falling as rain in the early spring, although the northern Black Hills can have a heavy winter snowpack (Driscoll et al., 2000). The northern Black Hills are generally cooler and wetter than in the southern parts, a trend that is consistent throughout the year (Froiland, 1990). Climatic conditions in the Black Hills are especially favorable for rapid ponderosa pine regeneration and growth (Graham et al., 2021) that was historically moderated by periodic fires in the forested interior (Brown and Sieg,1996) with more frequent fires at the forest-grassland ecotone at lower elevations (Brown and Sieg,1999). Brown and Cook (2006) described the higher elevation pre-settlement forest structure of the Black Hills as a complex mosaic consisting of non-forested meadows and open stands with large trees intermixed with a few dense stands with closed canopies. However, post-settlement fire exclusion and a favorable climate for ponderosa pine growth quickly resulted in larger and more uniform stands of dense trees leading to increased risk of intense wildfires (Brown and Cook, 2006) and mountain pine bark beetle (*Dendroctonus ponderosae*) outbreaks (Negron et al., 2017).

Vegetation of the Black Hills is dominated by ponderosa pine (*Pinus ponderosa* C. Lawson), with white spruce (*Picea glauca* [Moench] Voss), and occasional deciduous trees such as bur oak (*Quercus macrocarpa* Michx.), quaking aspen (*Populus tremuloides* Michx.) and paper birch (*Betula papyrifera* Marsh.) (Larson and Johnson, 2007). The unique geographical location of the Black Hills contributes to the understory plant diversity with 30 % of the species having origins from the Rocky Mountains, 17 % from the Great Plains, 16 % from other parts of North America, and the remaining from other places around the world (Larson and Johnson, 2007). Typical native understory species include poverty oatgrass (*Danthonia spicata*), upland sedges (primarily *Carex inops* subsp *heliophila*), white coralberry (*Symphoricarpos albus* [L.] S.F. Blake), creeping barberry (*Mahonia repens* [Lindl.] G. Don), kinnikinnick (*Arctostaphylos uva-ursi* [L.] Spreng.), and native wheatgrasses (*Elymus* spp.). Common exotic grass species include bluegrasses, (*Poa pratensis* L. and *P. compressa* L.), smooth brome (*Bromus inermis* ssp. *inermis*), and timothy (*Phleum pratense* L.). Plant nomenclature and nativity follows USDA Plants Database (www.plants.usda.gov).

We selected 12 cutting units within ten timber sale areas in which pre-harvest data could be collected in the same growing season as the harvest (Appendix Table A). A timber sale is the formal process for an individual or company to buy a contract to cut and remove timber (www.fs.usda.gov/managing-land/forest-management/products/timber-sales). A cutting unit is a subset parcel of a timber sale that may have specific harvest requirements for meeting stewardship policies and goals unique to the Black Hills National Forest such as protection of rare species, archeological sites, water resources, or other natural or cultural features.

Five timber sales were selected in 2007, two in 2008, two in 2010, and one in 2011. Size of the individual cutting units within a timber sale area ranged from 31 to 70 ha (mean = 45.1 ha). The sites are on three different soil series that are predominately typic Eutroboralfs, loamy and fine-loamy, skeletal, with mixed mineralogy. All study sites have well drained, non-ponding soils with root-restrictive soils >150 cm deep below ground level, with slopes that ranged from 10 % to 40 % (USDA Natural Resources Conservation Service, 2012). Elevation ranged from approximately 2056 m for plots established in the northern Black Hills (Elk Hay) to approximately 1233 m for plots established in the southern Black Hills (Dark Canyon).

2.2. Treatments

Three different tree harvesting methods that differ in disturbance intensity were used in this study: whole tree skidding (8 timber sales), cut-to-length (1 timber sale), and conventional (1 timber sale). The main differences between the types of harvest methods were how the slash (de-limbed tree branches) was managed. Whole tree skidding involves dragging cut trees, with limbs attached, to a landing site for delimbing. In conventional harvesting, trees were manually cut with chainsaws, delimbed where they fell, and then the boles were skidded to a landing site. Cut-to-length harvesting involved a large machine that cuts the tree at ground level, delimbs, and cuts to length, all in situ. In contrast to whole tree skidding, only the boles are dragged in conventional and cut-to-length harvesting. In both conventional harvesting and cut-to-length harvest, the slash resulting from the delimbing process remains on the forest floor where the trees were cut. None of the slash was masticated, either in-situ or at landing sites.

2.3. Data collection

We used a variant of the modified-Whitaker plot (Stohlgren et al., 1999) to collect vegetation data. Each plot consisted of a 0.1 ha macroplot (20 m X 50 m) and 12, 1 m^2 subplots (0.5 m X 2 m). Four ground level metal markers were established at each corner of the macroplot and two metal markers were established in opposite corners of a 5 m X 20 m reference plot that was located in the center of the macroplot.

Global Positioning System coordinates were recorded for all permanent metal markers. Locating the metal markers was facilitated by using a metal detector. Eight subplots were established around the inside perimeter of the macroplot while four subplots were located around the outside perimeter of the center reference plot. Using a tape measure and moving clockwise around the perimeter of the macroplot (140 m perimeter) beginning with the zero corner where the first subplot was placed, additional subplots were placed at eight meters, 29 m, 41 m, 70 m (opposite corner from zero plot), 80 m, 101 m, and 125 m. Beginning with the zero corner of the 5 × 20 m reference plot, (50 m perimeter), and moving clockwise, the remaining four subplots were placed at two meters, eight meters, 25.5 m, and 37 m of a tape measure.

Macroplots were subjectively established to capture as complete a set of the physical and biological characteristics of Black Hills ponderosa pine forests as possible. Individual plots were placed to maximize within plot homogeneity with respect to slope (short axis of the plot), aspect (long axis of the plot), and general soil characteristics. The number of permanent macroplots within individual timber sales generally varied with the size and heterogeneity of the timber sale. We specifically placed some macroplots directly adjacent to timber sale roads (roads used exclusively during timber harvest) and numbered Forest Service roads to potentially capture additional levels of disturbance. We also established a series of plots adjacent to but outside the cutting unit boundary (unlogged plots) by at least 100 m for seven timber sales. Unlogged plots could not be established in three timber sale areas (Elk Hay, Goat, and Scott) because of land ownership boundaries or the potential unlogged plots had physical or biological characteristics outside the variation of the logged plots. Selection of plots outside and adjacent to active cutting units followed the same procedures as within the cutting unit. A total of 96 logged and 42 unlogged macroplots were established.

We collected information on slope (%), aspect, and topographic position (summit, midslope, draw bottom) at each macroplot. Aspect was transformed to a radiation index that varies continuously between 0 and 1 [TRASP = 1 - cos((pi / 180)(aspect – 30)) / 2; Roberts and Cooper, (1989)]. The transformation assigns a value of zero to slopes oriented in a north-northeast direction, which are usually the coolest and wettest slopes. In contrast, a value of one is assigned to drier south- and west-facing slopes.

We estimated foliar cover of each species in the understory vegetation in each subplot using cover classes: Trace < 1 %, 1 = 1–5 %, 2 = 5–25 %, 3 = 25–50 %, 4 = 50–75 %, 5 = 75–95 %, and 6 = 95–100 %). All herbaceous species, and tree species ≤ 30 cm in height, were considered part of the understory. Shrub species, regardless of height, were also evaluated as part of the understory vegetation. We used the mid-point of the cover classes ("Trace" was analyzed at 0.05 %) to calculate mean cover for each species, bare soil, litter, and coarse woody debris (CWD) for each macroplot ($n = 12$). Separate plant species lists were made for each 0.1 ha macroplot and each set of 12 subplots. Specimens of plants that could not be identified in the field were collected and compared to herbarium specimens. With a few exceptions, all plants were recorded to species level whenever possible. Some species were lumped into genera, e.g. *Chenopodium*, *Ribes*, and *Carex*, due to lack of consistent identifying characteristics at the time of data collection. We used the USDA PLANTS database to group species by origin (Native versus Exotic) and for common names.

We calculated pre-harvest and post-harvest Exotic Vegetation Index (EVI) for each 0.1 ha macroplot by averaging relative exotic species richness from the macroplot and relative exotic species foliar cover. Relative exotic species richness for each macroplot was calculated by dividing the total number of exotic species by the total number of all species recorded in the macroplot. Relative exotic species cover for each macroplot was calculated by first summing individual exotic species cover values in each of the 12 subplots then dividing that value by the sum cover of all species in each subplot. The exotic relative cover of the 12 individual subplots was then averaged and used as an estimate of exotic relative cover for the macroplot. Because total cover for each subplot was estimated by summing individual cover estimates for each species occurring in the subplot, total cover could exceed 100 % because of overlapping canopies.

All trees > 30 cm tall and < 10 cm DBH were counted as saplings. We recorded the DBH for all trees ≥ 10 cm DBH and used these data to calculate Quadratic Mean Density (QMD) and Stand Density Index (SDI) for each macroplot (Reinke, 1933): QMD = $\sqrt{(\sum DBH^2)}$ and SDI = $= \mathrm{TPH}(\mathrm{QMD} / 25.4)^{1.605}$, where DBH is in centimeters, TPH is the number of trees per hectare with DBH values ≥ 10 centimeters and 1.605 is an exponent of Reineke's equation. Stand Density Index (SDI) values are presented as trees per hectare with a quadratic mean diameter of 25.5 cm (see VanderSchaaf, 2013).

All logged and unlogged plots were sampled immediately before harvest and again each subsequent year for 1, 2, 3, and 7-years post-harvest. Additional trees were harvested from two timber sales/cutting units (Goat and Lost Park) either at post-harvest year 4 or 5 (commercial logging contracts on a timber sale in the Black Hills National Forest expire at 5 years). Because of resource limitations in 2009 only logged plots were re-sampled 3 years post-harvest and 37 of the 46 plots established in 2007 were evaluated in 2009 (3 years post-harvest for these plots).

2.4. Data analysis

We investigate overall exotic vegetation responses (EVI) to harvest and time effects using a Before-After Control-Impact (BACI) analysis using a beta distribution and repeated measures on plots (Morrison et al., 2001). Repeated measures were modelled with a spatial exponential covariance function of time. The fixed effect of harvest had two-levels (unlogged ~ control, logged ~ impact). The fixed effect of time had four-levels (0, 1, 2, and 7 years post-harvest). Because no unlogged plots were evaluated at 3 years post-harvest, 3 years post-harvest was not included in this analysis. Time was nested within period of harvest (i.e. before ~ time 0 and after ~ times 1, 2, and 7). Denominator degrees of freedom were approximated using Kenward-Roger (Kenward and Roger, 1997). We assess model goodness-of-fit by ensuring the deviance was at or near 1. To control the type II error rate of multiple comparisons, Bonferroni's method was applied.

We used the difference between pre-harvest and post-harvest EVI as the response variable (abbreviated Diff-EVI) to construct variable importance plots using Random Forest (RF) algorithms available in R-statistical software (R Core Team, 2024). Importance plots help identify the relative strength of the response variable to a variety of predictor

variables that potentially indicate levels of disturbance, overstory and understory characteristics, and topographic features. Disturbance variables include harvest intensity (SDI harvested) and post-harvest cover values for bare soil, litter, and coarse woody debris (CWD). We also included the distance of each plot to the nearest timber sale road (roads used exclusively for harvest) and the distance to the nearest Forest Service numbered road as potential indicators of disturbance. We defined harvest intensity for one-year, 2-years, and 3-years post-harvest time periods as the difference between pre-harvest Stand Density Index values and one-year post-harvest Stand Density Index values. Because two timber sales, Goat and Lost Park, had at least one additional harvest period (at 4 or 5 years after pre-harvest sampling), harvest intensity for 7-years post-harvest was modified to include the additional harvest periods. The effects of any post-harvest overstory increases on exotic responses were evaluated by including post-harvest SDI for each post-harvest time period. Understory variables were macroplot and subplot native species richness and average subplot native species cover, shrub cover and Pre-harvest EVI. Topographic features included slope and TRASP (a transformed aspect value; Roberts and Cooper, 1989). The importance of each predictor variable was evaluated based on the percentage increase in the model's mean squared error as the values of each predictor variable were randomized across the samples (Pierce et al., 2012). Random Forest makes no distributional assumptions about the predictor or response variables. Although the approach is not suitable for hypothesis testing, it is a powerful tool for identifying and visualizing interactions among variables that are likely correlated (Cutler et al., 2007). We also created partial dependence plots using the difference between pre-harvest and post-harvest (Diff-EVI) as the response variable and the two most important predictor variables identified in the Importance Plots. Partial dependence plots provide a graphical illustration of the "average" trend of exotic responses to a predictor variable after factoring out all other predictor variables in the model (R Core Team, 2024). Individual partial dependence plots were created for the two most important predictor variables identified by the Importance Plots for each post-harvest time period. Resulting plots were individually digitized using WebPlotDigitizer (Rohagi, 2022) to create multiple year graphs for each of the two most important predictor variables.

We used a Conditional Inference Tree (ctree) model in Random Forest (R Core Team, 2024) to regress the macroplot level difference between pre-harvest EVI and the 7-years post-harvest EVI as the response variable. We used 7-years post-harvest because it likely expressed the most long-term effects of logging. Predictor variables were the same as those used to develop the importance plots. Conditional Inference Tree models are applicable to various types of regression problems (e.g., nominal, ordinal, numeric, etc.) and involve recursively partitioning data within a conditional inference framework in sequential steps (Hothorn et al., 2006). The first step tests the global null hypothesis of independence between the input variables and the response variable, and the procedure stops if the hypothesis is not rejected using Monte Carlo procedures ($p \leq 0.05$). However, if the hypothesis is rejected, then the algorithm selects the input variable with strongest association with the response variable and calculates a corresponding p-value. The second step involves a binary split on the selected input variable. Step three involves repeating Steps 1 and 2 until the model stops when $p > 0.05$ (Hothorn et al., 2006). Potential linear relationship between the difference between pre- and post-harvest EVI and harvest intensity identified by the conditional inference tree analysis were further examined using best-fit regression that maximized coefficients of determination. We then explored specific patterns in the pre-harvest and 7-year post-harvest understory composition and abundance using indicator species analysis (Dufrane and Legendre, 1997) using PC-ORD Version 7.0 (MjM Software, Glenden Beach, Oregon, USA). We used the nodes identified by the conditional inference tree analysis as the grouping variable. Mean foliar cover for each species within a macroplot was used to estimate the within-node mean abundance of each species. Relative abundance was then calculated by dividing mean abundance for each species within a node by the sum of node means. The frequency of occurrence for each species within each node, based on at least one occurrence in the 12 microplots per macroplot, was used as a measure of species constancy in each node. An Importance Value (IV) for each species for each of the four nodes was then calculated by multiplying relative abundance and relative frequency. Importance Values range from zero (no indication) to 100 (perfect indication) and both relative abundance and frequency must be high for the IV to be high. The statistical significance ($p \leq 0.05$) of the highest IV (IVmax) for a given species across nodes was evaluated using Monte Carlo simulation (5000 randomized runs). Species with significant IV max values ≥ 25 were then selected to serve as Indicator Species for pre- and post-harvest within that node.

3. Results

3.1. General responses

Pre-harvest sampling was conducted in 2007, 2008, 2010, and 2011. Collectively, over that time frame, a total of 278 species of vascular plants were recorded within the understory layer in the 96 logged and 42 unlogged plots. This included 23 species of native trees and shrubs (no exotic trees or shrubs species were found). Sixty-eight species were recorded in the logged plots that were not recorded in the unlogged plots of which 62 occurred in less than 5 % of the logged plots. Twenty-three species were recorded in the unlogged plots that were not found in the logged plots. These 23 species occurred in less than 5 % of the unlogged plots. About 80 % of the 255 herbaceous species recorded in the logged macroplots were recorded in the logged subplots.

At pre-harvest, exotic species comprised about 15 % (39 species) of the 255 herbaceous species recorded in logged and unlogged macroplots (Appendix Table B). Exotic species were absent in 3 of the 96 logged macroplots and 5 of the 42 unlogged plots at pre-harvest. Twenty-two of the 39 exotic species were recorded in both logged and unlogged macroplots. Nine of the 39 exotic species recorded in the 0.1 ha plots were not recorded in the series of 1 m^2 subplots. Three of the 39 species were found only during pre-harvest sampling. Mean macroplot richness and mean total cover of exotic species in the subplots, as a functional group ($\pm$ 1 SD) in the 96 logged plots (42 unlogged plots), was 5.1 $\pm$ 3.1 (4.6 $\pm$ 3.1) and 8.3 $\pm$ 14.7 % (13.5 $\pm$ 23.8 %), respectively. In comparison, macroplot native species richness for logged (unlogged plots) averaged ($\pm$ 1 SD) 30.7 $\pm$ 10.3 (29.5 $\pm$ 10.5) species per plot and native species total cover averaged 38.6 $\pm$ 25.2 % (42.1 $\pm$ 29.3).

Tragopogon dubius was the most common exotic species in the

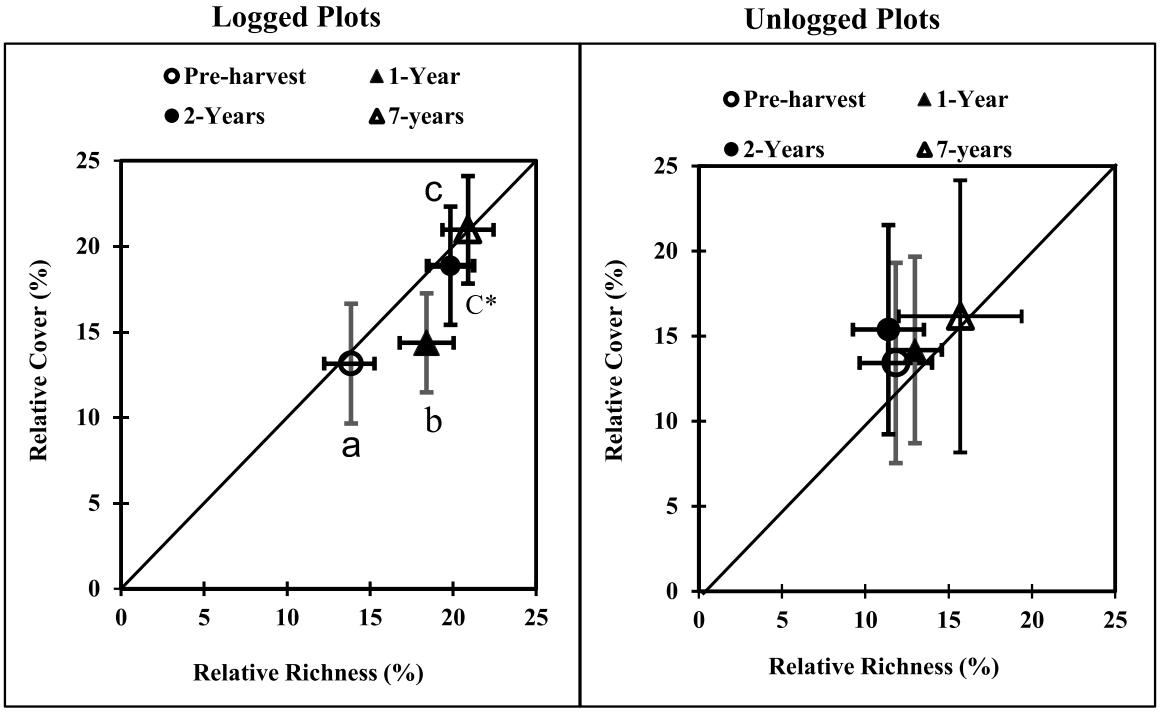

Fig. 1. Mean exotic species relative cover and relative richness for logged and unlogged plots at pre-harvest and 1-, 2-, and 7-years post-harvest (horizontal and vertical bares are 95 % confidence intervals). Mean Exotic VI values for logged plots followed by different letters are significantly different ($p < 0.05$) and letters highlighted by * indicate significant differences ($p < 0.05$) in Exotic VI values between logged and logged plots for that time period.

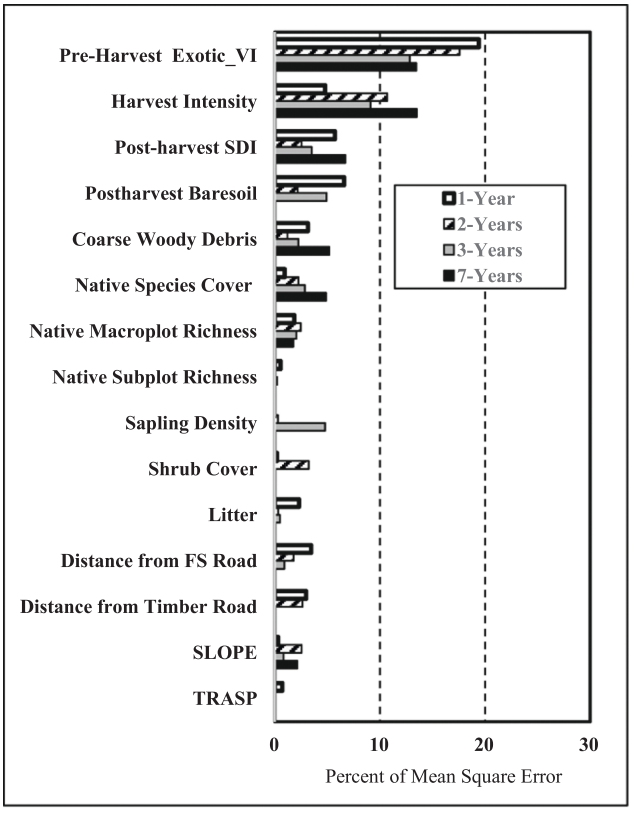

Fig. 2. Variable Importance Plot for 15 predictor variables in the Random Forest (RF) classification models that used the difference in Exotic VI between pre-harvest and 1-, 2-, 3-, and 7-years post-harvest as the response variable. Negative importance values are shown as zero.

annual/biennial group at pre-harvest occurring in 27 % of the logged plots. *Poa pratensis, Taraxacum officinale,* and *Trifolium spp.* were perennial exotic species recorded in 83 %, 60 %, and 49 % of the pre-harvest logged plots, respectively. *Cirsium arvense* and *Cynoglossum officinale* were the most common species listed as noxious weeds for the Black Hills occurring in 37 % and 38 % of the logged plots, respectively, during pre-harvest sampling.

At pre-harvest, mean SDI (± 1 SD) in logged plots was 416 trees (± 1.6) (Appendix Table A). At one-year post-harvest mean SDI was 163 (± 103.3) resulting in average SDI harvest intensity of 265 (± 153). Following harvest, an additional 24 species of herbaceous exotic plants were recorded in post-harvest logged macoplots (10, 6, 4, and 4 species at 1-year, 2-, 3-, and 7-years post-harvest, respectively) bringing the total number of exotic species recorded throughout the 10-year study to 63. One species, *Hieracium caespitosum*, was a new record for the Black Hills (Dickerson et al., 2016) and was recorded in one plot at 7-years post-harvest. Fourteen of the 63 exotic species that occurred in the 0.1-ha plots were not recorded in the 12 1-m^2 subplots.

At the plot level, pre- and post-harvest exotic species relative cover values were generally more variable than relative richness across the entire study period (Fig. 1). However, with the exception of values recorded at one-year post-harvest, relative cover and richness made equal contributions to Exotic VI at the landscape scale throughout the study. At one-year post-harvest, relative exotic richness was higher than relative cover, which represented a significant post-harvest increase from pre-harvest EVI ($p < 0.05$). Mean EVI increased significantly from 0.13 at pre-harvest to 0.19 at 2-years post-harvest ($p < 0.05$), which was significantly different from unlogged plots that increased from a value 0.12 at pre-harvest to a mean value of 0.14 during the same time frame. Mean EVI increased 1.2 % in the five-year time frame between 2-years post-harvest and 7-years post-harvest.

3.2. Specific responses

Importance plots show that pre-harvest EVI and harvest intensity

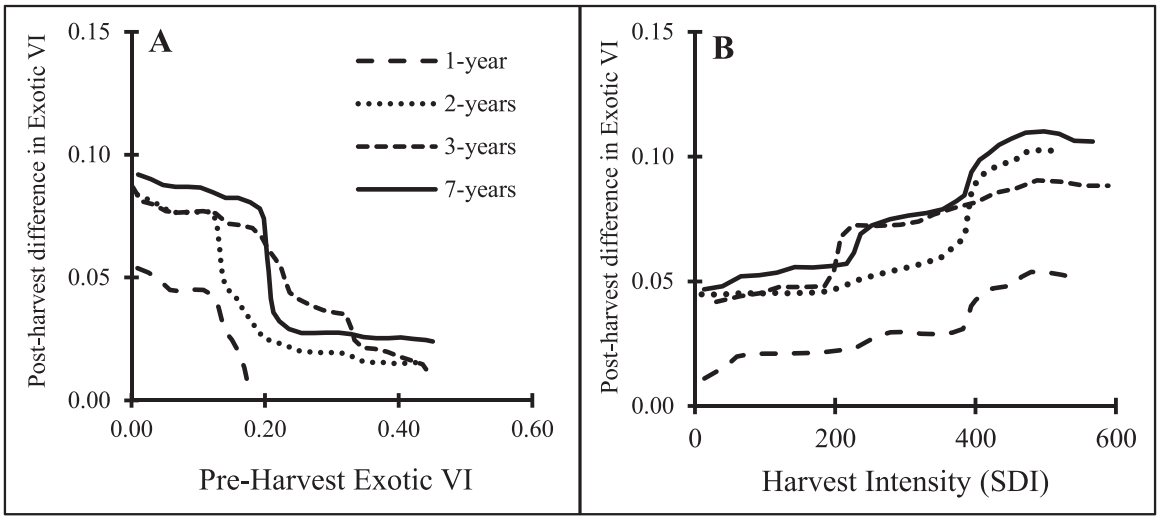

Fig. 3. Partial Dependence Plots for selected post-harvest predictor variables used in the Importance Plots (A. Pre-harvest Exotic VI and B. Harvest Intensity). Partial dependence is the dependence of the difference between pre- and post-harvest Exotic VI values (expressed as a decimal) after averaging out the effects of the remaining predictor variables used in the model.

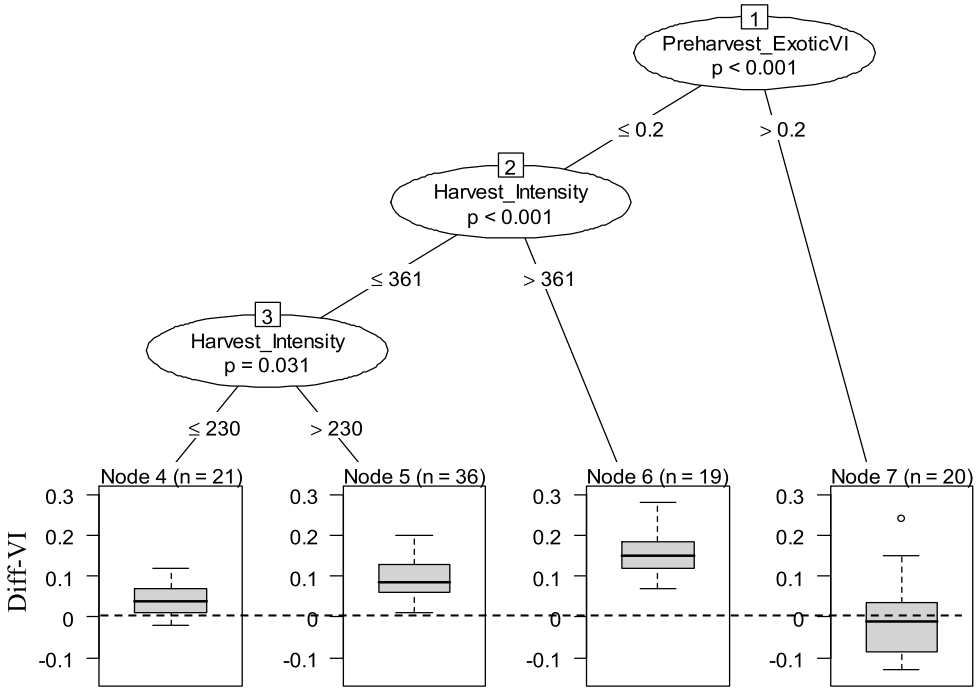

Fig. 4. Conditional inference tree (ctree) to test the hypothesis of independence between the difference in Exotic VI values between pre-harvest and 7-years post-harvest (Diff-VI, expressed as decimals) as the response variable and the predictor variables used in developing the importance plots in Fig. 2. n = the number of plots classified into each node. Dashed horizontal line represents no difference in Exotic VI between pre-harvest and 7 Years post-harvest.

were the most consistent predictors of responses in Diff-EVI values across the four sampling periods (Fig. 2). Pre-harvest EVI values had their strongest influence on the response variable (Diff-EVI) during the first post-harvest sampling period (one-year post-harvest), which decreased somewhat in years 2 and 3 then increased at 7-years post-harvest. The influence of harvest intensity on exotic species' responses was lowest the first year following harvest and highest at 7-years post-harvest. The increased influence in harvest intensity at 7-years post-harvest may reflect the re-entry harvests in the Goat and Lost Park timber sales in post-harvest periods 4-year or 5-year. The influence of post-harvest ponderosa pine sapling density (sapling = individual trees $>$ 30 cm in height and $<$ 10 cm DBH) was greatest three years post-harvest while the influence of post-harvest ponderosa pine SDI (trees $\geq$ 10 cm DBH) was highest at 7-years post-harvest. The amount of bare soil was highest the first year following harvest and was an important predictor of exotic species' responses the first two years following harvest. Field notes suggest that skid trails were the main factors influencing the amount of bare soil recorded at one-year post-harvest.

After factoring out the other 13 predictor variables, partial dependence plots (Fig. 3) were created using pre-harvest EVI and harvest intensity as the major predictor variables and Diff-EVI as the response variable. The observed responses further illustrate the temporal dynamics of exotic species in relation to pre-harvest EVI and harvest intensity. Pre-harvest EVI had its largest influence on Diff-VI the first year

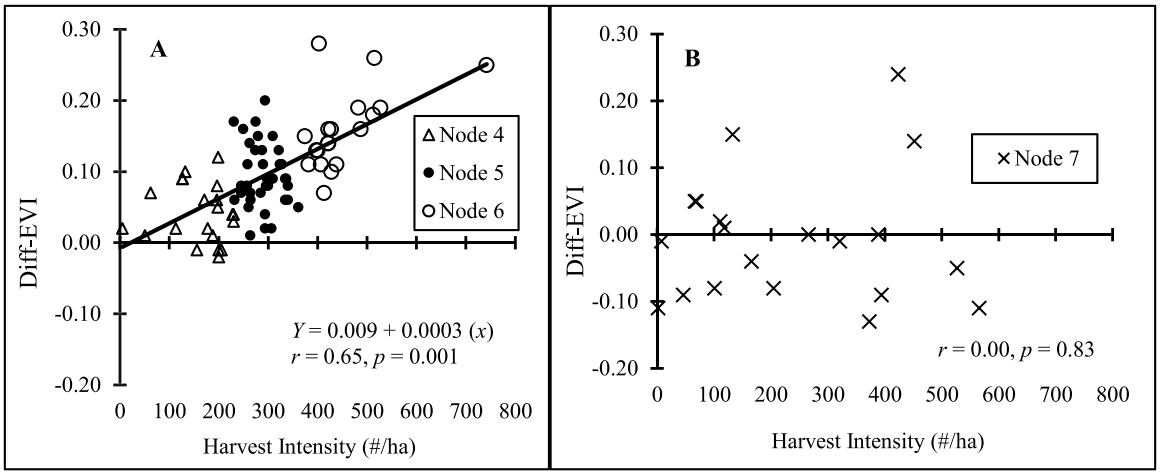

Fig. 5. Relationship between the difference in pre- and 7-years post-harvest exotic vegetation index (Diff-EVI) and harvest intensity (harvest intensity = difference between pre- and 7-years post-harvest stand density index) for Nodes 5, 5, and 6 (A) and Nodes 7 (B) that were identified in the Conditional Inference Tree analysis (Fig. 4).

Table 1
Number of plots in each classification and regression tree node at 7-years post-harvest.

	(Center text) Classification and Regression Tree Nodes			
Timber Sale	Node 4	Node 5	Node 6	Node 7
Dark Canyon	3	5	4	
Elk Hay	1	2	2	9
Fanny	5	4		2
Goat		2	3	1
Jewel	2	4	2	3
Lost Park	1	4	3	
Mercedes	3	6	1	
Power Pole	3	5		1
Scott	2	2	2	2
Thrall	1	2	2	2

Table 2
Results of Indicator Species Analysis conducted separately by preharvest and 7 years post-harvest using the four nodes identified by conditional inference tree analysis as the classification variable. Species highlighted in bold are exotic species for the Black Hills and species followed by a pound sign (#) are listed as noxious weeds for the Black Hills. (Pre-harvest IV/7 years post-harvest IV, n = number of macroplots for each node, * = significant Indicator Species value $p < 0.05$).

Species	Node 4 (n = 21)	Node 5 (n = 36)	Node 6 (n = 19)	Node 7 (n = 20)
Amelancher alnifolia	38 * /31	16/14	1/4	4/3
Apocynum androsamifolium	39 * /30	9/10	6/24	3/1
Astragalus flexuosus	26 * /12	2/6	0/2	1/0
Campanula rotundifolia	61 * /10	8/36	6/22	9/4
Symphotricuhum laevis	31 * /25	16/8	1/4	7/3
Cirsium arvense#	**0/3**	**0/14**	**0/37** *	**24/14**
Lactuca serriola	**0/0**	**0/1**	**0/32** *	**0/0**
Verbascum thapsus#	**5/1**	**0/2**	**0/37** *	**0/0**
Achilliea millefolium	23/16	18/21	7/15	31 * /37 *
Antennaria neglecta	6/2	5/6	14/4	22/25 *
Danthonia spicata.	17/15	11/16	14/32 *	38 * /26
Cynoglossum officinale#	**0/0**	**1/6**	**0/17**	**56 * /30 ***
Leucanthemum vulgare#	**0/0**	**0 /0**	**0/0**	**25 * /25 ***
Phleum pratense	**1/2**	**1/3**	**0/3**	**61 * /64 ***
Poa pratensis	**11/8**	**6/10**	**2/12**	**64 * /57 ***
Trifolium spp[a]	**1/3**	**1/4**	**1/17**	**69 * /30 ***
Taraxacum officinale	**5/4**	**6/19**	**3/13**	**40 * /25**

[a] Includes T. hybridum, T. pratense, and T. repens.

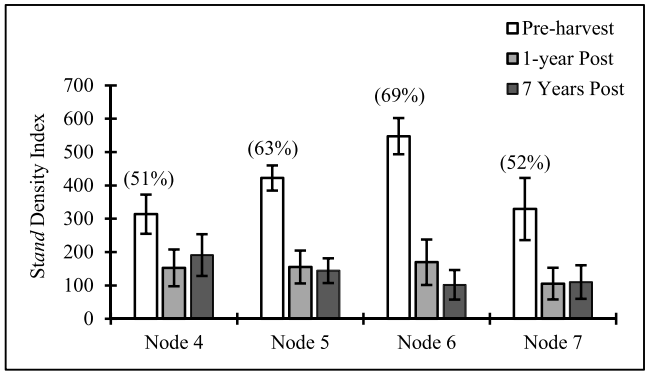

Fig. 6. Mean Stand Density Index values recorded pre-harvest, 1-year post-harvest, and 7-years post-harvest for the four nodes identified in the conditional inference tree model (± 95 % confidence intervals). Numbers in parenthesis are percent harvest at 1-year post-harvest.

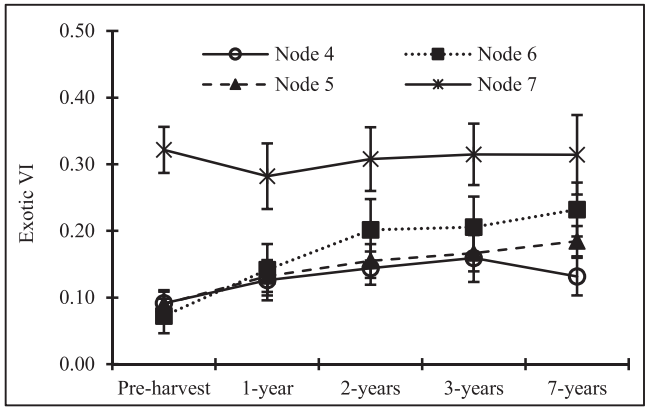

Fig. 7. Mean Exotic Vegetation Index (± 95 % confidence intervals for the four nodes identified by the ctree analysis at pre-harvest and at 1-, 2-, 3-, and 7-years post-harvest.

following harvest in plots with pre-harvest EVI values greater than 20 % showing the least change in the response variable (Fig. 3A). Fig. 3B illustrates somewhat linear post-harvest increases in Diff-EVI with increasing harvest intensity in all post-harvest time periods, with one-year post harvest plots showing the smallest level of increases.

The observed patterns were further explored with conditional inference trees using the same predictor variables used to develop the importance plots for all post-harvest time periods. The results for the 7-years post-harvest period are presented (Fig. 4) because this time period likely reflects more long-term responses of the understory. In the 20 plots with pre-harvest EVI values greater than 20 % (range = 22–45 %), 7-year post-harvest exotic responses were highly variable (Node 7). However, Fig. 4 shows that the average difference between pre-harvest and 7-year post-harvest EVI is close to zero. In plots with pre-harvest EVI values $\leq$ 20 %, plots were classified into three nodes (Nodes 3, 4 and 5) characterized by increasing Diff-EVI with increasing harvest intensity. Regression analysis showed a significant

relationship ($p < 0.001$) of increasing post-harvest Diff-EVI values with increasing harvest intensity for plots in nodes 4, 5, and 6 (Fig. 5A) that was not found for plots classified into Node 7 (Fig. 5B). Some plots in Nodes 4 and 7 that show negative Diff-EVI values indicate a reduction in exotic species composition at 7-years post-harvest below pre-harvest values. With the possible exception of the Elk Hay timber sale, individual plots with different pre-harvest EVI values and under different harvest intensities were widely scattered among the 4 nodes (Table 1). All of the timber sales had at least 2 plots in Node 5. Four timber sales had plots in all four nodes and all timber sales had at least one representative plot in 3 nodes.

At total of 17 species met the criteria established to serve as indicator species in at least one node either at pre-harvest or at 7-years post-harvest (Table 2). Nine of the 17 Indicator Species (IS) are described as exotic species for the Black Hills with three species (*Cynoglossum officinale, Cirsium arvense, and Verbascum thapsus*) listed as noxious weeds for the area. Only one shrub species (*Amelancher alnifolia*) was identified as an Indicator Species. Plots in Node 7 were the only set of plots that had Indicator Species that included both native and exotic species for both pre-harvest and 7-years post-harvest. Patterns in pre-harvest and 7-years post- harvest Indicator Species among nodes 4, 5, and 6 may be related to pre-harvest SDI and harvest intensities among the 3 nodes (Fig. 6). The difference between one-year post-harvest

SDI and 7-year post-harvest SDI for plots in Node 6 likely reflects the harvest reentries that occurred in post-harvest periods year 4 or year 5 in the Goat and Lost Park timber sales. The difference in one-year post-harvest and 7-year post-harvest SDI values shown for Node 4 suggests some ponderosa pine regeneration over the 7-year post-harvest time period. Although IS analysis suggests considerable differences in pre-harvest composition among nodes 4, 5, and 6, Exotic VI values were similar among the three nodes at pre-harvest and at one-year post harvest (Fig. 7). Post-harvest Exotic VI values remained similar among plots in these nodes until 7-year post-harvest when mean Exotic VI decreased between 3-year post-harvest and 7-years post-harvest for plots in Node 4. Exotic VI values within plots in Node 7 averaged 0.28–0.32 throughout the study.

4. Discussion

In this study, the pre-harvest Exotic Vegetation Index (EVI) values served as a valuable reference point in evaluating exotic species responses to large-scale commercial logging. The long disturbance history of the Black Hills likely played a large role in the development of the pre-harvest exotic plant communities examined in this study. The total number of exotic species we recorded at pre-harvest is about twice the number of exotic species reported by Abella et al. (2012) for a variety of classified ecosystems within a ponderosa pine landscape in northern Arizona. However, exotic species composition reported in Arizona is similar to what we report in the Black Hills National Forest. We recorded six of the seven grass species and nine of the 13 forb species (two additional species were matched by their respective genus) reported by Abella et al. (2012). The similarity in composition between the two forest systems illustrates the wide geographic distribution of several common exotic species found in ponderosa pine forests. These geographically wide-spread species appear to have overcome the various transport, colonization, establishment, and landscape level spread hurdles generally associated with exotic species expansion into new sites (Theohardes and Dukes, 2007).

Variations in historic and current land uses in the Black Hills, coupled with temporal and spatial precipitation patterns and differences in plot-level site characteristics across the widely distributed timber sales, likely contributed to a diverse pre-harvest understory. To help dampen the variability across our study sites and across sampling periods, we used the average of relative species richness and cover as an index of exotic species contributions to the understory plant community (Guo and Symstad, 2007). In our study, relative exotic richness and cover both averaged 13 % in pre-harvest evaluations. At one-year post-harvest, mean relative exotic richness increased to 17 % while mean relative cover remained at 13 %. However, beginning in the second post-harvest year, mean exotic relative richness and relative cover tended to increase in tandem in later post-harvest evaluations. The potential for a time-lag effect in cover at one-year post-harvest was expected because of the more rapid increases in richness by colonization relative to the slower responses of detectable increases in cover (Theohardes and Dukes, 2007). Dodson and Fiedler (2006), using sampling methods similar to our study, evaluated exotic species richness and cover responses to restoration treatments in ponderosa pine forests in Montana. They reported significant increases in species richness relative to control plots the first year following treatment application, while significant increases in exotic cover were not found until the second post-treatment year. In the northern Black Hills, Wienk et al. (2004) also observed a one-year lag before detecting significant increases in understory biomass during the second year following various overstory reduction treatments combined with burning.

Guo and Symstad (2007) presented broad evidence to describe an inflection point for relative exotic species richness of 15 % where increases in relative richness beyond this point increases the probability that exotic species will become a larger portion of total cover or biomass. Once exotics are introduced following disturbance related increases in resources (Davis et al., 2000), high exotic propagule pressure (Sutherland and Nelson, 2010) may make a local site more susceptible to further invasion through the combined competitive impacts of exotic species on native species (Daehler 2003; Vila and Weiner, 2004; Maron and Marler, 2008). In this context, exotic species, as a functional group, may be self-facilitating their expansion that was initiated by the disturbances associated with commercial logging. However, the small landscape level change in relative exotic species richness and cover between 2-years post-harvest and 7-years post-harvest suggest limits to the post-harvest co-occurring pulse of elevated propagule pressure and resource enrichment (Davis et al., 2000) that was initiated by commercial logging.

Results of our study are generally consistent with the considerable

synthesized evidence that show positive correlations between increases in exotic species composition and disturbance intensity in a variety of forested ecosystems (Sutherland and Nelson, 2010; Duguid and Ashton, 2013; Abella and Springer, 2015; Willms et al., 2017). Our experimental approach adds to the information base by using large-scale commercial logging as a tool to investigate understory responses to a broad the range of thinning intensities on a relatively large spatial scale. A key result of our analysis is the separation of the 96 logged plots into two groups based on a pre-harvest mean EVI value of 20 %. In the 76 plots with pre-harvest EVI values $\leq$ 20 % we found a significant positive relationship between Diff-EVI and harvest intensity with an intercept close to zero. While the scatterplot shows a continuum of Diff-EVI responses to harvest intensity (Fig. 5A), with no obvious breaks associated with the three identified nodes (Fig. 4), there may be several underlying factors in addition to harvest intensity that influenced post-harvest response patterns in this set of plots. For example, within the group of 21 plots with the lowest SDI harvest intensities ($\leq$ 230) mean Exotic VI increased the first three years following harvest, but then decreased between 3-years post-harvest and 7-years post-harvest (Fig. 7). The observed decreases in exotics in this set of plots following initial colonization may be related to the recorded increases in ponderosa pine stand densities between one-year and 7-years post-harvest. Relatively rapid tree growth is possible in the Black Hills because of the favorable climatic conditions for ponderosa pine (Boldt and Van Deusen, 1974; Hoffman and Alexander, 1987; Shepperd and Battaglia, 2002). Reduced competition resulting from tree harvesting coupled with favorable site and climatic conditions has the potential of moving un-harvested saplings into the tree layer rather quickly. In these plots, low harvest intensity combined with a diverse pre-harvest native understory, as identified by Indicator Species Analysis (Table 2), may have limited the initial establishment of exotic species. Additionally, the relatively rapid post-harvest increase in stand density may have prevented some exotic species from developing self-sustaining, expanding populations. Several plots in this group of plots had lower Diff-EVI values at 7-years post-harvest compared to pre-harvest (i.e. mean Diff-EVI values at 7-years post-harvest were negative).

Within the 36 plots with moderate levels of harvest intensity (stand density harvest $>$ 230 and $\leq$ 361), mean responses of exotics were initially identical to plots with lower harvest intensities for the first three post-harvest time periods. However, while exotic composition under relatively low harvest intensity decreased between 3-years and 7-years post-harvest, mean exotic richness and cover values under comparably higher harvest intensities continued to increase beyond the 3-year post-harvest time period. This pattern is likely a reflection of exotic species expansion under the combined conditions of higher harvest intensities, limited increase in post-harvest stand density, and a diverse pre- and post-harvest herbaceous understory with no significant dominant native or exotic species (Table 2).

In plots with the highest harvest intensities (stand density greater than 361, 19 plots), there was a sharp and complex shift in the 7-years post-harvest understory composition that included increases in a wide variety of exotic species commonly associated with heavily disturbed habitats (*Cirsium arvense, Cynoglossum officinaale, Verbascum thapsus*, and *Lactuca serriola*) (Table 2). This pattern was probably exacerbated by harvest re-entries in two timber sales that occurred in 4- or 5-years post- harvest time periods. At the same time, however, the native grass *Danthonia spicata* along with several native forbs, were also more prevalent at 7-years post-harvest compared to pre-harvest. This complex post-harvest understory of mixed origin species within this group of plots may have played an interactive role in preventing the more aggressive exotic species from dominating the post-harvest understory in some plots. Bernard-Verdier and Hulme (2015), in a landscape-level study in New Zealand, investigated how native and exotic species abundance and richness varied across a broad range of focal native and exotic species. They found that the abundance of native and exotic focal species had contrasting effects on species richness. In general, high abundance of a focal native species was often associated with increasing richness of native species and decreasing richness of exotic species. They observed the opposite pattern for exotic species where the abundance of a single exotic species was associated with lower native species richness but higher exotic species richness. The contrasting patterns in native and exotic species described by Bernard-Verdier and Hulme (2015) may be especially pertinent for the 20 plots with a pre-harvest EVI greater than 20 %. In this set of plots, we found considerable variation in Diff-EVI responses to harvest intensity that ranged from −13% to24 %, with an average of zero. Although exotic species comprised a large portion of the pre-harvest understory vegetation in these plots, indicator species analysis showed that there was no post-harvest change in mean post-harvest indicator values for three of the seven exotic species in this set of plots (*Leucanthemum vulgare*, *Phleum pratense*, and *Poa pratensis*) with four of the seven species showing post-harvest decreases in indicator values (*Cirsium arvense*, *Cynoglossum officinale*, *Trifolium* spp., and *Taraxacum officinale*) (Table 2). A somewhat similar pattern was observed for a small group of native species (*Achilliea millefolium*, *Antenaria neglecta*, and *Danthonia spicata*). Bernard-Verdier and Hulme (2015) discussed that, on a broad scale, contrasting patterns in native and exotic species abundance may essentially cancel each other out and often indicated no difference in exotic and native species abundance when averaged. Detailed analysis of native and exotic species composition at smaller spatial may provide valuable insight into contrasting native and exotic species associations in mixed-origin plant communities (Smith and Côté, 2019).

5. Conclusions

Commercial logging is a viable industry in the Black Hills (on both private and public lands) that offers a cost-effective approach to reducing the potential for wildfires and insect outbreaks in the face of high tree regeneration and rapid growth rates. Exotic plant expansion is generally a predictable side-effect of both small- and large-scale thinning efforts. Our study shows that predictions of understory responses following mechanical thinning can be improved by large spatial and temporal scale studies that include a wide range of harvest intensities implemented in complex plant communities of mixed-origin. Results of this study show that post-harvest responses of exotic plants in the Black Hills National Forest were largely driven by two independent factors: (1) the relative contribution of the pre-harvest exotic plant community; and (2) harvest intensity. Collectively, our results suggest three potential management thresholds. One threshold may exist based on a highly variable understory plant community identified as having a pre-harvest exotic species composition greater than 20 %. While the relative post-harvest contribution of exotic species to the understory increased in some plots and decreased in others relative to pre-harvest values, the average post-harvest change in exotics in this set of plots was essentially

zero at 7-years post-harvest. Further investigating the contrasting responses of understory communities of mixed origin to harvest would likely require examining species to species interactions at smaller spatial scales than presented in this paper.

In the remaining larger set of plots, post-harvest changes in the relative contribution of exotics were strongly correlated with harvest intensity. A relatively low harvest intensity threshold may exist (reduction in SDI $\leq$ 230 in this study) where overall disturbance was low, which likely reduced the potential for exotics to colonize. In addition, exotic species that did colonize some plots may not have persisted to form self-perpetuating populations because of a well-developed native understory and the relatively rapid tree growth observed in saplings and unharvested trees by the 7-years post-harvest time period. A second threshold may exist at high harvest intensities ($>$ 361 reduction in SDI) that increased the probability of aggressive exotic species, which included several species listed as noxious weeds for the Black Hills, becoming established and persisting. Between low-intensity and high-intensity harvest levels identified in this study is a third group of plots that were characterized by highly variable understory responses that involved both native and exotic species. While the post-harvest plant community in this group of plots did include two aggressive noxious weeds, understory competition may have limited these species to a relatively small component of understory. Exploring species to species interactions at smaller spatial scales may provide valuable insight into how understories of mixed origin respond to what may be classified as a moderate level of harvest intensity.

While commercial logging may be a viable cost-effective tool in reducing tree densities on a large spatial scale, our results show that, depending on harvest intensity, some increase in exotic species composition in the understory is a fairly predictable outcome. Further, a complete reliance on logging in historical-logged forests where fire is excluded may have future unforeseen consequences. Naficy et al. (2010) reported that historical logged and fire-excluded ponderosa pine forests in the northern Rocky Mountains had higher stand densities of live and dead trees and increased dominance of fire-intolerant trees compared to unlogged and fire-excluded forests. Similar increases in ponderosa pine tree density have been observed in the Black Hills (Brown and Cook, 2006), however, the increased dominance of fire-intolerant trees is not an issue. Nevertheless, the potential that the higher stand densities in a historically logged forest may be at a higher risk for high-intensity wildfires compared to forests without historical logging (Naficy et al., 2010) since tree diameters are often smaller and surface fuels are increased. The removal of fire from the landscape has demonstrated that the ponderosa pine forests in the Black Hills have become increasingly dependent on large-scale thinning to reduce the risk of wildfire and insect outbreaks with the chronic side-effect of possibly increasing opportunities for exotic plants. Further studies are needed to address the impact that mechanical thinning followed by prescribed fire will have on exotic plants.

CRediT authorship contribution statement

Jack L. Butler: Project administration, Conceptualization, Supervision, Investigation, Funding acquisition, Methodology, Data collection, Data curation, Formal analysis, Writing – original draft, Writing – review and editing. **Stefanie D. Wacker:** Conceptualization, Investigation, Methodology, Data collection, Data curation, Writing – original draft, Writing – review and editing. **Jacqueline P. Ott:** Investigation, Data collection, Formal analysis, Writing – review and editing. **Scott Baggett:** Formal analysis, Writing – review and editing. **Michael A. Battaglia:** Formal analysis, Writing – review and editing.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix Table A.

Select characteristics for 10 timber sale cutting units in the Black Hills National Forest, South Dakota used to evaluate exotic plant response following commercial timber harvest. Number of logged (unlogged) plots is the number of plots established at pre-harvest. SDI is Stand Density Index (number of trees $^{-0.1}$ ha @ 25.5 cm dbh). Previous harvest year is for timber sale area

Timber Sale (unit)	Harvest Year[1]	Previous Harvest Year	Year of last data collection	Area (ha)	Number of Logged (Unlogged) Plots	Pre-harvest Mean SDI (#/$^{-1}$ ha $\pm$ SD)	1 Year Post-harvest Mean SDI (#/$^{-1}$ ha $\pm$ SD)
Dark Canyon (9)	2007	1981–1987	2014	64	12 (7)	349 $\pm$ 116.2	54 $\pm$ 43.3
Elk Hay (8, 10, 11)	2010	Unk[2]	2017	34	14 (0)	413 $\pm$ 91.5	101 $\pm$ 94.2
Fanny (7)	2008	1973	2016	23	11 (6)	368 $\pm$ 152.6	158 $\pm$ 98.3
Goat (11)	2008*	1998	2016	51	6 (0)	585 $\pm$ 180.2	121 $\pm$ 124.0
Jewell (41)	2010	Unk[2]	2017	71	11 (3)	376 $\pm$ 142.8	116 $\pm$ 64.4
Lost Park (22)	2007*	1985	2014	30	8 (2)	514 $\pm$ 132.5	178 $\pm$ 164.7
Mercedes (17)	2007	1987–1988	2014	40	10 (8)	320 $\pm$ 71.6	42 $\pm$ 30.2
Power Pole (2)	2007	Unk[2]	2017	37	9 (10)	316 $\pm$ 142.6	119 $\pm$ 84.7
Scott (20)	2011	2005	2017	31	8 (0)	515 $\pm$ 180.0	250 $\pm$ 85.3
Thrall (4)	2007	1994	2014	70	7 (6)	409 $\pm$ 207.7	130 $\pm$ 85.8
				Total = 451 ha	**Total = 96 (42) plots**	**Mean = 416 $\pm$ 91.6**	**Mean = 163 $\pm$ 103.3**

* Because additional harvest re-entries occurred in either post-harvest year 4 or 5 for Goat (4 plots) and Lost Park (3 plots) post-harvest SDI values are given for 7-years post-harvest for these two sites.

[1] Permanent plots were established and pre-harvest data were collected during the harvest year.

[2] Unk = Unknown

Appendix Table B.

Mean macroplot frequency and mean foliar cover (n = 10 timber sales) recorded in logged and unlogged (in parentheses) timber sales in the Black Hills National Forest (Pre- = Pre-harvest, Time 1 = 1 Year Post-harvest, Time 2 = 2 Years Post-harvest, Time 3 = 3 Years Post-harvest, Time 7 = 7 Years Post-harvest). Unlogged plots were not sampled 3 years post-harvest. T = mean foliar cover < 1 %. *Species that were not recorded in microplots at any time period. Species highlighted in bold are on the invasive species list for the Black Hills National Forest. ^Range expansion (Dickerson et al., 2017). Species sorted by lifespan and macroplot frequency

	Logged Macroplot Frequency (Unlogged)					Logged Percent Foliar Cover (Unlogged)				
Annual/Biennial Species	**Pre-**	**Time 1**	**Time 2**	**Time 3**	**Time 7**	**Pre-**	**Time 1**	**Time 2**	**Time 3**	**Time 7**
Tragopogon dubius	27 (31)	38 (31)	53 (23)	55	65 (29)	T (T)	T (T)	T (T)	T	T (T)
Chenopodium album	8 (9)	21 (9)	23 (11)	20	22 (15)	T (T)	T (T)	T (T)	T	T (0)
Medicago lupulina	7 (13)	19 (13)	24 (13)	21	19 (17)	T (T)	T (T)	T (T)	T	T (T)
Lactuca serriola	0 (2)	30 (16)	22 (4)	20	23 (17)	0 (0)	T (T)	T (0)	T	T (0)
Logfia arvensis	**0 (0)**	21 (0)	25 (0)	25	18 (0)	0 (0)	T (0)	T (0)	T	T (0)
Bromus arvensis	3 (4)	6 (4)	10 (0)	8	8 (0)	T (T)	T (0)	T (0)	T	T (0)
Descurainia sophia	2 (7)	8 (5)	7 (0)	4	4 (0)	0 (T)	T (0)	T (0)	T	0 (0)
Polygonum convolvulus	T (0)	10 (2)	3 (0)	13	0 (0)	T (0)	T (0)	T (0)	T	0 (0)
Thlaspi arvense	T (1)	8 (1)	4 (3)	3	4 (3)	0 (0)	T (T)	T (T)	0	T (0)
Silene noctifloria	0 (0)	0 (0)	1 (0)	1	11 (12)	0 (0)	0 (0)	T (0)	0	T (T)
Capsella bursa-pastoris	1 (1)	3 (4)	4 (0)	3	1 (3)	0 (T)	T (0)	T (0)	T	T (0)
Bromus tectorum	**T (0)**	6 (0)	4 (0)	7	2 (0)	T (0)	T (0)	T (0)	T	T (0)
Camelina microcarpa	0 (2)	0 (2)	T (5)	T	3 (0)	T (T)	0 (0)	T (T)	0	T (0)
Buglossoides arvensis	**0 (0)**	0 (0)	5 (0)	4	0 (0)	0 (0)	0 (0)	T (0)	T	0 (0)
Matricaria discoidea	T (0)	4 (2)	1 (0)	1	0 (0)	0 (0)	T (0)	0 (0)	T	T (0)
Veronica persica	**0 (0)**	5 (0)	2 (0)	0	1 (0)	0 (0)	T (0)	0 (0)	0	T (0)
Sisymbrium altissimum	**0 (0)**	0 (0)	2 (0)	0	T (0)	0 (0)	0 (0)	T (0)	0	0 (0)
*S. loeselii**	0 (0)	0 (2)	0 (0)	0	T (0)					
*Chorispora tenella**	**0 (0)**	1 (0)	0 (0)	0	0 (0)					
Lolium temulentum	**0 (0)**	0 (0)	0 (0)	1	0 (0)	0 (0)	0 (0)	T (0)	T	0 (0)
*Tripleurospermum perforatum**	**0 (0)**	0 (0)	0 (0)	1	0 (0)					
Echinochloa crus-galli	**T (0)**	0 (0)	0 (0)	0	0 (0)	T (0)	0 (0)	0 (0)	0	0 (0)
*Hyoscyamus niger**	**0 (0)**	0 (0)	T (0)	0	0 (0)					

	Logged Macroplot Frequency (Unlogged)					Logged Foliar Cover (Unlogged)				
Biennial/Perennial Species	Pre-	Time 1	Time 2	Time 3	Time 7	Pre-	Time 1	Time 2	Time 3	Time 7
Poa pratensis	83 (81)	85 (82)	90 (74)	90	93 (75)	3 (5)	4 (5)	4 (5)	5	9 (3)
Taraxacum officinale	60 (42)	79 (71)	80 (56)	88	85 (61)	T (T)	T (1)	T (1)	T	T (2)
Trifolium spp.[1]	49 (52)	55 (52)	62 (58)	64	63 (31)	T (T)	T (T)	T (T)	T	T (T)
Cirsium arvense	**37 (43)**	**53 (40)**	**68 (35)**	**72**	**82 (48)**	**T (T)**	**T (1)**	**T (1)**	**T**	**3 (1)**
Cynoglossum officinale	**38 (37)**	**51 (32)**	**76 (30)**	**76**	**70 (52)**	**T (T)**	**T (T)**	**T (T)**	**T**	**T (T)**
Phleum pratense	46 (42)	48 (41)	48 (35)	52	61 (31)	1 (2)	1 (3)	1 (3)	2	3 (6)
Bromus inermis sub. inermis	24 (20)	39 (30)	34 (25)	45	44 (34)	T (T)	T (T)	T (T)	1	2 (T)
Verbascum thapsus	**8 (13)**	**35 (17)**	**35 (17)**	**31**	**43 (22)**	**T (T)**	**T (T)**	**T (T)**	**T**	**T (T)**
Cirsium vulgare	6 (0)	18 (0)	37 (2)	36	33 (0)	T (0)	T (0)	T (0)	T	T (0)
Agrostis stolonifera	16 (8)	11 (13)	17 (10)	6	20 (11)	T (T)	T (T)	T (T)	T	T (T)
Linaria vulgaris	**13 (8)**	**11 (5)**	**13 (6)**	**13**	**16 (6)**	**T (0)**	**T (T)**	**T (T)**	**T**	**T (T)**
Poa compressa	11 (0)	19 (1)	19 (11)	15	10 (0)	T (0)	T (T)	T (T)	T	T (0)
Thinopyrum intermedium	7 (4)	7 (9)	0 (0)	2	6 (6)	T (T)	T (T)	T (T)	T	T (T)
Elymus repens	10 (14)	T (1)	2 (3)	4	0 (0)	T (T)	T (T)	T (T)	T	0 (0)
Silene latifolia ssp. alba	0 (5)	8 (9)	0 (0)	4	4 (3)	0 (0)	T (0)	T (0)	0	0 (0)
Leucanthemum vulgare	**6 (0)**	**4 (0)**	**5 (0)**	**6**	**6 (0)**	**T (0)**	**T (0)**	**T (0)**	**T**	**T (0)**
Convolvulus arvensis	3 (0)	3 (0)	8 (2)	0	4 (3)	T (0)	0 (0)	0 (0)	T	T (0)
Dactylis glomerata	1 (0)	2 (0)	4 (1)	6	2 (3)	T (0)	T (0)	T (0)	T	T (0)
Melilotus officinalis	4 (2)	T (0)	8 (2)	0	2 (0)	T (0)	0 (0)	0 (0)	T	0 (0)
Carduus nutans	**0 (0)**	**2 (3)**	**2 (0)**	**3**	**5 (0)**	**0 (0)**	**T (0)**	**T (0)**	**0**	**T (0)**
Rumex acetosella	2 (0)	T (2)	2 (0)	3	3 (3)	T (0)	0 (0)	0 (0)	T	T (T)

	Logged Macroplot Frequency (Unlogged)					Logged Foliar Cover (Unlogged)				
Perennial Species	Pre-	Time 1	Time 2	Time 3	Time 7	Pre-	Time 1	Time 2	Time 3	Time 7
Silene vulgaris	0 (0)	0 (0)	0 (0)	1	2 (11)	0 (0)	0 (0)	0 (0)	0	0 (T)
Potentilla recta	**0 (0)**	**0 (7)**	**0 (0)**	**0**	**0 (0)**	**0 (0)**	**0 (T)**	**0 (0)**	**0**	**0 (0)**
Tanacetum vulgare	**0 (0)**	**0 (0)**	**5 (2)**	**T**	**T (0)**	**0 (0)**	**0 (0)**	**T (0)**	**0**	**0 (0)**
*Rumex crispus**	0 (0)	1 (2)	1 (2)	0	0 (0)					
*Schedonorus pratensis**	T (0)	0 (0)	T (0)	T	0 (6)					
*Lonicera tatarica**	0 (0)	3 (0)	2 (0)	0	0 (0)					
Cerastium fontanum	0 (0)	0 (0)	0 (0)	0	3 (0)	0 (0)	0 (0)	0 (0)	0	T (0)
Lolium perenne	0 (0)	1 (0)	2 (0)	0	0 (0)	0 (0)	T (0)	T (0)	T	0 (0)
Sonchus arvensis	0 (0)	3 (0)	0 (0)	0	0 (0)	0 (0)	T (0)	0 (0)	0	0 (0)
*Barbarea vulgaris**	0 (0)	0 (2)	0 (0)	0	0 (0)					
*Medicago sativa**	0 (0)	0 (0)	0 (0)	2	0 (0)					
Nepeta cataria	T (0)	0 (0)	2 (0)	T	0 (0)	0 (0)	0 (0)	T (0)	0	0 (0)
*Hesperis matronalis**	1 (0)	0 (0)	0 (0)	0	T (0)					
*Poa bulbosa**	0 (0)	0 (0)	1 (0)	0	0 (0)					
*Ranunculus acris**	1 (0)	0 (0)	0 (0)	0	0 (0)					
*Agropyron cristatum**	T (0)	0 (0)	0 (0)	0	0 (0)					
Hieracium caespitosum^*	0 (0)	0 (0)	0 (0)	0	T (0)					

[1]Includes T. hybridum, T. pratense, and T. repens.

Data availability

Data will be made available on request.

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