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Shifts in plant communities caused by vegetation clearing in power-line corridors

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M.Sc. Thesis

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Damilola Eyitayo

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Abstract

Habitat modification has been identified as one of the major threats to biodiversity and has increased over recent years due to growing human population and demand for resources. One omnipresent form of habitat modification is the creation of transmission lines by electricity distribution companies. Due to safety concerns, the vegetation beneath electricity transmission lines is maintained at young succession stages by frequent cutting.

Data was collected from 51 study sites in the south east Norway. All the sites were situated in forests intersected by a power-line corridor, and represent a range of boreal forest types. I focused on plant responses that may be of particular interest for managers, such as differences in plant community composition between the corridor and forest, total plant species richness and diversity, richness and relative abundance of some functional groups or single keystone species. Likewise, I focused on environmental factors that may be influenced by managers and planners.

Greater diversity of species was found to exist in the power-line corridors compared to the interior of the forest. Specifically, I found an increase in richness and relative abundance of forbs, shrubs, light tolerant species and species preferring nutrient rich soils. Although some other species such as bilberry and mosses decreased in the corridors. More importantly, it suggests that the corridors are able to resist plant invasions and support a greater diversity of cultural landscape species.

The site index was found to be highly significant in influencing the species composition of the corridor habitats. Total species richness and relative abundance of forbs, shrubs, and cultural landscape species increased with the site index, while the stage of succession and landscape heterogeneity also proved to be significant in shaping the species composition of the power-line corridors.

Synthesis and applications. This study confirms that less severe forms of disturbance such as the creation of power-line corridors can cause species diversity to increase and as such provide an opportunity to conserve biodiversity in the face of increasing human land use changes.

Key-words: power-line corridors, disturbance, functional groups, species richness, relative abundance, environmental variables

1.0 Introduction

On a global scale forests are increasingly subjected to human interference due to an increasing demand for natural resources and growing world population (Ellis 2011). Habitat modification caused by human land use changes have been implicated as the most important driver of biodiversity changes (Pimm et al. 1995). In addition to the effect of natural disturbances on plant communities (Turner 2010), human land use practices such as grazing activities, forestry and agricultural practices also influence the species composition and structure of an ecosystem (Chase 2003). Thus, there has been a growing interest among ecologists in understanding the nature of the effects of habitat modification on biodiversity and how conservation approaches could effectively cope with the growing threats of land use changes. One notable form of habitat modification which is yet to be fully studied is the construction of transmission lines by electricity distribution companies (Sala et al. 2000). In forested landscapes, power-line corridors are maintained to avoid contact between transmission lines and vegetation and are thus continually reset to earlier successional stages.

The boreal forest is a broad circumpolar biome comprising cool coniferous and deciduous tree species. It covers 14.7 million km^2 or 11% of the earth surface (Bonan & Shugart 1989), making it the world's largest terrestrial biome. In Norway, the boreal forests cover about 38% of the entire land surface (Nordic Family Forestry 2014). These forests serve as a source of timber, domestic animal grazing and game hunting. The forest vegetation also helps with carbon assimilation thereby reducing greenhouse effect and also has aesthetic value. The dominant plant species include: conifers such as Scots pine *Pinus sylvestris*, Norway spruce *Picea abies*, and dwarf shrubs like bilberry *Vaccinium myrtillus*, heather *Calluna vulgaris* and lingonberry *Vaccinium vitis-idaea* (Esseen et al. 1997). There is considerable spatial overlap between bilberry and heather, but their relative abundances vary. Parlane et al. (2006) found that the percent cover of bilberry was highest at intermediate irradiance, whereas heather cover increased linearly with increasing irradiance.

The creation of power-line corridors by frequent cutting to accommodate the traversing electric transmission lines represents a major ecological disturbance in the boreal forests of south eastern Norway. This is aptly described as an ecological disturbance because the observable differences between the corridor vegetation and adjoining forest vegetation fits the definition of disturbance provided by (Rykiel 1985). Rykiel (1985) defined a disturbance as 'a physical force or process, either abiotic or biotic, causing a perturbation (which includes stress) in an ecological component or system'. The power-line corridors are maintained through cycles of frequent cutting (Clarke & White 2008), usually alternating between 5 to 10 years (Berg et al. 2011; Russell et al. 2005). This form of human-induced disturbance can be distinguished from the more common form of anthropogenic disturbance involving forest clear-cuts in that it is relatively more frequent but less severe. This suggests that the ecological effects of the power-line corridors would differ from those expected from the better known forms of anthropogenic disturbance, such as timber logging, in terms of alterations in species composition, ecosystem structure and functioning. Ecological disturbances often have varying degrees of impact on species composition and ecosystem structure, depending on the frequency and severity of the disturbance. For example, while strong winds might destroy the vegetation of an area, mild forest fires have been shown to induce forest regeneration and enhance diversity (Pickett 1985).

The openings in the boreal forest resulting from the creation and maintenance of power-line corridors, creates semi-open shrub-rich habitats (e.g. Berg et al. 2011). This may in turn facilitate browsing by ungulates such as moose (*Alces alces*) and roe deer (*Capreolus capreolus*) in the corridor areas, which will contribute to prevent re-growth of forest and maintenance of a low-stature plant community. This might allow species associated with cultural landscapes to use the corridor habitats as refuges just as grassland species have been found to make use of utility corridors as refuges in forested landscapes (Nekola 2012). The culminating effect of the disturbance may be to create opportunities for invasive species to colonize the corridor areas (Hobbs & Huenneke 1992). Changes in microclimate (Pohlman et al. 2009) and increased irradiance in open-canopy corridor habitat should also benefit the species which are drought and light tolerant such as heather (*Calluna vulgaris*) while excluding or causing decline in relatively shade adapted species such as bilberry (*Vaccinium myrtillus*) (Parlane et al. 2006). Although the colonization potential of the different species depend on other factors such as the nearness of suitable seed and spores to the open corridor areas immediately after clearing, hence the consideration of heterogeneity in the entire landscape. The process by which the plant communities of the disturbed corridor habitats responds can be described in terms of succession, a concept that is closely linked with disturbance in plant ecology (Johnson & Miyanishi 2007). This is because the disturbance has a long-term influence on the corridor habitats by acting to initiate and terminate plant succession (Rydgren et al. 2004).

It can be predicted that the corridor vegetation would differ in species composition from the forest habitats due to alterations in biophysical factors such as solar irradiance and soil conditions occasioned by frequent cutting of the vegetation (Folke et al. 2004). It may also be asserted that greater biodiversity would be found in the corridor vegetation relative to the forest habitats. With regard to plants, the decay of plant matter that is left to on the ground after clearing favours nutrient cycling and renewal of the corridor vegetation. The increase in plant diversity might in turn facilitate an increase in animal species especially insect pollinators in power-line corridors (Komonen et al. 2013). Another reason for the assertion of greater plant diversity is that the open areas in the corridor areas immediately after a cutting of the vegetation may serve as suitable habitats for seeds and spores of many different species to disperse and colonize, while the resulting landscape heterogeneity from the disturbance may also enhance species richness (Redon et al. 2014).

My decision to consider the effects of this disturbance on both the individual species level and plant functional types level is informed by well known ideas regarding the individualistic response of plant species to environmental changes (Gleason 1926) and the need to overcome the difficulty of discussing these effects for each individual plant species in the community. Moreover, such endeavour would be of limited ecological use without the ingenious characterisation of plant communities, which the various plant functional type classifications provide. The individual species considered are heather and bilberry which are keystone species in the boreal forest ecosystem in that bilberry is a source of food, berries and plant fodder, for many fauna species (Ferris & Humphrey 1999) and heather provides nectar and pollen resources for insects such as bees and butterflies (Goulson et al. 2005). An assessment of the effects of the maintenance of power-line corridors in boreal forests with regard to plant functional types leads to a coherent understanding of the ecological effects of this form of disturbance in that in points out changes in the morphological, physiological and /or life history traits of broad classes of species in the plant communities being investigated (Duckworth et al. 2000; Lavorel et al. 1997). In addition, a consideration of functional groups of species leads to a holistic understanding of the plant communities and changes in traits that are important for ecosystem functioning (Cadotte 2011).

In order to preserve vital ecosystem processes and conserve biodiversity, management approaches might consider prioritizing any of the following, bearing in mind that the managers of the power-line corridors might decide to pursue different goals at different times depending on changes in the corridor habitats (Lindenmayer et al. 2008). To minimize the differences between the corridor and forest vegetation; maximize total plant species richness and richness of specific functional groups such as forbs, many of which are flowering plants that attract pollinators; maximize the cover of heather and bilberry in the corridor areas; preserve the capacity of the corridors to act as suitable habitats for species associated with cultural landscapes; and limit the potential for colonization of the corridor areas by invasive species.

This study can be viewed as a large-scale natural experiment intended to evaluate the ecological effects of the creation of power-line corridors on plant communities. In that, the forest habitats serve as the control, while the corridor habitats are the experimental treatment effects. Most available literature on power-line corridors as an ecological disturbance have sought to explain the ecological effects of this disturbance on animals, especially birds and insects, thus making this evaluation with respect to plants timely (Askins et al. 2012; Berg et al. 2011). In addition, those literatures often relate to Australian and North American ecosystems, whose degree of human interference are relatively recent compared to cultural landscapes of many European countries including Norway, which have been shaped by humans over several centuries. My consideration of the implications of the power-line corridors in the boreal forest ecosystem focuses on the understorey vegetation, which has been identified as a key driver of forest ecosystems (Nilsson & Wardle 2005).

This study aims to investigate the ecological effects of power-line corridors on species diversity and ecosystem structure of boreal forests of south eastern Norway. Specifically, I aim to: (1) evaluate effects on the abundance and diversity of some key plant species and plant functional groups (2) examine the degree of vegetation changes, measured in terms of compositional dissimilarities (3) determine whether the power-line corridors support invasive species and cultural landscape species and (4) explore the environmental factors influencing the observed changes in the power-line corridor vegetation, focusing on factors that may be influenced by managers and planners. This understanding would be useful in formulating effective conservation and management strategies for the power-line corridors.

2.0 Materials and Methods

2.1 Study sites and sampling design

The study system comprises the area traversed by the main power line grid in southeast Norway. In order to ensure that the study sites are representative of a broad range of environmental conditions present in the entire study region, 84 candidate sites were initially selected across by placing crosses on a general map of the main power line grid at regular intervals. Thereafter 51 study sites were randomly selected from the 84 candidate sites by drawing lots.

The study sites lie between latitude 58°N-61°N and longitude 8°E-11°E and represent a range of boreal forest types from north-boreal to boreo-nemoral forests with altitudes ranging from 25 to 1055 metres above the sea level. The sites were established in the field by determining the accurate geographical position of the closest location along the power line corridor's longitudinal direction with adjoining forest of at least 200 metres wide. This was done with the aid of aerial photos obtained from http://www.norgeibilder.no/. Stratified sampling technique was then used to designate two habitat types within each site and these are the forest habitat (100m from the forest or corridor edge), and the power-line corridor habitat. The forest habitat refers to the interior of the forest adjoining a sited electricity transmission line and is denoted as Fi, while the power-line corridor habitat refers to plant communities directly beneath an overhead electricity transmission line and is denoted as C (Fig. 1). At each site four plots were marked within each of these two groups of habitats, each of which was further divided into five 1 m^2 subplots. effects on the plant community are generally reported to dissipate within 50m from the forest edges, although in some plant communities theses could extend up to 100m (Harper et al. 2005) and hence the reason for the stratified sampling approach. A minimum distance of 50m was maintained between the plots in the same habitat, and between plots and adjacent power line posts. Each of the 51 study sites had forests on both sides of the corridor except for one which had clear cut on the opposite side of the corridor.

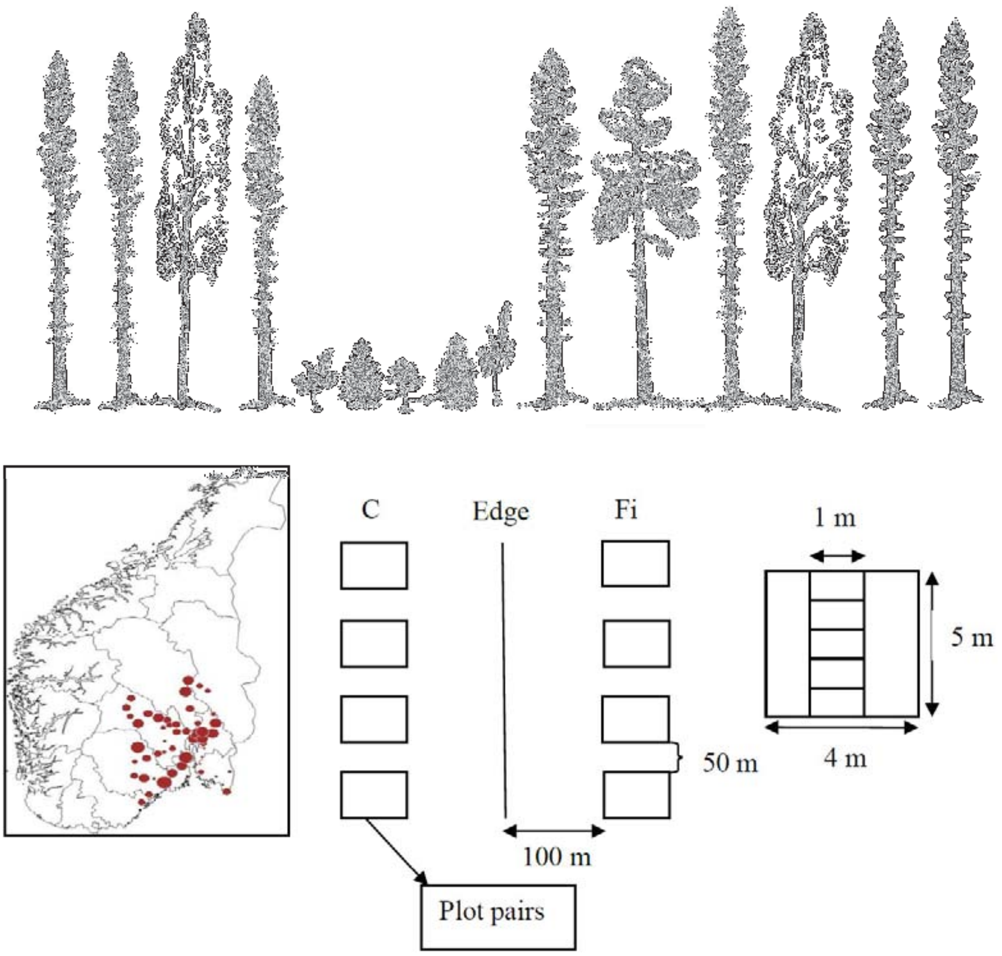

Fig. 1. Geographic distribution of the study sites and a schematic illustration of a study site. The schematic illustration shows a typical layout of a study site, where four plots each further containing five 1 m^2 subplots were marked for sampling in each of the two habitat types. C represents the power-line corridor habitat, edge refers to the edge of the forest and Fi denote the interior of the adjoining forest, which is designated as the forest habitat.

2.2 Field data collection

Field data on under-storey vegetation and habitat characteristics were collected for 20 sites in 2009 and for the remaining 31 sites in 2010. The relative abundance (percent cover) of plants in the understorey vegetation including shrubs, dwarf shrubs, forbs and graminoids in each of the five subplots (1m x 1m) for each of the 8 plots at each site was measured by visual estimation. The percent cover for lichens and mosses was also estimated, but these were not identified up to species level. Within the 4 m × 5 m plots, all trees greater than 1m in height were recorded and the tree height and crown width was measured by visual estimation. The basal areas (relascope sum) of tree trunks were measured at breast height from the centre of the plots through a technique known as angle count sampling (Schreuder et al. 1987). In addition, soil depth at the middle of each plot was measured and designated as shallow soil when ≤ 30cm and deep soil when > 30cm. The vegetation site index for each plot was determined based on the dominating tree species, soil depth, terrain slope, height above sea level, northern latitude, and the vegetation type that dominated an area of 100 m^2 (radius 5.64 m) from the centre of the plot (Nilsen & Larsson 1992). Site index is a measure of the productivity or fertility of plots (Nilsen & Larsson 1992) and it ranges from 6.0 to 23.1 for the plots included in this study.

2.3 Extraction of data from digital sources and the literature

Following the establishment of the study sites, I extracted data for the corridor width from aerial photos (http://www.norgeibilder.no/), while the age of the corridors was obtained from the grid owner (Statnett). Data on the growth season (June, July and August) temperatures and precipitation were obtained from the Norwegian Meteorological Institute (monthly averages in the 30 year period 1951-1991). The average growth season temperature is the average daily temperatures in June, July and August and the average growth season precipitation is the average monthly precipitation in June, July and August. Data on the density of forest edges, i.e., length of forest edges at increasing radius around each site (Table 1) were extracted by the use of ArcGIS (ESRI 2011) and Ar5 digital maps (http://www.skogoglandskap.no/temaer/ar5).

Each of the plant species in the field data was assigned Ellenberg indicator values (Hill et al. 1999). This data was analysed based on three of the original Ellenberg indicator factors. These are light, moisture, and nitrogen. Species with Ellenberg indicator values for light from 1 to 4 were categorized as shade-tolerant (L1), those with values from 5 to 7 were categorized as “intermediate” (L2) and those with values from 8 to 9 were categorized as light-tolerant (L3). Species with Ellenberg indicator values for moisture from 1 to 3 were categorized as drought-tolerant (F1), those with values from 4 to 7 were categorized as “intermediate” (F2) and those with values from 8 to 9 were categorized as preferring wet soils. Species with Ellenberg indicator values for nitrogen from 1 to 3 were categorized as surviving in nutrient poor soils (N1), those with values from 4 to 6 were categorized as “intermediate” (N2) and those with values from 7 to 9 were categorized as preferring nutrient rich soils (N3). Likewise, species associated with cultural landscapes were extracted from the field data according to a publication on habitat types in Norway (Halvorsen et al. 2009). Plants species in the field data were also classified according to their growth forms as shrubs, dwarf shrubs, forbs and graminoids (sedges and grasses). The richness and abundance data for the Ellenberg groups, cultural landscape species and ach of the functional groups based on growth forms were then derived by summarizing the original raw species data.

Table 1. The explanatory variables included in analyses of all the response variables.

Variables in analyses	Variables included in PCA	Spatial scale	Range/levels
Corridor age		Site	24-81 years
Corridor width		Site	Wide, Narrow
Site index		Plot	6-23.1
*PC1 trees			
	Relascope sum	Plot	0-36 trees
	Number of trees>1m	Plot	0-66 trees
	Number of conifers>1m	Plot	0-25 trees
	Number of deciduous trees>1m	Plot	0-61trees
	Number of spruce trees>1m	Plot	0-10 trees
	Number of pine trees>1m	Plot	0-25 trees
	Mean tree height of trees>1m	Plot	0-7 m
	Max tree height	Plot	0-8 m
	Sum crown width of trees>1m	Plot	0-47.9 m
	Species richness trees>1m	Plot	0-5 species
‡PC1 landscape edges			
	Within 150 m radius from site	Site	0-966 m
	Within 300 m radius from site	Site	0-3082 m
	Within 500 m radius from site	Site	0-6758 m
	Within 1000 m radius from site	Site	373-26088 m
	Within 2000 m radius from site	Site	6567-94826 m
Average temperature in growth season (June-August)		Site	9.9-13.3 °C
Average monthly precipitation in growth season (June-August)		Site	66-105 mm

*Variability accounted for by PC1:51%, ‡PC1:66%

2.4 Statistical Analysis

2.4.1 Response variables

Plant data from the five 1m × 1m subplots within each plot were first averaged (Fig. 1) and from these mean values, I calculated the total species richness and Shannon diversity (Kindt & Coe 2005) for each plot, as well as the species richness and percentage cover of shrubs, forbs, heather, bilberry and cultural landscape species. As a measure of the differences in species composition between the corridor and forest habitats, I calculated compositional dissimilarity (Chao et al. 2008). Compositional dissimilarity values (1-CqN) range from 0 to 1, with 1 being absolutely dissimilar and 0 being identical, that is not dissimilar.

2.4.2 Explanatory variables

The explanatory variables (Table 1) considered are environmental variables which can potentially influence the vegetation structure and which can be modified by planners and managers such as the age of the corridors, width of the corridors, the successional stage of the forest, landscape heterogeneity, and site index. In addition, climatic variables, i.e. average growth season temperature and precipitation were also considered. To deal with co-linearity among measurements of the tree layer and landscape heterogeneity, respectively, the compound variables PC1trees and PC1landsc were derived from PCA analyses of a host of related explanatory variables (Table 1). The PC1 axis scores (Table 1) are used as covariates in the statistical analyses. PC1 trees was used to denote the stage of succession of the vegetation based on a PCA analysis of composite variables listed in table 1, while PC1landscape was used to denote the degree of landscape heterogeneity based on a PCA analysis of the composite variables listed in Table 1. The PCA covariates were zero-skewness-standardized before analyses and ranged on a common 0–1 scale (Økland et al. 2001).

Data exploration was carried out based on the protocol recommended by (Zuur et al. 2010). A multi-panel scatter plot was made to explore collinearity among the various explanatory variables. The multi-panel scatter plot showed significant collinearity between site elevation and average growth season temperature and so the average growth season was preferred in the modelling. Corridor width was categorized as either wide when > than 45m (in which case had dual transmission towers) or narrow otherwise (when with a single transmission tower) modelled as a categorical variable. Out of the 51 study sites, 16 fell under the category of wide corridors, while the remaining 35 were categorized as narrow corridors. The exploratory analyses did not indicate heterogeneity between groups of categorical variables or extreme outliers in the data. Relationships between the response variables and explanatory variables were then subsequently explored using generalized linear mixed modelling (GLMM) techniques (Zuur et al. 2013). For each of the response variables, the main terms A, B, C, etc were included as explanatory variables, and the interactions A × B etc were included in the full (most complex) models. Model selection was carried out by backward elimination of terms based on the p-values (Crawley 2012), retaining terms with $p < 0.05$. Model adequacy was checked by use of graphical validation techniques (Zuur et al. 2013).

3.0 Results

3.1 Effect on total species richness and richness of forbs species

The total species richness increased in the corridor habitats compared to the forest habitats (Fig. 3) and the main environmental variables driving this increase were the succession stage of the vegetation, denoted as PC1trees, landscape heterogeneity, site index and the average growth season temperature (Table 2). The parameter estimates for these environmental variables indicate that the total species richness increases with increasing site index and vegetation successional stage up until a certain point and then declines thereafter (Table 2). These estimates also suggest that the total species richness in the corridor habitats decreases with increasing average growth season temperature (Table 2).

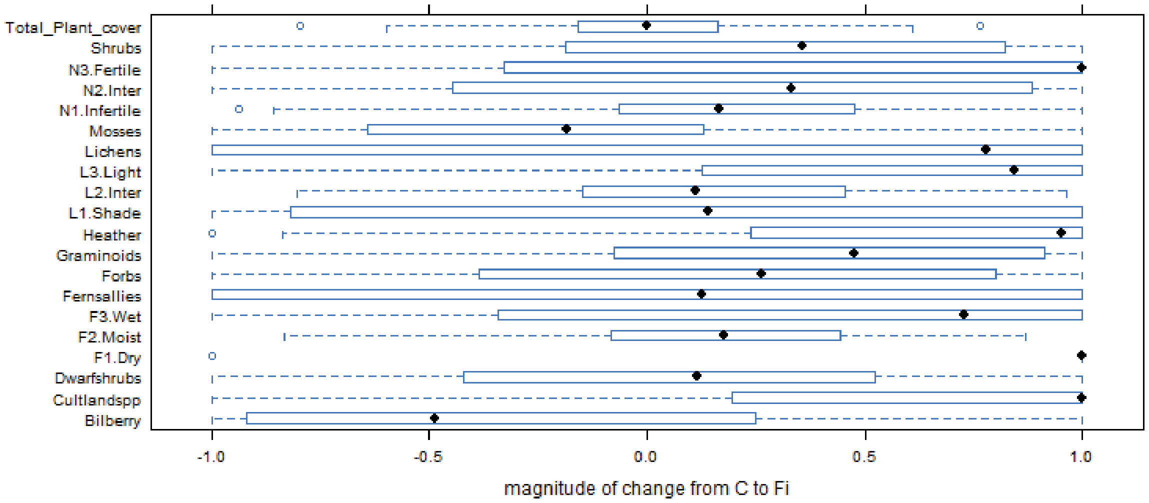

Fig. 2. Box and whisker plot showing changes in the corridor vegetation (see Fig. 1 for sampling design), with regard to total plant cover and cover of heather, bilberry and some plant functional types. The response variables with alphanumeric names are derived from the Ellenberg indicator factors described in the materials and methods.

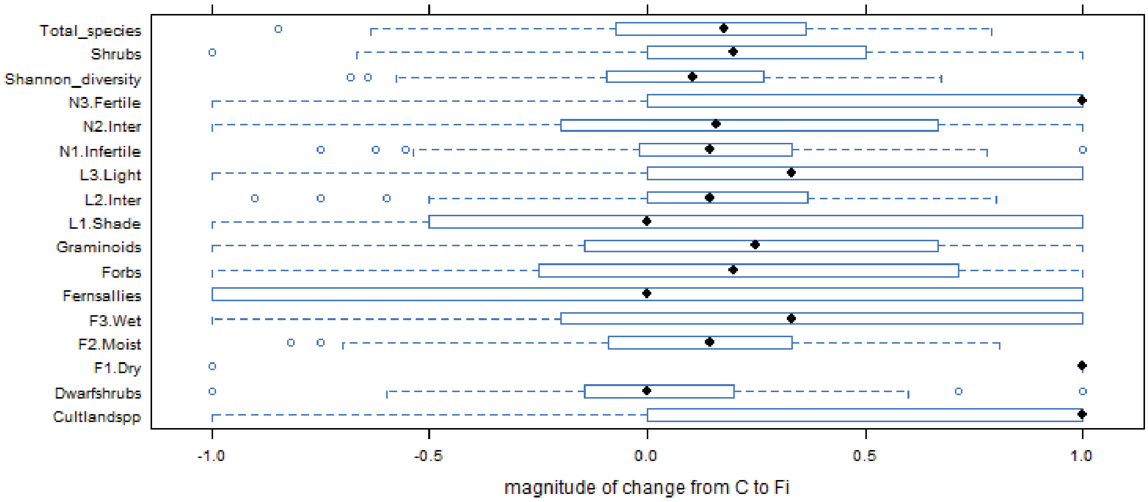

Fig. 3. Box and whisker plot showing changes in the corridor vegetation (see Fig. 1 for sampling design) relating to Shannon diversity, total species richness, and richness of some plant functional types. The response variables with alphanumeric names are derived from the Ellenberg indicator factors described in the materials and methods.

Table 2. Influence of environmental variables on the total number of vascular plant species and the number of forbs species in power-line strips (that is C plots in Fig. 1) dissecting boreal forests. PC1trees denote the stage of vegetation succession, PC1landscape is a measure of landscape heterogeneity and site index is a measure of soil fertility.

	Explanatory variables	β*	SE	z	p
Total species richness	Fixed effects				
	Intercept	2.6	0.30	8.8	<0.0001
	PC1trees	0.63	0.23	2.8	0.0053
	$(\text{PC1trees})^2$	-0.57	0.25	-2.3	0.020
	PC1landscape	0.30	0.11	2.6	0.0093
	Site index	0.17	0.040	4.2	<0.0001
	$(\text{Site index})^2$	-0.0026	0.0013	-2.0	0.048
	Average growth season temp.	-0.16	0.017	-9.1	<0.0001
	Random effect	σ	SD	Plots	Sites
	Site	0.016	0.13	204	51
Species richness of forbs	Fixed effects				
	Intercept	1.2	0.68	1.8	0.079
	PC1landscape	0.72	0.36	2.0	0.044
	Site index	0.49	0.078	6.2	<0.0001
	$(\text{Site index})^2$	-0.0095	0.0025	-3.8	0.0002
	Average growth season temp.	-0.38	0.041	-9.3	<0.0001
	Corridor width (wide vs narrow)	-0.99	0.41	-2.4	0.016
	PC1landscape × Corridor width	1.2	0.59	2.1	0.036
	Random effect				
	Site	σ	SD	Plots	Sites
		0.10	0.32	204	51

[*] Parameter estimates and standard errors of fixed effects for the final generalized mixed model (GLMM) for each response variable. For each response variable, the most complex model included the main effects of all the explanatory variables listed in Table 1, and the Corridor age × Corridor width, PC1trees × Corridor width, Site index × Corridor width, and PC1landscape × Corridor width interactions. In addition, a quadratic term was added for PC1trees and Site index. The GLMM modelling was performed as recommended in (Bolker et al. 2009), using the lme4 package in R (R Core Team 2013). For each response variable, terms were retained in the final model through backward selection (Crawley 2007) if their effect on the response was statistically significant (*P*<0.05). For both response variables, a model with log link function and Poisson distribution was fitted.

A prediction of the amount of influence of the environmental variables on the total species richness based on the model indicates that site index has the strongest positive effect on the total species richness, whereas the effect of PC1 trees and landscape heterogeneity is more moderate (Fig. 4). However, in spite of the greater species richness only one invasive species (*Epilobium ciliatum*) was found in the power-line corridors and this was present in only 1 of the 204 plots studied. The richness of forbs species also increased in the corridor habitats compared to the forest habitats (Fig. 3). The environmental factors having significant effect on the forbs species increase were landscape heterogeneity, site index, average growth season temperature, corridor width and the interaction between landscape heterogeneity and corridor width (Table 2). There was significant interaction between landscape heterogeneity and corridor width (Table 2, Fig. 5). The parameter estimates for these variables suggest that forbs species richness decreases with increasing average growth season temperature and corridor width, but increases with site index and landscape heterogeneity (Table 2), although the increase with landscape heterogeneity was steeper for wide corridors (Fig. 5). Predictions based on the model when plotted indicate that site index has a pronounced effect on the richness of forbs species (Fig. 5).

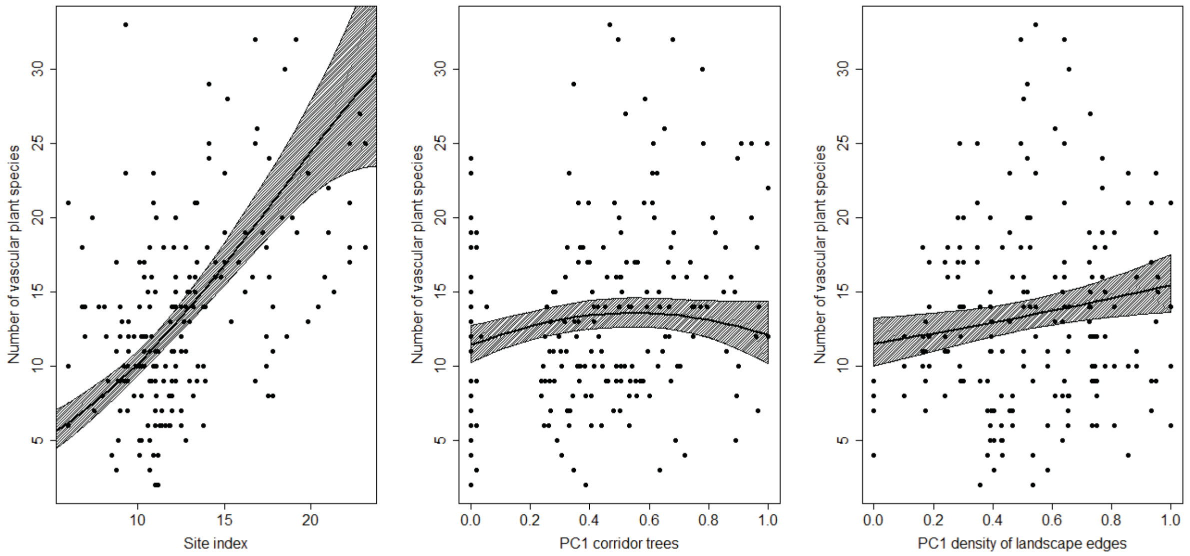

Fig. 4. Fitted relationships between the total species richness of corridor habitats and soil fertility which is designated as site index, vegetation succession stages denoted as PC1trees, landscape heterogeneity designated as PC1landscape, respectively. Hatched polygons are 95% confidence intervals. Dots are observed values (from 204 plots at 51 sites: fig.1). For details and results of the statistical modelling, see Table 2.

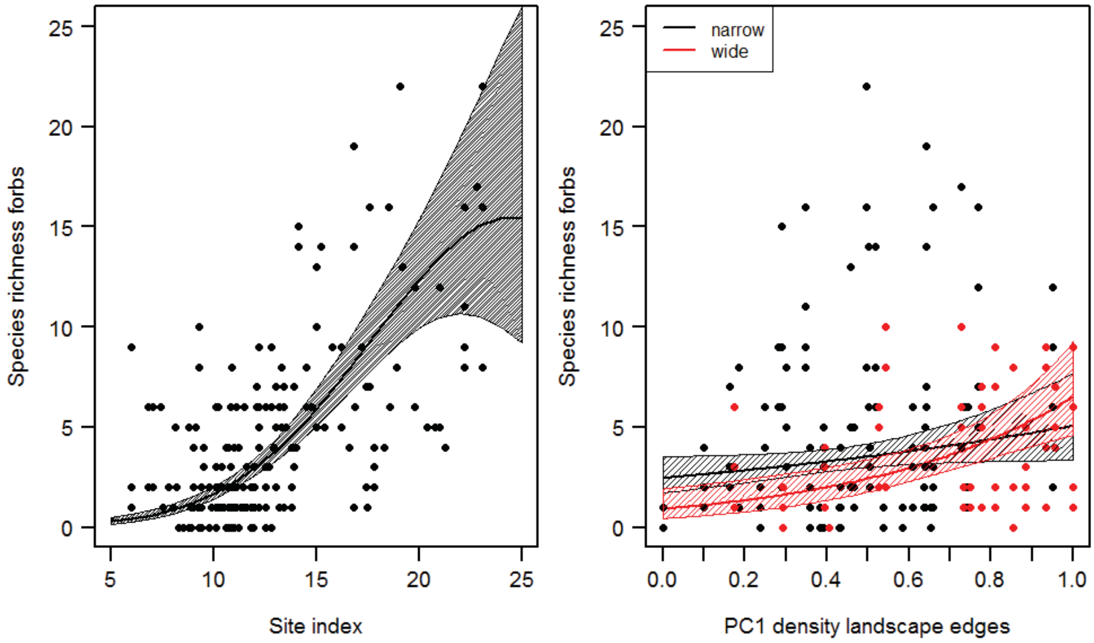

Fig. 5. Fitted relationships between the species richness of forbs in corridor habitats and soil fertility which is designated as site index, landscape heterogeneity designated as PC1landscape, respectively. Hatched polygons are 95% confidence intervals. Dots are observed values (from 204 plots at 51 sites: fig.1). For details and results of the statistical modelling, see Table 2.

3.2 Effect on diversity and compositional dissimilarity

Species diversity also increased in the corridor habitats (Fig. 3) and the environmental variables having significant effect on the Shannon diversity index were the vegetation successional stage, site index and the average growth season temperature. The parameter estimates for these variables indicate that the Shannon diversity index increases with vegetation successional stage and site index, but decreases with increasing average growth season temperature (Table 3). Estimates from the model also indicate that site index has the strongest positive effect on Shannon diversity index (Fig. 6).

Table 3. The effects of environmental variables on the Shannon diversity index of the corridor habitats and compositional dissimilarity between corridor and forest habitats (that is C and Fi plots in Fig.1). PC1trees denote the stage of vegetation succession and site index is a measure of soil fertility.

Measure	**Variable**	**β***	**SE**	**t**	**p**
Shannon diversity index	Fixed effect				
	Intercept	2.76	0.28	9.96	<0.0001
	PC1trees	0.27	0.13	2.14	0.0343
	Site index	0.09	0.01	8.06	<0.0001
	Average growth season temp.	-0.17	0.024	-7.3	<0.0001
Compositional dissimilarity	Fixed effect				
	Intercept	0.28	0.25	1.15	0.25
	Average growth season temp.	0.05	0.018	2.69	0.0099
	Corridor width (wide vs narrow)	0.16	0.07	2.22	0.0310

[*] Parameter estimates and standard errors of fixed effects for the final generalized mixed model (GLMM) for each response variable. For each response variable, the most complex model included the main effects of all the explanatory variables listed in Table 1, and the Corridor age × Corridor width, PC1trees × Corridor width, Site index × Corridor width, and PC1landscape × Corridor width interactions. In addition, a quadratic term was added for PC1trees and Site index. The GLMM modelling was performed as recommended in (Bolker et al. 2009), using the lme4 package in R (R Core Team 2013). For each response variable, terms were retained in the final model through backward selection (Crawley 2007) if their effect on the response was statistically significant ($P<0.05$).

Compositional dissimilarity was found to be influenced by the average growth season temperature and corridor width, and it increases with increasing average growth season temperature and corridor width (Table 3). Greater compositional dissimilarity in the wider corridor is depicted in figure 7.

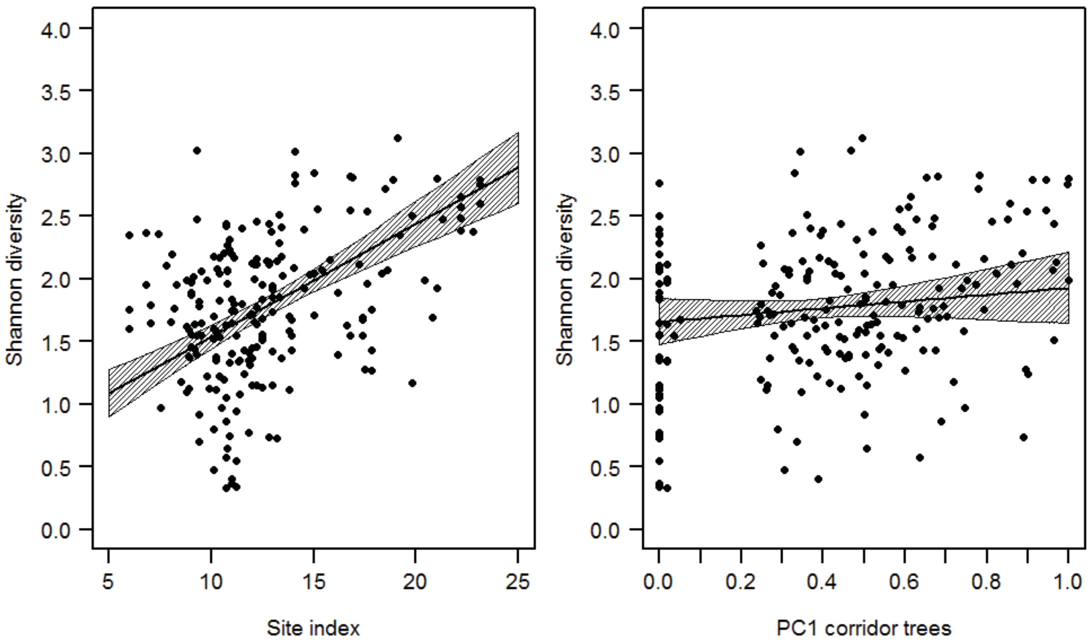

Fig. 6. Fitted relationships between the Shannon diversity index of corridor habitats and soil fertility which is designated as site index, vegetation succession stages denoted as PC1trees, respectively. Hatched polygons are 95% confidence intervals. Dots are observed values (from 204 plots at 51 sites: Fig. 1). For details and results of the statistical modelling, see Table 3.

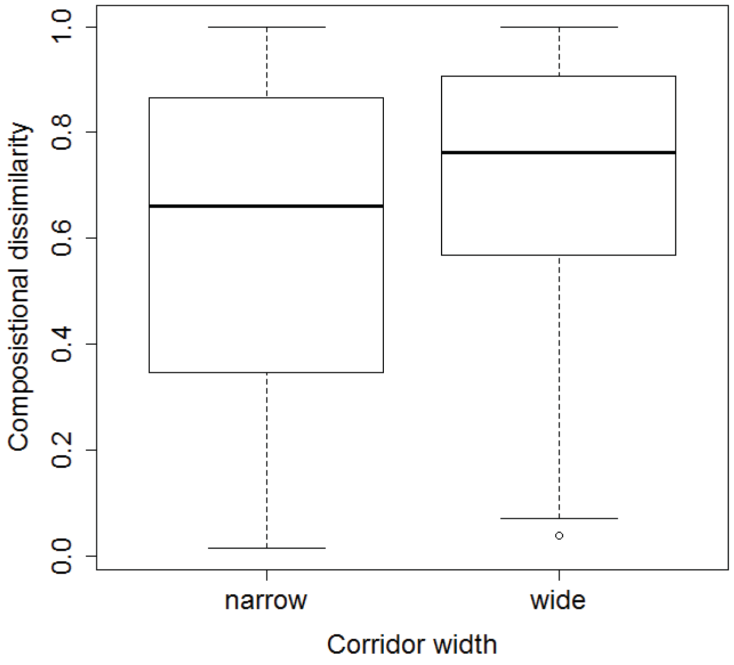

Fig. 7. Box and whisker plots showing difference in plant community composistion (compositional dissimilarity) between corridor and forest plots (see Fig. 1), for narrow (<45 m) and wide (>45 m) power-line corridors.

3.3 Effect on the cover of forbs, shrubs, heather, bilberry and cultural landscape species

The cover of forbs species increased in the corridor habitats compared to the forest habitats (Fig. 2) and this was influenced significantly by landscape heterogeneity, site index and average growth season temperature. The cover of forbs species increases with increasing landscape heterogeneity and site index, up to a certain level and then declines, while it decreases with increasing average growth season temperature (Table 4). The site index also appear to be the dominant factor affecting forbs species cover based on predictions from the model (Fig. 8). The cover of shrubs species increased in the corridor habitats compared to the forest habitats (Fig. 2) and this was influenced by vegetation successional stages, landscape heterogeneity and site index. The cover of shrubs species increased with increases in each of the environmental variables stated earlier (Table 4). Estimates from the model indicate that the three environmental variables influencing the cover of shrubs species are of equal in significance (Fig. 9).

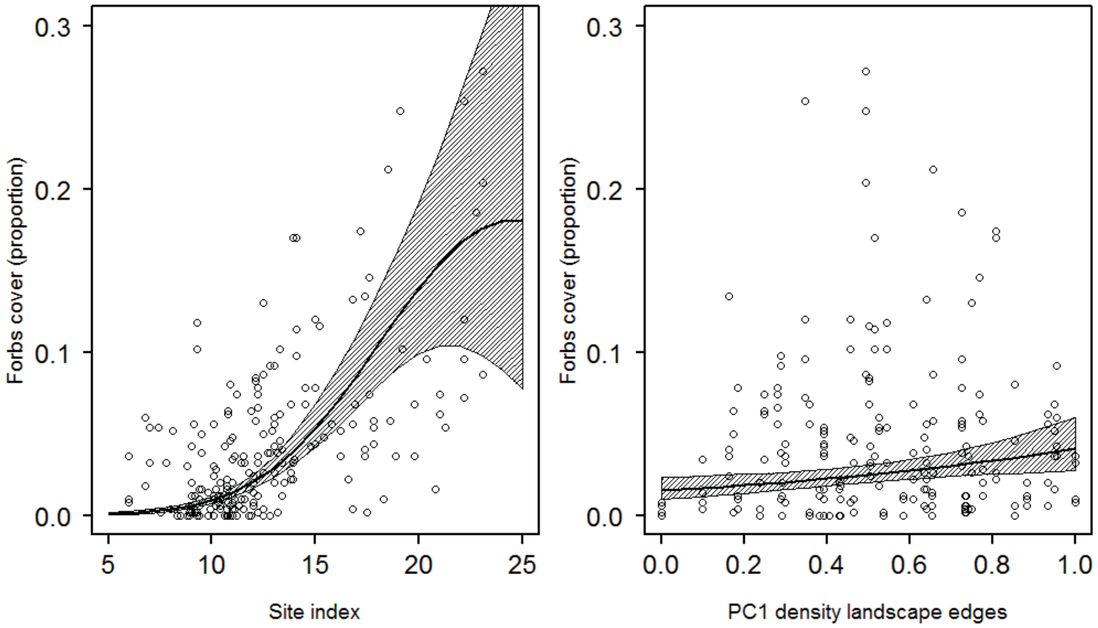

Fig. 8. Fitted relationships between forbs cover of corridor habitats and soil fertility which is designated as site index, landscape heterogeneity designated as PC1landscape, respectively. Hatched polygons are 95% confidence intervals. Dots are observed values (from 204 plots at 51 sites: Fig. 1). For details and results of the statistical modelling, see Table 4.

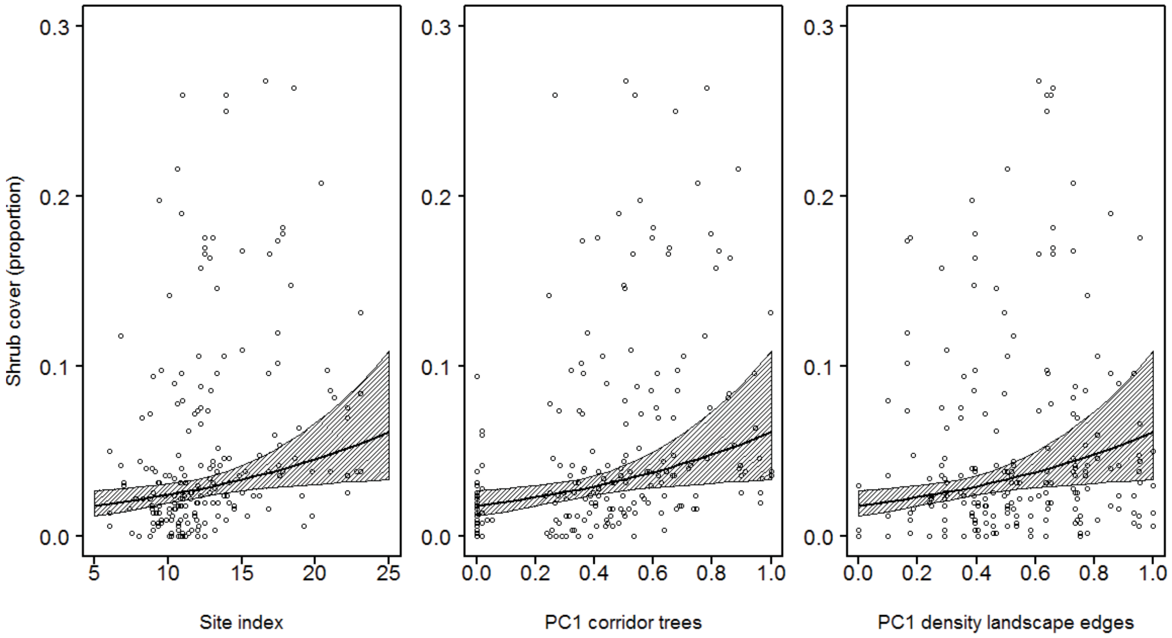

Fig. 9. Fitted relationships between shrubs cover of corridor habitats and soil fertility which is designated as site index, vegetation succession stages denoted as PC1trees, landscape heterogeneity designated as PC1landscape, respectively. Hatched polygons are 95% confidence intervals. Dots are observed values (from 204 plots at 51 sites: Fig. 1). For details and results of the statistical modelling, see Table 4.

Table 4. Influence of environmental variables on relative abundance. PC1trees denote the stage of vegetation succession, PC1landscape is a measure of landscape heterogeneity and site index is a measure of soil fertility.

Relative abundance	Explanatory variables	β*	SE	z	p
Cover of bilberry	Fixed effects				
	Intercept	-10.27	1.94	-5.28	<0.0001
	PC1trees	1.35	0.58	2.34	0.0195
	Site index	0.94	0.29	3.29	0.001
	(Site index)^2	-0.04	0.01	-3.86	0.0001
	Random effect	σ	SD	Plots	Sites
	Site	1.359	1.166	204	51
Cover of heather	Fixed effects				
	Intercept	-8.57	2.36	-3.63	0.0003
	PC1trees	-1.65	0.82	-2.02	0.043
	PC1landscape	-2.73	1.26	-2.17	0.03
	Site index	-0.68	0.09	-7.5	<0.0001
	Average growth season temp.	1.03	0.19	5.51	<0.0001
	Random effect	σ	SD	Plots	Sites
	Site	2.94	1.71	204	51
Cover of shrubs	Fixed effects				
	Intercept	-5.58	0.39	-14.42	<0.0001
	PC1trees	1.85	0.31	6.00	<0.0001
	PC1landscape	0.89	0.42	2.12	0.034
	Site index	0.06	0.02	2.58	0.010
	Random effect	σ	SD	Plots	Sites
	Site	0.276	0.525	204	51
Cover of forbs	Fixed effects				
	Intercept	-4.58	1.01	-4.56	<0.0001
	PC1landsc	1.02	0.38	2.72	0.0065
	Site index	0.73	0.14	5.36	<0.0001
	(Site index)^2	-0.01	0.005	-3.29	0.0010
	Average growth season temp.	-0.50	0.06	-8.68	<0.0001
	Random effect	σ	SD	Plots	Sites
	Site	0.195	0.441	204	51
Cover of species associated with cultural landscapes	Fixed effects				
	Intercept	-6.25	1.64	-3.81	0.00014
	PC1landsc	2.40	0.92	2.60	0.0094
	Site index	0.38	0.06	6.13	<0.0001
	Average growth season temp.	-0.48	0.14	-3.38	0.0007
	Random effect	σ	SD	Plots	Sites
	Site	1.13	1.06	169	51

[*] Parameter estimates and standard errors of fixed effects for the final generalized mixed model (GLMM) for each response variable. For each response variable, the most complex model included the main effects of all the explanatory variables listed in Table 1, and the Corridor age × Corridor width, PC1trees × Corridor width, Site index × Corridor width, and PC1landscape × Corridor width interactions. In addition, a quadratic term was added for PC1trees and Site index. The GLMM modelling was performed as recommended in (Bolker et al. 2009), using the lme4 package in R (R Core Team 2013). For each response variable, terms were retained in the final model through backward selection (Crawley 2007) if their effect on the response was statistically significant ($P<0.05$). For both response variables, a model with logit function and binomial distribution was fitted.

The cover of heather increased in the corridor habitat compared to the forest habitats (Fig. 2) and was found to be significantly influenced by vegetation successional stages, landscape heterogeneity, site index and average growth season temperature. The parameter estimates indicate that the cover of heather decreased with increasing vegetation successional stage, landscape heterogeneity and site index; while it increased with increasing average growth season temperature (Table 4). Estimates from the model clearly show decreasing heather cover with increasing vegetation successional stage; landscape heterogeneity and site index (Fig. 10). Conversely, the cover of bilberry decreased in the corridor habitats compared to the forest habitats (Fig. 2) and the environmental factors having significant effect on the cover of bilberry were the vegetation successional stages and site index. The parameter estimates indicate that the cover of bilberry tends to increase with increasing vegetation successional stages and site index (Table 4). Estimates from the model show that the vegetation successional stage has a more pronounced effect on the cover of bilberry (Fig. 11). The cover of cultural landscape species increased in the corridor habitats compared to the forest habitats (Fig. 2) and the environmental factors having significant effect on the cover of this group of species were landscape heterogeneity, site index and average growth season temperature. The cover of the cultural landscape species increases with increasing landscape heterogeneity and site index, but decreases with increasing average growth season temperature (Table 4). Estimates from the model indicate that the site index has the most pronounced effect on the cover of cultural landscape species. The results of the data analysis show that site index was the ever-present explanatory variable with significant effect on the response variables, except for compositional dissimilarity (Tables 2, 3 and 4).

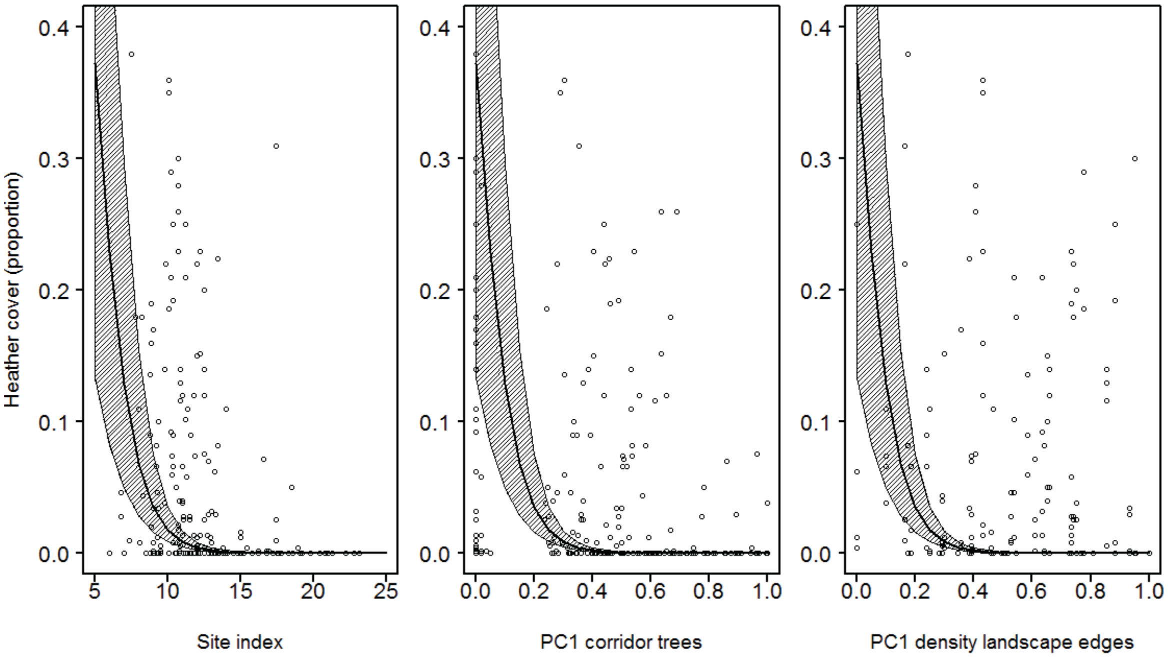

Fig. 10. Fitted relationships between heather (*Calluna vulgaris*) cover of corridor habitats and soil fertility which is designated as site index, vegetation succession stages denoted as PC1trees, landscape heterogeneity designated as PC1landscape, respectively. Hatched polygons are 95% confidence intervals. Dots are observed values (from 204 plots at 51 sites: Fig. 1). For details and results of the statistical modelling, see Table 4.

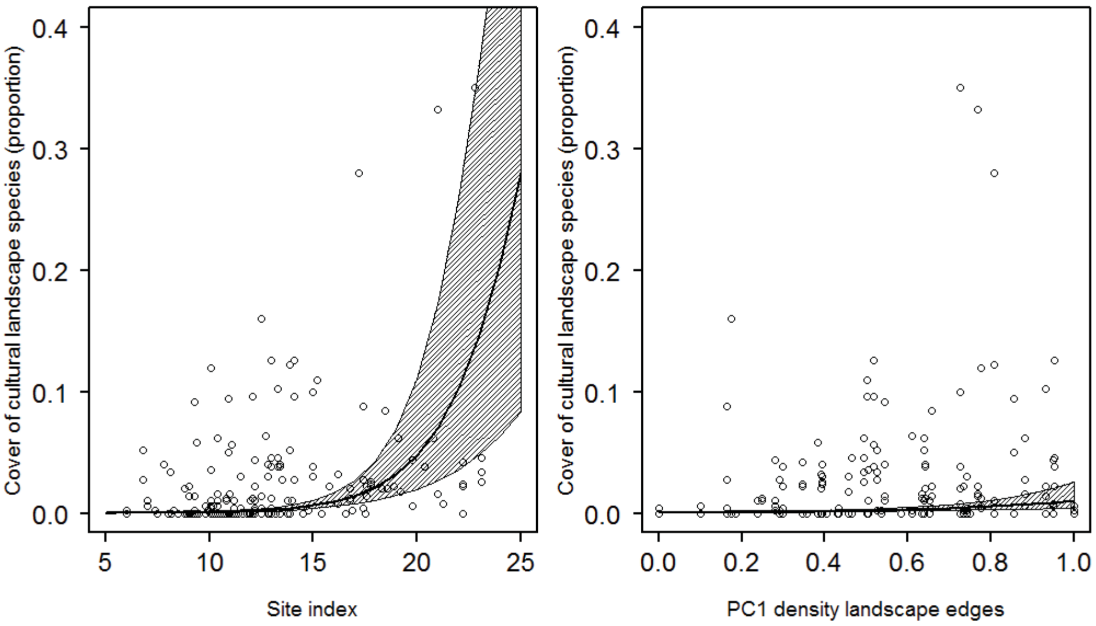

Fig. 11. Fitted relationships between the cover of cultural landscape species in corridor habitats and soil fertility which is designated as site index, landscape heterogeneity designated as PC1landscape, respectively. Hatched polygons are 95% confidence intervals. Dots are observed values (from 204 plots at 51 sites: see Figure 1). For details and results of the statistical modelling, see Table 4.

3.4 Effect on mosses, lichens and ellenberg functional groups

As expected, mosses declined in relative abundance in the corridor habitats (Fig. 2). Lichens on the other hand increased in relative abundance in the corridor habitats (Fig. 2). Among the Ellenberg functional groups, the most significant results are that drought tolerant species (L1) and species preferring nutrient rich soils (N3) increased in both richness and relative abundance in the corridor habitats (Figs. 2 and 3).

4.0 Discussion

This study highlights the implication of the maintenance of power-line corridors in boreal forests on species richness and relative abundance of two species and some plant functional types. Total species richness and diversity increased in the corridor habitats compared to the forest habitats (Fig. 3). The increase in total species richness in the corridor habitats is consistent with results from previous studies (Armesto & Pickett 1985; Belsky 1992) on the effects of disturbance on species richness in plant communities. My explanation for the greater overall diversity in the corridor habitats is that changes in the biophysical factors such as soil nutrients from the decay of plant matter from the cleared vegetation, increases soil fertility thereby favouring plant regeneration while the creation of gaps in the landscape subsequent to immediate clearing allows for recruitment of other species which were hitherto not present in the corridor habitats. The high species richness can also be explained by the coexistence of species adapted to both early and late successional stages. This is supported by the environmental factors- site index and landscape heterogeneity, being found to have significant positive effect on the total species richness (Table 2), with the total species richness increasing with increasing site index, which is a measure of soil fertility and increasing landscape heterogeneity which could be a surrogate for dispersal distances of different species to the corridor habitats. It is important to note that areas covered by the power-line corridors are mostly those with low to intermediate soil indices. This is because the electricity distribution companies often tries to avoid highly productive sites, so as not to get into conflict with the private landowners (Kristian Sommerstad, personal communication).

Another important result relating to greater biodiversity in the corridor habitats is that both indices of biodiversity used, total species richness and Shannon diversity, increased with increasing successional stage of the vegetation. However, this increase in species richness is only up until a certain point as shown by the second degree variable of succession, which decreased with increasing successional stages. This means that species richness increased during the early stages of succession due to increased availability of limiting resources, but decreased during the late stages of succession due to increased competition for the resources. Similar results were reported by (Burt & Rice 2009) who compared cleared and graded ski slopes. They found the cleared ski slopes, which are maintained in a similar manner to power-line corridors, to have greater species richness and mostly of intermediate successional stages in contrast to graded slopes with lower species richness and mostly of early successional stages. These results are quite remarkable, not the least because it demonstrates that power-line corridors are not a threat to biodiversity and could in fact be utilized as conservation areas for preserving biodiversity. These results can be considered to be significant with the models for total species richness and Shannon diversity explaining as much as 54% and 39% of the variability in each, respectively.

However, my results indicate the potential of the disturbed corridor habitats to resist biological plant invasions. This is evidenced by the occurrence of only one invasive species - *Epilobium ciliatum* in only 1 of the 204 plots. The rarity of invasive species in the corridors is contrary to one of the widely held themes in invasive ecology that disturbances would enhance invasibility based on the theory of fluctuating resource availability (Davis et al. 2000). In other words, the clearing of the corridor habitats would reduce light and water uptake and thus causing an increase in unused resources which in turn makes the community susceptible to invasion by exotic species. Conversely, my results doesn't entirely contradict this assumption but reflects another dimension to invasion ecology, in that the increase in unused resources tend to lead to greater diversity of native species which in fact makes the corridor habitats resistant to plant invasions based on the diversity-invasibility hypothesis (Tilman 1999), which suggests that the increase in competition for resources following increased biodiversity would further preclude invasibility. A better understanding of how the corridor habitats has been able to resist invasion would be obtained if other dimensions to invasion ecology such as propagule pressure and properties of potential invasive exotic species are studied with regard to the corridor habitats (Lonsdale 1999).

Considering the significance of monitoring responses to ecological disturbances with regard to plant functional types as opposed to taxonomic groups, the results indicate that functional groups based on life forms such as forbs (McIntyre et al. 1999) increased in both richness and relative abundance in the corridor habitats compared to the forest habitats (Figs. 2 and 3) This is consistent with the results of a previous study (Conde et al. 1983) suggesting that forbs are able to withstand disturbance. My explanation for this is that many of the these species have long been exposed to other similar forms of disturbance such as wildfires and clear-cutting and may have been able to develop some resilience over time (Peltzer et al. 2000). My results indicate that site index and landscape heterogeneity have a positive significant effect on the richness and relative abundance of forbs (Tables 2 and 4). This means that herbaceous flowering plants such as forbs prefer rich soils and can adapt to heterogeneous landscapes resulting from a disturbance. The degree of variability in richness and abundance of forbs species that is explained by the models for the two response variables is 61% and 21%, respectively and this suggests that the models can be relied upon. While this study focuses on plant species, it is important from an ecological stand-point to consider the consequences of the changes in plant species being studied on animal species with which they interact. Forbs are of great ecological importance in that they serve as a source of pollen and nectar for pollinating insects, such as bees, (Russell et al. 2005) and as such electricity transmission corridors has been recognized as being potentially suitable for pollinator conservation (Wojcik & Buchmann 2012). Management approaches should therefore consider the need to preserve the diversity and abundance of forbs in the corridors in order to conserve a vital ecosystem service in pollination.

Shrubs increased in abundance and richness in the corridor habitats compared to the forest habitats (Figs. 2 and 3). This appears to be the case because management approaches are aimed at preventing the corridor vegetation from reaching its full growth potential so as prevent interference by trees with electric transmission lines. This in turn allows functional groups such as shrubs, with lower heights compared to trees, to dominate the corridor habitats (Berg et al. 2011). My analysis reveals that the relative abundance of shrubs increased in the corridor habitats with increasing site index, landscape heterogeneity and successional stage (Table 4). This implies that shrubs tend to prefer rich soils and can survive competition from other species in the corridor habitats even as the community approaches late successional stages. There is however a large unexplained variation in the model for the cover of shrubs (R^2 = 9%) thereby limiting the reliability of the conclusion based on the results. The unexplained variability may be due to the large among and within site variation in the cover of shrubs.

This unexplained variability may also be due to the nature of the response variable, which groups together different species that may respond differently to changes in the environment and also because factors such as herbivory have not been considered in the modelling. The dominance of shrubs in the corridors has significant ecological implications. First, it plays a role in affecting ecosystem dynamics by delaying succession (Meilleur et al. 1994) and second it provides suitable habitats for nesting birds (Marshall & VanDruff 2011). This result is important in that it points to the possibility of a win-win situation between managers and electricity transmission companies, which can be achieved by preserving the relative abundance of shrubs in the corridor habitats. This will help fulfil the objective of delaying reforestation with its attendant cost saving for electricity distribution companies in maintaining the power-line corridors while also helping to conserve birds by providing them with suitable habitats.

My analysis reveals opposing effects on the relative abundance of two keystone species in the boreal forest- bilberry and heather (Mallik 2003). Bilberry is an important food source (leaves, stem and berries) for many organisms including humans, while the pollen and nectar of heather are eaten by insects. While the abundance of heather increased in the corridor habitats, the abundance of bilberry decreased. This is consistent with the observation of (Parlane et al. 2006) that heather increases in abundance with increasing irradiance while bilberry decreases in abundance at higher levels of irradiance. Therefore, increased exposure to irradiance due to the creation of power-line corridors serves as a bottleneck for bilberry, while enabling heather to occupy areas vacated by bilberry. From the results of my analysis, the abundance of bilberry increases with increasing site index and succession stage and the abundance of bilberry can be increased by minimizing clearing of areas with high site index and at higher succession stages. A substantial amount of the variability in the cover of heather is explained the model for heather cover (R^2 = 21%) thereby suggesting that the conclusion regarding heather can be relied upon, while the degree of variability in the cover of bilberry that is explained by the model for bilberry cover is 11%. The large unexplained variation here is due to other possible effects such as grazing activities in the corridor habitats which are difficult to quantify.

My results also indicate differences in species composition of the corridor habitats compared to the forest habitats. More importantly it shows that the width of the corridors has a significant effect on compositional dissimilarity, with wider corridors being more dissimilar to the forest habitats compared to narrow corridors (Fig. 7). This may be that certain groups of species, such as those preferring rich soils (N3) from the Ellenberg functional group classification, are better represented in the corridor habitats compared to the forest habitats (Fig. 3). Furthermore, the results suggest that the dissimilarity in species composition might be influenced by climatic conditions, with the average growth season temperature being one of only two factors having a significant effect on compositional dissimilarity. The degree of variability in compositional dissimilarity that is explained by the model for compositional dissimilarity is 6.3%. The large unexplained variation in compositional dissimilarity may be due to the large among and within site variation in the compositional dissimilarity data.

Another important effect of the maintenance of power-line corridors is that these corridors favour the colonization of species associated with cultural landscapes. This is because remnants of these cultural landscape species in the boreal forest would prefer to migrate to modified areas such as the corridor habitats which have similar characteristics to agricultural landscapes for which they are adapted (Hamre et al. 2010). With many of these cultural landscapes been threatened, the corridors might be preserved as a haven for these species by conserving areas where they occur, which are those with high landscape heterogeneity (Redon et al. 2014) and high site index (Table 4). The degree of variability in the cover of cultural landscape species that is explained by the model for this response variable is 20% and this suggests that the model is somewhat reliable. The plant-animal interactions in these corridors might also account for increased soil fertility from animal defecation and dispersal of seeds of plants both of which help explain the greater diversity and abundance of species in the corridor habitats. Although trampling and grazing by ungulates such as moose and roe deer in the corridor habitats are other forms of disturbance which have profound effects on plant community composition (Augustine & McNaughton 1998).

The creation of power-line corridors has a mixed effect on non-vascular plant species. This is based on the observation of an increase and decrease in relative abundance of lichens and mosses, respectively. The decrease in moss cover can be attributed to water loss and increased temperature (Michel et al. 2011; Proctor et al. 2007) resulting from the opening up of the corridor vegetation. The ability of lichens to survive these conditions and increase in abundance can be explained with reference to its symbiotic nature, with the algal component being able to survive extreme temperatures (Scherer et al. 1984). Reduced competition for resources during the early succession stages of the corridor habitats help explain the increase in richness and relative abundance of species preferring nutrient rich soils while desiccation due to increased irradiance helps explain the increase in richness and abundance of drought tolerant species.

5.0 Conclusion

The main findings from this study can be summarized as follows. (1) The power-line corridors have greater species diversity than the adjoining forests. (2) The corridor habitats differ slightly from the forest habitats in its species composition and more importantly is that the difference between these two habitats is greater in wider corridors. Thus, wider corridors should be avoided where management objective is to minimize compositional dissimilarity (3) The power-line corridors are able to resist invasion by exotic plant species and provide refuges for cultural landscape species and (4) environmental factors particularly site index plays a significant role in shaping the species composition of power-line corridors. More importantly, this thesis discusses the ecological significance of these findings and offers the following suggestions for management consideration. The power-line corridors should be maintained with the aim of preserving the richness and abundance of shrubs and forbs which provide food and shelter for several other species, while also helping to increase the corridor habitat's resistance to invasion by exotic species.

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