

Nature Conservation – achievements and challenges within its first four years

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To be effective, research on natural resource management and conservation must be communicated to practitioners involved in hands-on conservation efforts and to policy makers. However, the results of scientific research are often not readily applied in management. Likewise, many applied conservation schemes do not reflect current research knowledge. The “knowledge-implementation-gap” (Knight et al. 2008) is becoming increasingly obvious. As a consequence, the 10th Party of the Convention on Biological Diversity, in Nagoya held in October 2010, identified a strengthened link between science and policy as an explicit target (<http://www.cbd.int/sp/targets/>). This requires new alliances between science, economics, policy makers, and natural resource managers (Briggs and Knight 2011).

Four years ago the journal *Nature Conservation* was established to address these challenges (Henle et al. 2012). It had and still has as a major goal to support synergistic interactions among scientists, policy-makers and managers. This is a practical task. The knowledge base of conservation biologists is already extensive, and the numbers of experienced practitioners are increasing around the world. The task is to bring different specialists together and create a forum that supports knowledgeable practices, and to learn from the experience – successes and failures – of all parties. The journal specifically aims at strengthening the link between science, policy and management by publishing timely, innovative papers with clear practical relevance.

Strengthening the link between science, policy and management is not only a major challenge for applied biodiversity conservation (Carmen et al. 2015) but also for journals, especially new ones. Scientists are primarily rewarded for the number of publications in international journals with high impact factors (Alberts 2013). However, many of these journals are not accessible to those working in nature conservation management or policy. Achievements in the transfer of knowledge from science to policy and applied nature conservation are more difficult to measure. These difficulties were faced initially also by *Nature Conservation*. In the first three years it was challenging to obtain a sufficiently larger number of articles that were both scientifically of high quality and at the same time highly relevant for nature conservation.

Despite these challenges *Nature Conservation* managed to publish 12, 13, and 13 articles in 2012, 2013, and 2014. The number grew to 21 in 2015, and after the acceptance for tracking by two of the largest abstract and citation databases of peer-reviewed literature Thomson Reuters' Web-of-Science and Scopus, the number of submissions has recently increased. The rejection rate in the first four years was 60%. Most of the published articles were research articles (Fig. 1). Although there is some geographic bias in the submission of articles, we are proud that the geographic origin of authors is very broad, already comprising authors from 38 countries (Fig. 2). An overview of the most productive authors can be found at http://natureconservation.pensoft.net/most_productive_authors.

The two most frequently viewed articles published in the first four years is by van Sway et al. (2012) on the Dos and Don'ts for the butterflies of the Habitats Directive of the European Union (viewed 17978 times) and by Kideghesho et al. (2013) on

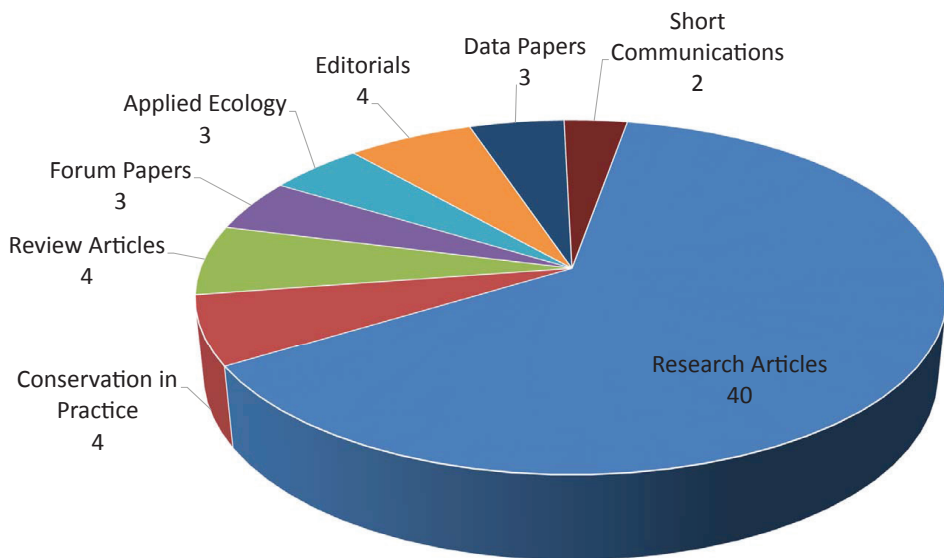


Figure 1. Distribution of published articles by type.

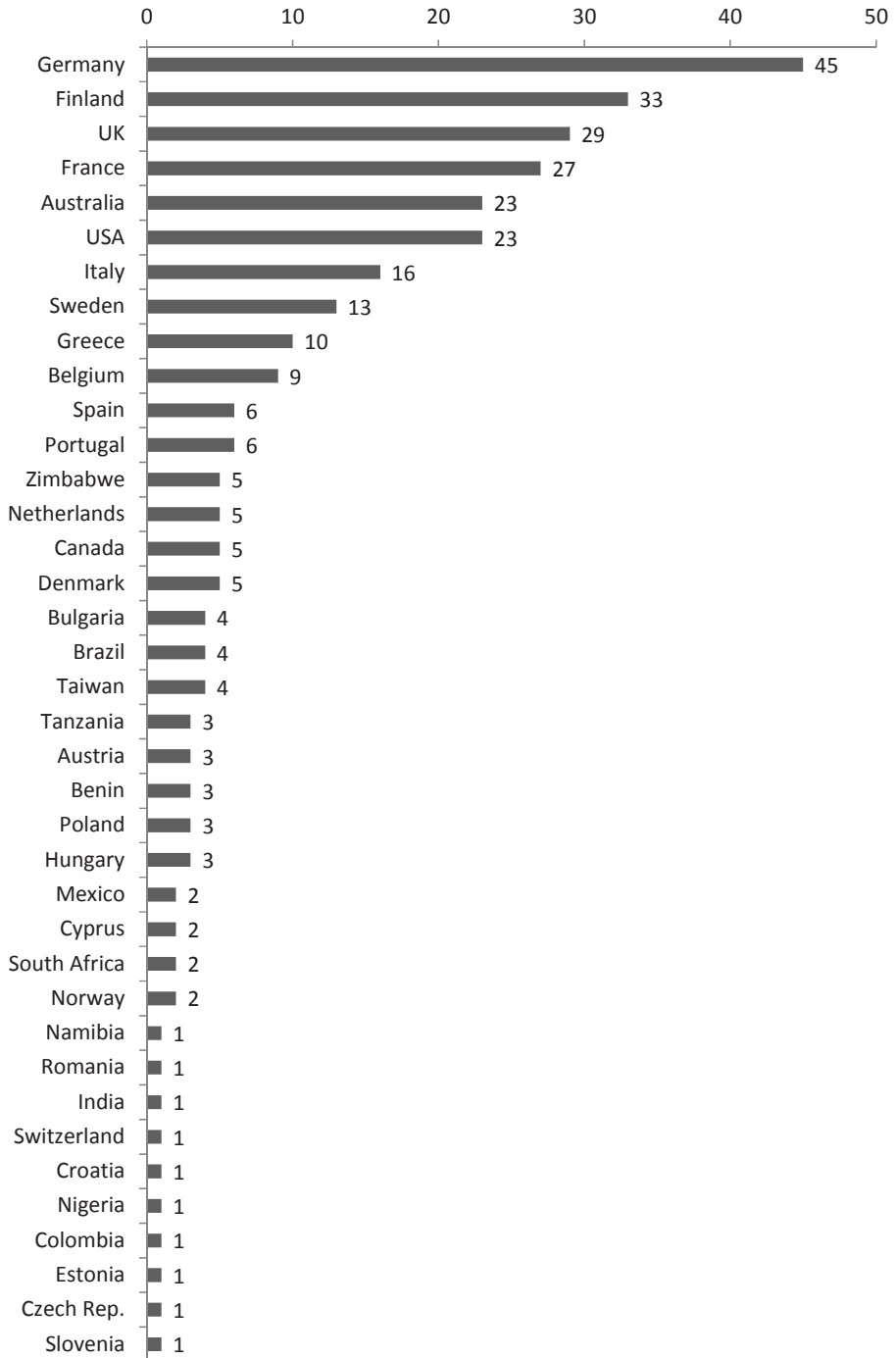


Figure 2. Distribution of authors by origin.

challenges in biodiversity conservation in Tanzania (viewed 14384 times). A further list of frequently viewed articles can be found at http://natureconservation.pensoft.net/browse_journal_articles?sortby=1. The large number of views indicate that these papers are viewed also by those working in or interested in applied nature conservation.

Nature Conservation specifically facilitate authors in generating impacts in applied nature conservation. The Public Relation team of Pensoft supports authors in generating news stories on papers that appeared in *Nature Conservation*. For example news stories on the illegal trade of the Indian star tortoises were published, among others in *The Guardian*; *National Geographic*; *Science News Magazine* and the butterfly publication of van Sway et al. (2012) featured in the European Research and Development Information Service (CORDIS). Similarly, press releases on Eurekalert obtained high numbers of views by science journalists e.g., “Aging nestling Carnaby’s cockatoo, *Calyptrorhynchus latirostris*, and estimating the timing and length of the breeding season” (2,932 views) and “A critical review of the Mediterranean sea turtle rescue network: a web looking for a weaver” (2,612 views).

Quality journals cannot exist without authors profiting from publishing in the journal and without the voluntary work of reviewers and editors. We are deeply grateful to all the reviewers and editors (named at http://natureconservation.pensoft.net/most_active_reviewers and http://natureconservation.teodor.pensoft.dev/most_active_editors) that helped achieving a timely evaluation of all published articles.

We hope that our readers enjoy the publications in *Nature Conservation* and will consider submitting manuscripts that may make a difference for biodiversity policy and management and nature conservation at large.

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Tree species composition rather than biodiversity impacts forest soil organic carbon of Three Gorges, southwestern China

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Abstract

Forest soil represents an important resource for mitigating the climate change. Besides, plant composition and diversity and their roles in ecosystem functioning are becoming a central issue in forest soil organic carbon (SOC) research. The primary objective of this research is to investigate the effects of tree species diversity and composition on potential of C sequestration of forest soil in Three Gorges area and provide basic information to future research on climate change. Two dominant forest ecosystems were selected: mixed conifer-broadleaf forest (F_m) and evergreen broadleaf forest (F_b). Then study transects were established and investigated. Soil samples were collected and determined for bulk density, SOC concentration and stock, nitrogen (N) concentration and C:N ratio. The results showed that the statistical differences of SOC concentrations and stocks between F_m and F_b were caused by tree species composition rather than the tree species diversity. And the most significant differences were found in the first two soil horizons (0–15 cm and 15–30 cm). The average C:N values of four different horizons in F_m were decreased with increasing soil depth as well as F_b . Not only SOC concentrations but also stocks of the two studied forests were decreased with increasing soil depth. However, F_m showed a larger capacity to store SOC with an average stock of 183.50 t/ha than that of F_b (100.44 t/ha) in study area. Thus, forest which is composed of conifer and evergreen broadleaf tree species may be the best choice for local afforestation and reforestation aimed at alleviating climate change in Three Gorges region.

Keywords

soil organic carbon, tree species composition, biodiversity, C:N ratio, Three Gorges

Introduction

Scientists have long been concerned with soil carbon (C), because it is often the master variable determining soil fertility (Malhi et al. 1999; Johnson and Curtis 2001; Johnson et al. 2002). C enters the soil through both litterfall and rhizodeposition and leaves the soil mainly as CO₂ via root and microbial respiration (Sulzman et al. 2005; Cleveland et al. 2010; Díaz-Pinés et al. 2011; Sayer et al. 2011). However, as one of the most important green house gases, the tightly relationship between CO₂ and soil C is generally accepted in the context of global climate change. Thus, known soil C stock has become very important for assessing changes in atmospheric CO₂ concentrations and of global climate (Dixon et al. 1994; Schimel 1995; Sørensen et al. 2004). As the largest pool of terrestrial organic carbon in the biosphere, more C is stored in soil than is contained in plants and the atmosphere combined (Jobbágy and Jackson 2000). Global surveys of mineral soil organic carbon (SOC) indicate that the soil holds about 1500 Pg C in the upper meter of soil (Post et al. 1982; Eswaran et al. 1993; Jobbágy and Jackson 2000), and most of this SOC (roughly 70% of all SOC) is contained in forest soils (Dixon et al. 1994; Batjes 1996; Jandl et al. 2007).

The potential C sequestration of forest ecosystems is widely accepted (Batjes 1996; Jandl et al. 2007). In fact, by sequestering large amounts of atmospheric C, forest plays an essential role in the global C cycle and is thought to offer a mitigation strategy to reduce global warming (Dixon et al. 1994; Chiti et al. 2012). However, the extent to which the vegetation layer influences SOC stocks in natural mountain forest land of Three Gorges area is still poorly understood. Moreover, many articles about SOC have been focusing on its stock of a large area, for example, global scale, hemispheric scale or national scale (Eswaran et al. 1993; Dixon et al. 1994; Batjes 1996; Fang et al. 2001; Goodale et al. 2002; Li et al. 2004; Chiti et al. 2012). The SOC of smaller scale, such as forest communities and ecosystems, is not considered enough, especially in the aspect of relation between SOC and forest composition and diversity. Composition of tree species has a pivotal effect on soil processes, including the cycling and accumulation of C (García-Oliva et al. 2006; Díaz-Pinés et al. 2011). For example, trees drive litterfall inputs, rhizodeposition, animal manure and rainfall distribution, soil temperature, and consequently they shift soil microbial quantity and activity (Simón et al. 2013). Based on composition, the forest area can be classified into various types. Proportion of different species in the same plant community can be quantified through species composition investigation. Therefore, it is important to consider the influence of tree species composition on SOC stock at given sites, as it may provide a basis for quantifying C pool in forest, which plays a relevant role in the global C cycle (Mathers and Xu 2003; Chen et al. 2004). Our study will be added to the growing body of information on soil C storage in subtropical mountain forest of China. But beyond

that, the relationship between plant diversity (i.e.: totality of genes, species, and ecosystems of a region) and biogeochemical process that regulates the ecosystem has been a central issue in both ecological and environmental sciences recently (Bunker et al. 2005; Chen 2006). Many studies have suggested that plant communities with high species diversity may promote more efficient use of resources compared with those of less species diversity and thus lead to greater net primary production, and consequently higher C sequestration (Saha 2008; Saha et al. 2009; Meier and Bowman 2010; Wang et al. 2011). However, Huston and Marland (2003) indicated that ecosystems with multiple species are not necessarily more productive than ecosystems with few species. Many natural ecosystems with low plant diversity, even near monocultures, are highly productive. Nevertheless, ecosystems with multiple species indeed provide some insurance that they may be steadier and continue to perform a particular function even if one of the species is lost. However, quantitative estimates of effects of tree species composition on SOC stocks under natural forest ecosystems remain scarce (Chapin III et al. 2000; Berger et al. 2002; Díaz-Pinés et al. 2011). Therefore, forest composition or biodiversity, which can be confirmed as the dominant effect on ecosystem C sequestration? The issue needs further researches.

In natural ecosystem, nitrogen (N) is a primary nutrient that limits vital activities of plant and microbe (Vitousek and Howarth 1991; Hu et al. 2001; LeBauer and Treseder 2008; Wei and Sun 2009). C cycling is consequently influenced by soil N and C:N ratio (Cleveland and Liptzin 2007; Cleveland et al. 2011), and both factors partly indicate activity of microbe and level of soil C decomposition by respirations of roots and microbes. Thus, both C stock and effects of C:N ratio have been hot spots of scientific interest in global change (Hungate et al. 2003; Chen 2006; Davidson and Janssens 2006).

Because of the alleviation effect on global warming, C sequestration ability of forest is expected for more and more focus (Wu et al. 2003; Lal 2004; Bonan 2008; Tarnocai et al. 2009). Especially in China, the large developing country all through the world, the conflict between environment and develop is becoming sharper and sharper. In order to reduce the green house gas, Chinese government has been struggling since a long time ago. Many measures have been conducted particularly in forestry. In last decades, although millions of hectares were planted (afforestation and reforestation) per year, making a huge C pool, the SOC stocks of forests in China have not restored from the continuously forestry C sequestration reducing since late 1940s (Fang et al. 2001). However, little attention was paid on the composition of tree species during silviculture and afforestation. It may cause inefficient C sequestration and cause unintended disastrous environmental consequences, especially in arid and semiarid regions (Gao et al. 2011). Nevertheless, C sequestration of natural forest should be studied in detail for “close-to-nature” afforestation and reforestation and finding the best forest management plan. Moreover, about 28 to 35% of forest C storage occurs in the southwestern region (including the provinces of Sichuan, Chongqing, Tibet, Yunnan, Guangxi, and Guizhou) which is the largest in China (Fang et al. 2001). Thus, forest C sequestration study in this region is important for

afforestation and reforestation aiming at reducing green house gas in China. Since the end of 20th century, the Natural Forest Protection Project has been conducted in Three Gorges of southwestern China (<http://english.forestry.gov.cn/index.php/information-services>). Vegetation coverage in this area was 35.62% by the end of 2007 and it was far greater than average of China (20.36%) (Zhao 2007). The natural forest ecosystems of the area are great potential for C stock. But study on this is still rarely showed. Therefore, the aims of this study are as follows: (1) Study the effects of forest composition on SOC concentration and stock. (2) Analyze the effects of tree species diversity on SOC concentration and stock. (3) Difference of SOC decomposition in different forests is showed by C:N ratio.

Materials and Methods

Description of research area

Our study was carried out at Jinyun Mountain, Three Gorges area, southwestern China. The forest area is totally 1112.7 ha which accounts for 96.6% local land area, and typical subtropical forest species are abundant. The study area is bounded by the two major river systems of the region, i.e., the Yangtze River and the Jialingjiang River. Elevation ranges from 350 to 952 m. This region has a subtropical monsoon climate with long warm to hot humid summers and short cool to cold and cloudy winters with the lowest total number of sunshine days in China (about 1000 hours per year). The mean annual temperature is 13.6 °C and the average annual precipitation is 1611.8 mm. Soil type is Kandihumults of Ultisols (Staff 2010).

Methods for investigating, sampling and determining

In our study, we investigated two natural forest ecosystems in April, 2011: the mixed conifer-broadleaf forest (F_m) and the evergreen broadleaf forest (F_b). These two forests are close to each other (separated from each other by approximately 100 m) and have similar elevation and same aspect. The basic information, including vegetation, soil and topography characteristics, is showed in Table 1. The total area of F_m was 17.3 ha, and the area of F_b is 12 ha. Transect method was performed to survey trees, shrubs and herbs. Soil samples were collected by establishing plots in transects. Parallel transects (100 × 40 m) separated by about 50 m, were established in forest F_m ($n=7$) and F_b ($n=5$). Then two 20 m × 20 m plots were randomly selected in each single transect. Unfortunately, only thirteen plots were set in F_m because of topographical reason. After this, the total inventory of all tree species was conducted in every plot. Shrub species were surveyed in three randomly selected 2 m × 2 m subplots involved in each 20 m × 20 m plot. And within each 2 m × 2 m subplot, herb species were recorded by setting one 1 m × 1 m quadrat. Plant species were recorded and counted. Biodiversity indices

Table 1. Basic information of studied forest ecosystems.

| Site | Dominant tree species | Shrub species | Herb species | Soil type | Range of slope (°) | Aspect | Mean elevation (m) | Range of canopy density | Total area (ha) |
|----------------|---|--|---|--------------------------|--------------------|--------|--------------------|-------------------------|-----------------|
| F _m | <i>Pinus masoniana</i> Lamb. <i>Cunninghamia lanceolata</i> (Lamb.) Hook. <i>Symplocos setchuensis</i> Brand <i>Lindera kuangtungensis</i> (Liou) Allen | <i>Maesa japonica</i> (Thunb.) Moritz. <i>Eurya nitida</i> Korthals <i>Sarcandra glabra</i> (Thunb.) Nakai <i>Smilax china</i> <i>Rosa multiflora</i> Thunb. <i>Eurya fangii</i> Rehd. <i>Elaeagnus bockii</i> Diels <i>Rubus assamensis</i> Focke <i>Rubus multifolius</i> Focke <i>Rubus corchorifolius</i> L. f. <i>Camellia cuspidata</i> (Köchs) Wright ex Gard. | <i>Woodwardia japonica</i> (L. f.) Sm. <i>Lophatherum gracile</i> Brongn. <i>Oplismenus compositus</i> (Linn.) Beauv. <i>Hicriopteris glauca</i> (Thunb.) Ching <i>Commelina communis</i> Linn. <i>Senoloma chusanum</i> Ching <i>Miscanthus sinensis</i> Anders. <i>Hemerocallis fulva</i> (L.) L. <i>Conyza canadensis</i> (L.) Cronq. <i>Phylla nodiflora</i> (L.) Greene | Kandihumults of Ultisols | 15–24 | N | 820 | 0.85–0.97 | 17.3 |
| F _b | <i>Lindera kuangtungensis</i> (Liou) Allen <i>Symplocos setchuensis</i> Brand <i>Castanopsis fargesii</i> Franch. <i>Adinandra bockiana</i> <i>Gordonia acuminata</i> Chang | <i>Maesa japonica</i> (Thunb.) Moritz. <i>Eurya nitida</i> Korthals <i>Sarcandra glabra</i> (Thunb.) Nakai <i>Smilax china</i> <i>Neolitsea aurata</i> (Hay.) Koidz <i>Eurya fangii</i> Rehd. <i>Symplocos lincifolia</i> Sieb. et Zucc. <i>Rubus multifolius</i> Focke <i>Camellia cuspidata</i> (Köchs) Wright ex Gard. | <i>Woodwardia japonica</i> (L. f.) Sm. <i>Oplismenus compositus</i> (Linn.) Beauv. <i>Hicriopteris glauca</i> (Thunb.) Ching <i>Commelina communis</i> Linn. <i>Senoloma chusanum</i> Ching <i>Miscanthus sinensis</i> Anders. <i>Hemerocallis fulva</i> (L.) L. <i>Conyza canadensis</i> (L.) Cronq. <i>Phylla nodiflora</i> (L.) Greene | Kandihumults of Ultisols | 12–28 | N | 822 | 0.88–0.94 | 12 |

were calculated according to the inventory process mentioned above. Then mineral soil samples were collected by depth (0–15, 15–30, 30–50 and 50–100 cm or bedrock when the profile is not deep down to 100 cm.) in all quadrates. Soils from 1 m × 1 m quadrates in the same 20 m × 20 m plot were mixed and homogenized by depth. Consequently, one composite sample of mineral soil of each single horizon was collected in a plot. The total number of soil composite samples of F_m was 50 and that of F_b was 34. These samples were transported to the lab shortly after sampling (Díaz-Pinés et al. 2011) and air dried in shade. Soil bulk density and volume proportion of gravel at each soil sampling horizon were determined according to Landsberg et al. (2003). The Kjeldahl method was carried out to obtain N concentrations of soil (Gong et al. 2012). The SOC concentrations were tested according to the dichromate acid wet oxidation method (Yeomans and Bremner 1988).

Calculation

SOC concentrations and stocks and their vertical distributions were studied. The statistical differences of SOC in 0–100 cm between the two studied forests were analyzed by T-test. The statistical differences of SOC in each horizon (i.e.: 0–15cm, 15–30 cm, 30–50 cm and 50–100 cm) between the two studied forests, as well as those among horizons, were analyzed by one-way ANOVA respectively. And this method was performed to test the differences between tree species diversity of the two researched forests. The results were summarized to explain the effects of tree species composition and diversity on SOC accumulation. In order to study the effect of tree diversity on SOC sequestration, the correlations between SOC and tree species diversity indices of F_m , as well as F_b , were then estimated by regression analysis. As an important controller of SOC decomposition, soil C:N ratio was also analyzed. One-way ANOVA was performed to evaluate the differences between C:N ratios of the two studied forests so as to understand the condition of SOC decomposition. Data analysis was implemented by using Microsoft Office Excel 2003 (Microsoft Corporation, US) and SPSS-17 (IBM Corporation, US).

SOC stock was calculated according to following formula:

$$ST = \sum_{i=1}^n C_i \times \rho_i \times h_i \times (1 - \theta_i) / 1 \quad (1)$$

Where ST is SOC stock (t/ha), i is soil horizon code, n is the number of soil horizons, C_i is SOC concentration (g/kg), ρ_i is soil bulk density (g/cm³), h_i is soil horizon thickness (cm), θ_i is volume proportion (%) of gravel with diameter (ϕ) >2 mm.

Tree species diversity was presented by following indices (Li and Li 2006):

Simpson's index (biodiversity index):

$$D = 1 - \sum_{i=1}^S \left(\frac{n_i(n_i - 1)}{N(N - 1)} \right) \quad (2)$$

Shannon - Wiener index (biodiversity index):

$$H = -\sum_{i=1}^S \frac{n_i}{N} \log_2 \left(\frac{n_i}{N} \right) \quad (3)$$

Margalef index (richness index):

$$R = \frac{S-1}{\ln N} \quad (4)$$

Where N is total number of trees in plot, i is tree species type, n_i is number of individuals of tree species i , S is number of tree species.

Results

SOC under the two studied forests

SOC concentrations of the studied forests remarkably decreased with increasing depth of mineral soil. These correlations could be simulated as follows:

$$SOC \text{ concentration}(F_m) = 19.967 \ln(\text{soil depth}) - 99.282 \quad (R^2 = 0.4712, n = 50, p < 0.001)$$

$$SOC \text{ concentration}(F_b) = 18.066 \ln(\text{soil depth}) - 79.115 \quad (R^2 = 0.5818, n = 34, p < 0.001)$$

SOC concentrations of F_m and F_b may be calculated by above empirical models. But it indeed needs more samples for accuracy.

Significant differences ($p < 0.001$) were found among the four soil horizons in both F_m and F_b (see Fig. 1A). The statistical difference of total SOC concentrations of 0–100 cm mineral soil between the two studied forests was significant with $p = 0.0016$ (see Fig. 1A and Table 2). It indicated that the average concentration of 0–100 cm SOC in F_m (85.62 g/kg) was remarkably larger than F_b (46.18 g/kg) (see Fig. 1A, Table 2). Compared with F_m , SOC of F_b (46.18 g/kg) only accounted for 53.94% of its SOC concentration. In the first two soil horizons, the SOC concentrations were even more remarkably different: 0–15 cm soil with $p = 0.0101$ and $p = 0.0338$ for 15–30 cm soil (see Fig. 1A, Table 2). In the other two horizons, that the p values were 0.2068 (30–50 cm) and 0.1539 (50–100 cm) respectively indicated insignificant differences between SOC concentrations of F_m and F_b . The most remarkable difference was found in the first mineral soil horizon. 0–15 cm SOC concentration of F_m (52.38 g/kg) was significantly larger than that of F_b (31.02 g/kg). The other horizons of F_m had greater SOC than F_b as well even though the statistical differences were not remarkable (see Fig. 1A, Table 2).

SOC stocks in 0–100 cm of F_m and F_b (Fig. 1B, Table 2) were statistically different ($p = 0.0052$). However, 0–15 cm SOC stocks of the two forests were not significantly different with a p value of 0.0843 in contrast with 0–15 cm SOC concentrations (see Fig. 1A, B and Table 2). But the statistical difference of SOC stocks in 15–30 cm soil was remarkable ($p = 0.0294$) (Fig. 1A). The SOC stocks, at the last two horizons, were

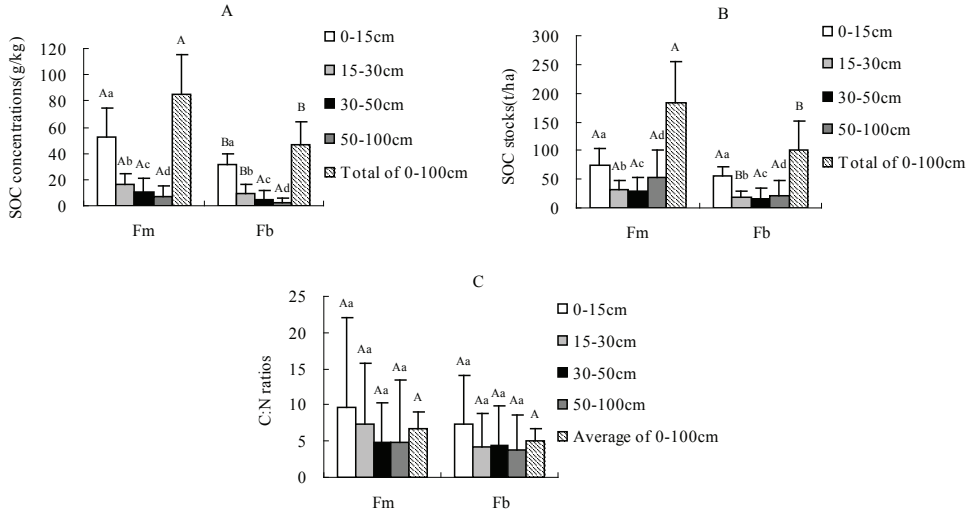


Figure 1. SOC concentrations (Fig. 1A), SOC stocks (Fig. 1B) and C:N ratios (Fig. 1C) of F_m and F_b . Where solid columns of different colors respectively show average values of each single horizon under the F_m and F_b . The columns with oblique lines are mean values of total SOC concentrations, total SOC stocks and average C:N ratios of 0-100 cm soil in F_m and F_b separately. Letters above each error bar indicate the statistical difference. The different capital letters show significant difference between value series of two forests ($p < 0.05$), for example, capital letters on top of the two white columns (A and B) show difference between average 0–15 cm SOC concentrations of F_m and F_b . The different lowercase letters present remarkable differences of values among different soil horizons within a studied forest ($p < 0.001$), for example, differences among SOC concentrations of 0–15 cm, 15–30 cm, 30–50 cm and 50–100 cm horizon of F_m were significant according to “a, b, c and d”.

both not statistically different between F_m and F_b as the same as those of SOC concentrations (see Fig. 1A, B and Table 2). Both SOC stocks of F_m and F_b along soil horizons were found to be significantly different ($p = 0.0013$ for F_m , $p = 0.00006$ for F_b , Fig. 1B). The change of SOC stocks from topsoil to bottom was performed as follows:

$$SOC\ stock(F_m) = 7.1205 \ln(\text{soil depth}) - 73.184 \quad (R^2 = 0.0325, n = 50, p < 0.5)$$

$$SOC\ stock(F_b) = 14.68 \ln(\text{soil depth}) - 87.567 \quad (R^2 = 0.2184, n = 34, p < 0.005)$$

SOC stocks of F_m and F_b were decreased from 0-15 cm to 30-50 cm firstly, then they were increased (Fig. 1B and Table 2).

As shown in Fig. 1B and Table 2, the average total SOC stocks (0-100 cm) ranged from 183.50 t/ha of F_m to 100.44 t/ha of F_b . The quantitative relationship between SOC stocks of F_m and F_b was consistent with that of SOC concentrations (see Fig. 1A, B and Table 2). In F_m , 62.91% of the total SOC down to 1 m was in the top 30 cm (Fig. 1B and Table 2). The proportion was even more in F_b (83.08%). However, the main difference of SOC stock was found in 15-30 cm soil rather than the first horizon (Fig. 1B and Table 2).

Table 2. Mean value (\bar{x}) \pm standard deviation (σ) of SOC concentration, SOC stock and C:N ratio. The p values which are less than 0.05 indicate significant difference between F_m and F_b .

| | F_m | F_b | | | | | |
|--------------------------|----------------------|----------------------|--------|--------|------------------------|--------|---------------------------|
| | $\bar{x} \pm \sigma$ | $\bar{x} \pm \sigma$ | p | F | Critical values of F | t | Statistical values of t |
| SOC concentration | | | | | | | |
| 0–15 cm | 52.38 \pm 22.49 | 31.02 \pm 8.88 | 0.0101 | 7.9880 | 4.3248 | — | — |
| 15–30 cm | 16.45 \pm 7.66 | 9.51 \pm 6.37 | 0.0338 | 5.1562 | 4.3248 | — | — |
| 30–50 cm | 10.62 \pm 9.77 | 5.19 \pm 6.31 | 0.2068 | 1.7229 | 8.3997 | — | — |
| 50–100 cm | 7.57 \pm 7.52 | 2.89 \pm 4.41 | 0.1539 | 2.2275 | 8.3997 | — | — |
| 0–100 cm | 85.62 \pm 30.17 | 46.18 \pm 18.44 | 0.0016 | — | — | 2.0796 | 3.6342 |
| SOC stock | | | | | | | |
| 0–15 cm | 75.23 \pm 29.09 | 56.96 \pm 14.58 | 0.0843 | 3.2841 | 4.3248 | — | — |
| 15–30 cm | 32.03 \pm 14.88 | 18.70 \pm 11.55 | 0.0294 | 5.4666 | 4.3248 | — | — |
| 30–50 cm | 29.00 \pm 24.38 | 15.09 \pm 18.60 | 0.2114 | 1.6865 | 4.4513 | — | — |
| 50–100 cm | 53.59 \pm 47.01 | 20.30 \pm 28.37 | 0.1092 | 2.8583 | 4.4513 | — | — |
| 0–100 cm | 183.50 \pm 71.59 | 100.44 \pm 50.38 | 0.0052 | — | — | 2.0796 | 3.1159 |
| C:N ratio | | | | | | | |
| 0–15 cm | 9.63 \pm 12.47 | 7.44 \pm 6.71 | 0.5853 | 0.3071 | 4.3248 | — | — |
| 15–30 cm | 7.30 \pm 8.41 | 4.27 \pm 4.45 | 0.3162 | 1.0544 | 4.3248 | — | — |
| 30–50 cm | 4.82 \pm 5.54 | 4.51 \pm 5.36 | 0.8910 | 0.0194 | 4.4513 | — | — |
| 50–100 cm | 4.89 \pm 8.53 | 3.83 \pm 4.72 | 0.7780 | 0.0821 | 4.4513 | — | — |
| 0–100 cm | 6.66 \pm 2.29 | 5.01 \pm 1.64 | 0.4464 | 0.8175 | 3.9574 | — | — |

C:N ratios of F_m and F_b

Concentrations of SOC and soil N of F_m ($r=0.6656$, $n=50$, $p<0.001$) were linearly and remarkably correlated as well as those of F_b ($r=0.5566$, $n=34$, $p<0.001$). The results showed that soil N may have important effects on SOC. However, as a metric of SOC quality, the soil C:N ratios of the studied forests were not statistically different (Fig. 1C and Table 2). The above results indicated that C:N ratio may not lead to the differences of SOC between the two studied forests. However, the C:N ratio was decreased with increasing soil depth (Fig. 1C and Table 2). From the first to the forth horizon, the average C:N values of F_m were respectively 9.63, 7.30, 4.82 and 4.89 (Fig. 1C and Table 2). And those of F_b were 7.44, 4.27, 4.51 and 3.83 (Fig. 1C and Table 2). Although the average C:N ratio of 30–50 cm soil under F_b was larger than 15–30 cm soil, and the mean C:N ratio of 50–100 cm under F_m was larger than 30–50 cm soil, the C:N ratio was also generally decreased from 0–15 cm with maximum to 50–100 cm with minimum.

Relationship between tree species diversity and C

Average values of tree species diversity indices were shown in Table 3. That the average values of diversity indices of F_m were greater than those of F_b except for D indicated

Table 3. Average values of tree species diversity indices.

| Forests | R (Margalef) | H (Shannon-Wiener) | D (Simpson's) |
|----------------|--------------|--------------------|---------------|
| F _m | 1.542 | 1.832 | 0.693 |
| F _b | 1.406 | 1.663 | 0.812 |

that the biodiversity (H) and richness (R) were the best in F_m, whereas another biodiversity index D of F_b was the greatest. And the one-way ANOVA analysis supplied the estimation: the statistical difference between biodiversity indices of F_m and F_b was not significant. According to the linear correlation analysis between SOC and tree species diversity of F_m and F_b, the SOC concentrations, including SOC in each individual horizon and total soil profile, were not significantly correlated with the three diversity indices as well as SOC stocks ($p>0.05$).

Discussions

The main differences of SOC concentrations of F_m and F_b were presented in 0–15 cm and 15–30 cm soil with p value of 0.0101 and 0.0338 respectively. The reason may be that roots are mainly distributed in 0–50 cm soil horizon (Waisel et al. 1991; Upson and Burgess 2013). However, the difference of SOC concentrations was insignificant in 30–50 cm horizon ($p=0.2068$). The SOC stocks of F_m and F_b were only significantly different in 15–30 cm horizon ($p=0.0294$). Totally, SOC concentrations of 0–100 cm in the two forests ($p=0.0016$) were significantly and statistically different as well as SOC stocks ($p=0.0052$) (Fig. 1A, B and Table 2). Thus, tree species compositions of forest ecosystems could be considered as a reasonable factor for distinguishing SOC from each other especially in surface soil (0–30 cm). Currently, Chinese government has been carried out many protection programs of forest in order to build a healthy natural ecosystem, for example, Land Conversion from Farmland back to Forestland Project, Wildlife Protection and Nature Reserve Development Program and Natural Forest Protection Project, etc (<http://english.forestry.gov.cn/index.php/information-services>). Simultaneously, afforestation and silviculture are implemented all through the country to decelerate global warming. Ecological conditions of China have a continual improvement and C sink potential keeps increasing. However, problems are also existed: monoculture afforestation, lack of forest management, and contradiction between food shortage and returning crop land to forest. Forest quality is influenced by those problems. These issues should be properly solved. In Three Gorge area, because of our SOC stock estimates, mixed conifer-broadleaf forest (F_m) with the largest soil C pool (183.50 t/ha) may be the best choice for local afforestation and reforestation aimed at alleviating climate change.

Liu (2005) suggested that broadleaf forest was climax communities in succession process of Mt. Jinyun with mixed conifer-broadleaf forest being inferior community. Several works (Malhi et al. 1999; Marín-Spiotta and Sharma 2013) suggested that both

land use change and forest succession gradient were generally thought to have effect on SOC stocks, especially for surface soil. Nevertheless, the successional effects of forest communities in Three Gorges are still not understood well. Thus, the specific studies under local conditions are very necessary. Not only the 0–15 cm SOC concentrations were remarkably different ($p=0.0101$) between F_m and F_b , but also the 15–30 cm SOC concentrations were significantly different from each other ($p=0.0338$). Although insignificant, the difference of 0–15 cm SOC stocks between forests was indeed existent with p value of 0.0843. Nevertheless SOC stocks of 15–30 cm soil were remarkable different between F_m and F_b ($p=0.0294$). And the differences of SOC concentrations and stocks between F_m and F_b became weaker and weaker with increasing soil depth where roots of bottom soil were far less than surface soil. The above analysis showed that succession effects may be another reason which could control SOC stock of forest by influencing tree species composition.

Our total C stock estimates of 0–100 cm mineral soil under the two forest ecosystems (100.44–183.50 t/ha, Fig. 1B and Table 2) were beyond the range of values estimated for the mineral soil under forests of Mt. Dinghu with the similar climate (30.90–127.90 t/ha) (Fang et al. 2003), but included the estimate for the Ultisols soil (144.80 t/ha) in Chongqing city (Huang et al. 2005) (Table 4). In Mt. Dinghu (Fang et al. 2003), the SOC stocks in mixed conifer-broadleaf forests (30.90–107.10 t/ha) were less than those of evergreen broadleaf forests (95.00–127.90 t/ha). Chen (2007) suggested that mixed conifer-broadleaf forest (92.33–127.13 t/ha) sequestered less C than evergreen broadleaf forest (151.63–290.82 t/ha) in Three Gorges region (Table 4). The results were contrary to our data. However, study in Spain showed that mixed conifer-broadleaf forest caught more SOC than evergreen broadleaf forest as well as in other regions of Mediterranean conditions which belongs to subtropics as the same as our research area (Díaz-Pinés et al. 2011) (Table 4). Li et al. (2004) also suggested that SOC stock of evergreen broadleaf forest (129.2 t/ha) in China was less than that of mixed conifer-broadleaf forest (225.70 t/ha) (Table 4). Ni (2001) estimated the SOC stocks of the two types of forests in China: 124.00–142.00 t/ha for evergreen broadleaf forest and 130.00–150.00 t/ha for mixed conifer-broadleaf forest (Table 4). Mixed forest caught more SOC than evergreen broadleaf forest. The differences of C stock among regions may be also due to climatic (Díaz-Pinés et al. 2011; Chiti et al. 2012) and geologic conditions etc (Schaefer et al. 2009). Besides, sampling time may also affect estimating value of C stock in forest soil. However, in southwestern China, the average forest biomass C stock was 60 t/ha which was the largest all through the country (Fang et al. 2001). And that the SOC is far more than biomass C is widely accepted. It indicated that natural forest in this region is a great container for C. Our results were greater than the average SOC stock of Ultisols soil on the Earth (Eswaran et al. 1993), which also showed the strong C sequestration of forests in the research area (Table 4). However, Woodwell (1984) indicated that Ultisols soil under virgin and secondary forests on Earth stored more SOC (180.00–240.00 t/ha) than the two forests in this study (Table 4).

Both the average soil C:N ratios of F_m and F_b were decreasing with increasing soil depth. And the average C:N ratios of F_b were less than those of F_m in each soil horizons.

Table 4. Published values of SOC in comparable mixed conifer-broadleaf forest and evergreen broadleaf forest in subtropical region.

| Locatoion | Vegetation type | SOC stocks (t/ha) | Soil type | Source |
|--|--------------------------------|-------------------|--------------------------|--------------------------|
| Earth | Virgin and secondary forests | 180.00–240.00 | Ultisols | Woodwell (1984) |
| Earth | —— | 83.00 | Ultisols | Eswaran et al. (1993) |
| China | Mixed conifer-broadleaf forest | 130.00–150.00 | —— | Ni (2001) |
| China | Evergreen broadleaf forest | 124.00–142.00 | —— | Ni (2001) |
| Fujian, China | Mixed conifer-broadleaf forest | 30.90–107.10 | Ultisols | Fang et al. (2003) |
| Fujian, China | Evergreen broadleaf forest | 95.00–127.90 | Ultisols | Fang et al. (2003) |
| China | Mixed conifer-broadleaf forest | 225.70 | —— | Li et al. (2004) |
| China | Evergreen broadleaf forest | 129.20 | —— | Li et al. (2004) |
| Chongqing, China | —— | 144.8 | Ultisols | Huang et al. (2005) |
| Three Gorges region, China | Mixed conifer-broadleaf forest | 92.33–127.13 | Ultisols and Alfisols | Chen (2007) |
| Three Gorges region, China | Evergreen broadleaf forest | 151.63–290.82 | Ultisols and Alfisols | Chen (2007) |
| Central Spain | Mixed conifer-broadleaf forest | 80.00–100.00 | Inceptisols and Alfisols | Díaz-Pinés et al. (2011) |
| Central Spain | Evergreen broadleaf forest | 40.00–70.00 | Inceptisols and Alfisols | Díaz-Pinés et al. (2011) |
| Three Gorges region, China (Chongqing section) | Mixed conifer-broadleaf forest | 183.50 | Ultisols | This study |
| Three Gorges region, China (Chongqing section) | Evergreen broadleaf forest | 100.44 | Ultisols | This study |

The C:N ratio provides some indication about the relative quality and biochemical stability of soil organic materials (Díaz-Pinés et al. 2011; Bui and Henderson 2013). The C:N ratio hinted a weak SOC decomposition in our studied forests. Therefore, SOC stock of study area was larger than the average value of Utisols soil on the earth (Eswaran et al. 1993) as well as forest soil in southwestern China (Fang et al. 2001). However, the C:N values of F_m and F_b were not statistically different ($p=0.3879$). On the other hand, SOC concentrations ($p=0.0474$) and stocks ($p=0.0116$) of F_m and F_b were remarkably different. The results showed that SOC differences between F_m and F_b were influenced by C:N ratio little, which indicated that SOC decompositions of F_m and F_b were similar. The composition of tree species may be a rational factor for distinguishing the differences between C sequestrations of forests in study area as above analysis.

The relationship between tree species diversity and SOC under studied forest ecosystems was not linear in our study. However, Chen (2006) suggested that SOC stocks were linearly increased with growing H indices of forests in Northeastern China. Nevertheless, in Sichuan Province of southwestern China (closely located in the west of

our study area), the correlations between SOC and R and H of forests were as the same as our findings (Zhang et al. 2011). Kirby and Potvin (2007) did not find any linear relationship at soil profiles under forests in Eastern Panama either. The environmental factors of different region (Ewel et al. 1991; Berendse 1998; Forrester et al. 2006) and various forest productivities (Vandermeer 1989; Tilman et al. 1997) may cause the different relationships. And more studies are needed to explain the correlation between biodiversity and SOC in order to develop forest management and establish forest with great C sequestration. However, the statistical difference between biodiversity indices of F_m and F_b was not significant (Table 3). Thus, in contrast to plant species composition, biodiversity may not make difference in forest soil C sequestrations.

Conclusions

Tree species composition significantly and statistically influenced SOC concentrations and stocks of F_m and F_b . In first two soil horizons (0-15 cm and 15-30 cm), these differences were even more significant. However, SOC of F_m and F_b were not influenced by tree species diversity due to the very low linear coefficients. And the statistical difference between biodiversity indices of F_m and F_b was not significant. Thus, in contrast to plant species composition, biodiversity may not make difference in forest soil C sequestrations. The average C:N values of F_m in four different horizons were decreased with increasing soil depth as well as F_b . And the values were larger in F_m . But the difference between C:N ratios of F_m and F_b was not remarkable. C:N ratio contributed little to the difference between SOC of the two studied forests. Not only SOC concentrations of F_m and F_b were decreased with increasing soil depth but also SOC stocks reduced from surface soil to bottom. F_m showed a large capacity to store SOC rather than F_b in the area. Thus, mixed conifer-broadleaf forest may be the best choice for local afforestation and reforestation aimed at alleviating climate change in Three Gorges region. However, conflict issues can still be found in the relation between SOC and tree species diversity in studies all over the world. It needs more detail researches in different scale to explain.

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An integrated approach for landscape contrast analysis with particular consideration of small habitats and ecotones

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Abstract

Habitat diversity is considered as an expression of biodiversity at landscape level in addition to genetic and species diversity. Thus, effective methods for measuring habitat pattern at landscape level are crucial for understanding the ecological processes. In this paper we propose to extend the commonly used model Patch Corridor Matrix Model (PCMM) for spatial pattern analysis. Originally, this model assumes discrete structures within the landscape without explicit consideration of “gradients” between patches. The gradients, often called “ecotones”, can be considered as “soft edges” which have a profound influence on adjacent ecosystems. Another part of information that has often been ignored are “small habitats” inside patches (e.g. hedgerows, tree rows, copse, and scattered trees), which leads to within-patch heterogeneity being underestimated. In this paper, the concept of landscape contrast is used to integrate the discrete and gradient landscape representations by incorporating small habitats and ecotones in methods to measure landscape heterogeneity. A height gradient is used to define the ecotones between forest and field. Then, patch contrast (i.e. Edge Contrast Index (ECON)) is calculated based on the height difference between adjacent vegetation patches. Artificial elements (e.g. traffic roads) are considered as barriers which are assigned with the highest edge contrast value. At the landscape level, a metric called Area-Weighted Edge Contrast (AWEC) is introduced to describe the landscape structure. The edge effects of ecotones, small habitats, and traffic roads are incorporated in the calculation of AWEC. Our test examples show that incorporation of ecotones and small habitats can smooth “edge effects” among patches and result in a more realistic quantification of habitat contrast. The contrast concept is especially useful in a vegetated landscape with less human impact. It could be understood as an additional interpretation to fragmentation of habitats with permeable edges among them. Consequently, this presented approach may enhance the understanding of the relationship between landscape pattern and process.

Keywords

Patch corridor matrix model; Gradient model; Ecotones; Small habitats; Habitat fragmentation

Introduction

Landscape metrics based on the mosaic model are often used in landscape analysis. In practice the application of this model may be over simplified by losing valuable information on the landscape structure, such as the terrain characteristics of landscape (Hoechstetter 2008, Walz et al. 2007), transitional areas between patches (Kent et al. 1997), and small habitats within patches (Hou and Walz 2013). Kent et al. (1997) defined transitional area as a subset of landscape boundaries that represents the zones between plant communities with some degree of naturalness, as opposed to the sharper demarcations that usually occur between land-use types. Ecotone as a type of transitional area indicates the overlap or zone of relatively rapid change between two plant communities (Forman 1995, Kent et al. 1997). It has a profound influence on adjacent ecosystems, for example, ecotones control the flux of materials and energy between ecosystems (Fortin et al. 2000), functioning as ecological boundaries that contribute to the spatial heterogeneity of the landscape (Cadenasso et al. 1997, Fagan et al. 2003, Holland et al. 1991, Senft 2009, Strayer et al. 2003). Small and linear vegetation patches (e.g. scattered trees, hedgerows, tree rows and groves) are of high natural value for the conservation of biodiversity (Ernault and Alard 2011, Forman 1995, Morelli 2013). The main functions of these small habitats in ecosystem are either providing habitats for some edge species or forming a network to strength the species movement, such as hedgerow network (Burel and Baudry 1995, Forman and Baudry 1984).

With the development of remote sensing technology, it is possible to direct map the small habitats or discriminate different types of habitats occurring in spatially contiguous units (Bunting and Lucas 2006, Corbane et al. 2015, Hill et al. 2007, Hirschmugl et al. 2007). Especially the combination of LiDAR (Light Detection And Ranging) data and high resolution images has been proved to be useful in mapping tree crowns and measuring individual tree structure (Holmgren et al. 2008, Hou and Walz 2014, Morsdorf et al. 2004, Smart et al. 2012). However, the advantages of remote sensing technology in habitat mapping are not fully utilized. Among the large amount of existing landscape metrics, there is still lack of metrics which can fully incorporate ecotones and small habitats in the landscape structure analysis. The metrics used for analyzing landscape structure are dependent on the conceptual model for representing the landscape. McGarigal et al. (2009) introduced surface metrics as an alternative to patch metrics for the quantification of landscape gradient structure. Hoechstetter et al. (2011) used lacunarity analysis to analyze gradual value progressions in landscape systems. The both methods consider the landscape as a continuous surface instead of the patch mosaic model. The surface metrics are derived from a raster based data in which the only discrete unit is a pixel or grid cell (Lausch et al. 2015). In this paper,

the landscape is regarded as a mosaic with discrete patches and permeable boundaries between them (intermediate edge contrast). The focus is on developing or adapting suitable metrics to incorporate the ecotones and small habitats as crucial factors in the analysis of landscape structure.

Landscape heterogeneity has been integrated into metapopulation theory by incorporating habitat fragmentation and landscape contrast (Biswas and Wagner 2012). The fragmentation indices consider both composition and spatial pattern of landscape, but the boundaries between patches are either regarded as not permeable (highest contrast) or as full permeable (no contrast). In other words the patch borders are abrupt in the use of fragmentation concept. In contrast, landscape contrast has been considered as a crucial factor for assessing habitat pattern across different scales (Biswas and Wagner 2012, Schindler et al. 2008). Edge contrast affected the magnitude of edge effects, with a tendency for stronger responses to old and tall plantations (hard edges) than to young and short plantations (soft edges) (Reino et al. 2009). For example, in the form of passive dispersal, seeds will accumulate on the forest boundary as plants dispersed by wind; or the “terrain barriers” can act as obstacles for the movement of certain species. Ecotones can reduce the edge contrast value on both edges of plant communities. In particular, the degree of patch contrast may influence species dispersal patterns, and thus indirectly affect the degree of patch isolation. The objective of this research is to integrate both ecotones and small habitats in landscape contrast analysis which results in a detailed and comprehensive description of landscape pattern.

Methods

Applied concepts and test sites

In this paper, the ecotone is defined at a detailed spatial level as a “soft” boundary between forest and field. It has a three dimensional structure appearing as gradual blending of the two plant communities on the boundary area, where the third spatial dimension (vegetation height) is used to constrain the transition zone on forest-field boundary. It refers to mixed vegetation above the field layer but below the overstory formed by a combination of side branches of canopy trees, small trees, lianas, and shrubs. The small habitats (including single trees, tree rows, hedges, and copses) are defined by an area less than 0.5-1 hectare, a minimum width of 5 m and the occurrence in the field, isolated from forest (BfN 2002). The small habitats can be distinguished from their shape features. For example, hedges are defined as shrub-dominated structures, while a copse is characterized by several or dominating trees in the vegetation stand. A tree row is a line of trees exhibiting a long and narrow outline.

Two test sites with varied landscape structure are selected from the German national park “Saxon Switzerland”, which is located in south-eastern Germany (Figure 1). It is a mountainous area largely covered by forest, encompassing several types of land use structures and classes, mainly including rural settlements and surrounding agricultural

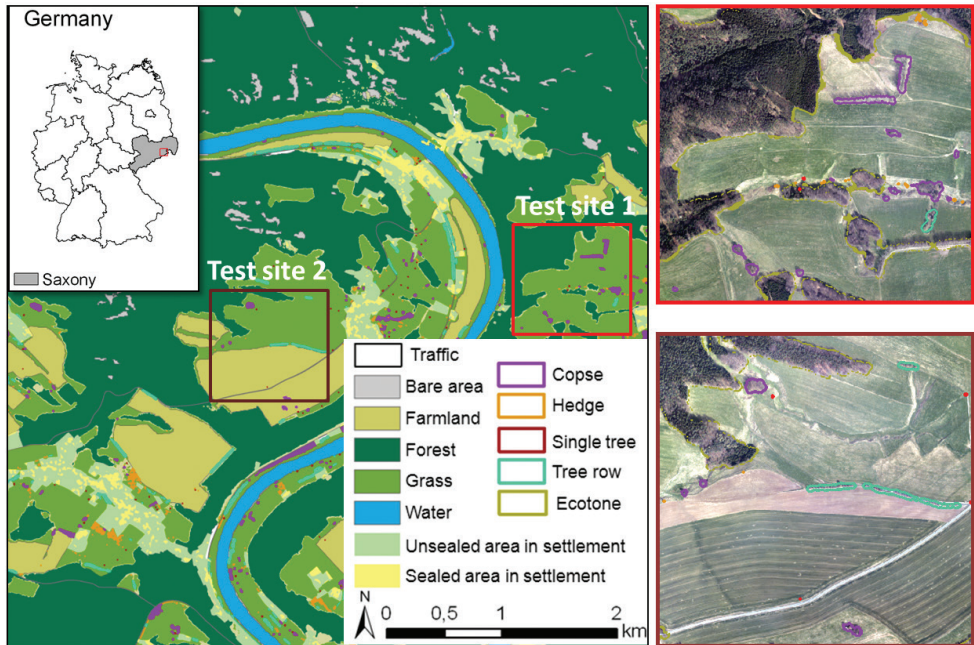


Figure 1. Classified land use maps including small habitats and ecotones in two test sites located in Saxon Switzerland.

land. A very detailed land cover map of this region including ecotones and small habitats was produced from the combination of RapidEye remote sensing images and a high resolution normalized Digital Surface Model (nDSM, 1 m resolution) which was derived from LiDAR data (see detail in Hou and Walz 2014).

Patch contrast

Patch contrast is used to describe the relative difference between patches or patch classes; for example “edges” have a kind of “contrast effect”. A strong contrast value means that adjacent patches differ strongly and the transitions between them are narrow or even absent (Forman 1995). Of relevance to the contrast of vegetation cover, the “dissimilarity” or “edge contrast weight” can be derived from the difference in height among habitats. The contrast value is highly related to the conceptual model used for simulating the landscape. Categorical landscape models ignore within-patch heterogeneity and emphasize contrast between adjacent patches. Specifically in this research, ecotones between forest and field are defined as height gradient and the boundary behavior is related to the transition forms, such as a thin border or a broad transition zone with mixed vegetation. In this case, the vertical structure is used as a means that integrates discrete and gradient forms of spatial heterogeneity. Such differences have rather easily deducible ecological consequences. The forest along an ecotone is less iso-

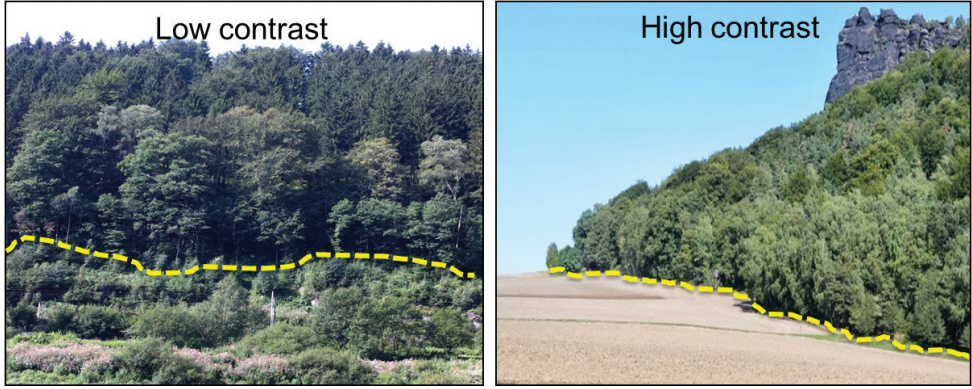


Figure 2. Examples of contrast magnitude along patch edges in “Saxon Switzerland”, Germany (Photos: Hou).

lated than along the bare soil (agriculture land) (Figure 2.2). Artificial elements, such as traffic roads, are considered as barriers which present high contrast and can be assigned with the highest edge contrast value.

In this context, a height-based variant of the Edge Contrast Index (ECON) (Hoechstetter 2009) has been used for characterizing patch contrast (Formula 1). ECON equals the sum of the patch perimeter segment lengths p_k multiplied by their corresponding contrast weights (d_k), divided by the total patch perimeter (P), then converted into a percentage value (multiplied by 100). The dissimilarity value d_k assigns values between 0 and 1, with a value of 0 being assigned to the minimum difference in mean height between two adjacent patches. Conversely, a value of 1 is assigned to the maximum difference in mean height between two adjacent patches, edge segments along the landscape boundary are assigned $d_k = 0$. An in-between dissimilarity value is assigned according to the proportion of height difference to the maximum difference. In this paper, the minimum and maximum height differences are set in 0 m and 20 m.

$$\text{Formula 1: } ECON = \frac{\sum_{k=1}^m (p_k \times d_k)}{P} \times 100 ,$$

p_k : edge length of segment k ;

d_k : contrast weight of segment k ;

P : total patch perimeter;

m : number of patch segments.

Range: $0 < ECON \leq 1$

Landscape contrast

At a higher organizational level, it could be misleading to simply calculate the mean edge contrast for a particular patch type (class level) or for all patches (landscape level). An irregular-shaped small patch may play a disproportionately role in the overall land-

scape contrast. Therefore, a metric which refers to the patch proportions has been developed at the landscape level: Area-Weighted Edge Contrast (AWEC) (Formula 2). It is not only an accumulation of the edges' contrast value; meanwhile the area proportion of each patch is also incorporated. Based on the modified ECON, AWEC can be understood as average dissimilarity in vertical structure of habitats. The lowest value of AWEC is 0 when the whole landscape is considered as one patch (landscape boundary is assigned with dissimilarity of 0), and the highest value is 1 as all patches have hard edges (maximum dissimilarity).

$$\text{Formula 2: } AWEC = \frac{\sum_{i=1}^n (a_i \times ECON_i)}{A}$$

n: number of patches in the landscape;

a_i : area of patch i ;

$ECON_i$: the edge contrast value of patch k , see Formula 1.

A: area of the total landscape.

Range: $0 \leq AWEC \leq 1$

Results

Comparison of contrast analysis with and without consideration of ecotones and small habitats

The contrast indices are calculated firstly in a vegetated area (test site 1) from a section of Saxon Switzerland (using the nDSM with horizontal resolution of 1 m and the land cover data). The results of landscape contrast analysis are shown in Figure 3. Having a look at the first case (a), the indices are calculated based on the land cover classes including forest and field without consideration of small habitats and ecotones. In the second case (b), the land covers are at a more detailed level of the land surface including small habitats and ecotones. In the outcome of the calculation of Edge Contrast Index (ECON), Patch A in case (a) is considered as a whole forest patch adjacent to the field and has an edge contrast value of 81.25 %. In case (b), it is an assembly of two small forest patches connected by an ecotone. Compared to Patch A, Patch A_1 shows a lower ECON of 78.07 % and Patch A_2 has also a lower ECON of 58.28%. The reason is the existing ecotones around Patch A that act as buffer area between forest and field, resulting in a lower average height contrast of Patch A_1 and Patch A_2 from their surrounding patches. Patch B is also divided into two separate parts in case (b). The Patch B_1 has a lower value of ECON compared to Patch B, while patch B_2 shows a higher ECON value of 83.46%. Patch B in case (a) is a representation of the average ECON value of two patches with different vertical structures. It shows that the strictly categorical model neglect the inner heterogeneity of patches. The large forest Patch C shows also a decreased value of ECON from case (a) to case (b). This is due to the detection of ecotones which can lower the height contrast between forest and field. Small habitats

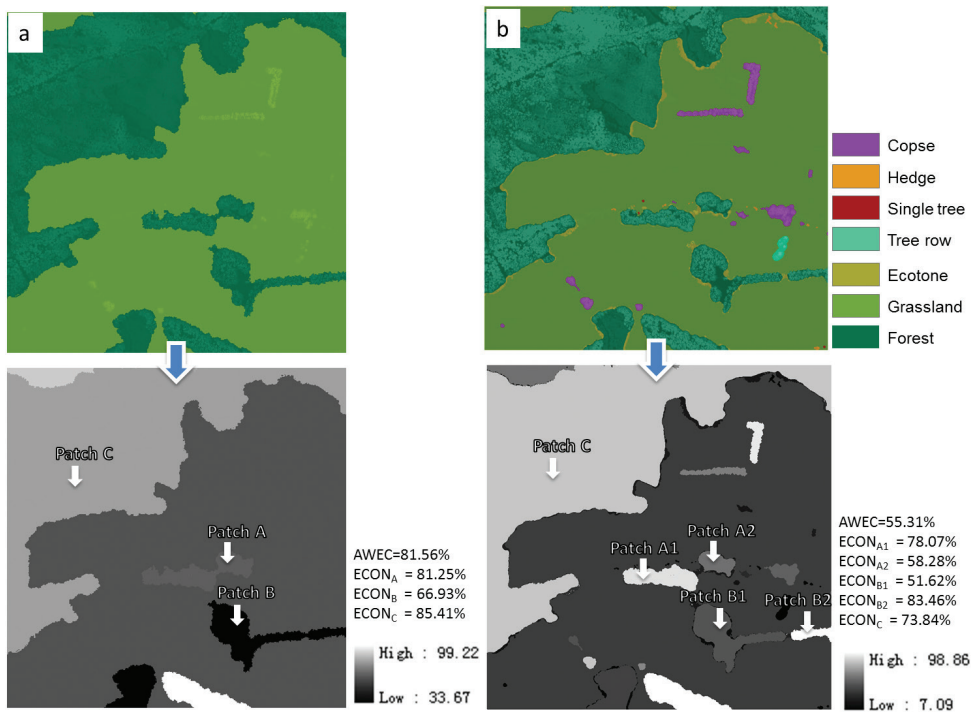


Figure 3. The applications of the adjusted Edge Contrast Index (ECON) and Area-Weighted Edge Contrast (AWEC) in the test site 1 (case (a) shows the results of contrast metrics without small habitats and ecotones; case (b) shows the results of contrast metrics including small habitats and ecotones).

mostly show lower contrast values than the large forest and field patches (Figure 3 (b)). They can partially alleviate the contrast value for the whole landscape, but the alleviation is limited due to their small area proportions. Although more patches are delineated in the case (b), the Area-Weighted Edge Contrast (AWEC) is still lower than in case (a). It means both ecotones and small biotopes possess low edge contrast values that can reduce the overall edge contrast of the whole landscape.

Comparison of contrast analysis with and without consideration of artificial elements

The patch contrast (ECON) is defined based on the vegetation height difference. But in reality there are often artificial elements existing in a vegetated landscape, e.g. traffic roads. We assume that the edges of artificial elements have the highest (100%) contrast weight to neighboring patches. A comparative test is exemplified in test site 2 (see Figure 4). The edge contrast values and landscape contrast are compared in two cases. Case (c) eliminates the traffic road and it is assumed as a vegetated area without artificial elements. Case (d), in contrast, shows a mixed landscape including both natural and artificial elements. Applying ECON and AWEC in both cases, the results

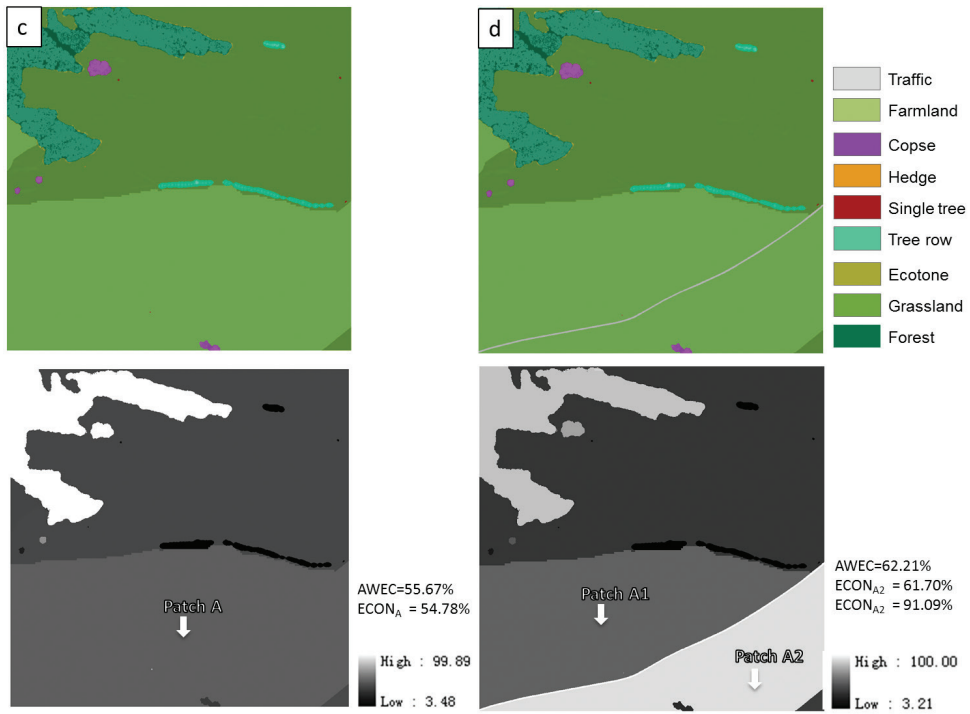


Figure 4. The applications of the adjusted Edge Contrast Index (ECON) and Area-Weighted Edge Contrast (AWEC) in the test site 2 (case (c) shows the results of contrast metrics without consideration of traffic road; case (d) shows the results of contrast metrics including traffic road).

are shown in Figure 4. In case (d), Patch A is dissected by a traffic road which increases the ECON of both Patch A₁ and A₂. This results in a higher AWEC on landscape level. The increased value of AWEC from case (c) to case (d) shows the effect of traffic road on landscape contrast.

Discussion

The use of models for quantifying landscape patterns

Patchiness and gradients are the concentrated expressions of spatial heterogeneity in the landscape (Wu 2007). Correspondingly two types of model have been used to represent landscape structure: Gradient Model (GM) and Patch Corridor Matrix Model (PCMM). Lausch et al. (2015) have concluded that the characteristics of research area and research objective are the decisive factors for choosing the appropriate model representing the landscape pattern. Landscapes under low human pressure are recommended for using the GM approach; anthropogenic-dominated landscapes should preferably be represent-

ed with the PCMM model. Furthermore, the research objective requires specific landscape metrics to be derived from relevant landscape model. In a natural landscape, the borders among heterogeneous vegetation are ambiguous. Thus, the surface metrics based on gradient model (e.g. Normalized difference vegetation index (NDVI), Topographic wetness (TWI), Greenness, etc.) are useful to capture the high transitory heterogeneity (McGarigal et al. 2009). In a cultural landscape, the land surface has been intensively used and managed by human, rendering the landscape in homogenous patches with distinct borders. The resulting landscape structure is therefore best represented with the PCMM approach, delineating patches of land-cover or land-use types by sharp borders.

In reality there is nearly no place without human impacts (Walz and Stein 2014). Official land use data (e.g. ATKIS (Amtliches Topographisch-Kartographisches Informationssystem), the official German nation-wide digital database for topographic spatial data) often ignore the small habitats and ecotones. It seems that the remote sensing technology remains an experimental tool used in focal areas requiring standardized scientific methodologies for detailed habitat monitoring at the regional and national levels (Corbane et al. 2015). This could be the reason why the PCMM approach has been more frequently used in landscape structure analysis far beyond the GM. Since PCMM is originated from the human perception of landscape, it is straightforward, understandable and easy to use. Quantitative metrics can be easily established and a variety of software (Baker and Cai 1992, McGarigal et al. 2012, Rempel et al. 2012) based on PCMM has emerged and facilitated the knowledge transfer from theoretical model to practice. However, applying PCMM in a semi-natural landscape could be oversimplified by losing ecotones between patches. A possible solution for such problem could be incorporating the gradient concept into mosaic model to distinguish inner core patch and its transitional boundary. In this paper we apply the gradient concept in the mosaic model to represent the ecotones between forest and field, i.e. the interior of forest is regarded as the core patch, and the height gradient of vegetation between forest and field is considered as an ecotone. This applied landscape model represents the spatial heterogeneity in a more realistic condition. Human boundaries (i.e. traffic roads) can be integrated in landscape contrast analysis as barriers with highest contrast value. But this may arise further complexities (see below for the comparison between (Figure 3b) and (Figure 4d)). It makes more sense to differentiate the edge effect and barrier effect by using the concept of landscape contrast and fragmentation. As shown in Figure 5, the concept of landscape contrast unifies discrete and continuous landscape representations (GM and PCMM) (Biswas and Wagner 2012) and would be better applied in a vegetated landscape with intermediate edge contrast. In contrast, the concept of fragmentation is applied in a binary model, which highlights the edge contrast between patches, assuming both ECON and AWEC equal to 1.

The use of contrast metrics in different landscapes

At the patch level, the modified edge contrast index (ECON) measures the degree of height contrast between a patch and its immediate neighborhood. ECON is a relative

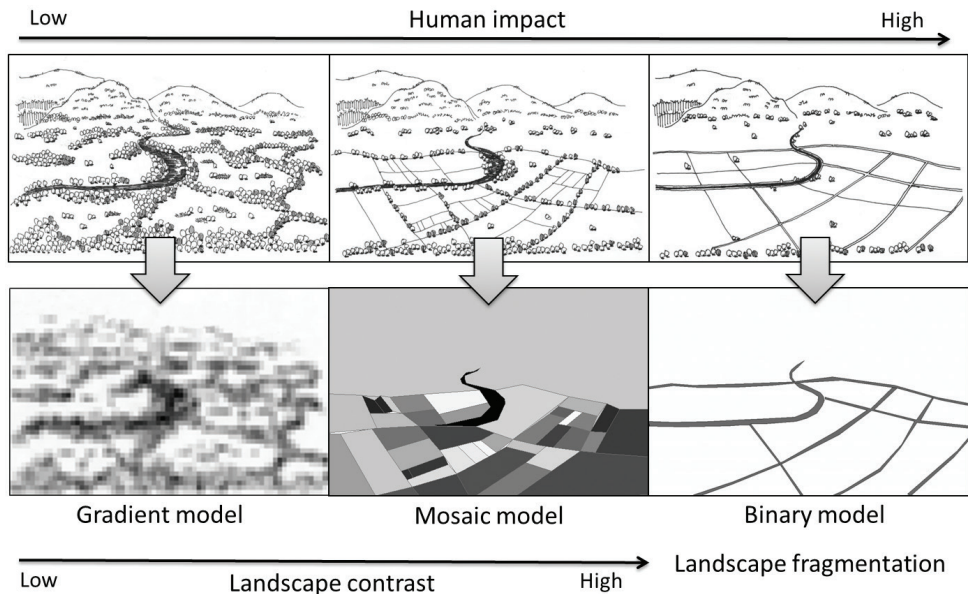


Figure 5. The conceptual models and metrics applied in different landscapes.

measure at patch level and stands for the degree of contrast in patch edge regardless of how big the patch is. At the landscape level, the edge effect have been often measured by Total Edge Contrast Index (TECI) or Contrast-Weighted Edge Density (CWED) (McGarigal et al. 2012), which count for all patches' edges multiplied by the corresponding contrast weight, divided by the total length of edge or total area in the landscape. The both indices only concern on the length of edges, regardless of the patch proportion. It would be helpful to quantify the edge contrast from the perspective of landscape configuration by using the Area-Weighted Edge Contrast (AWEC). This area-weighted index may be more appropriate than the unweighted mean index, since larger patches play a dominant role in the landscape dynamics. This index can also be applied in landscapes differing in total size and with differing proportions of habitat patches.

The examples shown in test site 1 demonstrate that the existence of small habitats and ecotones can reduce the landscape contrast as they possess the characteristic of lower edge contrast than the patch interior. In addition, the ecotones function as buffer areas or “soft boundaries” which reduce the height contrast between forest and field. The small habitats which are normally neglected in landscape structure analysis also account for the average height of matrix. For this reason, it is necessary to incorporate these small habitats and differentiate the patch interior and its exterior, such as ecotones. Attempts that incorporation of ecotones in fragmentation metrics (e.g. effective mesh size (MESH) (Jaeger 2000)) have been made to show the alleviation effects of landscape fragmentation by ecotones (Hou and Walz 2013). However, using the fragmentation metrics, the ecological functions of small habitats may be regarded as the perforation phase of fragmentation process according to their geometric characteristics

(Forman 1995, Jaeger 2000). This is contradicted to the perception that losing small habitats leads to landscape fragmentation (Jongman 2004). From the third dimension, the concept of landscape contrast can bring the ecological function of ecotones and small biotopes together. If the height difference among patches is considered as “terrain barrier”, the ecotones or small habitats can be recognized as shift areas that influence transboundary movements.

Artificial elements (e.g. traffic roads) affect not only regional or metapopulation dynamics but also have a direct effect on local population dynamics (Pontoppidan and Nachman 2013). The examples shown in test site 2 present the effect of the traffic road in the analysis of landscape contrast. The results show that the traffic road has direct impact on its neighboring patches and leads to an increase of ECON values. As a result, the AWEC of the whole landscape has increased. In an anthropogenic-dominated landscape, the value of AWEC should be approaching to its maximum value 1. Comparing case (b) in test site 1 (Figure 3) and case (d) in test site 2 (Figure 4), test site 2 shows a higher landscape contrast value. But only using contrast metrics, it is hard to see the structure variation between the two test sites. It would be necessary to use the fragmentation metrics as an additional indicator to describe the dissected landscape by a traffic road in test site 2. A binary model can be used for this purpose, for example, all artificial elements will be assigned 1, other patches should be merged and assigned 0. Both landscape contrast and fragmentation metrics are needed to compare the habitat pattern of two test sites.

Conclusions

In this paper we present an integrated approach to analyze the landscape contrast as a means to describe landscape heterogeneity. Incorporation of gradient concept in landscape structure analysis helps to overcome the limitation of PCMM that valuable information on patch boundary is missing. Not like the gradient model, the integrated approach is still based on a classified map which contains an additional category of gradient elements, such as ecotones on forest/field boundary. Therefore, the robust metrics derived from PCMM can be adapted to quantify the landscape structure including gradients. Similar to PCMM, this approach has also limitations as the simplification of land surface may be affected by the classification schemes of land cover.

The modified contrast metrics in this study show different sensitivities to different landscape compositions. Comparison of applying contrast metrics in a vegetated landscape (Figure 3) has revealed that the introduced measures can full account for the effects of ecotones and small habitats and lead to improvements for characterizing the vegetation heterogeneity from the third dimensional perspective. Artificial elements with highest contrast weight can also be incorporated in the modified contrast metrics. They can significantly increase the contrast value of the landscape (Figure 4). Generally the introduced contrast metrics are more applicable for characterizing the landscape pattern with an intermediate human impact (Figure 5). As the human impact

increases, the landscape pattern would be better represented by the categorical model with strict borders and the fragmentation metrics are likely to be applied in this case.

Ecotones and small habitats are often ignored in landscape structure analysis. This may due to the fact that there is lack of suitable conceptual model and metrics to integrate them. Our experimental results have shown that the discussed approach (contrast metrics based on an integrated model) is efficient for implementation under different landscape composition. We suggest that greater attention should be paid to these detailed landscape elements at the local level.

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Butterfly monitoring using systematically placed transects in contrasting climatic regions – exploring an established spatial design for sampling

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Abstract

Butterfly monitoring schemes are recording programs initiated to monitor nationwide butterfly abundance and distribution patterns, often with help from volunteers. The method generates high-resolution data, but may be associated with a degree of habitat sampling bias if volunteers prefer to survey areas perceived to be high-quality butterfly habitats. This can result in habitats becoming underrepresented in the data set, leading to less information about the butterfly populations there. In the present study, we investigate the possibility of applying a spatial design used by the Swedish Bird Survey for nationwide, grid-based sampling, with a goal to get butterfly monitoring data covering a representative sample of different habitats. We surveyed four 2×2 km sampling squares, split into 100 m segments, in the southernmost region of Sweden (Scania) and four in the northernmost region (Norrbotten). The grid-based transects were compared with volunteer-selected transects in a GIS analysis using a refined Swedish version of CO-RINE land cover data to see how well these two transect designs represent true habitat coverage. A total of 53 km transect was monitored, resulting in 490 individuals and 29 different species recorded. We found that transect cover correlated significantly with overall land cover using both monitoring methods, though standardised transects outperformed volunteer-selected transects in habitat representation in Scania, but not in Norrbotten. Butterflies were found to aggregate significantly in specific habitats, but with contrasting results for the two geographically different regions. Grasslands in both regions generated a high number of recorded butterflies, although so did clear-cut and residential areas in Norrbotten as well. The

highest number of individuals recorded per transect was found in bogs in Scania. This study emphasises the value of complementing free site selection monitoring schemes with spatially representative schemes such as the Swedish Bird Survey, and sheds some light on general habitat preferences for Swedish butterflies in two contrasting climatic regions.

Keywords

butterflies, monitoring, biodiversity, habitat, sampling, transects, boreal, continental, populations, GIS, CORINE

Introduction

Butterflies are the most widely studied of all insect groups (Dennis et al. 2006, van Swaay et al. 2008), and their sensitivity to environmental change together with the availability of butterfly data makes this group very useful as indicators for biodiversity (New 1997, Thomas 2005, Dennis et al. 2006, van Swaay et al. 2015). Furthermore, since many butterfly species require a warm microclimate for optimal growth and development (Wallisdeveries and Van Swaay 2006, Eilers et al. 2013), they can also serve as indicators for climatic change (Thomas 2005, van Swaay et al. 2008, Betzholtz et al. 2013). While birds, another popular group of biodiversity indicators, range more widely and over larger areas, butterflies provide important additional and complementary area-specific information since they are more likely to reflect environmental changes occurring on a more detailed scale (van Swaay et al. 2006).

Habitat loss and fragmentation is a major driver behind the decline of many butterfly species worldwide (Bergman et al. 2004, Ekroos et al. 2010). Due to changes in land-use and intensification of agriculture throughout the last century there has been a loss of many open and half-open natural and semi-natural habitats (Nilsson et al. 2013, Cousins et al. 2015). This has had dramatic effects on many insect groups, and there has been a decline in butterfly numbers all over Europe (van Swaay et al. 2006, Konvicka et al. 2008, Van Dyck et al. 2009, Dover et al. 2011). The recently developed “European Grassland Butterfly Indicator” (van Swaay et al. 2015) based on butterfly data from 22 European countries suggests that grassland butterfly populations may have declined as much as 30% since 1990.

In order to effectively monitor population trends, butterfly monitoring schemes now run in several European countries, e.g. the UK, Finland, Germany and the Netherlands (van Swaay et al. 2008, 2015, Kühn et al. 2012, Botham et al. 2013, Heliölä et al. 2013). Schemes vary in the way monitoring sites are selected (van Swaay et al. 2015), with some schemes using random site placement whereas others have sites selected by coordinators. However, the majority of schemes use the more flexible approach where butterfly recorders can place their monitoring sites freely (van Swaay and Warren 2012, van Swaay et al. 2015). Free site selection is generally appreciated by recorders as they can be involved in the site selection process by influencing choice of site characteristics, accessibility, and being able to relate more closely to the sites that they monitor (van Swaay and Warren 2012). The main disadvantage of free site

selection is that the geographic coverage commonly becomes non-random and hence not representative of habitats and butterfly populations in general (Dover et al. 1997, van Swaay and Warren 2012, van Swaay et al. 2015). The concern that open areas and in particular semi-natural grasslands tend to become overrepresented in monitoring data and that forests and other parts of the wider countryside are less well covered (van Swaay and Warren 2012) have recently inspired strategies that substantially reduce sampling bias (e.g. Brereton et al. 2011, Lang and Buhler 2012).

In the UK, species that are widespread across the general countryside compromise half of the butterfly fauna (Asher et al. 2001) but are underrepresented in traditional monitoring; declines for these species went largely undetected by monitoring schemes during the 20th century (Brereton et al. 2011). This led to the launch of the Wider Countryside Butterfly Survey, a scheme which uses the grid-based sampling design of the British Bird Survey (Greenwood et al. 1995) to get representative trends across the whole countryside (Brereton et al. 2011, Botham et al. 2013, Roy et al. 2015). Grid-based, spatially representative sampling is increasingly used in bird monitoring today (e.g. Greenwood et al. 1995, Davey et al. 2013, Lehtikoinen 2013) and offers an attractive development for butterfly monitoring (Kéry and Plattner 2007, Brereton et al. 2011, Lang and Buhler 2012, van Swaay and Warren 2012). However, there is a great need to evaluate how well such standardised designs suit butterfly monitoring in different climatic regions. Factors such as the openness of the countryside, topography, and the time and effort necessary to walk transects in different regions can differ markedly even within one single country (Brereton et al. 2011) and is particularly relevant in countries that span a large range of latitudes. For instance, a coniferous forest in Sweden's southern, continental region can be dense, dark and not particularly well suited for butterflies. In contrast, corresponding forests in the northern, boreal region can be much more open, sunlit, and hence attractive to butterflies.

The Swedish Butterfly Monitoring Scheme is a nationwide program with free site selection (Pettersson et al. 2011). Volunteers appreciate that they can place monitoring transects themselves and this flexible design has been central to the growth of the scheme. Because free site selection tends to result in some habitats being underrepresented (van Swaay and Warren 2012), it would be valuable to complement the Swedish scheme with a grid-based approach (cf. Lindenmayer and Likens 2009). A very promising way of doing so would be to count butterflies along transects that already form part of a standardised, nationwide monitoring design: the Swedish Bird Survey (Green and Lindström 2015). This scheme consists of 716 routes in a 25×25 km grid, covering Sweden as a whole. The layout ensures that all major habitats are proportionally represented. At the centre of each grid cell, the bird fauna is censused once per year along eight 1 km transects arranged in a 2×2 km square (Figure 1, Green and Lindström 2015). The standard method of butterfly monitoring, using "Pollard walk" transects, (Pollard and Yates 1993) could potentially be used along the bird monitoring transects (Brereton et al. 2011, Roy et al. 2015). Hence, the grid used by the Swedish Bird Survey offers an attractive design for obtaining butterfly monitoring data that cover major habitats proportionately.

With this study we have a twofold goal. The first is to evaluate if the large-scale, grid-based method used by the Swedish Bird Survey can be adapted and applied to butterfly monitoring. The second aim is to use the collected data to quantify butterfly abundance in relation to habitat characteristics in two contrasting climatic regions of Sweden: Scania and Norrbotten.

Methods

GIS analysis of land cover

While transects based on standardised grid-based designs are generally assumed to outperform free site selection in terms of true habitat representation, this assumption is rarely evaluated. Here, we quantified land cover in the Swedish Bird Survey transects visited in the present experiment ($N = 4$ in each region, total length: 32 km per region) as well as land cover in volunteer-selected free transects in the same regions ($N = 5$ in each region, total length 12.92 km in Scania and 12.22 km in Norrbotten). The free transects that were analysed comprised all sites in Norrbotten and a random, corresponding subsample of the transects in Scania. The free transects that we analysed covered less distance than the standardised transects and hence had smaller areas. To allow direct comparison despite differing total areas, we recalculated land cover to the smallest area (free transects, Scania: 4.45 km²; Appendix: Table S3). Land cover data was obtained from the SMD database, which is a refined, Swedish version of the CORINE land cover database with the smallest mapping unit 1–25 ha, map resolution 25×25 m, and 2000 as reference year (Swedish Environmental Protection Agency, 2014). To quantify land cover class coverage, we clipped the SMD using a 200 m buffer around each transect following Davey et al. (2013) using QGIS (v.2.12, QGIS Development Team 2016).

Field data collection

Butterfly monitoring took place during the summer of 2010 from June 17 to July 7 in Scania, the southernmost part of Sweden, and from July 20–28 in Norrbotten, the northernmost part of Sweden. Each site consisted of the four outer sections of 2×2 km squares used by the Swedish Bird Survey (Figure 1), equalling an 8 km long transect. The survey squares were selected with the intention to cover different habitats along the transects, in order to thoroughly test the sampling method. The four sites visited in Scania were called Hyby, Tjörnarps, Slätteberga and Kongaö, and the four sites in Norrbotten were Sundom, Rosfors, Långberget and Bergnäset (Appendix: Table S1, Figure 1). Two persons (EV together with a colleague) walked all transects. Recording only took place during sunny weather, and monitoring did not start earlier than 10 am and ended not later than 5 pm (Central European summer time, UTC +2), which is the time most suitable for butterfly activity (van Swaay et al. 2008). Average tempera-

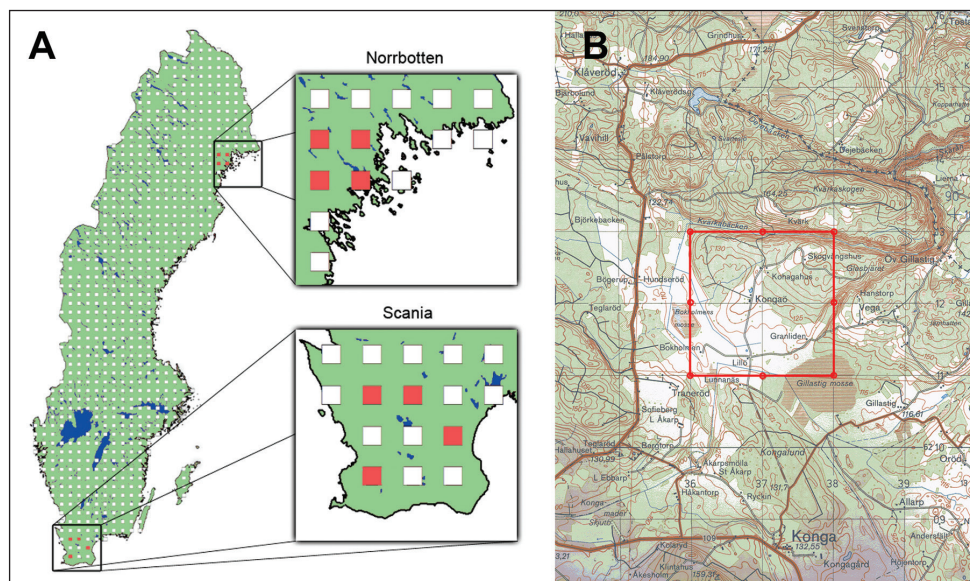


Figure 1. **A** Map over Sweden with systematically placed bird monitoring squares throughout the country, used by the Swedish Bird Survey. The two regions featured in this study are shown, with monitoring squares visited portrayed in red. The squares are 2×2 km wide, indicate location, and not shown to scale **B** Example map of a bird monitoring square: site Kongaö in Scania, Sweden, indicated as a red, 2×2 km square. Butterfly transects followed the periphery of bird monitoring squares ($N = 8$ sites) as closely as possible for as long as weather conditions and time of day permitted (Pettersson et al. 2011). © Lantmäteriet, I2014/00579.

ture throughout the day ranged from 17° to 25° C, and the wind varied in different habitats, but never exceeded 5 in the Beaufort scale, which is acceptable in terms of butterfly monitoring (van Swaay et al. 2008).

All transects monitored were divided into segments of approximate 100 m length with an accompanying description for the habitat surrounding the transects. The distance of the transect segments were estimated with the help of maps. The different habitat categories were: deciduous forest, coniferous forest, grassland, residential area, fen/bog, and clear-cut area (examples of three habitats can be seen in Figure 2). Road was added as an additional unique category since much monitoring had to be performed on small paths and roads due to accessibility. The habitat description of each 100 m segment consisted of percentages of each habitat category that the transect crossed (e.g. deciduous forest 80%, grassland 20%), in order to reflect the surroundings as closely as possible. As each butterfly individual was attributed to a specific 100 m segment, this typically resulted in non-integer numbers for butterfly individual and species counts per habitat category (i.e. 0.8 butterflies in deciduous forest and 0.2 butterflies in grassland, based on the example above). Areas such as cultivated fields, highly dense forests and open water were avoided for reasons of safety, land-owner privacy, and accessibility.

Recordings were made of all butterflies (Rhopalocera) and burnet moths (Zygaenidae), as seen within an 'invisible box' of 5 m in front of the recorders, 2.5 m to each side

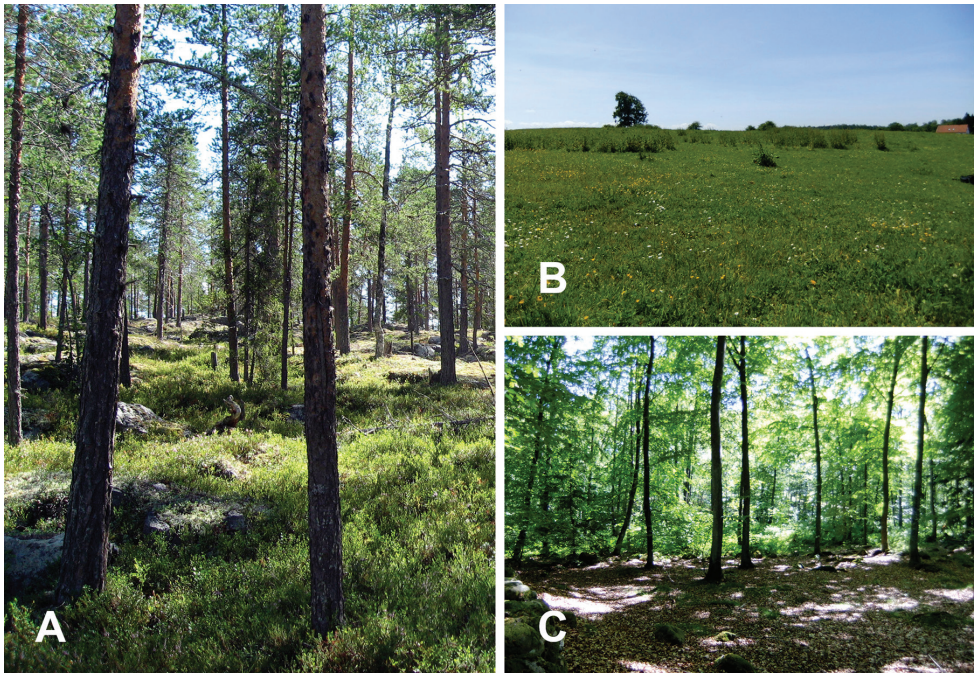


Figure 2. Examples of butterfly habitat categories: **A** coniferous forest in Rosfors (Norrbotten) **B** grassland in Hyby (Scania), and **C** deciduous forest in Slätteberga (Scania). Photos by Elin Videvall.

and 5 m above, according to the ‘Pollard walk’ method (Pollard and Yates 1993, Pettersson et al. 2011). The monitoring was paused during the time for identification of the species, and subsequently resumed. Observations were documented using a butterfly net and a camera, and validated to species level using colleagues and literature (Eliasson et al. 2005, Söderström 2006). The pace of walking depended on habitat, accessibility, and butterfly density, but was in general approximately 3 km/h. Double counting of individuals cannot be completely ruled out, but was avoided as far as possible. Individuals not caught and identified to species level were counted and included in the (total) butterfly abundance analyses but not in any species or biodiversity measurements. In total, 42 out of 490 individuals observed (8.6%) could not be identified to species level (Appendix: Table S2). Recorder bias should not be of major concern since one of the authors (EV) was present at all sites and single-handedly documented each individual recorded. Species names (English and scientific) follow Tolman (2001) and Eliasson et al. (2005), respectively.

Data analyses

For each region, we evaluated similarity in habitat coverage between the region as a whole and the two transect approaches using Spearman Rank Correlation tests within

each region followed by two-sample tests of correlation coefficients using Fisher z-transformed values (Zar 1999).

Butterfly data was compiled using values per 100 m segment as the basis for calculations. To quantify biodiversity, we used the Simpson's Diversity Index (Magurran 2004), where both the number of species as well as the abundance of the species is taken into account. The Simpson's Index (D) measures the probability that two individuals randomly selected from a sample will belong to the same species,

$$D = \sum_{i=1}^S \frac{n_i(n_i - 1)}{N(N - 1)}$$

where n_i = the number of individuals in the i th species, N = the total number of individuals and S = the number of species in the sample. We represent this biodiversity measurement as $1/D$, called the Simpson's Reciprocal Index. In this variant of D , 1 is the lowest possible value, representing a community containing only one species, and the maximum possible value is the number of species in the sample (Magurran 2004). Further, we used the Simpson's Evenness Index ($1/DS$) representing the species evenness in sites with values ranging from 0 (aggregation of species) to 1 (completely even; Magurran 2004).

Alpha (α) diversity generally measures species diversity of a defined area or habitat, whereas beta (β) diversity is used as a measure of the difference between two or more defined areas (Magurran 2004, Anderson et al. 2011). Following this, we use the term α diversity for the site-specific diversity measure (number of species per site), and β diversity for the difference between the number of species at each site and the number of species in the region). Each diversity measure results in four replicates per region. In order to assess whether α and β diversity differed between regions, we used a Wilcoxon two-sample test.

Chi-square tests of butterfly abundance were performed for each habitat category, to test if butterfly distributions differed from random expectations, i.e. if individuals distributed among habitat categories according to their relative coverage along the transects. All statistical analyses were performed in R (v. 2.15, R Core Team 2013).

Results

GIS land cover analysis

The land cover representation of the Swedish Bird Survey transects and the volunteer-selected transects correlated significantly with overall land cover within Scania (SMD land cover classes: SBS transects: $r = 0.890$, $N = 14$, $p < 0.001$; in free transects: $r = 0.524$, $N = 16$, $p = 0.04$, Appendix: Table S3) as well as in Norrbotten (SMD land cover classes: SBS transects: $r = 0.733$, $N = 20$, $p < 0.001$; free transects: $r = 0.422$, $N = 23$, $p = 0.045$, Appendix: Table S3). In Scania, the correlation between overall land

cover and Swedish Bird Survey transects was significantly higher than that between overall land cover and free selection transects ($Z = 2.054$, $p = 0.03$). There was no difference in the relation between overall land cover and the two transect categories in Norrbotten ($Z = 1.472$, $p = 0.14$).

Use of bird monitoring squares

The systematically placed 2×2 km squares used by the Swedish Bird Survey correspond to 8 km butterfly transect per site (80 segments). In total we monitored 53 km transect (83%) out of the 64 km transect present in the eight squares combined. Using transects along the borders of Bird Survey squares proved to be slightly more than what was normally possible to cover within one day of butterfly recording. The landscape along the transects was sometimes difficult to traverse, and some parts of the transects were completely inaccessible. The overall distance monitored in the two regions was very similar, with 27 km in Scania and 26 km in Norrbotten. Even though some of the squares were not completely surveyed, we got a substantial amount of data with an average of 6.6 km transect monitored per site.

Habitat coverage in the two regions

Field estimates of total habitat coverage was divided fairly equal between deciduous forest (25%), coniferous forest (26%), and grassland (30%), among all transects monitored (Figure 3). The remaining three habitat categories had less coverage, with clear-cut area at 4%, residential area had 9%, and fen/bog covered 6% of all transects. A large part (39%) of all monitoring was performed alongside smaller roads or paths. Scania had a higher percentage of deciduous forest while the dominant habitat in Norrbotten was coniferous forest (Figure 3).

Butterfly monitoring

A total of 490 butterfly individuals were recorded, with 250 counted in Scania (9.3 individuals per km transect) and 240 in Norrbotten (9.2 individuals per km transect). We recorded 29 different butterfly species (Appendix: Table S2), but no burnet moths. We found 22 species in Scania and 16 species in Norrbotten. Of the 22 species recorded in Scania, 13 were uniquely found in this region, and in Norrbotten, 6 out of 16 species were only found in this region. The site with the highest number of species recorded was Långberget in Norrbotten with 12 different species, whereas Kongaö in Scania was the site with most individual recordings (147 individuals) (Appendix: Table S1).

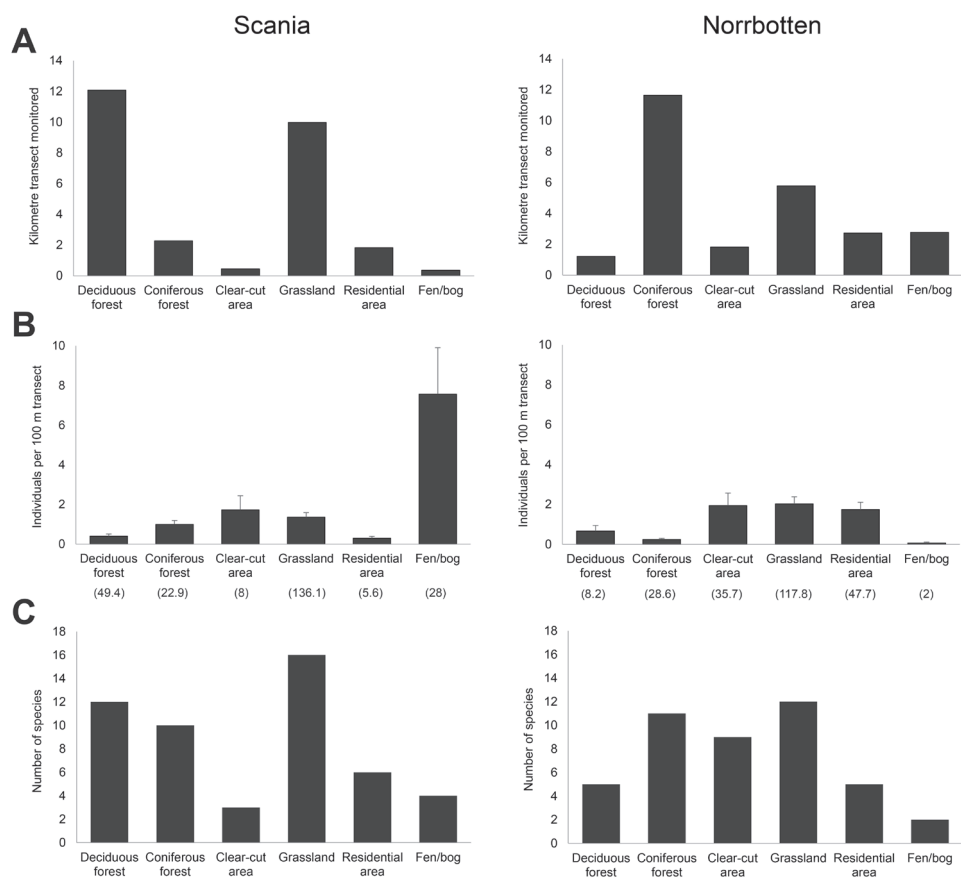


Figure 3. **A** Transect monitored (km) in different habitats for two geographically different regions in Sweden: the southernmost region, Scania, and the northernmost region, Norrbotten **B** Number of butterfly individuals recorded per 100 m transect in different habitats (mean values \pm SE). Numbers in parentheses indicate total number of individuals monitored per habitat (non-integers due to proportional habitat coverage per 100 m transect) **C** Total number of butterfly species recorded in the different habitats.

Butterfly abundance in different habitats

Butterfly abundance in different habitats was compared to the expected number of individuals relative to habitat coverage in each region (Table 1 and Figure 3). Within the habitat category deciduous forest, a significantly lower number of individuals was observed in the region of Scania than expected if the butterflies had distributed randomly over the transects ($\chi^2 = 24.2$, $p < 0.001$) (Table 1), but they were not relatively fewer than expected in the deciduous forests of Norrbotten ($\chi^2 = 0.48$, $p = 0.488$). The opposite was true for coniferous forests: in Norrbotten we observed significantly fewer individuals than expected ($\chi^2 = 45.8$, $p < 0.001$), but in Scania the number of

Table 1. Butterfly abundance in different habitats for two geographically different Swedish regions, Scania and Norrbotten.

| | Number of individuals recorded | Mean nr of individuals per 100 m | χ^2 -value | p-value | Significance ¹ | Abundance relative to expected value |
|-------------------|--------------------------------|----------------------------------|-----------------|---------|---------------------------|--------------------------------------|
| Scania | | | | | | |
| Deciduous forest | 49.4 | 0.4 | 24.2 | < 0.001 | *** | Lower |
| Coniferous forest | 22.9 | 1.0 | 0.07 | 0.787 | ns | No difference |
| Clear-cut area | 8.0 | 1.7 | 1.1 | 0.286 | ns | No difference |
| Grassland | 136.1 | 1.4 | 8.4 | 0.004 | ** | Higher |
| Residential area | 5.6 | 0.3 | 5.7 | 0.017 | * | Lower |
| Fen/bog | 28.0 | 7.6 | 19.2 | < 0.001 | *** | Higher |
| Norrbotten | | | | | | |
| Deciduous forest | 8.2 | 0.7 | 0.48 | 0.488 | ns | No difference |
| Coniferous forest | 28.6 | 0.3 | 45.8 | < 0.001 | *** | Lower |
| Clear-cut area | 35.7 | 2.0 | 6.7 | 0.009 | ** | Higher |
| Grassland | 117.8 | 2.0 | 24.2 | < 0.001 | *** | Higher |
| Residential area | 47.7 | 1.8 | 6.9 | 0.008 | ** | Higher |
| Fen/bog | 2.0 | 0.07 | 20.2 | < 0.001 | *** | Lower |

¹Chi-square tests between observed and expected number of individuals indicate significant differences in habitats denoted with asterisks: ns (non-significant), * ($p < 0.05$), ** ($p < 0.01$), and *** ($p < 0.001$).

individuals recorded in this habitat matched the expected number very well ($\chi^2 = 0.07$, $p = 0.787$). The habitat category fen/bog inhabited significantly more individuals than expected in Scania ($\chi^2 = 19.2$, $p < 0.001$), but significantly fewer individuals in Norrbotten ($\chi^2 = 20.2$, $p < 0.001$) (Table 1 and Figure 3).

In Norrbotten, we found a significantly higher number of individuals in clear-cut areas ($\chi^2 = 6.73$, $p = 0.009$) and in residential areas ($\chi^2 = 6.94$, $p = 0.008$) than expected, but not in the clear-cut areas of Scania ($\chi^2 = 1.14$, $p = 0.286$), and in Scania’s residential areas we even recorded significantly fewer individuals than expected ($\chi^2 = 5.71$, $p = 0.017$) (Table 1). Not surprisingly, significantly more individuals were found in grasslands than expected, in both Scania ($\chi^2 = 8.35$, $p = 0.004$) and Norrbotten ($\chi^2 = 24.2$, $p < 0.001$) (Figure 3), though the aggregation of butterflies in grasslands were denser in Norrbotten.

Butterfly species richness in different habitats

The habitat with highest number of species recorded in total was grassland, with 16 different species recorded in Scania and 12 species in Norrbotten, however this might be partly due to relatively high coverage of grassland monitored, with 10 km grassland visited in Scania (37%), and 5.8 km in Norrbotten (22.3%) (Figure 3). Nonetheless, grassland displayed significantly higher number of species in Norrbotten than expected ($N = 12$, $\chi^2 = 4.58$, $p = 0.032$; Table 2). Despite lower habitat coverage of coniferous

Table 2. Species richness in different habitats in two geographically different Swedish regions, Scania and Norrbotten.

| | Number of species recorded ¹ | χ^2 -value | p -value | Significance ² | Diversity relative to expected value |
|-------------------|---|-----------------|------------|---------------------------|--------------------------------------|
| Scania | | | | | |
| Deciduous forest | 12 | 0.21 | 0.644 | ns | No difference |
| Coniferous forest | 10 | 5.59 | 0.018 | * | Higher |
| Clear-cut area | 3 | 2.05 | 0.152 | ns | No difference |
| Grassland | 16 | 2.57 | 0.109 | ns | No difference |
| Residential area | 6 | 2.72 | 0.099 | ns | No difference |
| Fen/bog | 4 | 3.18 | 0.074 | ns | No difference |
| Norrbotten | | | | | |
| Deciduous forest | 5 | 3.14 | 0.076 | ns | No difference |
| Coniferous forest | 11 | 0.81 | 0.369 | ns | No difference |
| Clear-cut area | 9 | 6.11 | 0.013 | * | Higher |
| Grassland | 12 | 4.58 | 0.032 | * | Higher |
| Residential area | 5 | 1.65 | 0.199 | ns | No difference |
| Fen/bog | 2 | 0.02 | 0.880 | ns | No difference |

¹Excluding unidentified species, see Appendix: Table S2,

²Chi-square tests between observed and expected number of species indicate significant differences in habitats denoted with asterisks: ns (non-significant), * ($p < 0.05$).

forests and fens/bogs in Scania, these habitats still harboured several butterfly species ($N = 10$ and $N = 4$, respectively). The same was true for Norrbotten's clear-cut areas and deciduous forests ($N = 9$ and $N = 5$, respectively; Figure 3 and Table 2). The habitat with the lowest species diversity, despite moderate habitat coverage, was fens/bogs in Norrbotten ($N = 2$, Figure 3).

Biodiversity measures

The biodiversity in the two regions was measured using the Simpson Reciprocal Diversity Index ($1/D$) and the Simpson Evenness Index ($1/DS$). The Simpson Reciprocal Index for Scania (6.86) was slightly higher (although non-significantly) than the index for Norrbotten (5.10) (this difference was tested between the sites using a Wilcoxon rank sum test: $W = 12$, $p = 0.31$). The Simpson Evenness Index for Scania (0.31), was similar and not significantly different than the corresponding number for Norrbotten (0.32) (tested between the sites with a Wilcoxon rank sum test: $W = 11$, $p = 0.47$).

The mean species number within a region, the α diversity, did not differ significantly between Scania and Norrbotten (Wilcoxon rank sum test: $W = 10$, $p = 0.661$). The β diversity, defined here as the mean difference in species number between each site and the total species number of that region, was significantly higher in Scania than in Norrbotten (Wilcoxon rank sum test: $W = 16$, $p = 0.028$).

Discussion

Butterfly recording using systematically placed transects

In this study we have tested the possibility of using systematically placed transects in butterfly monitoring schemes in order to get butterfly recordings with reduced volunteer habitat bias. Volunteer recorders are most often free to select the location for monitoring (Roy et al. 2007, van Swaay et al. 2008), which may result in taxonomically and geographically biased records. Dennis and Thomas (2000) and Dennis et al. (1999) demonstrated that species richness and occurrence are positively correlated with recording intensity. The volunteers' visits are biased by access (e.g. the distance from their home), the location of potential 'butterfly hot spots' (either diversity or rarity hot spots), and areas with a greater number of butterfly resources (such as semi-natural grasslands). This type of bias allows for good coverage of environmentally protected sites but may not provide trends representative for species in other habitats (e.g. woodland species, Roy et al. 2007).

Our GIS analyses showed that standardised transects mirrored overall land cover better than free transects in Scania, but not significantly better in Norrbotten. The Norrbotten landscape is generally less urbanised than Scania (cf. Appendix: Table S3), and one likely explanation that free transects do not represent true habitat coverage as well in Scania is that such transects are more likely to be placed near where volunteers live (cf. Dennis and Thomas 2000; Pettersson et al. 2011) and human influence on adjacent habitats is more pronounced in Scania than in Norrbotten. This is true not only for the extent of urban structures but also for semi-natural grasslands and other habitats heavily influenced by human activities.

We found that the transects used by the Swedish Bird Survey provided good coverage of traditionally underrepresented butterfly habitats such as forests, clear-cuts and wetlands in both regions (Figure 3, cf. Brereton et al. 2011, Roy et al. 2015). Even though the habitat categorisation was broad, our results clearly indicate that there are several habitats other than grasslands harbouring butterflies. Similarly, Berg et al. (2011) demonstrated that several typically overlooked habitats in forest-dominated landscapes can have at least as high numbers of butterfly species as semi-natural grasslands. Thus, it is evident that systematically placed transects throughout the country can be an important consideration in order to reduce bias in habitat coverage. It is highly likely that some of the habitats monitored, such as deciduous and coniferous forests, are strongly underrepresented in traditional butterfly monitoring.

During the year that the present study was performed, there were still relatively few free transects in Scania and Norrbotten and a direct comparison of simultaneously collected butterfly recordings from multiple standardized and free transects was not possible. Now substantially more free transects are monitored and such a direct comparison of observations made throughout the season would be a logical next step.

Butterfly abundance and species richness in different habitats

We found major differences in butterfly abundance and species richness in the different habitats and between the two regions. In grasslands, we found more butterfly individuals than expected in both regions (Table 1), and the total number of species found was the highest in this habitat, for both regions (Table 2). Although grassland was also the second most monitored habitat in terms of total transect length for both regions (Figure 3), the number of species recorded per 100 m transect was still relatively high, and our results confirm the importance of grassland habitats for both butterfly abundance and species richness.

The clear-cut and residential areas in Norrbotten harboured more butterfly individuals than expected, but not in Scania; this region had instead significantly fewer individuals recorded in residential areas (Table 1). For the two forest habitat categories, we found fewer individuals in deciduous forests in Scania (but not in Norrbotten), and fewer individuals in coniferous forests in Norrbotten (but not in Scania) (Table 1). The contrasting differences between the regions in butterfly abundance in different habitats may be due to the drastic latitudinal and climatic differences of the two regions. Interestingly, we recorded significantly more individuals and species than expected in the habitat fen/bog in Scania, where a relatively small area harboured a substantial number of Cranberry Blues (*Plebeius optilete*), but also other species. These results stress the value of using representative, grid-based geographic sampling so that diversity hotspots and common habitats become neither over- nor underrepresented in monitoring schemes.

The β diversity was significantly different between the regions Scania and Norrbotten, indicating that the sites in Norrbotten harboured many of the same species, as opposed to Scania where the sites often had different species composition. This is most likely because Norrbotten has a smaller species pool compared to Scania, which harbours several rare and local butterfly species (Eliasson et al. 2005, Öckinger et al. 2006). Alternatively, the 2×2 km squares in Norrbotten could be more similar to each other in terms of habitat composition compared to the Scanian study squares.

Some butterfly species that we recorded in Scania were not seen in Norrbotten ($N = 13$), and in Norrbotten 6 out of 16 species were not found in Scania. This is likely due to the climate differences resulting in different species distributions, but it is also plausible that many species were not detected by chance. We monitored the two regions intentionally during different dates to take into account their differences in spring arrival. Recordings in Scania were performed between June 17th and July 7th, and recordings in Norrbotten in late July (20th – 28th). Because of the large latitudinal differences between the regions (Appendix: Table S1), spring can arrive approximately seven weeks later in Norrbotten than in Scania (Alexandersson 2002), so we believe the later monitoring in Norrbotten would take most of this into account.

Implications for future butterfly monitoring

A ‘reduced effort’ monitoring scheme is based on a higher number of transects and counted only a few times per year, as opposed to the more traditional scheme, which are to a greater extent based on more regular visits and free site selection (van Swaay et al. 2008). A reduced effort scheme with fewer visits per year but more sites monitored might make volunteers more willing to record in areas where they are less likely to record high numbers of individuals or many species (Roy et al. 2007). Stratified random sampling of sites or systematically placed sites (like in this study), will generate a lot of data for widespread butterfly species but fewer records of rare or localised species (van Swaay et al. 2008). This can, however, be mitigated by adding additional transects at sites where these species are known to occur (van Swaay et al. 2008, van Swaay and Warren 2012).

A Swedish butterfly monitoring scheme with systematically placed transects throughout the country with the intention to cover different habitats would produce a representative picture of the nation’s butterfly population without introducing bias from habitat choice by the recorder. Free and systematic site selection should however not be seen as mutually exclusive. The Swedish Bird Survey started with free site selection in 1969 and added its grid-based network of geographically representative transects to the monitoring scheme in 1996. Similarly, the UK butterfly monitoring scheme has added a complementary, Wider Countryside monitoring scheme (Brereton et al. 2011, Roy et al. 2015). In other words, free selection transects and geographically representative ones can certainly exist side by side. In fact, grid networks encouraging spatially representative selection of free sites are used by some monitoring schemes (e.g. Åström et al. 2014).

Following this, we suggest that it would be valuable to complement monitoring schemes with free site selection such as the Swedish butterfly monitoring by adding standardised, grid-based sampling schemes. An exciting possibility resulting from joint monitoring of different organisms in a grid-based design is that more general biodiversity trends such as those indicated by Thomas et al. (2004) could be addressed at high spatial resolution. In general, we believe that transects for recording butterflies need to be smaller than the 2×2 km squares used by the Swedish Bird Survey. If national butterfly monitoring relies on recordings by volunteers, the transect cannot be too long as it might deter volunteers from participating. Monitoring squares the size of 1×1 km or 750×750 m, as used in study by Jonason et al. (2010), is a much more reasonable size for a butterfly monitoring scheme.

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Appendix

Table S1. Geographical coordinates (Swedish grid, RT 90 2.5 gon V) for butterfly monitoring sites.

| Site | Municipality | Region | Recording | Latitude (N) | Longitude (E) |
|-------------|--------------|-------------|------------|--------------|---------------|
| Hyby | Svedala | Scania | 2010-06-17 | 55.5595° | 13.2235° |
| Tjörnarps | Höör | Scania | 2010-06-24 | 56.0170° | 13.5922° |
| Slätteberga | Tomelilla | Scania | 2010-06-29 | 55.7985° | 14.0016° |
| Kongaö | Svalöv | Scania | 2010-07-07 | 56.0089° | 13.1913° |
| Sundom | Luleå | Norrbottnen | 2010-07-20 | 65.7750° | 22.0709° |
| Rosfors | Piteå | Norrbottnen | 2010-07-23 | 65.5780° | 21.4869° |
| Långberget | Boden | Norrbottnen | 2010-07-24 | 65.8021° | 21.5372° |
| Bergnäset | Luleå | Norrbottnen | 2010-07-28 | 65.5614° | 22.0342° |

Table S2. Butterfly species recorded in each region.

| English name | Scientific name | Scania | Norrbottn | Total |
|---------------------------------|------------------------------|--------|-----------|-------|
| Large Skipper | <i>Ochlodes sylvanus</i> | 59 | 4 | 63 |
| Black-veined White | <i>Aporia crataegi</i> | 2 | 0 | 2 |
| Large White | <i>Pieris brassicae</i> | 2 | 0 | 2 |
| Green-veined White | <i>Pieris napi</i> | 7 | 2 | 9 |
| Moorland Clouded Yellow | <i>Colias palaeno</i> | 0 | 1 | 1 |
| Brimstone | <i>Gonepteryx rhamni</i> | 6 | 1 | 7 |
| Idas Blue | <i>Plebejus idas</i> | 0 | 10 | 10 |
| Cranberry Blue | <i>Plebejus optilete</i> | 24 | 12 | 36 |
| Silvery Argus | <i>Aricia nicias</i> | 0 | 10 | 10 |
| Amanda's Blue | <i>Polyommatus amandus</i> | 2 | 1 | 3 |
| Common Blue | <i>Polyommatus icarus</i> | 6 | 2 | 8 |
| Scarce Copper | <i>Lycaena virgaureae</i> | 0 | 49 | 49 |
| Green Hairstreak | <i>Callophrys rubi</i> | 1 | 0 | 1 |
| Dark Green Fritillary | <i>Argynnis aglaja</i> | 0 | 3 | 3 |
| High Brown Fritillary | <i>Argynnis adippe</i> | 0 | 1 | 1 |
| Lesser Marbled Fritillary | <i>Brenthis ino</i> | 10 | 9 | 19 |
| Pearl-bordered Fritillary | <i>Boloria euphrosyne</i> | 1 | 0 | 1 |
| Small Pearl-bordered Fritillary | <i>Boloria selene</i> | 16 | 1 | 17 |
| Map Butterfly | <i>Araschnia levana</i> | 2 | 0 | 2 |
| Red Admiral | <i>Vanessa atalanta</i> | 1 | 0 | 1 |
| Peacock Butterfly | <i>Aglais io</i> | 6 | 0 | 6 |
| Small Tortoiseshell | <i>Aglais urticae</i> | 7 | 49 | 56 |
| Heath Fritillary | <i>Melitaea athalia</i> | 2 | 0 | 2 |
| Speckled Wood | <i>Pararge aegeria</i> | 1 | 0 | 1 |
| Wall Brown | <i>Lasiommata megera</i> | 1 | 0 | 1 |
| Large Wall Brown | <i>Lasiommata maera</i> | 1 | 0 | 1 |
| Small Heath | <i>Coenonympha pamphilus</i> | 16 | 0 | 16 |
| Ringlet | <i>Aphantopus hyperantus</i> | 51 | 0 | 51 |
| Arran Brown | <i>Erebia ligea</i> | 0 | 69 | 69 |
| Unknown Black | | 1 | 0 | 1 |
| Unknown White | | 7 | 2 | 9 |
| Unknown Blue | | 6 | 8 | 14 |
| Unknown Orange | | 12 | 6 | 18 |
| | Total number of individuals: | 250 | 240 | 490 |
| | Total number of species: | 22 | 16 | 29 |

Table S3. Land cover information for sites in the two regions: a) area in km² for Scania and Norrbotten as a whole, for Swedish Butterfly Monitoring Scheme (free) transects ($N = 5$ in each region), and Swedish Bird Survey (standardised) transects (see Supplementary Material Table S1, $N = 4$ in each region), b) Land cover data following the SMD database which is a refined Swedish version of the CORINE land cover database (Swedish Environmental Protection Agency, 2014). Code denotes SMD raster code, CLC Code the corresponding CORINE code, SMD Code the refined Swedish Code, CLC/SMD Description explains the SMD code. Numbers presented are coverage in each habitat category for each subgroup, recalculated for ease of comparison to the smallest area of all six subgroups (Free Transects, Scania: 4.45 km²)

| | | | | Scania | | | Norrbotten | | |
|------|----------|-----------|---|------------|----------------|------------------------|------------|----------------|------------------------|
| | | | | Total area | Free transects | Standardised transects | Total area | Free transects | Standardised transects |
| a) | | | | 15995 | 4.45 | 12.80 | 33570 | 4.52 | 12.80 |
| b) | | | | | | | | | |
| Code | CLC Code | SMD Code | CLC/SMD Description | | | | | | |
| 1 | 1.1.1 | 1.1.1 | Continuous urban fabric | 0.028 | | | 0.0078 | | |
| 2 | 1.1.2 | 1.1.2.1.1 | Discontinuous urban fabric, > 200 inhabitants and limited green areas | 0.46 | 0.76 | | 0.047 | | |
| 3 | 1.1.2 | 1.1.2.1.2 | Discontinuous urban fabric, > 200 inhabitants and larger green areas | 1.6 | 0.73 | | 0.22 | 2.4 | |
| 4 | 1.1.2 | 1.1.2.2 | Discontinuous urban fabric, < 200 inhabitants | 0.27 | 0.79 | | 0.070 | 1.4 | |
| 5 | 1.1.2 | 1.1.2.3 | Solitary houses and farm yards | 0.10 | 0.58 | | 0.10 | 0.94 | 1.8 |
| 6 | 1.2.1 | 1.2.1 | Industrial or commercial units | 0.47 | 2.9 | | 0.10 | | |
| 7 | 1.2.2 | 1.2.2 | Road and rail networks and associated land | 0.14 | | | 0.0076 | | |
| 8 | 1.2.3 | 1.2.3 | Port areas | 0.10 | | | 0.0022 | | |
| 9 | 1.2.4 | 1.2.4 | Airports | 0.092 | | | 0.019 | | |
| 10 | 1.3.1 | 1.3.1.1 | Sand and gravel pits | 0.048 | | | 0.028 | | 0.21 |
| 11 | 1.3.1 | 1.3.1.2 | Other mineral extraction sites | 0.044 | 3.8 | | 0.0045 | | |
| 12 | 1.3.2 | 1.3.2 | Dump sites | 0.035 | | | 0.0068 | | |
| 13 | 1.3.3 | 1.3.3 | Construction sites | 0.015 | | | | | |
| 14 | 1.4.1 | 1.4.1 | Green urban areas | 0.76 | | | 0.11 | 0.43 | |
| 15 | 1.4.2 | 1.4.2.1 | Sport facilities, shooting grounds etc | 0.12 | | | 0.021 | | |

| | | | | Scania | | | Norrbotten | | |
|----|-------|-------------|---|------------|----------------|------------------------|------------|----------------|------------------------|
| | | | | Total area | Free transects | Standardised transects | Total area | Free transects | Standardised transects |
| 16 | 1.4.2 | 1.4.2.2 | Airfields (grass) | 0.0013 | | | 0.0015 | | |
| 18 | 1.4.2 | 1.4.2.4 | Golf courses | 0.22 | | 3.6 | 0.010 | | |
| 19 | 1.4.2 | 1.4.2.5 | Non-urban parks | 0.087 | 3.1 | 1.7 | 0.0079 | | |
| 20 | 1.4.2 | 1.4.2.6 | Camping sites and holiday cottages | 0.094 | | | 0.011 | | |
| 30 | 2.1.1 | 2.1.1 | Non-irrigated arable land | 31 | 43 | 16 | 1.4 | 19 | 9.6 |
| 31 | 2.2.2 | 2.2.2 | Fruit trees and berry plantations | 0.15 | | | 0.0017 | | |
| 32 | 2.3.1 | 2.3.1 | Pastures | 5.9 | 14 | 8.4 | 1.0 | 2.8 | 0.57 |
| 40 | 3.1.1 | 3.1.1.1 | Broad-leaved forest, not on mires or bare rock | 8.7 | 18 | 40 | 2.1 | 3.4 | 2.0 |
| 41 | 3.1.1 | 3.1.1.2 | Broad-leaved forest on mires | 0.21 | | 2.6 | 0.74 | 3.3 | 0.76 |
| 43 | 3.1.2 | 3.1.2.1.1 | Coniferous forest, lichen dominated forest floor | | | | 3.6 | 0.53 | 5.4 |
| 56 | 3.1.2 | 3.1.2.1.2 | Coniferous forest, not on lichen dominated forest floor | 3.3 | 1.3 | 3.1 | 16 | 20 | 17 |
| 44 | 3.1.2 | 3.1.2.1.2.1 | Coniferous forest, 7-15 m | 6.7 | 3.4 | 7.7 | 10 | 12 | 18 |
| 45 | 3.1.2 | 3.1.2.1.2.2 | Coniferous forest, >15 m | 1.2 | | 3.0 | 3.6 | 0.042 | 1.3 |
| 46 | 3.1.2 | 3.1.2.2 | Coniferous forest on mire | 0.0026 | | | 0.63 | 1.2 | 2.7 |
| 47 | 3.1.2 | 3.1.2.3 | Coniferous forest on bare rock | 1.771 | 2.9 | 3.29 | 7.4 | 10 | 8.7 |
| 48 | 3.1.3 | 3.1.3.1 | Mixed forest, not on mires or bare rock | 0.17 | | 0.65 | 1.2 | 0.84 | 0.034 |
| 49 | 3.1.3 | 3.1.3.2 | Mixed forest on mire | | | | 0.00059 | | |
| 50 | 3.1.3 | 3.1.3.3 | Mixed forest on bare rock | 0.022 | | | 0.082 | | |
| 51 | 3.2.1 | 3.2.1 | Natural grassland | 0.038 | | | 0.015 | | |
| 52 | 3.2.2 | 3.2.2 | Moors and heathland | 0.11 | | | 0.13 | 0.042 | |
| 53 | 3.2.4 | 3.2.4.1 | Thickets | 1.7 | 1.2 | 6.3 | 6.3 | 0.83 | 7.7 |
| 54 | 3.2.4 | 3.2.4.2 | Clear-cuts | 2.1 | 3.9 | 3.5 | 11 | 14 | 8.8 |
| 58 | 3.3.1 | 3.3.1 | Beaches, dunes, and sand plains | 0.047 | | | 0.032 | | |
| 59 | 3.3.2 | 3.3.2 | Bare rock | 0.010 | | | 0.013 | | |
| 70 | 4.1.1 | 4.1.1 | Inland marshes | 0.12 | | | 0.16 | 0.25 | 0.19 |
| 71 | 4.1.2 | 4.1.2.1 | Wet mires | 0.049 | | 0.069 | 1.6 | | |
| 72 | 4.1.2 | 4.1.2.2 | Other mires | 0.49 | | | 13 | 1.1 | 2.3 |

| | | | | Scania | | | Norrbotten | | |
|----|-------|---------|-----------------------------------|------------|----------------|------------------------|------------|----------------|------------------------|
| | | | | Total area | Free transects | Standardised transects | Total area | Free transects | Standardised transects |
| 73 | 4.1.2 | 4.1.2.3 | Peatbogs | 0.13 | | | 0.022 | | |
| 74 | 4.2.1 | 4.2.1 | Salt marshes | 0.0032 | | | 0.0057 | | |
| 80 | 5.1.1 | 5.1.1 | Water courses | 0.043 | | | 0.80 | | |
| 81 | 5.1.2 | 5.1.2.1 | Water bodies, open | 1.8 | 0.34 | | 3.0 | 4.1 | 1.3 |
| 82 | 5.1.2 | 5.1.2.2 | Water bodies, closing vegetation | 0.072 | | | 0.044 | 0.50 | 0.010 |
| 83 | 5.2.1 | 5.2.1 | Coastal lagoons | 0.0074 | | | 0.057 | | |
| 84 | 5.2.2 | 5.2.2 | Estuaries | 0.0059 | | | 0.40 | | |
| 85 | 5.2.3 | 5.2.3.1 | Sea and ocean, open | 29 | | | 16 | 0.91 | |
| 86 | 5.2.3 | 5.2.3.2 | Sea and ocean, closing vegetation | 0.012 | | | 0.0094 | | |