Vascular plant species richness
at the landscape scale:
Patterns and processes

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Inga Schmiedel
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Betreuungsausschuss

Prof. Dr. E. Bergmeier, Abt. Vegetationsanalyse & Phytodiversität, Georg-August-Universität Göttingen

Prof. Dr. C. Leuschner, Abt. Ökologie & Ökosystemforschung, Georg-August-Universität Göttingen

Dr. H. Culmsee, Abt. Vegetationsanalyse & Phytodiversität, Georg-August-Universität Göttingen

Mitglieder der Prüfungskommission

Referent: Prof. Dr. E. Bergmeier, Abt. Vegetationsanalyse & Phytodiversität, Georg-August-Universität Göttingen

Korreferent: Prof. Dr. C. Leuschner, Abt. Ökologie & Ökosystemforschung, Georg-August-Universität Göttingen

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Abbreviations

ATKIS   Authorative Topographic-Cartographic Information System, *Amtliches Topographisch-Kartographisches Informationssystem*

Az.   Reference number, *Aktenzeichen*

BfN   Federal Agency for Nature Conservation, *Bundesamt für Naturschutz*

BMU   Federal Ministry for the Environment, Nature Conservation, Building and Nuclear Safety, *Bundesministeriums für Umwelt, Naturschutz, Bau und Reaktorsicherheit*

BNatSchG   Federal Nature Conservation Act, *Bundesnaturschutzgesetz*

BÜK 50   Soil map, scale 1:50,000, *Bodenübersichtskarte 1:50,000*

CaCO$_3$ Calcium carbonate

CBD   Convention on Biological Diversity

COP   Convention of the Parties

DBU   German Federal Foundation for the Environment, *Deutsche Bundesstiftung Umwelt*

DEM   Digital Elevation Model

DHDN   *Deutsches Hauptdreiecksnetz* (coordinate system)

DLM   Digital Landscape Model

DWD   German Weather Service, *Deutscher Wetterdienst*

EEA   European Environment Agency

e. g.   For example [*lat.* exempli gratia]

EIV   Ellenberg Indicator Values

EU   European Union

FFH   Fauna-Flora-Habitat (Habitats Directive)

FG   Fragmentation geometry

FGSV   Road Research Institute, *Forschungsgesellschaft für Straßen- und Verkehrswesen*

GBIF	Global Biodiversity Information Facility
GIS	Geographic Information System
GLM	Generalised Linear Model
GSPC	Global Strategy for Plant Conservation
i. e.	that means [*lat.* id est]
IPA	Important Plant Areas
LCT	Land cover type
LGN	Lower Saxon Authority for Land-surveying and Geoinformation, *Landesvermessung und Geobasisinformation Niedersachsen*
m$_{eff}$	Effective Mesh Size Index
n/a	not available
NLfB	Lower Saxon Regional Authority for Mining, Power and Geology, *Landesamt für Bergbau, Energie und Geologie*
NLÖ	Lower Saxon Regional Authority of Ecology, *Niedersächsisches Landesamt für Ökologie*
NLWKN	Lower Saxon Water Management, Coastal Defence and Nature Conservation Agency, *Niedersächsisches Landesamt für Wasserwirtschaft, Küsten und Naturschutz*
NMUK	Lower Saxon Ministry for the Environment and Climate Protection, *Niedersächsischen Ministerium für Umwelt, Energie und Klimaschutz*
NR	Geographical regions, *naturräumliche Regionen*
NSG	Nature protection area, nature reserve, *Naturschutzgebiet*
NW-FVA	Northwest German Forest Research Station, *Nordwestdeutsche Forstliche Versuchsanstalt*
RL	Red List
UG	Study area, *Untersuchungsgebiet*
UNECE	United Nations Economic Commission for Europe
VEAC	Victorian Environmental Assessment Council
z. B.	German for *zum Beispiel* (for example)

Summary

The heterogeneous geographic distribution of species richness has fascinated and challenged biologists over centuries. As a result, a large number of possible hypotheses that aim at explaining these patterns have been proposed. Currently, the consensus is that multiple processes simultaneously affect species richness patterns, with the influence of each process differing between taxa, landscapes and periods in time. Generally, environmental conditions are assumed to be the main drivers underlying species richness patterns at large scales. However, as many landscapes and thus species' habitats are strongly influenced by humans, particularly in Central Europe, the effect of human land use on richness patterns cannot be neglected. The severe anthropogenic changes in land cover, increasing landscape fragmentation, degradation and intensification have already led to large losses in (semi-)natural vegetation, biodiversity and ecosystem functions and are expected to cause even more losses in the future.

Protected areas play a major role in the alleviation of anthropogenic pressures on species and their habitats. However, as site selection for many of these areas has not been based on knowledge on species occurrences, their effectiveness might be limited.

The present thesis, conducted in the federal states of Lower Saxony and Bremen in northern Germany, therefore aims at 1.) identifying the richness patterns of vascular plant species in the two states; 2.) testing the effectiveness of the protected area network for the conservation of threatened species in the lowland region of the study area and 3.) analysing the relationship between species richness patterns and their underlying processes with special regard to the human influence.

In order to address these aims, analyses were carried out on grid-based floristic data obtained from the vascular plant survey of Lower Saxony (*Niedersächsisches Pflanzenartenerfassungsprogramm*, NLWKN 1982-2003) that were combined with high-resolution data on land cover and information on environmental conditions and the location and extent of protected areas. In doing so, the thesis represents the first comprehensive analysis of the extensive floristic data base of Lower Saxony.

To analyse the patterns of vascular plant species richness across the study region, the overall species richness as well as the richness of five sub-groups of species aggregated using information on residence and threat status of the species inside each landscape unit (1,762 grid cells of c. $30\,km^2$) was assessed. Richness of all species groups was found to be heterogeneously distributed across the region showing an increase in richness from north to south and – to some extent – also from west to east. The centres of high plant species richness of the different groups largely correlated with each other but also showed some interesting divergences.

The effectiveness of the protected area network in the lowland region of the state was shown to be generally high: 'Hotspots' of threatened species were found to be largely covered by nature reserves (German: *Naturschutzgebiete*) and Natura 2000 sites with the latter areas expediently complementing the former. It remained unclear, however, if the effective 'territorial' protection is reflected in an effective species protection.

The anthropogenic influence on the landscape of the study region was assessed using two separate analyses. These two studies focussed on the relationship between species richness and the degree of landscape fragmentation on one hand and landscape modification on the other hand.

The degree of landscape fragmentation was assessed by applying the *Effective Mesh Size Index* (m_{eff}). The relationship between m_{eff} and species richness of the total number of species as well as five sub-groups of vascular plants aggregated using their residence and threat status was assessed using a variation partitioning approach that separated the effects of landscape fragmentation, environmental conditions and spatial autocorrelation. Landscape fragmentation was found to significantly affect richness of all species groups. However, the strength of the effect varied between species, being largest for neophytes and smallest for threatened species.

A clustering analysis on landscape metrics describing landscape composition and configuration revealed a six-level gradient of landscape modification for the study region. The gradient spanned from highly fragmented urban landscapes to little fragmented landscapes with a high proportion of forest. Species richness of the total number of species as well as seven sub-groups of vascular plants aggregated using the residence and threat status as well as habitat preferences of the species were found to significantly differ along the gradient. For the entire and native species pool as well as for forest, low-nutrient indicating and threatened plants, species numbers peaked at the opposite ends of the modification gradient. For neophytes, urban plants and high-nutrient indicators richness decreased along the gradient from high species numbers in urbanised to low numbers in less modified landscapes.

The results obtained from the investigations in this thesis provide a basis for future landscape and conservation planning approaches in the study region by facilitating the identification of landscapes of high conservation value and/or concern. Provided that appropriate, high quality data like those used in the present thesis are available, the methodologies applied and developed in context of the thesis can be transferred to other regions to accordingly support conservation and landscape planning.

Zusammenfassung

Die auf verschiedenen Skalen-Ebenen heterogene Verteilung des Artenreichtums fasziniert Naturforscher und Biologen seit Jahrhunderten, so dass eine Vielzahl von Hypothesen zur Erklärung dieser Muster aufgestellt wurde. Es besteht breiter Konsens darüber, dass eine Vielzahl verschiedener Prozesse für die Variation des Artenreichtums verantwortlich ist. Je nachdem welche Taxa, welche Orte und welche räumlichen und zeitlichen Skalen betrachtet werden, wird den verschiedenen Faktoren dabei eine unterschiedlich große Bedeutung zugemessen. Umweltgradienten gelten in diesem Zusammenhang als die wichtigsten, den Artenreichtum bestimmende Faktoren. Da jedoch die Landschaften und damit Habitate der Arten weltweit und vor allem in Mitteleuropa stark anthropogen überprägt sind, sollte der auf den Artenreichtum wirkende Einfluss anthropogener Interventionen in diesem Zusammenhang nicht vernachlässigt werden. Die durch den Menschen und seine Aktivitäten verursachten Landnutzungsänderungen, eine steigende Landschaftsfragmentierung und -degradation sowie die Intensivierung der Landnutzung haben sich bereits im Verlust (halb-)natürlicher Landschaftselemente, dem Rückgang der Biodiversität und der Verschlechterung von Ökosystemfunktionen niedergeschlagen; eine weitere Verschlechterung der Zustände wird vorhergesagt.

In diesem Zusammenhang dient die Ausweisung von Schutzgebieten vielfach dazu, den auf die Arten und ihre Habitate wirkenden Druck zu vermindern. Da die Lage dieser Gebiete jedoch häufig nicht auf Grundlage der Kenntnis über das Vorkommen von Arten bestimmt wurde, dürfte die Effektivität der Schutzgebiete für den Artenschutz vielfach eingeschränkt sein.

Die vorliegende, die norddeutschen Bundesländer Niedersachsen und Bremen umfassende Arbeit untersucht 1.) die Muster des Artenreichtums der Gefäßpflanzenarten in den beiden Ländern; 2.) die Effektivität des Schutzgebietsnetzwerks des norddeutschen Tieflands für den Schutz seltener und bedrohter Gefäßpflanzenarten und 3.) den Zusammenhang zwischen Mustern des Artenreichtums und den ihnen zugrundeliegenden Prozessen unter besonderer Berücksichtigung des anthropogenen Einflusses. Die Arbeit basiert auf einem umfangreichen, im Rahmen des Niedersächsischen Pflanzenartenerfassungsprogramms (NLWKN 1982-2003) erhobenen Datensatzes, der räumlich explizite Informationen zur Verbreitung aller in den Bundesländern vorkommenden Arten liefert. Die floristischen Daten wurden für die durchgeführten Analysen mit hochaufgelösten Daten zu Landbedeckung und Umweltbedingungen kombiniert. Damit repräsentiert die vorliegende Arbeit die erste umfassende Auswertung des umfangreichen floristischen Datensatzes.

Für die Untersuchung der Muster des Gefäßpflanzenreichtums innerhalb der Untersuchungsregion wurden für die Gesamtzahl der Arten sowie für fünf auf Grundla-

ge des floristischen Status' und des Gefährdungsgrades der Arten zusammengestellte Untergruppen der Artenreichtum pro Landschaftseinheit (1.762 Messtischblatt-Quadranten à ca. $30\,km^2$) berechnet. Der Artenreichtum aller Gruppen zeigte eine heterogene Verteilung in der Untersuchungsregion, wobei eine Zunahme der Artenzahlen von Nord nach Süd und zum Teil auch von West nach Ost erkennbar war. Die Zentren hoher Artenvielfalt der verschiedenen Gruppen korrelierten miteinander.

Die Analyse der Effektivität der Schutzgebiete für den Schutz der seltenen und bedrohten Gefäßpflanzenarten zeigte, dass die Artvorkommen im niedersächsischen Tiefland zu einem relativ hohen Anteil durch Naturschutz- und Fauna-Flora-Habitat-Gebiete abgedeckt sind, wobei letztere Gebiete die erstgenannten sinnvoll ergänzten. Im Rahmen der Analyse konnte jedoch nicht untersucht werden, inwiefern sich der nachgewiesene effektive Gebietsschutz in einem effektiven Artenschutz niederschlägt.

Der anthropogene Einfluss auf die Landschaft der Untersuchungsregion wurde im Rahmen zweier separater Analysen untersucht, wobei der Zusammenhang zwischen Artenreichtum und Landschaftsfragmentierung einerseits und dem Grad der Landschaftsmodifikation andererseits analysiert wurde.

Die Landschaftsfragmentierung wurde in der vorliegenden Arbeit mittels des Landschaftsmaßes der 'Effektiven Maschenweite' (m_{eff}; engl. *Effective Mesh Size Index*) untersucht. Der Zusammenhang zwischen m_{eff} und dem Artenreichtum aller Gefäßpflanzenarten sowie fünf weiteren nach ihrem floristischen bzw. Gefährdungsstatus definierten Gruppen wurde mittels einer Varianzpartitionierung ermittelt, die die Effekte der Landschaftsfragmentierung von den durch Umweltvariablen und räumliche Autokorrelation der Daten verursachten Effekten separiert. Der Grad der Landschaftsfragmentierung erklärte einen signifikanten, jedoch unterschiedlich großen Anteil des Artenreichtums aller untersuchten Gruppen, wobei der stärkste Effekt für die Gruppe der Neophyten und der geringste für die Gruppe der seltenen und bedrohten Arten erkennbar waren.

Eine Cluster-Analyse auf Grundlage verschiedener, die Komposition und Konfiguration der Landschaft beschreibender Landschaftsmaße identifizierte für die Modellregion einen sechsstufigen 'Landschaftsmodifikations'-Gradienten. Dieser Gradient reichte von stark fragmentierten urbanen hin zu wenig fragmentierten Landschaften mit hohem Waldanteil. Der Gesamtartenreichtum sowie die Artenzahlen von sieben verschiedenen, nach ihrem floristischen Status, ihrer Gefährdung bzw. ihrer Habitat-Bindung bestimmten Artengruppen zeigten signifikante Unterschiede entlang des Gradienten. Die Gesamtartenzahl wie auch der Reichtum der indigenen, der bedrohten sowie der an Waldlebensräume und nährstoffarme Habitate gebundenen Arten war jeweils an den Enden des Gradienten am höchsten. Der Reichtum der Neophyten sowie der an urbane Räume und nährstoffreiche Habitate gebundenen Arten nahm dagegen von den urbanen zu den wenig beeinflussten Landschaften hin ab.

Die im Rahmen der Arbeit erhaltenen Ergebnisse können zukünftig als Grundlage für naturschutzfachliche Planungen in der Untersuchungsregion dienen, indem sie

die Ausweisung von für den Naturschutz relevante Landschaftsräume unterstützen. Sofern entsprechende, qualitativ mit denen der vorliegenden Arbeit vergleichbare Grundlagendaten vorliegen, eignen sich die im Rahmen der Arbeit angewendeten und entwickelten Methoden, um auf andere Regionen übertragen zu werden und auch dort naturschutzfachliche und landschaftsplanerische Prozesse zu unterstützen.

1 | Introduction

1.1 Patterns of species richness at the landscape scale

The heterogeneous geographic distribution of species richness has fascinated biologists for centuries (RICKLEFS, 2004; see BARTHLOTT et al., 2005 for an overview of the history of the exploration of diversity patterns of vascular plants). The acceleration of extinction rates in recent decades (HODAPP et al., 2014) has meant that the identification and analysis of the spatial patterns of species richness is now more important than ever. Species richness is frequently referred to as an indicator for biodiversity (HODAPP et al., 2014) as it is comparatively easy to measure and does not depend on knowledge of the abundance of species. Biodiversity, in turn, is widely accepted as to play an important role for ecosystem functioning (SECRETARIAT OF THE CBD, 2006; CARDINALE et al., 2012; HODAPP et al., 2014). Therefore, species diversity has been used as a proxy for the ecological status of ecosystems (HODAPP et al., 2014).

The importance of biodiversity is widely acknowledged in science as well as in political decision making and therefore numerous measures to explore and protect it have been implemented. On the global scale, the Convention on Biological Diversity (CONFERENCE OF THE PARTIES, 1992) is ratified by 194 countries (status September 2014; SECRETARIAT OF THE CBD, 2014) and aims at the "conservation of biological diversity, the sustainable use of its components and the fair and equitable sharing of the benefits arising out of the utilization of genetic resources" (SECRETARIAT OF THE CBD, 2005, p. 5). These goals have been implemented in European legislation by the EU Biodiversity Strategy (EUROPEAN COMMISSION, 1998). In this context, the Habitats Directive (together with the Birds Directive) forms the "cornerstone of Europe's nature conservation policy" (EUROPEAN COMMISSION, 2014) and aims at the protection of biodiversity, i. e. the diversity in flora, fauna and habitats.

As only what is known can be effectively protected, the identification of species richness patterns is an important aspect of their effective protection. Plants are seen as the "key structural elements of terrestrial ecosystems" (KREFT & JETZ, 2007, p. 5925) as they play an important functional role as primary producers and provide habitats for animal species (CARDINALE et al., 2011; SCHETTER et al., 2013). Because of this, as well as the fact that data on plant species occurrences are relatively easy to collect compared to the collection of data of mobile organisms (SCHETTER et al., 2013), a lot of research has focused on these organisms. Patterns of vascular plant species richness have been investigated on the global (MYERS et al., 2000; BARTHLOTT et al., 2005; KIER et al., 2005) as well as on the European scale (ESSL et al., 2013) and have been used for global-scale conservation priority setting (MYERS et al., 2000).

Species richness patterns may be either directly derived by summarising the knowledge on the spatial distribution of single species (see e. g. HAEUPLER, 2000; GARVE, 2007) or – if reliable data are missing – estimated by applying modelling approaches (cp. KIER et al., 2005; MAES et al., 2005; KREFT & JETZ, 2007).

The availability of digital data on species occurrences on a regional and national level has increased rapidly in recent years. At the global scale, the Global Biodiver-

sity Information Facility (GBIF) provides access to more than 400 million spatially explicit records of several taxonomic groups, including vascular plants (GBIF, 2014). In Europe, most countries maintain species mapping projects that gather data on the distribution and presence or even abundance of plant and animal species (see e.g. ZAJAC & ZAJAC, 2001; LANDUYT et al., 2012; NETPHYD & BFN, 2013). The mapping of species occurrences in Germany is supervised by the federal states (see e.g. HARDTKE & IHL, 2000; KORSCH et al., 2002; HAEUPLER et al., 2003). The results of this mapping are then reported to the federal authority.

Data on plant species occurrences provide a good basis for conservation planning (BROOKS et al., 2004). Therefore information obtained from floristic mapping have been applied for various purposes. For example, HAEUPLER & VOGEL (1999) used data obtained from such a mapping to derive patterns of species richness for the whole of Germany. Based on species number of rare and threatened plants per landscape unit, the authors determined eight 'hotspots'[1] of phytodiversity distributed across the country. On a smaller scale, WINTER et al. (2010b) used information on the occurrences of threatened plant species in an administrative district of Germany to identify Important Plant Areas (IPAs) that form an integral part of the Global Strategy for Plant Conservation (THE SECRETARIAT OF THE CBD, 2002).

Species records have furthermore been used to analyse the indicative power of selected sub-groups of plant species for the overall species richness of a region: For north-west Germany, DIEKMANN et al. (2008) used records of forest species taken from a flora atlas of the Weser-Elbe region and related the richness of these species to that of the overall number of plants reported for the region. The authors showed that species numbers of this group are indicative for total species richness as well as for the number of threatened species. Similarly, KATI et al. (2004) found that woody plants acted as suitable indicators for the richness of other taxonomic groups in the Mediterranean.

In the study area of the present thesis, the federal states of Lower Saxony and Bremen, species records amongst other things serve as a basis for the creation of distribution maps[2], the compilation of Red Lists and as a contribution to landscape and environmental planning (SCHUPP et al., 2001).

1.2 Linking patterns and processes: What influences species richness patterns at the landscape scale?

The analysis of the underlying causes of species richness patterns is a major goal in ecological, evolutionary, and biogeographical research (GASTON, 2000; GODFRAY &

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[1]In the following the term 'hotspot' is used for landscape units hosting a high richness of threatened plant species (cp. definition of HAEUPLER & VOGEL, 1999).

[2]See GARVE (1994) for such maps for threatened plant species and GARVE (2007) for all vascular plant species of the state.

LAWTON, 2001; GOTELLI et al., 2009; WIENS, 2011) and has challenged researchers for centuries (RICKLEFS, 2004). In recent decades, researchers have provided many different ideas on the question of what influences richness patterns (see RICKLEFS, 2004 for an overview on the history of these concepts). In accordance with many researchers, COLWELL et al. (2004, p. E14) suggests that "multiple factors are surely at work and that any particular factor may be more or less important for different taxa, different places, and different spatial and temporal scales". Processes and factors influencing species richness patterns include speciation and extinction rates (COLWELL et al., 2004; WIENS, 2011), geometric constraints (i. e. mid-domain effect, MDE, COLWELL & HURTT, 1994) as well as geographic patterns of temperature, productivity, potential evapotranspiration, topography, formation history (COLWELL et al., 2004) and human activities (WHITE & KERR, 2007; WANG et al., 2011). The following two subsections will focus on the role of abiotic environmental conditions and the human influence shaping species richness patterns. It has to be kept in mind, though, that the underlying processes influencing these patterns are far too complex to be fully disentangled. Thus, usually the processes are approached by using indicators or surrogates.

1.2.1 Environmental conditions as key drivers for richness patterns of plants

The role of the abiotic environment in shaping species richness patterns has been analysed on various spatial scales ranging from global (KREFT & JETZ, 2007) to local assessments (HEIKKINEN, 1996). BARTHLOTT et al. (2007, p. 308) describes 'geodiversity', i. e. "the heterogeneity of topography, geology, soils, or climate", as the main driver for global spatial patterns in species richness. More specifically, KREFT & JETZ (2007) showed that for vascular plants potential evapotranspiration, the number of wet days per year, and measurements of topographical and habitat heterogeneity are core predictors of global richness patterns. Similar factors were found to explain species richness on a smaller scale: Elevation range followed by minimum monthly potential evapotranspiration was identified as to be the most important factor determining woody plant species richness of the Iberian Peninsula VETAAS & FERRER-CASTÁN (2008). Similarly, climatic variables like temperature and potential evapotranspiration were found to be by far the most important predictors for vascular plant species richness in Austria (MOSER et al., 2005). For alien plant species richness in the European Alps, environmental heterogeneity proved to be the best surrogate (DAINESE et al., 2014). The same factor in combination with the presence of calcareous rock outcrops and threshold values of minimum precipitation was found to also determine species richness in Switzerland (WOHLGEMUTH, 1998). For Germany, native and alien plant species richness was found to be largely dependent on the diversity of geological substrates (KÜHN et al., 2003). Similarly, on the local scale the overall habitat diversity was identified as the main driver for species richness of vascular plants in a nature reserve in northern Finland (HEIKKINEN, 1996).

All these findings promote the hypothesis that overall species richness tends to be higher in a varied environment compared to a monotonous one, as heterogeneous environmental conditions provide a greater diversity of niches potentially suitable for plant species (HONNAY et al., 2003; ADAMS, 2009).

1.2.2 The human impact on richness patterns of plants

The previous subsection dealt with the role of the abiotic environment in shaping species richness patterns. Undoubtedly, environmental conditions are the main drivers underlying species richness patterns on a large scale. However, as many landscapes and thus species' habitats worldwide and particularly in Central Europe are strongly influenced by humans (KOWARIK, 1990), the shaping effect of human interventions on richness patterns should not be overlooked.

Over the last few decades, much attention has been focused on the analysis of the effects of human interventions at the landscape scale. In this context, a large number of studies have searched for surrogates describing the human impact on landscapes and biodiversity.

Human-induced threats to and protection of plant species richness

Habitat loss, fragmentation and degradation have been identified as being the main factors threatening species richness (FAHRIG, 2001; AGUILAR et al., 2006; KRAUSS et al., 2010). Both habitat fragmentation and loss – the latter being an "important by-product of habitat fragmentation" (McGARIGAL, 2014, p. 89) – are known to increase the isolation between plant or animal sub-populations, thereby impeding the genetic exchange between individuals. The resulting loss in genetic variation may increase the probability of extinction by demographic and environmental stochasticity (BOOY et al., 2000). In many cases, habitats are not immediately fully destroyed but undergo a gradual degradation that, however, ultimately leads to their complete destruction.

A multitude of landscape-level processes are responsible for habitat loss, fragmentation and degradation in Western Europe, including – but not being limited to – urbanisation, agricultural intensification, eutrophication and climate change (cp. Fig. 1.1).

Urbanisation[3] has been identified as one of the leading causes of species extinction (McKINNEY, 2006; ARONSON et al., 2014). Urbanisation affects biodiversity in many ways, e. g. through the fragmentation and destruction or degradation of species' habitats and by altering the precipitation and temperature regime as well as air, soil and water quality (SUKOPP & STARFINGER, 1999). In their review on causes of species endangerment in the United States, CZECH et al. (2000) found that urbanisation in the country endangers more species than any other human activity although barely 5 %

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[3]i. e. "the process by which towns and cities are formed and become larger as more and more people begin living and working in central areas" (Definition taken from MERRIAM-WEBSTER (2014).

of the total surface area of the United States is covered by urban and other built-up areas (McKinney, 2002). In Germany, urban and traffic areas cover about 14 % of the landscape (Statistische Ämter des Bundes und der Länder, 2012), and this proportion is increasing: Each day about 80 ha of undeveloped land are transferred to building land (UBA, 2014) resulting in an enormous loss of unsealed surface area and a fragmentation and loss in species' habitats. Despite the destructive effect of urbanisation on native habitats, cities are found to be generally richer in plant species than surrounding areas (Honnay et al., 2003; Wania et al., 2006; Lippe & Kowarik, 2008). This pattern might arise from different aspects: Generally, niche availability in urban areas is known to be rather high (Deutschewitz et al., 2003; Honnay et al., 2003; Wania et al., 2006). Kühn et al. (2004) furthermore suggested that urban areas are commonly situated in naturally diverse ecoregions resulting in an overall increased species diversity. Also, alien species are known to show increased numbers in urban areas, adding to the urban species pool (Roy et al., 1999; Kühn et al., 2004).

Agricultural intensification has been proposed as one further main cause of biodiversity loss (Tilman et al., 2001; Tscharntke et al., 2005). In Central Europe it accelerated in course of industrialisation in the late 19th and early 20th century (Tscharntke et al., 2005; Krause & Culmsee, 2013) and has been reinforced by the advent of the common agricultural policy of the EU since the 1950s (Stoate et al., 2001; Krause et al., 2011). Conventional farming practices affect species diversity mainly through the conversion of non-arable land that increases habitat loss and fragmentation (Tscharntke et al., 2005). However, not only the conversion of non-arable to high-productive cropland but also the abandonment of marginal sites, the removal of small-scale landscape elements and the application of pesticides and fertilisers have an effect on species (Tilman et al., 2001). Studies on the landscape scale in Europe have identified large losses in plants, invertebrates and farmland birds following agricultural intensification (Andreasen et al., 1996; Donald, 1998; Donald et al., 2001; Butler et al., 2010; Storkey et al., 2012; Meyer et al., 2013).

The **eutrophication** of ecosystems, i. e. primarily the input of nitrogen, phosphorous and sulphur compounds, has been identified to reduce plant species richness in a broad range of European ecosystems (Dise, 2011). As high fertiliser inputs largely add to the eutrophication of a landscape, the process is tightly linked to that of agricultural intensification. Additional inputs adding to the eutrophication of ecosystems come from industry and transport (UNEP, 2014). Due to strict regulations in Europe (see 'Convention on Long-range Transboundary Air Pollution' and its eight subsequent protocols; UNECE, 1979), emissions of air pollutants, especially sulphur, have been reduced in recent years (Dise, 2011). However, critical loads are still exceeded in many semi-natural ecosystems in Europe (Hettelingh et al., 2008; Dise, 2011). Eutrophication affects species mainly through the degradation of their habitats – in the worst case habitats will become uninhabitable for the species under consideration which results in habitat loss. The negative impacts of eutrophication on species

richness have been reported for all kinds of habitats across Europe, e. g. for acid grass-lands in the Atlantic biogeographic region (Stevens et al., 2010), southern Swedish temperate oak forests (Brunet et al., 1998), Swiss mountain grasslands (Roth et al., 2013) and British heathlands as well as acid and calcareous grasslands (Maskell et al., 2010).

Based on different scenarios, the impact of **climate change** on the distributions of plant and animal species has been frequently modelled in recent years (e. g. Bakkenes et al., 2002; Thuiller et al., 2005; Pompe et al., 2008). These analyses have shown that many European plant species are at risk from climate change (Thuiller et al., 2005). Thomas et al. (2004) predicted that due to climate change 6–8 % of all plant species in Europe could become extinct by 2050. For Germany, Pompe et al. (2008) suggested that – even under moderate climate change assumptions – negative impacts on the German flora will arise. The authors point out that due to climate change severe shifts in the species pools will occur.

Climate change scenarios up to the year 2080 for Germany have identified a definite warming trend of 1.6–3.8 °C and small overall changes in annual precipitation (<10 %) (Zebisch et al., 2005). However, changes in seasonal precipitation will occur with an increase in winter and a decrease in summer precipitation (Zebisch et al., 2005). Due to their particular environmental characteristics, different regions in Germany are assumed to be affected by climate change by a varying degree (Zebisch et al., 2005). Highest vulnerability to climate change is expected for south-west and the central parts of eastern Germany as well as the Alps. The low mountain ranges of Germany as well as the coastal regions show medium vulnerability whereas north-west Germany was identified to be least vulnerable (Zebisch et al., 2005). Therefore, impacts of climate change on the study area of this thesis, the federal states of Lower Saxony and Bremen, can be assumed to be small compared to other German states.

The processes discussed in the previous paragraphs all add to the modification of a landscape and by doing so pose a threat to species. As it is difficult, if not impossible, to consider all of these processes separately as their effects on biodiver-sity in a landscape are usually interlinked, multi-layered concepts combining them have been proposed by several researchers. These concepts aim at describing the degree of modification a landscape has undergone. Concepts proposed involve 'land-scape modification' (Forman & Godron, 1986) and the related concepts of 'landscape naturalness' (Machado, 2004; Liira & Sepp, 2009; Winter et al., 2010a), 'hemeroby' (Sukopp, 1976; Kowarik, 1988; Grabherr et al., 1998; Steinhardt et al., 1999; Wrbka et al., 2004; Walz & Stein, 2014), 'urbanity' (Hill et al., 2002; Wrbka et al., 2004) and the 'degree of culturalness' (Jansen et al., 2009). These concepts are used to assess the conservation value of landscapes (Wrbka et al., 2004) and have been proposed to al-low conclusions on biodiversity (Walz & Stein, 2014). However, until now no study has directly analysed the relationship between any of these concepts and species rich-ness/diversity.
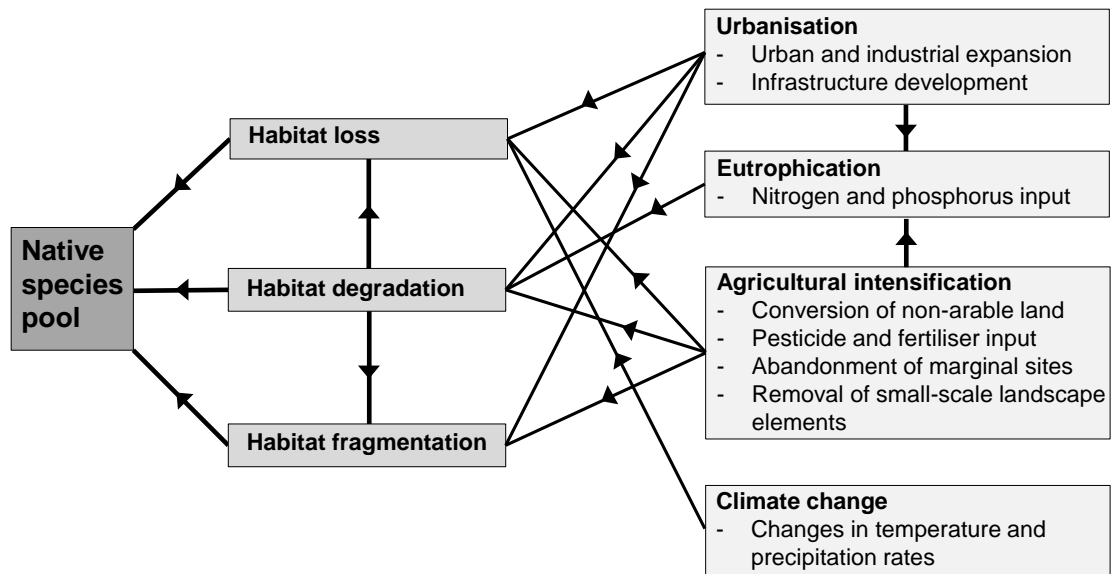
Figure 1.1: Human-induced threats to the local species pool (selection of main threats). Arrows indicate the main direction of the impact.

As discussed in the previous paragraphs, species in today's highly industrialised landscapes are threatened by a multitude of processes, most of them being linked to human-induced changes in the landscape. To reduce the negative impacts of these changes on species, a number of different concepts for the protection of species have been proposed that can generally be allocated into segregative and integrative measures (PLACHTER, 1991). These concepts have been developed for agricultural landscapes but can analogously be transferred to forests, water bodies and urban areas (PLACHTER, 1991). In segregative concepts, areas used for production are separated from those designated for the protection of species, habitats and ecosystems (JESSEL, 2012). In contrast, integrative concepts try to include both production and conservation in the same areas. Integrative measures reduce negative impacts on species by a reduction in the intensity of land use and the establishment of biotope networks (PLACHTER, 1991). This management can, amongst others, promote species tolerant to land use (TSCHARNTKE et al., 2012), improve ecosystem services and reduce isolation of habitats (JESSEL, 2012; see also "dual conservation strategy", JACKSON et al., 2009). However, it can not promote the protection of species linked to (semi-)natural ecosystems (e. g. mires, rocky outcrops). For the protection of such species the segregative concept might be useful. Segregation is mainly enforced by the designation of protected areas inside which land use is forbidden, restricted or regulated to provide habitats for species otherwise lost from a region.

In Germany, there are various categories of protected areas, of which national parks, nature reserves (German: *Naturschutzgebiete*) and Natura 2000 sites (EU Habitats Directive) are those covering the largest areas. Protected areas are widely ac-

cepted as being essential for the long-term conservation of biodiversity (Evans et al., 2006). However, their effectiveness in protecting species remains largely unclear as the extent and location of many protected areas have not been defined using knowledge on species occurrences (Winter et al., 2010b; Jedicke, 2012). Determining the effectiveness of such areas "in representing and maintaining biodiversity is a core issue in systematic conservation planning" (Gaston et al., 2006, p. 77). Nevertheless, few studies until now have focused on this topic. Regional studies that analysed the effectiveness of protected areas have identified numerous gaps in the existing protected area network (see e. g. Scott et al., 2001 for the USA, Andelman & Willig, 2003 for the Western Hemisphere and Deguise & Kerr, 2006 for Canada). An analysis by Rodrigues et al. (2004) uncovered such gaps on a global scale. In contrast, for Great Britain, Jackson et al. (2009) found the current protected areas to cover the occurrences of a large majority of Red List plant species. However, despite the good overall species coverage, less than one third of the total number of occurrence records were covered. It is obvious that results obtained for one particular region cannot be directly transferred to other areas and that further research in this field is strongly needed.

Surrogates for the human influence on species richness patterns

As mentioned earlier, much attention has focused on the analysis of the effects of human interventions on species richness patterns in recent decades. Studies on this topic usually have searched for surrogates describing and quantifying the human impact on landscapes and biodiversity. For this purpose, land cover data that reflect the influence of humans on the landscape as well as landscape metrics used to quantify the magnitude of this impact have been widely applied. Using landscape metrics either the composition (i. e. the abundance of patch types in a landscape) or configuration (i. e. the spatial arrangement of patches) of a landscape can be assessed (cp. subsection 2.2.5). Indices used include measures of landscape diversity, patch shape complexity, edge length and density, core area sizes as well as metrics used to quantify landscape fragmentation, landscape/habitat connectivity and isolation. Below, some of these measures and their relevance for species richness and diversity are discussed.

   Landscape diversity – analogical to 'geodiversity' (cp. subsection 1.2.1) – was identified to be a convenient surrogate for species richness (Luoto, 2000; Honnay et al., 2003). Different measures have been used to express landscape diversity, the simplest merely reflecting the number (i. e. patch richness) or proportions of different land cover types (Deutschewitz et al., 2003; Kühn et al., 2003; Wania et al., 2006), the more complex ones combining richness and evenness of land cover types (e. g. Simpson's and Shannon's Diversity Index; see e. g. Deutschewitz et al., 2003; Honnay et al., 2003; Kumar et al., 2006).

Several studies have analysed the relationship between species richness and landscape diversity. For Germany, a high structural and habitat diversity in combination with moderate levels of natural and/or anthropogenic disturbance was found to increase richness of native and alien plant species (DEUTSCHEWITZ et al., 2003). Similarly, regional plant species richness of Flanders, Belgium was found to be positively correlated with landscape diversity expressed in terms of high patch richness and evenness and irregular patch shapes (HONNAY et al., 2003). As landscape diversity in the study area was found to go along with the degree of urbanisation also a positive correlation between the percentage of built-up area and species richness was stated. As previously discussed in the paragraph on the effects of urbanisation, the latter has been frequently reported by researchers and has been attributed to the increased niche availability in urban areas.

However, not only landscape diversity measures have been applied to explain species richness patterns. Several studies have used **patch shape complexity** as a surrogate for species richness as they assumed that simple patch shapes will be dominant in landscapes strongly influenced by humans, whereas more natural landscapes should be characterised by irregular shaped patches (MOSER et al., 2002; YAMAURA et al., 2008). In the study of MOSER et al. (2002) an index of patch shape complexity developed by the authors that is based on the number of shape-characterising points along a polygon's boundary proved to be a good predictor for the species richness of vascular plants and bryophytes in Austria: the authors found species richness to be highest in landscapes for which irregular patch shapes were reported. Similarly, for a Mediterranean forest landscape SCHINDLER et al. (2013) identified patch shape to be a good indicator for the diversity of woody plants. In contrast, by using a shape index independent of area for Japan, the study of YAMAURA et al. (2008) found the richness of forest-floor plants to be influenced by patch shape to only a limited extent whereas local forest structure had a much larger effect.

As **landscape fragmentation** resulting in habitat loss and isolation is supposed to be one of the largest threats to biodiversity worldwide (JAEGER, 2000; see also previous section on threats to species) its indicative power for species richness has been analysed by numerous studies at the local and landscape scale (see for an overview DEBINSKI & HOLT, 2000; FAHRIG, 2003). The simplest way in which the fragmentation of a landscape can be expressed is by using measures of patch numbers or patch density (McGARIGAL, 2014). However, as these simple fragmentation measures are supposed to be insensitive and inconsistent in their behaviour (McGARIGAL, 2014), attempts have been made to develop alternative and more explicit measures that "behave in a consistent and logical manner across all phases of the fragmentation process" (McGARIGAL, 2014, p. 131). Still, despite their qualities, these indices (i. e. *Landscape Division Index*, *Splitting Index* and *Effective Mesh Size*; JAEGER, 2000) until now have not been applied as potential surrogates for species richness. Instead, frequently, the size and the **isolation** of patches or the **connectivity** between them were used as (inverse)

measure of fragmentation (e. g. HELM et al., 2006; COUSINS et al., 2007; KOLB, 2008; ZIMBRES et al., 2013). RAATIKAINEN et al. (2009) for extensive grasslands in southern Finland used a connectivity index taking into account the distances to all possible source populations ('IFM measure', cp. MOILANEN & NIEMINEN, 2002) to analyse how richness of spring ephemerals is related to habitat area and connectivity. They found the number of ephemerals to be highest in larger and well-connected study patches. In the study of ADRIAENS et al. (2006) a similar measure was applied to analyse how vascular plant species richness of calcareous grasslands in southern Belgium is influenced by landscape configuration. The authors found that the connectivity between grassland patches only affected the number of specialist species but not that of generalists. The opposite result was obtained by CRISTOFOLI et al. (2010) for heathlands in the high Ardenne (Belgium): By using measures of patch connectivity and isolation the authors found generalist, not specialist species richness to be correlated with connectivity. Interestingly, for islands in a large lake in the south-east of China, island isolation measured as the Euclidean distance to the mainland did not significantly explain vascular plant species richness (HU et al., 2011).

The aforementioned examples show that effects of landscape fragmentation and habitat isolation may vary depending on the species groups and habitats considered (EWERS & DIDHAM, 2006; RODRIGUEZ-LOINAZ et al., 2012). Therefore, further studies on this important topic using different measures of fragmentation and isolation/connectivity should be undertaken for different landscapes and habitats.

1.3 General objectives and thesis outline

The present thesis applies spatially explicit data on the distribution of vascular plant species in combination with various information on land cover and environmental conditions to identify processes shaping species richness patterns. In doing so, the analyses focus on the significance of the anthropogenic influence and only marginally consider the importance of environmental conditions. All analyses were conducted for the federal states of Lower Saxony and Bremen, northern Germany, for which spatially explicit area-wide data on the occurrences of all vascular plant species (plant survey of Lower Saxony, NLWKN 1993–2003) as well as information on land cover and environmental conditions are available.

The main goals of the thesis are:
   a) To identify and describe the patterns of vascular plant species richness in Lower Saxony and Bremen, Germany.
   b) To evaluate the adequacy of the protected area network for the protection of rare and threatened plant species in the lowland region of the study area.
   c) To analyse the relationship between richness patterns and the human influence on the landscape.

The thesis is subdivided into four main data chapters (chapters 3 to 6) briefly presented in the following paragraphs. These chapters are preluded by a chapter that introduces the reader to the study area and gives an overview on the data used in the analyses (chapter 2). The thesis closes with chapter 7 which summarises and synthesises the main findings of the previous chapters and highlights open questions and future research needs.

As the key conservation work and data collection is conducted by local authorities and the large number of volunteers working for the plant survey of Lower Saxony, considerable parts of the thesis (Chapter 3 and 4) were published in German to make the results accessible to these people in charge, of which not all are familiar with the English language. This audience-focused approach guarantees that the published research can be directly implemented within the study area. The German articles are preluded by detailed English summaries to make the key results available to the global research community.

Patterns of vascular plant species richness   As pointed out in section 1.1, the identification of spatial patterns of species richness provides an important key aspect of conservation planning. The study presented in chapter 3 of this thesis makes use of the extensive species record database of the vascular plant survey of Lower Saxony and Bremen to derive richness patterns of six different species groups. The groups considered in the analysis comprise (a) the total number of species that has been reported for Lower Saxony and Bremen, (b) species grouped by their residence status (native species, archaeophytes, neophytes) and (c) species grouped by their threat status (non-threatened species, threatened species). The study serves as a starting point on which the studies presented in the following chapters are based.

The following research questions are answered by interpreting maps of the richness patterns of the aforementioned species groups and by taking into account the results obtained from a correlation analysis:

- Which patterns of vascular plant species richness are observable in Lower Saxony and Bremen and the three natural landscape regions, respectively?
- Do the different plant species (sub-)groups show richness patterns that are distinguishable from each other and how much do their richness centers overlap?
- In which areas do threatened species show highest species numbers and how do these hotspots differ from that of non-threatened species?

Adequacy of the protected area network   As knowledge on the spatial distribution of the rare and threatened species of Lower Saxony and Bremen is of major importance for their effective protection, occurrences of these species have been recorded at a much finer resolution (c. 2 km^2) than those of all other species. In the study presented in chapter 4 these high-resolution data were used to test to which extent the hotspots of rare and threatened species ('RL species') in the lowlands of Lower Sax-

ony are covered by protected areas. The analysis considers nature protection areas (*Naturschutzgebiete*, NSG) and NATURA 2000 sites. Additionally to the total number of threatened plant species Red List species linked to forest habitats ('RL forest species') are included in the analysis. The data are analysed using Generalised Linear Models (GLM) with Poisson distribution. The study aims at answering the following research questions:

- What patterns of diversity can be observed for the two species groups?
- Are occurrences of threatened species covered by nature protection areas?
- Does the designation of NATURA 2000 sites improve the situation?
- Are the results obtained for RL species transferable to RL forest species?

The human influence on species richness patterns As outlined in section 1.2, species richness patterns in industrialised countries are assumed to be strongly affected by human activities. To understand the magnitude and direction of this influence and the underlying regulatory processes is an important field of research. The two chapters dealing with this subject (chapter 5 and 6) use different methodological approaches to illuminate the relationship between species richness patterns and human influence.

The study presented in chapter 5 investigates whether landscape fragmentation expressed by means of the *Effective Mesh Size Index* is related to the richness of six different groups of vascular plants aggregated using information on their residence and threat status, respectively. The index has frequently been suggested to be ecologically meaningful (Jaeger, 2000; Jaeger et al., 2008; Girvetz et al., 2008) and is widely applied in studies on landscape fragmentation. However, its relevance for biodiversity has not been tested and its suitability as an indicator of biodiversity has even been questioned (Esswein, 2007). The presented study aims to shed light on this topic by applying variation partitioning techniques used to disentangle the relationship between species richness, landscape fragmentation and environmental variables. Spatial autocorrelation in the data is considered by using PCNM analysis. It is hypothesised that:

- Landscape fragmentation, expressed as Effective Mesh Size (m_{eff}), varies in space, which results in a gradient in the degree of fragmentation within the sample of landscape units used in this study.
- The choice of fragmentation geometries that are used for the calculation of m_{eff} is of major importance for explaining plant species richness patterns on the landscape scale.
- Plant species richness varies between landscape units along the fragmentation gradient, with the effect size differing between different groups of species dependent on their residence (natives, archaeophytes and neophytes) and threat status (threatened vs. non-threatened).

The study presented in chapter 6 makes use of k-means clustering to derive a gradient of 'landscape modification' (cp. subsection 1.2.2) for 1386 landscape units of Lower Saxony and Bremen. The analysis is based on landscape metrics quantifying landscape composition and configuration. Clusters differing in terms of their degree of landscape modification are set up and placed along a 'gradient of landscape modification' according to the one proposed by FORMAN & GODRON (1986). The gradient is then linked to richness patterns of the total number of vascular plant species as well as that of seven sub-groups that were aggregated using information on residence and threat status and habitat preferences of the species. The study seeks to verify the following hypothesis:

- Based on landscape metrics used to assess the composition and configuration of a landscape, clusters of landscapes with similar characteristics can be differentiated. These can be placed along a gradient of landscape modification as proposed by FORMAN & GODRON (1986), spanning from (semi-)natural (mostly forested) to urbanised landscapes.
- The landscape clusters are meaningful for vascular plant species richness, i. e. species numbers vary significantly along the gradient of landscape modification intensity.
- Species numbers of different vascular plant species groups show different patterns along the gradient of landscape modification intensity.

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2 | Description of data

2.1 Study region

The study region comprises the German federal states of Lower Saxony and Bremen. Lower Saxony encompasses an area of 47.624 km^2 whereas Bremen as the smallest federal state of Germany has a size of only 404 km^2. As landscape formation history and natural environmental configuration, colonisation history and current land use vary widely across the state, Lower Saxony harbours a high number of different habitats. The area may roughly be subdivided into three regions, the coastal, the lowland and the upland area (Fig. 2.1), that largely differ in their characteristics.

The **coastal area** is dominated by marine deposits from the Holocene (HEUNISCH et al., 2007). It comprises the Wadden Sea, its islands and salt marshes, the estuaries of the three largest rivers of Lower Saxony (Ems, Weser and Elbe) as well as impoldered tidal wetlands. It shows an oceanic climate with mean annual precipitation rates of about 700–900 mm and mean annual temperatures of c. 9 °C. Main soil types occurring in this area are Fluvisols, Leptosols and Gleysols (NLFB, 1997[1]). The costal area is predominantly covered by grasslands with a proportion of 60 % whereas cropland areas contribute 20 % and forests merely 1 %.

The **lowland area** was formed during the Pleistocene and harbours various types of landscapes ranging from intensively managed agricultural landscapes in the western part to more wooded landscapes in the eastern part. From West to East, the climate changes from oceanic (annual precipitation rate c. 900 mm) to suboceanic-subcontinental (annual precipitation rate c. 550 mm). Mean annual temperatures in this area fluctuate around 8 °C. The climatic gradient is reflected in the soil characteristics: In the West the dominating soil types are those affected by seepage water, water logging and mass transfer (Histosols, Podzols); in the East primarily Cambisols and Leptosols are found. The area is dominated by croplands that cover about 40 % of this region; forests and grasslands are present at proportions of about 20 % each.

The **upland area** is a very heterogeneous region that is dominated by deposits from the Meso- and Palaeozoic (HEUNISCH et al., 2007). Due to its diverse topography, climatic conditions vary strongly. The uplands are dominated by aeolian silt as well as clay, sand and lime stone deposits (NLFB, 1997). Luvisols and Cambisols are the most common soil types in this region. Mean annual temperatures range between 8–9 °C. Mean annual precipitation rates vary between 650–750 mm in the lower parts of the area and 800–900 mm in the elevated parts. The Harz mountains at the South-western border of Lower Saxony feature very specific climatic conditions: The most elevated parts (up to 971 m a.s.l.) experience a montane climate with an annual mean precipitation rate of up to 1,500 mm and annual mean temperatures of only 7 °C (NLFB, 1997). The Harz region is dominated by base poor silicate bedrock as well as argillaceous shale and greywacke. Main soil types occurring in this area are Cambisols and Podzols (NLFB, 1997). Out of the three regions, the upland area features the high-

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[1]Translation of German soil types follows IUSS WORKING GROUP WRB (2006).

est proportion of forests (c. 30 %); grasslands only prevail in about 9 % of the area whereas croplands dominate c. 45 % of the region.

The study region was subdivided into 1762 grid cells (*Messtischblatt-Quadranten*) with a size of about 5.5 km × 5.5 km (1/4 of a topographical map, scale 1 : 25,000; Fig. 2.1). Out of these 1386 grid cells are fully covered by land, 353 grid cells are located on the edge and 23 grid cells are fully covered by sea. For a more detailed study (see chapter 4, SCHMIEDEL et al., 2013), the lowland area of the state was overlaid with a more detailed grid of 15.729 cells with a size of about 1 km × 2 km (*Minutenfelder*, 1/60 of a topographical map, scale 1 : 25,000).

2.2 Data and data preparation

Data available for the study region included information on the distribution of vascular plant species, data on environmental variables as well as land-use types (Table 2.1). The following paragraphs give a detailed overview of these data.

Species data were analysed using MS Access (MICROSOFT CORPORATION, 2010) and ArcGIS-ArcInfo 9.2 (ESRI INC., 2007). ATKIS data, areal information as well as environmental variables – unless stated otherwise – were analysed using ArcGIS-ArcInfo 9.2, Hawth's Tools (BEYER, 2004) and Fragstats 3.3 (MCGARIGAL & CUSHMAN, 2002).

2.2.1 Species data

Occurrence data of all vascular plant species of Lower Saxony and Bremen were made available as an MS Access database by the vascular plant survey of Lower Saxony (*Niedersächsisches Pflanzenartenerfassungsprogramm*, NLWKN; Table 2.1). The data base (last update 15 December 2008) comprises 2,395,045 entries and covers the time span from 1993–2003 (first and second period of the state-wide mapping; third period is still underway). Out of all taxa listed in the data base only the 1879[2] non-erratic ones were considered[3]. Occurrence data were available on the grid basis mentioned in the previous chapter.

Prior to the analyses, species were aggregated on the highest taxonomical level found in the data base, i. e. subspecies were assigned to the species level and species to the respective aggregate found in the data base[4]. This aggregation affected 213 taxa (see Table A.3 in Appendix A for an overview). Thus, in total, 1666 taxa remained in the analysis out of which 72 were occurring in grid cells fully covered by sea or located on the edge of the study area only[5] (cp. section 2.1) and thus were excluded

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[2]Due to errors in the database that were detected just after the first article (SCHMIEDEL et al., 2011) was published, the total number of species considered in this article was 1819 taxa only.

[3]In total, 2708 taxa are listed in the floristic list of the data base out of which 829 are extinct, erratic or do not occur.

[4]The exception being the first article (SCHMIEDEL et al., 2011) which did not apply any aggregation.

[5]Hereafter referred to as 'edge only' species.

for the analyses conducted in Schmiedel & Culmsee (submitted) and Schmiedel et al. (submitted).

For the analyses conducted in context of the present thesis, the species listed in the data base were aggregated into sub-groups using the residence and threat status and habitat preferences of the species, respectively (Table 2.2).
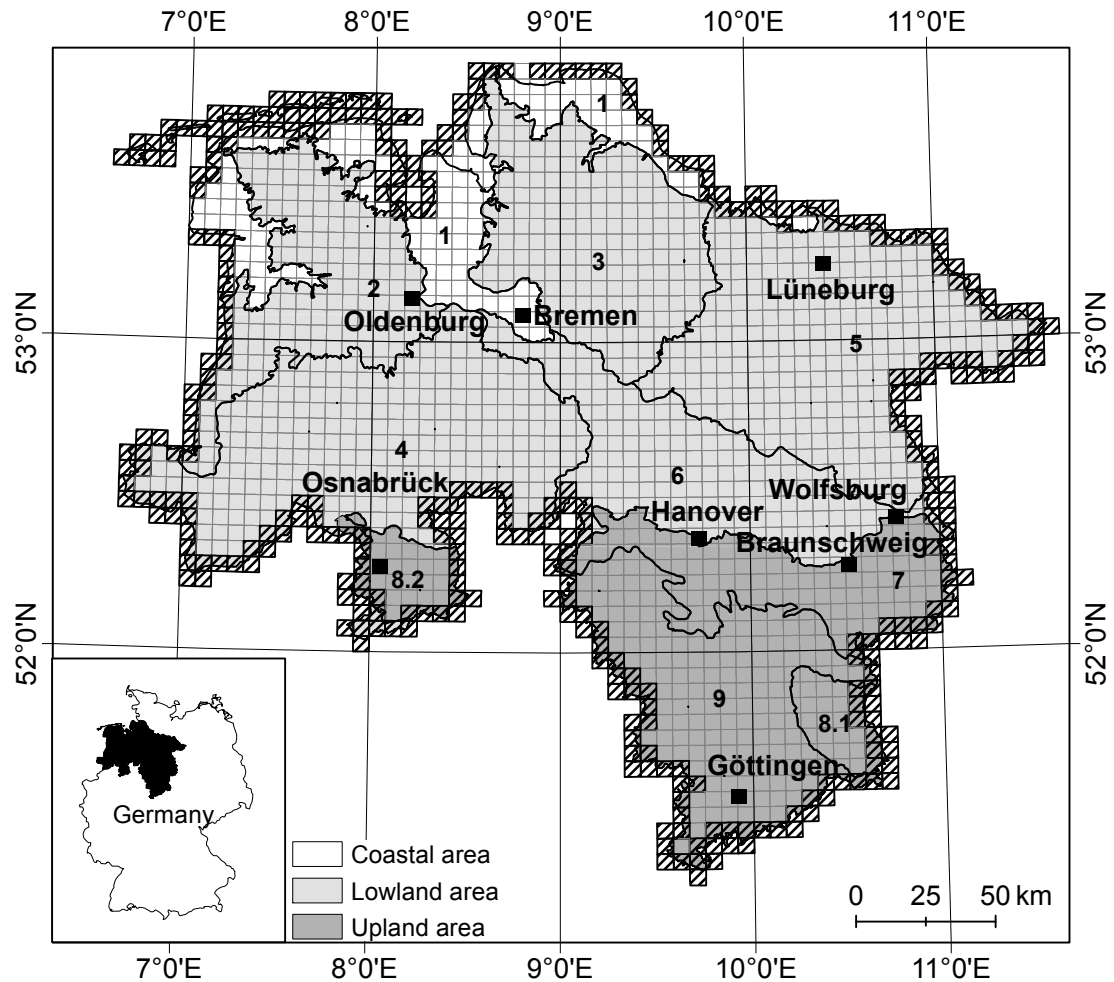
Figure 2.1: The study area encompassing the federal states of Lower Saxony and Bremen, with its three natural landscape regions (coastal, lowland and upland area). The region can be further subdivided into 10 bio-geographical regions (1 = Wadden Sea and marshes, 2 = East Frisian and Oldenburg moraine lowlands, 3 = Stade moraine lowlands, 4 = Ems, Hunte and Dümmer moraine lowlands, 5 = Luneburg Heath and Wendland, 6 = Weser and Aller plains, 7 = Old moraine loess landscapes, 8.1 = Uplands of Osnabrück, 8.2 = Uplands of the Weser and Leine valleys, 9 = Harz mountains.) Plant occurrence data from the plant survey of Lower Saxony were available for 1762 quadrants of the topographical map (scale 1:25,000, i. e. about 5.5 km × 5.5 km in size). The area of hatched quadrants has only been partly inventoried, because they are either only partially covered by land area or are situated in neighbouring federal states.

The residence status of species was determined by using the information provided by the data base itself and information from KLOTZ et al. (2002): Native species were distinguished from archaeophytes (species that became naturalised prior to the 16th century) and neophytes (species that became naturalised in the 16th century or later) (see Table 2.2). Information on the threat status of species was derived from the Red List of Lower Saxony and Bremen (GARVE, 2004): Threatened species (Red List status 1, 2, 3, G and R) were distinguished from non-threatened species (Red List status V and *) (see Table 2.2).

Information on habitat preference, i. e. the linkage to urban and forest habitats were taken from KLOTZ & KÜHN (2002), BFN (2013) and SCHMIDT et al. (2011), respectively (Table 2.2). Urban species were defined as those species with an urbanity indicator value of 4 (urbanophilic) and 5 (moderately urbanophilic). Forest species are those species categorised as to occur in closed forests only (category 1.1, SCHMIDT et al., 2011) as well as forest edges and clearings (category 1.2, SCHMIDT et al., 2011). Information on Ellenberg Indicator Values (EIV) for nutrients (N) were taken from BFN (2013). Those species having a N value of 1 to 3 were defined as species of nutrient-poor habitats, those with a N value of 7 to 9 as species of nutrient-rich habitats (Table 2.2).

Table A.2 in Appendix A lists all species, along with their residence and threat status as well as their habitat preferences and furthermore denotes the 'edge only' species.

Pivot tables in MS Access were used to compute species richness per grid cell. One query was conducted for each species group (Table 2.2).

2.2.2 ATKIS data

Information on the extent and spatial configuration of different land use types were derived from the ATKIS Digital Landscape Model (DLM, LGN, 2008) of Lower Saxony and Bremen. Overall, the DLM differentiates c. 90 different 'ATKIS object types'. These were combined into 8 different land cover types (LCTs), i. e. industrial, urban and traffic areas, croplands, forests, grasslands, water bodies and "other LCTs" as listed in Table 2.3. Linear elements represented as polylines in the DLM (roads, rivers) were buffered by using either standard values (roads, FGSV, 1996, see. Table A.5 in Appendix A) or information provided by the dataset itself (rivers) in order to convert them into polygons (Hawth's Analysis Tools, BEYER, 2004). Subsequently, they were combined with the other polygon features to a non-overlapping feature layer using ArcGIS-ArcInfo 9.2 (ESRI INC., 2007).

Information on average altitude, inclination and exposure per grid cell were derived from the ATKIS Digital Elevation Model (DEM, LGN, 2008). The DEM data were further used to calculate the *Mean Heat Load Index* (MCCUNE & KEON, 2002, eq. 3) per grid cell by using the ArcScript 'Heat load index' (PARKS, 2004) in ArcView GIS 3.3 (ESRI INC., 2002).

Table 2.1: Overview of the data used in the analysis.

Description	Source	Data type & resolution	Derivable information
Species data			
Occurrence data of all 1879 vascular plant taxa of Lower Saxony and Bremen per grid cell	Vascular plant survey of Lower Saxony[a], NLWKN[b]	MS Access data base (1993–2003), 2,395,045 entries	Species distribution, species richness
ATKIS data			
Digital Landscape Model (DLM)	LGN (2008), ATKIS	Shape file, scale 1:25,000	Landscape composition and configuration
Digital Elevation Model (DEM)	LGN (2008), ATKIS	Raster file, c. 5 m × 5 m	Altitude, exposure, inclination, heat load index
Areal information			
Geographical classification of natural landscapes (Naturräumliche Gliederung)	NLWKN, after PILGRIM & FRANKE, 1993	Shape file, scale 1:50,000	Natural landscape units, bio-geographical regions
Spatial information on protected areas (Natura 2000, nature protection areas, national parks, forest reserves)	NLWKN, 2010a; NLWKN, 2010b, NW-FVA[c]	Shape files, scale 1:25,000	Location and extent of protected areas
Environmental variables			
Map of soil types of Lower Saxony (BÜK 50[d])	LBEG[e] 2003	Shape file, scale 1:50,000	Soil types (27 categories), soil humidity (5 categories), CaCO$_3$ content of the soil (3 categories), diversity of soil types
Climate data	DWD[f] 1961-1990	Raster files, 1 km × 1 km	Mean/min./max. annual temperature, mean/min./max. annual precipitation, no. of days with frost/snow/ice, no. of summer days, sunshine duration, phenological data

[a] Niedersächsisches Pflanzenartenerfassungsprogramm, [b] Niedersächsisches Landesamt für Wasserwirtschaft, Küsten und Naturschutz, [c] Nordwestdeutsche Forstliche Versuchsanstalt, [d] Bodenübersichtskarte 1:50,000, [e] Landesamt für Bergbau, Energie und Geologie, [f] German Weather Service, *Deutscher Wetterdienst*

Table 2.2: Groups of vascular plant species based on residence and threat status of the species as well as their habitat preferences as used in the analyses of the present thesis.

Levels	Definition
Threat status	
Threatened species	Species listed on the Red List of Lower Saxony and Bremen, categories 1, 2, 3, G, R (critically endangered, endangered, vulnerable, threatened, near threatened, extremely rare)[a]
Non-threatened species	Species not threatened according to the Red List of Lower Saxony and Bremen, categories *, V (not threatened, prewarned)[a]
Residence status	
Native species	Species native to Lower Saxony and Bremen[b]
Archaeophytes	Plant species not native to Lower Saxony and Bremen; established prior to 16th century[b]
Neophytes	Plant species not native to Lower Saxony and Bremen; established in 16th or later[b]
Habitat preference	
Forest species	Herbaceous and shrubby plant species linked to forest habitats[c]
Urban species	Plant species of urban habitats[d]
EIV species N1-3	Plant species of nutrient-poor habitats[e]
EIV species N7-9	Plant species of nutrient-rich habitats[f]

[a] GARVE (2004); [b] Information taken from the data base of he vascular plant survey of Lower Saxony (NLWKN 1993-2003), BFN (2013) and KLORZ et al. (2002); [c] Categories K1.1, K1.2, K1.2, S1.1, S1.2, SCHMIDT et al. (2011); [d] Urbanity indicator, categories 4 (urbanophilic) and 5 (moderately urbanophilic), KLORZ et al. (2002); [e] Ellenberg Indicator 1, 2, 3 (BFN, 2013); [f] Ellenberg Indicator 7, 8, 9 (BFN, 2013).

2.2.3 Areal information

Information on bio-geographic regions and the location and extent of protected areas were obtained from shape files provided by the NLWKN and the NW-FVA (Table 2.1).

The geographical classification of natural landscapes of Lower Saxony (PILGRIM & FRANKE, 1993) defines the borders of the natural landscape units of the state. Roughly, the state area can be divided into three regions (coastal, lowland and upland area, cp. section 2.1) which themselves can be further divided into 10 bio-geographical regions and 82 natural landscape units (see Table A.1 in Appendix A). Because the grid cells in some cases were placed on the border between two or several regions/units the following rule was applied: Each grid cell was defined to belong to that region/unit that holds the highest proportion of area of the respective grid cell.

Information on protected areas included the location and extent of Natura 2000 sites, nature protected areas (*Naturschutzgebiete*, NSG), national parks as well as forest reserves (*Naturwaldreservate*) (Table 2.1).

2.2.4 Environmental variables

Soils   Information on soil types, soil humidity, $CaCO_3$ content as well as the diversity of soil types were obtained from the map of soil types of Lower Saxony (BÜK 50, scale 1:50,000, vector data; Table 2.1). For all calculations the vector data were transformed into raster data of a pixel size of $50\,m \times 50\,m$.

BÜK 50 lists 126 soil types that were grouped into 27 soil type categories (see Table A.4 in Appendix A). Based on these categories, the diversity of soil types was calculated using Fragstats 3.3 (McGARIGAL & CUSHMAN, 2002).

Soil humidity in BÜK 50 is arranged in 11 levels ranging from arid to wet/open water conditions. An additional 12th level includes all soils influenced by backwater. The 12 levels were grouped into 5 categories: "dry" (BÜK 50, soil moisture levels 0–3), "mesic" (soil moisture levels 4–6), "moist" (soil moisture levels 7–9), "wet" (soil moisture levels 10–11) and "influenced by backwater" (soil moisture level 12). Finally, the surface ratio of each soil moisture category was calculated per grid cell.

The $CaCO_3$ content of the soil in BÜK 50 is arranged in six levels ranging from soils with no $CaCO_3$ content (BÜK 50, level c0) to soils with a high $CaCO_3$ content (level c5). These levels were grouped into three categories: "free from $CaCO_3$" (BÜK 50 $CaCO_3$ content level c0, c1), "medium $CaCO_3$ content" (c2, c3) and "high $CaCO_3$ content" (c4, c5). Then, the surface ratio of each $CaCO_3$ category was calculated per grid cell.

Climate   Information on regional climate were obtained from climate data of the German Weather service (*Deutscher Wetterdienst*, DWD; Table 2.1). Several different climatic parameters were derivable, i. e. mean, minimum and maximum annual temperature and precipitation, the number of summer days and days with frost, snow

Table 2.3: Land cover types (LCTs) derived from ATKIS-DLM. For more detailed information on the included ATKIS object types see Table A.6 in Appendix A.

LCT	Description
Industrial areas	Areas used for (non agricultural) production (e. g. supply and disposal facilities, exhibition centers, mining areas etc.)
Urban areas	Areas used for housing and recreation (e. g. residential areas, parks, campgrounds etc.)
Traffic areas	Areas used for transport (roads, railway lines, airports, etc.)
Croplands	Areas used for agricultural production
Forests	Forested areas
Grasslands	Grassland areas
Water bodies	Standing and flowing waters (e. g. lakes, streams, channels, springs)
Others	Areas not belonging to any of the above mentioned LCTs (e. g. tree rows, bogs, islands)

and ice as well as the sunshine duration. Furthermore, phenological data, e. g. the time of snowdrop (*Galanthus nivalis*) flowering, apple blossom etc. were available. All parameters were calculated as averages per grid cell.

2.2.5 Landscape metrics

Landscape metrics were used to quantify the composition and configuration of the landscape in the study area. These metrics are based on the patch-corridor-matrix concept by FORMAN (1995) which states that landscapes are composed of structural elements: the patches, the corridors and the matrix. The arrangement of these elements in the landscape creates a particular landscape pattern or patch mosaic (LANG & BLASCHKE, 2007). The term 'patch' in this context describes more or less homogeneous, non-linear landscape elements that differ from the surrounding landscape (LEITÃO et al., 2006). The matrix is defined by its proportion of the landscape ($\geq 50\,\%$), its connectivity as well as its control on landscape dynamics.

Landscape metrics that evolved in the 1980s and 90s (LANG & BLASCHKE, 2007) facilitate the description of patterns of a landscape with regard to its composition and configuration on the patch, class as well as landscape level. These indices describe and quantify the spatial structure of patches, classes and the landscape, respectively (LEITÃO et al., 2006). They provide the opportunity to measure geometric properties of landscape elements and their relative position and distribution in the landscape (LEITÃO et al., 2006). On the patch level, the geometric properties of individual patches can be assessed, whereas on the class level patches of the same type (class; in this study named land cover types, LCTs) are aggregated. On the landscape level the configuration of different types of classes is considered (LANG & BLASCHKE, 2007).

All landscape metrics were calculated using Fragstats 3.3 (MCGARIGAL & CUSHMAN, 2002). Because the software is limited to the use of raster data, the DLM vector

data were converted to raster format with a resolution of $5\,\mathrm{m} \times 5\,\mathrm{m}$ using ArcGIS-ArcInfo 9.2 (ESRI INC., 2007). A higher resolution could not be chosen because of hardware limitations. Because landscape metrics had to be assessed separately for each of the 1762 grid cells (see section 2.1) the raster data set was cut into 1762 separate rasters using Hawth's Tools (BEYER, 2004) in ArcGIS. In the following these 1762 tiles were considered as separate 'landscapes'. Landscape metrics were calculated on class and landscape level, respectively. The indices that were calculated describe the landscape composition (e.g. patch sizes, patch numbers) and configuration, i.e. the interactions and correlations between the landscape elements (e.g. expressed in terms of patch shape, complexity, fragmentation). The following paragraphs name and describe the landscape metrics used to quantify the composition and configuration of the landscape. Naming of the indices follows the nomenclature used in Fragstats 3.3 (MCGARIGAL et al., 2002).

Landscape metrics quantifying landscape composition

Class level   The landscape composition within the 1762 grid cells on the class level was assessed using the indices *Total Class Area* (CA) and *Number of Patches* (NP; Table 2.4). CA for each landscape reports the percentage of area of the particular classes (MCGARIGAL & CUSHMAN, 2002). NP counts the number of patches of each class under observation. The NP index together with CA represents a simple measure to quantify fragmentation of the classes in the landscape: If a class is characterised by high values of CA as well as NP the area belonging to the class is highly fragmented.

Landscape level   The landscape composition within the 1762 grid cells on the landscape level was assessed using the indices *Patch Richness* (PR), *Shannon's Diversity Index* (SHDI) and *Shannon's Evenness Index* (SHEI; Table 2.4). PR addresses the number of classes in the landscape and thus is an important basic measure for landscape diversity. If besides the pure number of classes in the landscape also their proportions shall be considered, the *Shannon's Diversity Index* is the method of choice: If the landscape is composed of one patch only, the value of SHDI equals zero. The more patch types are present and the more homogeneously distributed they are, the larger the value of SHDI becomes. SHEI, in turn, is a measure of the evenness of the distribution of patch types/classes (MCGARIGAL & CUSHMAN, 2002). The index can take values between zero and one: If a landscape consists of only one patch type, values of SHEI will be zero. If several LCTs are present, the value of SHEI increases the more evenly distributed they are.

Table 2.4 lists all landscape metrics used to assess landscape composition, their range of values as well as their units.

Table 2.4: Landscape metrics used to assess landscape composition with respect to their range of values, units and basis of computation (after MᴄGᴀʀɪɢᴀʟ & Cᴜsʜᴍᴀɴ, 2002). LCT = land cover type; ↑ = increase in value; n/a = not available.

Landscape parameter	Landscape metric	Range of values	Unit	Basis of computation
Class level				
Total class area	*Total Class Area*, CA	CA > 0	ha	All LCTs
Patch numbers/ Landscape diversity	*Number of Patches*, NP	NP ≥ 1 NP = 1, if the landscape consists of one single patch	n/a	grassland, cropland, forest
Landscape level				
Number of patch types (classes)	*Patch Richness*, PR	PR > 0	n/a	All LCTs
Landscape diversity	*Shannon's Diversity Index*, SHDI	SHDI ≥ 0 SHDI = 0, if the landscape consists of one patch/class only SHDI ↑, if the number of patch types/classes increases and/or the proportions of different classes is becoming more uniformly distributed	n/a	All LCTs
Uniform distribution of LCTs	*Shannon's Evenness Index*, SHEI	0 ≤ SHEI ≤ 1 SHEI = 0, if the landscape consists of one patch/class only SHEI↑, if the proportion of the patches of all classes are becoming more equal SHEI = 1, if the patches of all classes are present at the same proportions	n/a	All LCTs

Landscape metrics quantifying landscape configuration

Class level   Landscape configuration on the class level was assessed only for selected LCTs, i.e. forest, grassland and cropland areas. For these LCTs several configuration indices, namely the *Area-weighted Mean Patch Area Distribution* (AREA_AM), the *Area-weighted Mean Shape Index* (SHAPE_AM), the *Total Edge Index* (TE), the *Mean Proximity Index* (PROX_MN), the *Total Core Area Index* (TCA) as well as the *Area-weighted Mean Core Area Distribution* (CORE_AM) were calculated (Table 2.5). In the present study metrics based on area-weighted mean (AM) patch characteristics were favoured over those describing the average (MN) patch characteristics because the former provide a more landscape-centric instead of patch-centric perspective and thus better reflect the average conditions of a randomly chosen pixel in the landscape (McGarigal, 2014).

AREA_AM calculates the area-weighted mean size of all patches of one class. Thus, the index allows to discriminate between LCTs present at a large vs. small scale (McGarigal & Cushman, 2002).

SHAPE_AM quantifies the complexity of patches. In raster data the simplest shape is a square; the more complex a patch becomes, i.e. the more it deviates from the shape of a square, the stronger the value of the *Shape Index* increases.

PROX_MN quantifies the isolation or rather connectivity of the patches of one class. For the calculation of the index a 'search radius' has to be defined; patches within this distance are regarded as connected. The search radius in the study was set to 100 m. If a patch has no neighbours of the same patch type within this distance the *Proximity Index* is 0; the index increases as the neighbourhood of the patch is increasingly occupied by patches of the same type; the closer these patches become and the less scattered they occur, the larger the value of the index becomes (McGarigal, 2014).

TE sums up the length of the edges of all patches belonging to one LCT (Lang & Blaschke, 2007). A high value of TE can – depending on the quality of the LCT under consideration – either be an indicator of a high structural diversity or identify a high degree of landscape fragmentation (Lang & Blaschke, 2007).

TCA sums up the core areas of all patches belonging to one class. The core area is defined as the area within a specified core area distance, i.e. that area that is not influence by edge effects (McGarigal & Cushman, 2002; Lang & Blaschke, 2007). The core area of patches, different from patch size, is influenced by the shape of the patches. Thus, a patch can be potentially large enough to support particular species but because of its shape can be influenced by edge effects to such a large extent that the habitat becomes unsuitable for species (McGarigal & Cushman, 2002). For the calculation of TCA the 'edge depth' has to be defined. In the present study TCA was calculated for forests only; edge depth was defined as being 100 m (cp. Murcia, 1995; Harper et al., 2005; Hennenberg et al., 2005; Wei & Hoganson, 2005; Hennenberg et al., 2008a,b). CORE_AM resembles TCA but measures the average core area of all patches belonging to one class.

Table 2.5: Landscape metrics used to assess landscape configuration with respect to their range of values, units and basis of computation (after MᴄGᴀʀɪɢᴀʟ & Cᴜsʜᴍᴀɴ, 2002). FGs = fragmentation geometries. LCT = land cover type; ↑ = increase in value; n/a = not available.

Landscape parameter	Landscape metric	Range of values	Unit	Basis of computation
Class level				
Patch complexity	*Shape Index Distribution*, SHAPE_AM	SHAPE_AM $\geq$ 1 SHAPE_AM = 1, if the patches are maximally compact (square) SHAPE_AM ↑, if the shape of patches becomes more irregular	n/a	grassland, forest, cropland
Mean patch size	*Patch Area Distribution*, AREA_AM	AREA_AM > 0	ha	grassland, forest, cropland
Total Core Area	Total Core Area, TCA	TCA $\geq$ 0 TCA = 0, if each location of all patches belonging tot he class lays within the specified edge depth	ha	forest
Mean core area	*Core Area Distribution*, CORE_AM	CORE_AM > 0	ha	forest
Total edge length	*Total Edge*, TE	TE $\geq$ 0	m	grassland, forest, cropland
Patch isolation	*Proximity Index*, PROX_AM	PROX $\geq$ 0 PROX = 0, if a patch has no neighbours of the same patch type within the specified search radius	n/a	grassland, forest, cropland
Landscape level				
Complexity of patches	*Shape Index Distribution*, SHAPE_AM	SHAPE_AM $\geq$ 1 SHAPE_AM = 1, if the patches are maximally compact (quadrat) SHAPE_AM ↑, if the shape of patches becomes more irregular	n/a	all LCTs
Mean patch size	*Patch Area Distribution*, AREA_AM	AREA_MN > 0	ha	all LCTs
Total edge length	*Total Edge*, TE	TE $\geq$ 0	m	all LCTs
Landscape fragmentation	*Effective Mesh Size*, MESH	2.5×10^{-3} ha [a] $\leq$ MESH $\leq$ size of the grid cell	ha	FGs & remaining landscape

[a] Area of one single pixel of the raster (pixel 5 m $\times$ 5 m = 25 m^2 = 2.5×10^{-3} ha).

Landscape level   Landscape configuration on the landscape level was assessed using the *Area-weighted Mean Shape Index* (SHAPE_AM), the *Area-weighted Mean Patch Area Distribution* (AREA_AM) as well as the *Effective Mesh Size Index* (MESH; later following JAEGER et al. (2006) also named m_{eff}) (Table 2.5).

As explained for the class level, SHAPE_AM quantifies the complexity of patches. On the landscape level this complexity is not averaged over the patches of one class but over all patches being present in the landscape. The same applies to AREA_AM that on the landscape level calculates the area-weighted mean size of all patches in the landscape and, in doing so, provides a measure of the overall fragmentation of LCTs.

MESH measures the fragmentation of a landscape and is frequently used in many studies quantifying landscape fragmentation (e. g. JAEGER et al., 2001; WALZ, 2005; ESSWEIN & SCHWARZ-VON RAUMER, 2006; MOSER et al., 2007; JAEGER et al., 2008; GIRVETZ et al., 2008; LI et al., 2010; EEA, 2011). The index denotes the size of the patches when the landscape is divided into S areas (each of the same size) with the same degree of landscape division as obtained for the observed cumulative area distribution (MCGARIGAL, 2014).

Table 2.5 lists all landscape metrics used to assess landscape configuration, their range of values as well as their units.

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3 | Patterns of vascular plant species richness at the landscape scale

Summary

The present study analyses the richness patterns of different groups of vascular plant species for the federal states of Lower Saxony and Bremen. For this purpose, all 1819 established species occurring in the two states as well as different sub-groups of species were considered. Sub-groups were defined by using information on residence (1509 native species, 160 archaeophytes, 145 established neophytes) and threat status (643 threatened vs. 1176 non-threatened species) of the species, respectively. Because of their contrasting demands regarding their habitats it was assumed that species belonging to the various groups would occupy different ecological niches in the landscape; therefore – on a large scale – their centres of diversity were assumed to be spatially segregated from one another.

Data on species occurrences were obtained from the floristic mapping of the two states (plant survey of Lower Saxony, NLWKN 1982–2003) and were available on a grid basis (1,762 grid cells of c. 30 km^2). The study aimed at answering the following questions: 1.) Which patterns of vascular plant species richness are observable in Lower Saxony and Bremen and the three natural landscape regions, respectively? 2.) Do the different plant species (sub-)groups show richness patterns that are distinguishable from each other and how much do their richness centers overlap? 3.) In which areas do threatened species show highest species numbers and how do these hotspots differ from that of non-threatened species?

Data were analysed by calculating the number of species per grid cell for each of the species groups mentioned above. Species richness per grid cell was then displayed using a geographical information system (GIS). To test if the centres of diversity of the species groups were overlapping, the pairwise non-parametric Spearman's rank correlation coefficient ρ was calculated.

Species richness of all groups was found to be heterogeneously distributed across the states with an observable increase of species numbers from north to south and – to some extent – also from west to east. The spatial distribution of the richness centres of the different plant species groups were largely correlated with each other but also showed some interesting divergences. Considering the total number of established species occurring in the study area an aggregated distribution pattern was observable: As site conditions in the Quaternary coastal and lowland areas are relatively homogenous these regions – with exception of the major stream valleys of the Weser, Aller and Elbe – were found to be relatively species poor. In contrast, the upland area with its heterogeneous site conditions, generally hosted a high overall species richness. The large group of native species showed a distribution very similar to that of the total number of species. Archaeophytes were concentrated in the coastal area of Bremen, the Weser and Aller plains, and the loess landscapes of the lowland area, as well as smaller cities. High richness of established neophytes was mainly found in congested urban areas. Richness centres of threatened species were spatially highly differentiated: at the coast only the islands in the North Sea repre-

sented small hotspots, while in the lowlands relatively large hotspots were situated in the Wendland, the Luneburg Heath and the Elbe-Weser triangle north of Bremen. In the uplands, the area around Göttingen, the uplands of the Weser and Leine valleys and the margins of the Harz Mountain represented well defined hotspots of Red List species.

Zusammenfassung

Voraussetzung für die Entwicklung von Schutzstrategien für den Pflanzenartenschutz ist die Kenntnis über die Verteilung der Zentren der Artenvielfalt im Raum. Je nach Einbürgerungsstatus und Gefährdungssituation kommt verschiedenen Artengruppen dabei eine unterschiedliche Bedeutung zu. In der vorliegenden Studie werden für die Gesamtfläche der Bundesländer Niedersachsen und Bremen die im Niedersächsischen Pflanzenarten-Erfassungsprogramm (1982-2003) auf Messtischblatt-Quadranten-Ebene erhobenen Verbreitungsdaten von Gefäßpflanzensippen unter Berücksichtigung der Gesamtflorenliste (1.819 Sippen), ihres Einbürgerungsstatus (1.509 Indigene, 160 Archäophyten, 145 etablierte Neophyten) und ihrer Gefährdungssituation (ungefährdete und gefährdete Arten; davon 643 Sippen mit Rote-Liste-Status 1, 2, 3, G oder R) ausgewertet. Auf Basis der Gesamtliste ergibt sich eine inhomogene Verteilung der Sippendichte im Gesamtuntersuchungsraum, wobei die standörtlich relativ homogene Küste sowie das Tiefland – mit Ausnahme der großen Stromtäler (Weser, Aller, Elbe) – relativ artenarm sind und das standörtlich sehr heterogene Hügel- und Bergland grundsätzlich die höchsten Sippendichten aufweist. Unter Berücksichtigung des Einbürgerungsstatus zeigen die Archäophyten jeweils die größten Überschneidungsbereiche zu den Indigenen und etablierten Neophyten. Die Verbreitungsmuster der großen Gruppe der Indigenen ähneln denen der Gesamtliste, während sich die Archäophyten auf den Bremer Küstenraum, das Weser-Aller-Flachland, die Börden und das südliche Weser-Leine-Bergland konzentrieren. Die Zentren der Sippenvielfalt der etablierten Neophyten liegen vor allem in städtischen Ballungsräumen und erscheinen oftmals sehr punktuell. Die Rote-Liste-Arten sind in der Mehrzahl indigen (91 %), 8 % von ihnen sind Archäo-, nur 1 % Neophyten. Ihre Diversitätszentren sind außerordentlich differenziert: An der Küste gehören nur die isoliert liegenden Nordsee-Inseln dazu, während im Tiefland das Wendland, die Lüneburger Heide und das Elbe-Weser-Dreieck großflächige Diversitätszentren aufweisen. Im Hügel- und Bergland finden sich vor allem im Raum Göttingen, dem Weserbergland und am Harzrand gut abgegrenzte Zentren der Rote-Liste-Artendiversität. Viele dieser bedrohten Sippen sind vermutlich Spezialisten, die an natürliche oder naturnahe Habitate angepasst und somit nur in den wenigen Landschaftsbereichen anzutreffen sind, die die entsprechenden Habitatbedingungen bieten.

Abstract

Distribution patterns of vascular plant taxa in the federal states of Lower Saxony and Bremen, Germany, with respect to their naturalisation and threat status

Detailed knowledge about the spatial distribution of richness centres is a pre-requisite for developing national strategies for the conservation of vascular plant species. Thereby, different groups of plant taxa may have different relevance depending on their naturalisation and threat status. In this study, we investigate the spatial distribution of plant species richness centres in the area of the federal states of Lower Saxony and Bremen based on data from the plant survey of Lower Saxony (Niedersächsisches Pflanzenarten-Erfassungsprogramm). Analyses include: 1.) The complete floristic list (1819 taxa) 2.) three groups of taxa defined by their naturalisation status (1509 indigenous taxa, 160 archaeophytes, 145 established neophytes), 3.) two groups of taxa defined by their threat situation (unthreatened versus vulnerable taxa, including 643 taxa which have been red listed with status 1, 2, 3, G or R). Based on the complete floristic list, a clumped distribution pattern was observed. Site conditions in the Quaternary coastal and lowland areas are relatively homogenous and, with exception of the major stream valleys of the Weser, Aller and Elbe, relatively species poor, while the upland area, with its heterogeneous site conditions, shows the overall highest taxonomic richness. The large group of indigenous taxa shows a distribution very similar to that of the complete floristic list. Archaeophytes are concentrated in the coastal area of Bremen, the Weser and Aller plains, and the Old moraine loess landscapes, as well as smaller cities. High phytodiversity of established neophytes is mainly found in congested urban areas and several other small-scale centres of diversity. Red List species are mainly indigenous (91 %), 8 % are archaeophytes and only 1 % are neophytes. Their centres of phytodiversity are spatially highly differentiated: at the coast only the islands in the North Sea are small hotspots, while in the lowlands relatively large hotspots are situated in the Wendland, the Luneburg Heath and the Elbe-Weser triangle north of Bremen. In the uplands, the area around Göttingen, the uplands of the Weser and Leine valleys and the Harz mountain margins are well defined hotspots of Red List taxa. Many of these endangered species are presumably specialists adapted and limited to natural and semi-natural habitats.

Keywords: Archaeophytes, plant diversity, phytodiversity, hotspots, neophytes, Red Lists, richness centres.

Einleitung

Das Verbreitungsmuster von Pflanzenarten im Raum wird durch deren Ausbreitungs-modi, ihr Konkurrenz- und Anpassungsverhalten, Wechselwirkungen mit der abiotischen Umwelt, durch evolutionäre und erdgeschichtliche, aber auch aktuelle, meist durch den Menschen verursachte Prozesse bestimmt (Haeupler, 1974). Landschafts-räume mit hoher struktureller und Habitat-Heterogenität, die sich aus einem Mosaik von künstlich geschaffenen Habitaten und Resten naturnaher Vegetation zusammen-setzen, weisen oftmals eine sehr hohe Pflanzenartenvielfalt auf (Deutschewitz et al., 2003). Insbesondere in städtischen Ballungszentren liegen die Artenzahlen meist höher als in dem umgebenden ländlichen Raum (Haeupler, 1974; Wania et al., 2006; Lippe & Kowarik, 2008).

Für die Erfassung von Zentren der Artenvielfalt auf Landschaftsebene kommt verschiedenen Artengruppen je nach Einbürgerungsstatus und Gefährdungssituation eine unterschiedliche Bedeutung zu. Insbesondere Diversitätszentren von indigenen und/oder gefährdeten Pflanzensippen ("Hotspots") sind für den Artenschutz von Bedeutung (Globale Strategie zur Erhaltung der Pflanzen, Sekretariat der CBD, 2007), während solche von gebietsfremden Arten, insbesondere von invasiven Neophyten, auf eine Gefährdung von heimischen Arten, ökologischen Kreisläufen und des bestehenden Landschaftsbildes hinweisen können (Hubo et al., 2007). Gefährdete Arten stammen fast ausschließlich aus den Gruppen der Indigenen und Archäophyten (Definition bei Wagenitz, 2003). In der mitteleuropäischen Landschaft sind die Hauptursachen für ihre Gefährdung die Intensivierung der Landnutzung und ihre Folgen (Eutrophierung, Entwässerung, Aufgabe extensiver Bewirtschaftungsweisen) sowie der gestiegene Flächenbedarf der Bevölkerung (Garve, 2004). Etablierte Neophyten sind dagegen unter Einfluss des Menschen seit ca. 1500 n. Chr. in ein Gebiet eingewandert und – nach zeitlichen und populationsbiologischen Etablierungskriterien (Schnittler & Ludwig, 1996) – fester Bestandteil der Flora geworden. Aufgrund der kontrastierenden Habitatansprüche der verschiedenen Artengruppen ist anzunehmen, dass sie wie die heimischen Arten unterschiedliche Nischen in der Landschaft besetzen, so dass bei großräumiger Betrachtung ihre Zentren der Artenvielfalt räumlich getrennt sein sollten.

In dieser Studie werden für die Bundesländer Niedersachsen und Bremen die Verbreitungsmuster verschiedener Gefäßpflanzengruppen untersucht: Es werden die Gesamtheit der Arten sowie weitere, nach ihrem Einbürgerungsstatus (Indigene, Archäophyten und etablierte Neophyten) bzw. ihrer Gefährdungssituation (gefährdete und ungefährdete Arten) definierte Artengruppen betrachtet. Die Analyse umfasst die von 1982 bis 2003 erhobenen Funddaten von Sippen der Roten Liste und Florenliste der Farn- und Blütenpflanzen Niedersachsens und Bremens (Garve, 2004) und basiert auf der floristischen Kartierung der beiden Bundesländer nach der Methodik des Niedersächsischen Pflanzenarten-Erfassungsprogramms (Schacherer, 2001). Dieses wird von der niedersächsischen Fachbehörde für Naturschutz (NLWKN) durchge-

führt und liefert auf Ebene von Messtischblatt-Quadranten flächendeckende Informationen über die Funde aller Gefäßpflanzensippen.

Die vorliegende Studie ist im Rahmen des von der Deutschen Bundesstiftung Umwelt geförderten Projektes "Identifizierung von Indikatorartengruppen für ein Biodiversitäts-Monitoring zur Bewertung von Grünland- und Waldlebensräumen" entstanden. Sie stellt eine grundlegende Auswertung des Pflanzenartendatensatzes dar, auf der weitere, speziellere Untersuchungen aufbauen. Auf Grundlage der Pflanzenartendaten sollen folgende Fragestellungen untersucht werden:

1. Welche Diversitätsmuster sind im gesamten Gebiet bzw. in den drei Naturräumen Küste, Tiefland sowie Hügel- und Bergland auf Grundlage der Gesamtliste der Farn- und Blütenpflanzen erkennbar?

2. Ergeben sich für die verschiedenen Gruppen je nach Einbürgerungsstatus unterschiedliche Muster der Phytodiversität und wie stark überschneiden sich die Zentren ihrer Diversität?

3. Wo befinden sich die Zentren der Sippenvielfalt der Rote-Liste-Arten (Hotspots im Sinne von SCHMITT & HAEUPLER, 2009) und wie unterscheiden sie sich von den Diversitätszentren der ungefährdeten Sippen?

Material und Methoden

Untersuchungsgebiet

Das Untersuchungsgebiet umfasst Niedersachsen und Bremen, wobei Niedersachsen mit einer Größe von 47.624 km^2 das zweitgrößte, Bremen mit einer Größe von 404 km^2 das kleinste Bundesland Deutschlands ist (NIEDERSACHSEN, 2008). Das Gebiet wird in drei Großregionen unterteilt: die Küste, das Tiefland und das Hügel- und Bergland (Abb. 3.1).

Die einzelnen Regionen unterscheiden sich stark in ihrer Naturraumausstattung: Die **Küste** wird geologisch vor allem durch marine Ablagerungen aus dem Holozän bestimmt (HEUNISCH et al., 2007). Ihr Klima ist ozeanisch geprägt, der mittlere Jahresniederschlag beträgt 700–900 mm, die mittlere Jahrestemperatur liegt bei durchschnittlich 9 °C. Vorherrschende Bodentypen sind Marschen, Syroseme, podsolige Ranker und Gleye (NLfB, 1997). Die Küstenregion wird dominiert von Grünländern, die etwa 57 % der Region einnehmen. Während Ackerflächen hier einen Anteil von 22 % besitzen, sind lediglich 1 % Wälder vorhanden.

Das niedersächsische **Tiefland** wird von pleistozänen Ablagerungen dominiert (HEUNISCH et al., 2007). Von West nach Ost bringt ein Kontinentalitätsgradient Jahresniederschlagsmittel von 800 mm im Westen bis etwa 550 mm im Osten sowie Jahresmitteltemperaturen von 8.0–8.5 °C mit sich. Entlang dieses Klimagradienten sind unterschiedliche Böden ausgebildet: Während im ozeanischen Westen von Sickerwasserbildung, Vernässung und Stoffverlagerung geprägte Böden (Moorböden, Podsole) vorherrschen, finden sich im subkontinentalen Osten vor allem Braunerden

und Ranker (NLFB, 1997). Das Tiefland wird dominiert von Ackerflächen (41 %), Wälder und Grünländer nehmen jeweils etwa knapp 1/4 der Region ein (21 bzw. 22 %).

Das standörtlich sehr heterogene **Hügel- und Bergland** wird vom Meso- und Paläozoikum (HEUNISCH et al., 2007) dominiert und weist aufgrund der stark variierenden Topographie besonders unterschiedliche klimatische Verhältnisse auf. Das Bergvorland ist geprägt von Löss- und Sandlössablagerungen sowie Ton-, Sand- und Kalkgesteinen (NLFB, 1997). Neben Parabraunerden und Braunerden kommen auch Übergänge zu Pseudogleyen vor (Bördenvorland). In der Lössbörde treten großflächig Schwarz- und Parabraunerden auf. Die Jahresdurchschnittstemperatur der tieferen Lagen (Becken) liegt 1–2 °C über der der Höhenzüge; die mittleren Jahresniederschläge sind auf den Höhenzügen deutlich höher (800–900 mm) als in den Beckenlagen (650–750 mm). Der Harz im Südosten Niedersachsens nimmt eine Sonderstellung ein: Der Hochharz (in Niedersachsen bis 971 m ü. NN) weist mittlere Jahresniederschläge von bis zu 1.500 mm und eine Jahresmitteltemperatur von nur 7 °C sowie lange, schneereiche Winter auf und ist somit (hoch-)montan geprägt (NLFB, 1997). Sowohl im Hochharz als auch im Oberharz, der submontane bis montane Klimaverhältnisse aufweist, kommen basenarme Silikatgesteine sowie Tonschiefer und Grauwacken vor. Vorherrschende Bodentypen sind hier Braunerden und Podsole (NLFB, 1997). Die Region weist von den drei Großregionen den höchsten prozentualen Waldanteil auf (32 %). Grünländer nehmen hier nur knapp 9 % der Fläche ein, während 45 % von Ackerflächen geprägt sind.

Untersuchungsraster

Das Untersuchungsgebiet ist auf Grundlage der Topographischen Karte 1:25.000 in ein Raster von Messtischblatt-Quadranten (MTB-Quadranten, im Folgenden nur Quadranten genannt) mit einer Größe von etwa 5.5 km × 5.5 km gegliedert. Für die vorliegenden Analysen wurden 1.762 Quadranten berücksichtigt, von diesen liegen 1.386 vollständig im Untersuchungsgebiet und sind gänzlich von Land bedeckt (Vollquadranten), weitere 376 liegen teilweise oder vollständig im Meer oder teilweise an der Grenze zu anderen Bundesländern (Randquadranten). Es wurden 255 Quadranten der Küste, 1.101 Quadranten dem Tiefland und 406 Quadranten dem Hügel- und Bergland zugeordnet. Dabei wurden Quadranten im Grenzgebiet zweier Regionen jeweils der Region zugeordnet, die den größeren Flächenanteil hatte.

Pflanzenartendaten

Die Gesamtliste der Farn- und Blütenpflanzen in Niedersachsen und Bremen richtet sich nach GARVE (2004). Verbreitungsangaben auf Quadranten-Ebene wurden der Datenbank des Niedersächsischen Pflanzenarten-Erfassungsprogramms entnommen (NLWKN 1982-2003, Stand 15.12.2008). Eine Sammlung von Artenverbreitungskar-
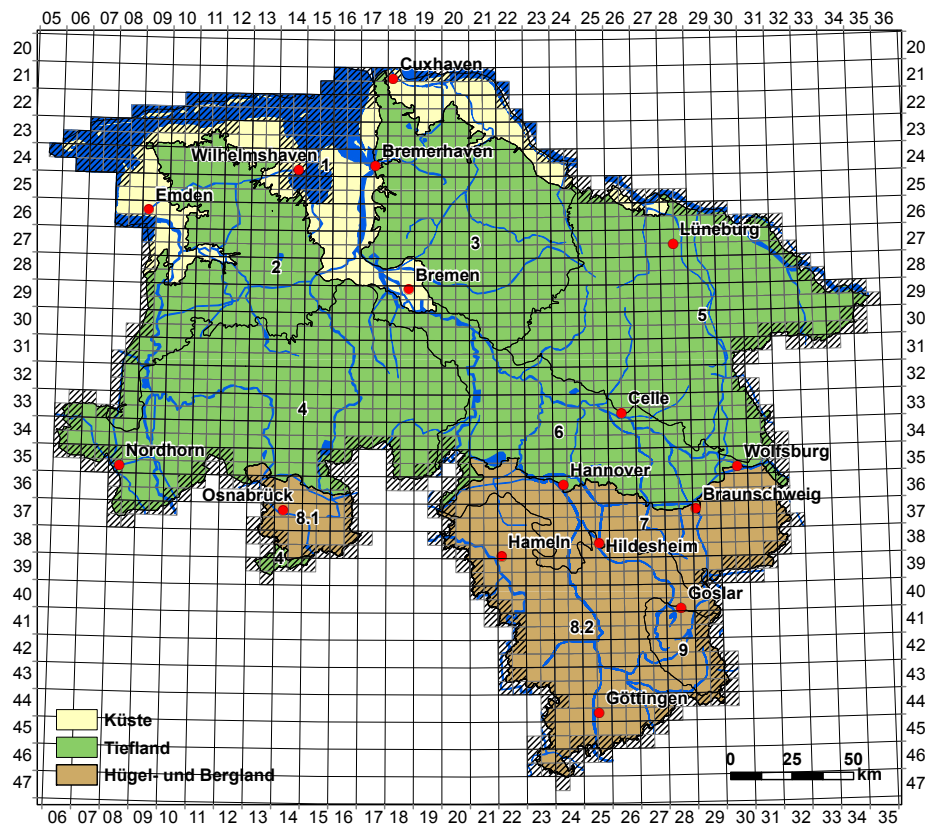
Abbildung 3.1: Das Untersuchungsgebiet umfasst die Gesamtfläche der Bundesländer Nie-
dersachsen und Bremen und wird in drei naturräumliche Großregionen unterteilt: die Küste,
das Tiefland und das Hügel-/Bergland. Eine weitere Untergliederung erfolgt in zehn natur-
räumliche Hauptregionen (1 = Watten und Marschen, 2 = Ostfriesisch-Oldenburgische Geest,
3 = Stader Geest, 4 = Ems-Hunte-Geest und Dümmer-Geestniederung, 5 = Lüneburger Hei-
de und Wendland, 6 = Weser-Aller-Flachland, 7 = Börden, 8.1 = Osnabrücker Hügelland, 8.2
= Weser- und Leinebergland, 9 = Harz). Die 1.762 Messtischblatt-Quadranten sind grundle-
gende Kartiereinheit des Niedersächsischen Pflanzenarten-Erfassungsprogramms (Zahlenan-
gaben am Rand entsprechen der TK25-Basisblatt-Nummer). Schraffierte Quadranten wurden
auf einer kleineren Fläche floristisch erfasst, da sie teilweise oder gänzlich im Meer bzw. in
angrenzenden Bundesländern liegen.

Figure 3.1: The study area the federal states of Lower Saxony and Bremen, with its three
natural landscape regions (Küste, coast; Tiefland, lowlands; Hügel- und Bergland, uplands).
The region can be further divided into ten natural regions (1 = Wadden Sea and marshes, 2 =
East Frisian and Oldenburg moraine lowlands, 3 = Stade moraine lowlands, 4 = Ems, Hunte
and Dümmer moraine lowlands, 5 = Luneburg Heath and Wendland, 6 = Weser and Aller
plains, 7 = Old moraine loess landscapes, 8.1 = Uplands of Osnabrück, 8.2 = Uplands of the
Weser and Leine valleys, 9 = Harz mountains.) Plant occurrence data from the plant survey
of Lower Saxony were available for 1762 quadrants of the topographical map (scale 1:25,000,
i. e. about 5.5 × 5.5 km in size, frame numbers equal ordnance map numbers). The area of
hatched quadrants has only been partly inventoried, because they are either only partially
covered by land area or are situated in neighbouring federal states.

ten mit Angaben zur Funddichte wurde von Garve (2007) veröffentlicht. Eine frühere Auswertung der Anzahl der Rote-Liste-Arten je Quadrant erfolgte durch Garve (1994).

Der Pflanzenartendatensatz wurde nach folgenden Kriterien gruppiert:

1. Gesamtliste: alle Sippen der Farn- und Blütenpflanzen mit floristischem Normalstatus, d. h. Indigene, Archäophyten und etablierte Neophyten, unter Ausschluss von unbeständigen und synanthropen Vorkommen sowie Sippen, die in Niedersachsen und Bremen ausgestorben oder verschollen sind (1.819 Sippen mit Einträgen in der Datenbank).

2. Einbürgerungsstatus: Indigene (1.509 Sippen), Archäophyten (160 Sippen) bzw. etablierte Neophyten (145 Sippen) nach der von Garve (2004) verwendeten Definition. Der Einbürgerungsstatus der Sippen wurde der Niedersächsischen Florenliste entnommen, wobei der Etablierungsstatus der Neophyten von Garve (2004) übernommen wurde. Die Unterscheidung von Indigenen und Archäophyten erfolgte durch Abgleich mit der BIOLFLOR-Datenbank (Klotz et al., 2002). Für fünf der den Indigenen/Archäophyten zugeordneten Sippen konnte kein Einbürgerungsstatus festgestellt werden (*Panicum riparium*, *Persicaria mitis*, *Potamogeton* x *undulatus*, *Rumex* x *pratensis*, *Silene* x *hampeana*).

3. Gefährdungssituation: ungefährdete Sippen inklusive der Arten der Vorwarnliste (1.176 Sippen) bzw. gefährdete Sippen (643 Sippen mit Rote-Liste-Status 1, 2, 3, G oder R), wobei sich der Gefährdungsstatus auf das Gesamtgebiet Niedersachsen und Bremen bezieht (Garve, 2004).

Datenanalyse

Für jede der untersuchten Gruppen wurde die Sippenzahl je Raumeinheit (Quadrant) erfasst und mithilfe eines Geographischen Informationssystems dargestellt (ArcGIS-ArcInfo 9.2; ESRI Inc., 2007). Bei der Visualisierung der Verbreitungsmuster wurden natürliche Grenzwerte verwendet, wobei jeweils fünf Sippenzahl-Klassen gebildet wurden. In den Abbildungen wurden negative Abweichungen vom Mittelwert mit abgestuften Grautönen und die Klassen um bzw. oberhalb des Mittelwerts in Farbabstufungen dargestellt.

Zusätzlich wurden die mittleren und maximalen Sippenzahlen je Gruppe und Quadrant für Niedersachsen/Bremen bzw. die einzelnen Regionen Küste, Tiefland und Hügel- und Bergland ermittelt. Räumliche Autokorrelationen von Quadranten mit hohen Sippenzahlen (Klumpungen der Diversitätszentren) wurden als Moran's Index berechnet, wobei Werte gegen -1 auf eine starke Streuung und Werte gegen $+1$ auf eine starke Klumpung von Quadranten mit hoher Artenvielfalt im Raum hinweisen.

Für die nach ihrem floristischen Status (Indigene, Archäophyten, etablierte Neophyten) bzw. ihrem Gefährdungsstatus (ungefährdete Arten, Rote-Liste-Arten) gruppierten Sippen wurde jeweils für Paare dieser drei bzw. zwei Gruppen mittels des Rangkorrelationskoeffzienten ρ nach Spearman für nicht normalverteilte Daten

(*Shapiro-Wilk Normality-Tests*) mit der R-Software (R DEVELOPMENT CORE TEAM, 2012) überprüft, ob ihre Diversitätswerte (Zentren der Sippen-Diversität) miteinander korreliert sind. Der Koeffizient ρ kann Werte zwischen -1 und $+1$ annehmen, je nachdem, ob eine negative, positive bzw. – bei einer Annäherung gegen Null – keine Korrelation vorhanden ist.

Ergebnisse

Muster in der Phytodiversität der Gefäßpflanzen: Sippenzahlen auf Basis der gesamten Florenliste

Von den insgesamt 1.819 in Niedersachsen und Bremen vorkommenden Gefäßpflanzensippen mit floristischem Normalstatus kommen im Mittel etwa ein Viertel (435 Sippen) je Quadrant vor (Tab. 3.1). Maximal wurden 747 Sippen innerhalb eines Quadranten gezählt. Tabelle 3.2 listet die fünf Quadranten mit den höchsten Sippenzahlen auf Basis der gesamten Florenliste auf. Die sippenreichsten Quadranten kommen verstreut vor. Phytodiversitätszentren lassen sich im südlichen und östlichen Niedersachsen sowie nordwestlich von Bremen identifizieren (Abb. 3.2). Vor allem im nördlichen und westlichen Niedersachsen ist die Gefäßpflanzendichte gering. Die deutliche räumliche Konzentration von Diversitätszentren kann auch durch die starke räumliche Autokorrelation belegt werden (Tab. 3.3).

Vergleicht man die Gefäßpflanzenvielfalt der drei naturräumlichen Großregionen, so weist das Berg- und Hügelland sowohl die höchste absolute als auch die höchste mittlere Sippenzahl je Quadrant auf (Tab. 3.1). Die Sippenzahl ist in allen Quadranten relativ gleichmäßig hoch, eine Ausnahme stellt hier nur der Solling dar. Das Tiefland nimmt eine Mittelstellung ein: Die mittlere Sippenzahl je Quadrant entspricht in etwa dem für die Gesamtfläche ermittelten Durchschnitt. Während das westliche Tiefland relativ artenarm ist, finden sich Zentren der Sippen-Vielfalt vor allem im Weser-Aller-Flachland – insbesondere die Flusstäler von Aller, Leine und Weser zeichnen sich deutlich ab –, in der Lüneburger Heide und im Wendland. An der Küste kommen im Mittel nur 306 Sippen je Quadrant vor, so dass die Sippenzahl wesentlich unter dem Landesdurchschnitt liegt. Lediglich im Raum Bremen und auf den Inseln sind relativ hohe Sippenzahlen je Quadrant zu verzeichnen.

Tabelle 3.1: Gesamtsippenzahl (Summe), mittlere Sippenzahl je Quadrant (MW) mit Standardabweichung (± Stabw) und maximale Sippenzahl pro Quadrant (Max) für verschiedene Artengruppen für Niedersachsen und Bremen (Gesamtfläche) bzw. die drei Regionen Küste, Tiefland und Hügel-/Bergland.

Table 3.1: Total number of vascular plant taxa (Summe), mean number of taxa per quadrant (MW) with standard deviation (± Stabw), and maximum number of taxa (Max) per quadrant for different plant groups considering the complete area of Lower Saxony and Bremen (Niedersachsen und Bremen), and the three regions coast (Küste), lowlands (Tiefland), and uplands (Hügel-/Bergland).

	Niedersachsen und Bremen			Küste			Tiefland			Hügel-/Bergland		
	Summe	MW±Stabw	Max	Summe	MW±Stabw	Max	Summe	MW±Stabw	Max	Summe	MW±Stabw	Max
Gesamtliste	1819	435±127	747	1133	306±152	651	1406	440±98	743	1484	503±119	747
Indigene	1509	347±101	611	888	239±116	482	1119	352±76	591	1206	401±97	611
Archäophyten	160	56±20	116	128	42±22	101	152	55±16	115	157	68±21	116
Etablierte Neophyten	145	32±13	84	115	25±16	80	132	33±11	79	119	34±13	84
Ungefährdete Arten	1176	408±114	668	935	292±142	601	1088	413±87	652	1017	469±104	668
Rote-Liste-Arten	643	27±19	127	266	14±13	62	437	27±17	111	526	34±23	127
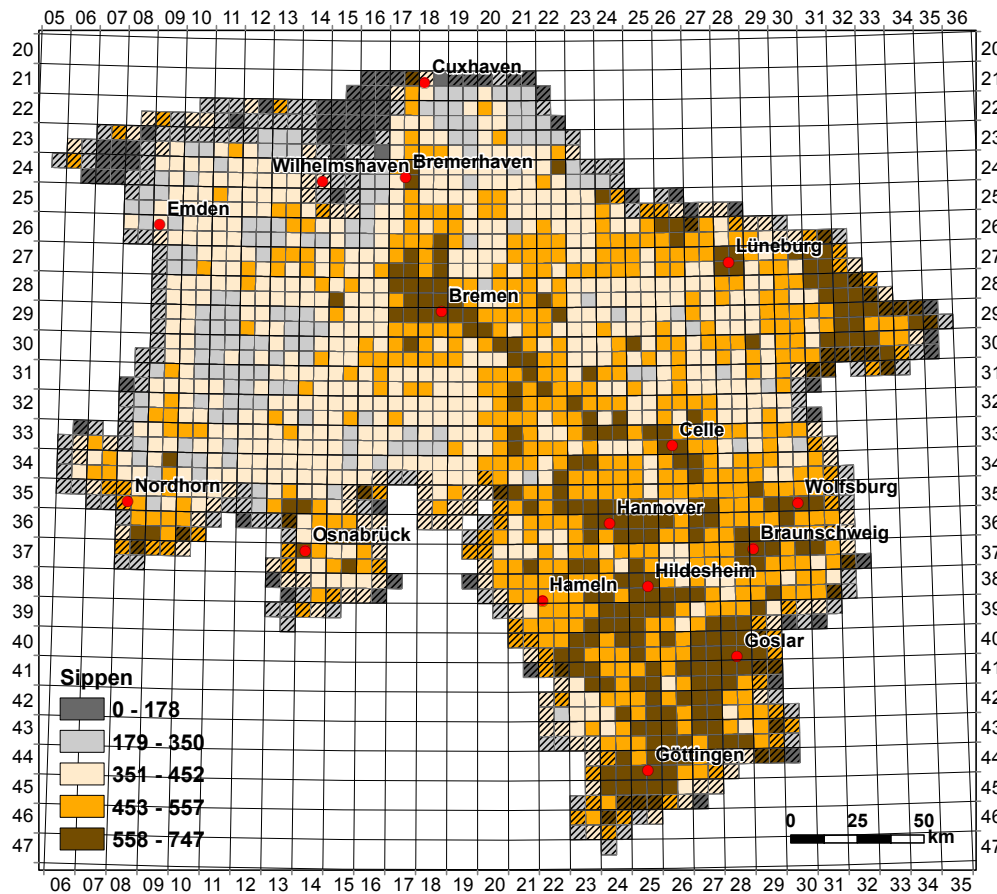
Abbildung 3.2: Anzahl der in Niedersachsen und Bremen vorkommenden Gefäßpflanzensippen mit floristischem Normalstatus je Quadrant basierend auf der Gesamtliste (1.819 Sippen).

Figure 3.2: Number of taxa (excluding erratic and synanthropic occurences) per quadrant in Lower Saxony and Bremen, considering the complete list of 1819 vascular plant taxa.

Muster in der Phytodiversität der Gefäßpflanzen unter Berücksichtigung ihres Einbürgerungsstatus

Die absolute Zahl der Indigenen ist mit 1.509 Sippen um ein Zehnfaches höher als die der Archäophyten (160 Arten) und der etablierten Neophyten (145 Arten) (Tab. 3.1). Dementsprechend unterscheiden sich auch die mittleren Sippenzahlen zwischen den Gruppen mit durchschnittlich 347 Indigenen, 56 Archäophyten und 32 etablierten Neophyten je Quadrant. An der Küste kommen in allen Gruppen weniger Sippen vor als im Tiefland bzw. Hügel- und Bergland, wobei letztere Region – jeweils nur mit geringem Unterschied zum Tiefland – etwas artenreicher ist. Lediglich die absolute Zahl der etablierten Neophyten ist im Tiefland höher als in den übrigen Regionen.

Tabelle 3.2: Auflistung der jeweils fünf sippenreichsten Quadranten je Artengruppe . Quadrant = Nummer das Quadranten, Sippen = Anzahl der Sippen innerhalb des Quadranten, Ort = Innerhalb des Quadranten gelegene Stadt/ Ortschaft.

Table 3.2: A list of the five quadrants with the highest number of taxa per species group. Quadrant = Number of the quadrant, Sippen = Number of taxa inside the quadrant, Ort = City/Town located inside the quadrant.

Artengruppe		1	2	3	4	5
Gesamtliste	Quadrant	4028/3	2832/1	3729/2	4129/1	3032/3
	Sippen	747	743	736	727	726
	Ort	Langelsheim	Hitzacker	Weddel	Bad Harzburg	Luckau
Indigene	Quadrant	4429/2	4028/3	3032/3	4129/1	2832/1
	Sippen	611	598	591	590	586
	Ort	Walkenried	Langelsheim	Luckau	Bad Harzburg	Hitzacker
Archäophyten	Quadrant	3624/4	3624/1	3625/1	3729/1	3825/2
	Sippen	116	115	114	114	108
	Ort	Laatzen	Hannover	Misburg	Braunschweig	Hildesheim
Etablierte Neophyten	Quadrant	3729/1	2818/4	2918/2	3624/1	2818/3
	Sippen	84	80	80	79	76
	Ort	Braunschweig	Gröpelingen	Bremen	Hannover	Grambke
Ungefährdete Arten	Quadrant	3729/1	3729/2	4129/1	4028/3	3032/3
	Sippen	668	663	655	654	652
	Ort	Braunschweig	Weddel	Bad Harzburg	Langelsheim	Luckau
Rote-Liste-Arten	Quadrant	4429/2	2934/4	2832/1	4429/1	2731/4
	Sippen	127	111	110	109	108
	Ort	Walkenried	Gartow	Hitzacker	Bad Sachsa	Glienitz

Die Phytodiversitätskarte der indigenen Sippen (Abb. 3.3) weist starke Parallelen zur vorher besprochenen Darstellung der Sippen der gesamten Florenliste (Abb. 3.2) auf. Im Vergleich der drei Regionen ist die Sippenzahl je Quadrant im Hügel- und Bergland meist überdurchschnittlich hoch. Lediglich im Solling und in den Börden entsprechen die Sippenzahlen etwa dem Durchschnitt. Im Tiefland sind insbesondere weite Teile der Ems-Hunte-Geest und Dümmer-Geestniederung sehr artenarm. Das Wendland, Teile der Stader Geest und des Weser-Aller-Flachlandes – insbesondere wieder das Aller- und Leinetal – sind dagegen ausgesprochene Diversitätszentren. Mit Ausnahme des Raumes Bremen ist die Küstenregion nahezu flächendeckend arm an Indigenen. Die fünf sippenreichsten Quadranten (Tab. 3.2) sind anders gelagert als die der Gesamtliste, weisen jedoch wie diese ebenfalls eine recht zerstreute Verteilung auf.

Tabelle 3.3: Moran's Index als Maß für die räumliche Autokorrelation der Phytodiversitäts-zentren für die verschiedenen Gruppen.

Table 3.3: Moran's Index as a measure for spatial autocorrelation of plant richness centres considering all plant groups.

Artengruppe	Moran's Index	Z-Wert	p-Wert
Gesamtliste	0,138	148,68	0,01
Region Küste	0,157	26,04	0,01
Region Tiefland	0,110	75,25	0,01
Region Hügel-/Bergland	0,062	18,30	0,01
Indigene	0,133	143,18	0,01
Archäophyten	0,166	178,18	0,01
Etablierte Neophyten	0,090	97,69	0,01
Ungefährdete Arten	0,142	152,39	0,01
Rote-Liste-Arten	0,097	104,77	0,01

Ein differenzierteres Muster zeigt die Phytodiversitätskarte der Archäophyten (Abb. 3.4). Hier sind insbesondere der Bremer Küstenraum, das Weser-Aller-Flachland, die Börden sowie das südliche Weser-Leine-Bergland ausgesprochene Phytodiversitätszentren. Diese überschneiden sich stark mit denen der indigenen Sippen (Tab. 3.4), treten aber noch stärker räumlich konzentriert auf (Tab. 3.3). Dies zeigt sich auch bei der Betrachtung der fünf sippenreichsten Quadranten (Tab. 3.2): Die drei artenreichsten Quadranten konzentrieren sich im Raum Hannover, weiterhin ergeben sich hohe Sippenzahlen um Braunschweig und Hildesheim.

Die Phytodiversitätskarte der etablierten Neophyten (Abb. 3.5) zeigt zum einen eine großflächige Konzentration im Bremer Küstenraum, dem Weser-Aller-Flachland und den Börden, zum anderen aber auch eine Vielzahl von zum Teil sehr isolierten Einzelquadranten mit überdurchschnittlich hohen Sippenzahlen. Dabei sind städtische Ballungsräume (Großstädte ≥ 100.000 Einwohnern; HEINEBERG, 2006) grundsätzlich Zentren der Neophyten-Diversität. Diese starke Streuung von großflächigen und punktuellen Zentren der Neophyten-Diversität führt zu einem relativ geringen räumlichen Klumpungseffekt (Tab. 3.3). Dies spiegelt sich auch in der zerstreuten Verteilung der fünf sippenreichsten Quadranten wider (Tab. 3.2). Gleichzeitig ist jedoch der räumliche Überschneidungsbereich zwischen Archäophyten und etablierten Neophyten sehr groß und auch größer als der zwischen Indigenen und Archäophyten bzw. Indigenen und Neophyten (Tab. 3.4).

Muster in der Phytodiversität der Gefäßpflanzen unter Berücksichtigung ihrer Gefährdungssituation

Etwa ein Drittel der Gefäßpflanzen der Gesamtflorenliste (634 Sippen) sind als selten oder gefährdet eingestuft (Tab. 3.1). Für diese Gruppe werden im Hügel- und Berg-

Tabelle 3.4: Ähnlichkeiten in den Verbreitungsschwerpunkten der Gruppen nach Einbürgerungsstatus bzw. Gefährdungssituation. Ergebnisse der Berechnung des Rangkorrelationskoeffzienten ρ nach Spearman. ***, signifikant auf dem Niveau p $\leq$ 0,001.

Table 3.4: Similarities in the distribution of species groups sorted by naturalization status (indigenous, archaeophytes, established neophytes) and threat situation (non-threatened vs. Red List taxa). Results based on the calculation of Spearman's Rank Correlation Coefficient ρ. ***, significant with p $\leq$ 0.001.

	Spearmans ρ	p-Wert
Indigene - Archäophyten	0,744***	< 0,001
Indigene - etablierte Neophyten	0,681***	< 0,001
Archäophyten - etablierte Neophyten	0,805***	< 0,001
Rote-Liste-Arten - ungefährdete Sippen	0,691***	< 0,001

land sowohl die höchsten absoluten als auch mittleren Zahlen pro Quadrant erreicht. Die Mehrzahl der Rote-Liste-Arten gehört der Gruppe der Indigenen an, jedoch sind auch fast ein Drittel der Archäophyten Rote-Liste-Arten, während nur 6 % der Neophyten als gefährdet eingestuft werden (Tab. 3.5).

Die räumlichen Phytodiversitätsmuster der ungefährdeten Sippen (Abb. 3.6) weichen von denen der Rote-Liste-Arten (Abb. 3.7) stark ab, wobei der Überschneidungsbereich aber dennoch relativ hoch bleibt (Tab. 3.4). Insgesamt sind die Rote-Liste-Arten stärker auf mehrere, kleinflächigere Raumeinheiten verstreut als die stark zusammenhängenden Zentren der ungefährdeten Arten (Tab. 3.3). In der insgesamt eher artenarmen Küstenregion bilden nur die Nordsee-Inseln isolierte Hotspots der Rote-Liste-Arten (zum Hotspot-Begriff siehe SCHMITT & HAEUPLER, 2009). Im Tiefland ist die Überlagerung von Verbreitungsmustern der ungefährdeten und gefährdeten Sippen relativ hoch, wobei insbesondere das Wendland (vgl. Tab. 3.2), die Lüneburger Heide und das Elbe-Weser-Dreieck nördlich von Bremen Hotspots der Rote-Liste-Arten sind. Während im Hügel- und Bergland die Anzahl der ungefährdeten Sippen fast überall überdurchschnittlich hoch ist, weisen die Rote-Liste-Arten dagegen stark abgegrenzte Hotspots im Raum Göttingen, dem Weserbergland und am Harzrand (vgl. Tab. 3.2) auf.

Diskussion

Die räumliche Verteilung der Pflanzensippenvielfalt in Niedersachsen und Bremen weist für alle betrachteten Gruppen sowohl von Nord nach Süd als auch in gewissem Maße von West nach Ost eine Zunahme in der Sippendichte auf. Dabei ist das Hügel- und Bergland grundsätzlich die Region mit der höchsten Sippendichte. Eine Ausnahme bilden hier nur die Neophyten.
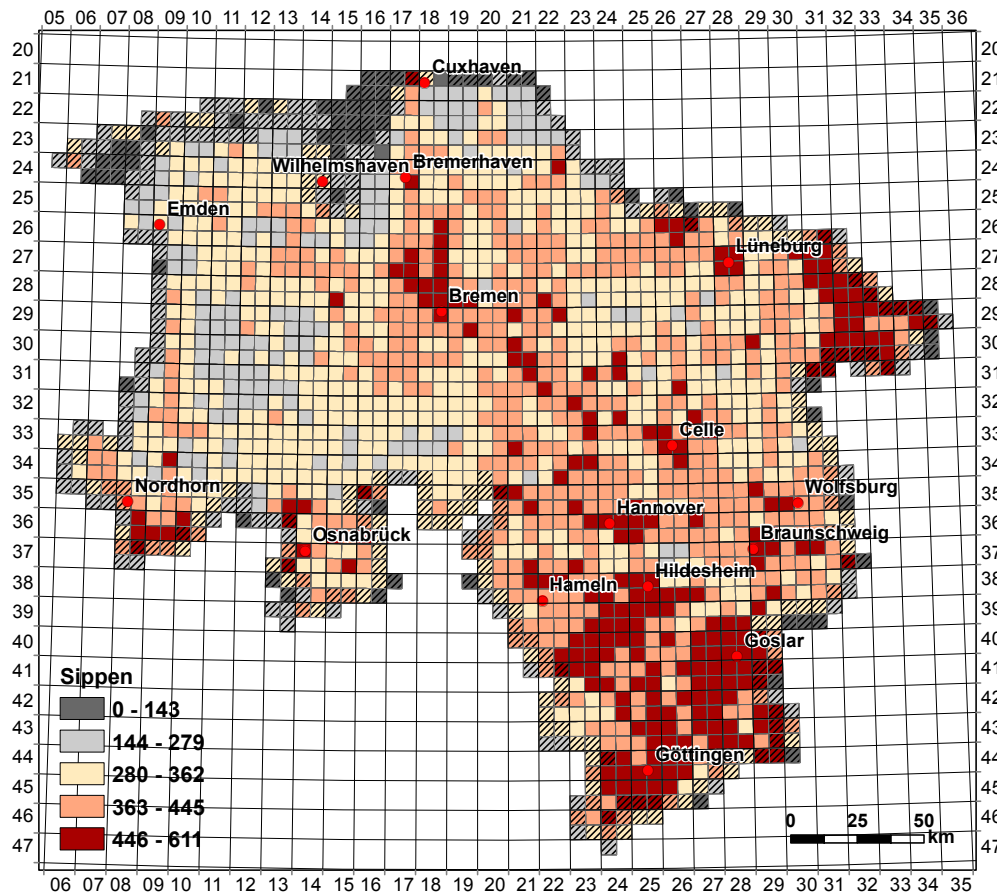
Abbildung 3.3: Anzahl der in Niedersachsen und Bremen vorkommenden 1.509 indigenen Gefäßpflanzensippen.

Figure 3.3: Number of taxa per quadrant in Lower Saxony and Bremen, considering 1509 indigenous vascular plant taxa.

Die relative Artenarmut des norddeutschen Tieflandes ist sicherlich u. a. dadurch bedingt, dass die landschaftliche Heterogenität (Klima, Relief, Gesteine, Böden) des Hügel- und Berglandes von den Lössbörden bis in den Hochharz wesentlich höher ist als die des relativ uniform vom Quartär geprägten Tieflandes. Die kleinräumigen ökologischen Gradienten dürften somit zusätzlich sowohl eine höhere absolute Sippenzahl als auch höhere Sippendichten im südlichen Niedersachsen bedingen. Das insbesondere im Tiefland ausgeprägte Ost-West-Gefälle verläuft parallel zum klimatischen subkontinental-ozeanischen Gradienten. Insbesondere im nördlichen Harzvorland und im Wendland liegen die westlichen Verbreitungsgrenzen vieler Gefäßpflanzenarten, die Vorposten einer östlich der Elbe reicheren, subkontinental getönten Flora sind.
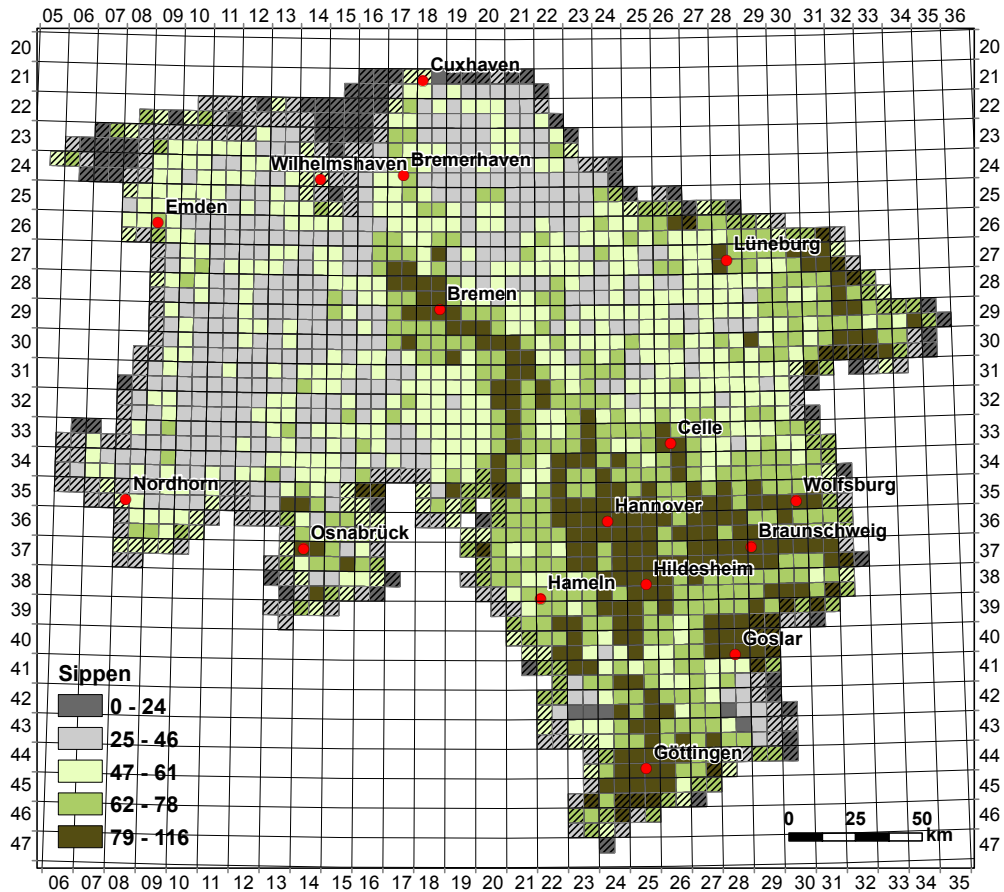
Abbildung 3.4: Sippenzahl je Quadrant der in Niedersachsen und Bremen vorkommenden 160 Archäophyten.

Figure 3.4: Number of taxa per quadrant in Lower Saxony and Bremen, considering 160 archaeophytes.

Die Verbreitungsmuster der Artengruppen nach Einbürgerungsstatus weisen große räumliche Überschneidungen auf. Während die Lage der Diversitätszentren von Indigenen und etablierten Neophyten im Tiefland viele Parallelen aufweist, sind sie im Hügel- und Bergland überwiegend negativ korreliert. Kühn et al. (2003) zeigten in einer deutschlandweiten Analyse, dass die geologische Heterogenität eines Gebietes der wichtigste Faktor für die Sippenzahlen sowohl der Einheimischen als auch der Gebietsfremden ist. Pyšek et al. (2002) erklärten die positive Korrelation von Indigenen und Neophyten in Naturschutzgebieten Tschechiens mit einer hohen Habitatdiversität in den entsprechenden Gebieten, die dazu führt, dass die beiden Gruppen nicht unbedingt miteinander in Konkurrenz stehen müssen. Dies mag auch für die positiv korrelierenden Vorkommen im niedersächsischen Tiefland zutreffen. Gerade in den Gebieten des Hügel- und Berglandes, in denen große, zusammenhängen-
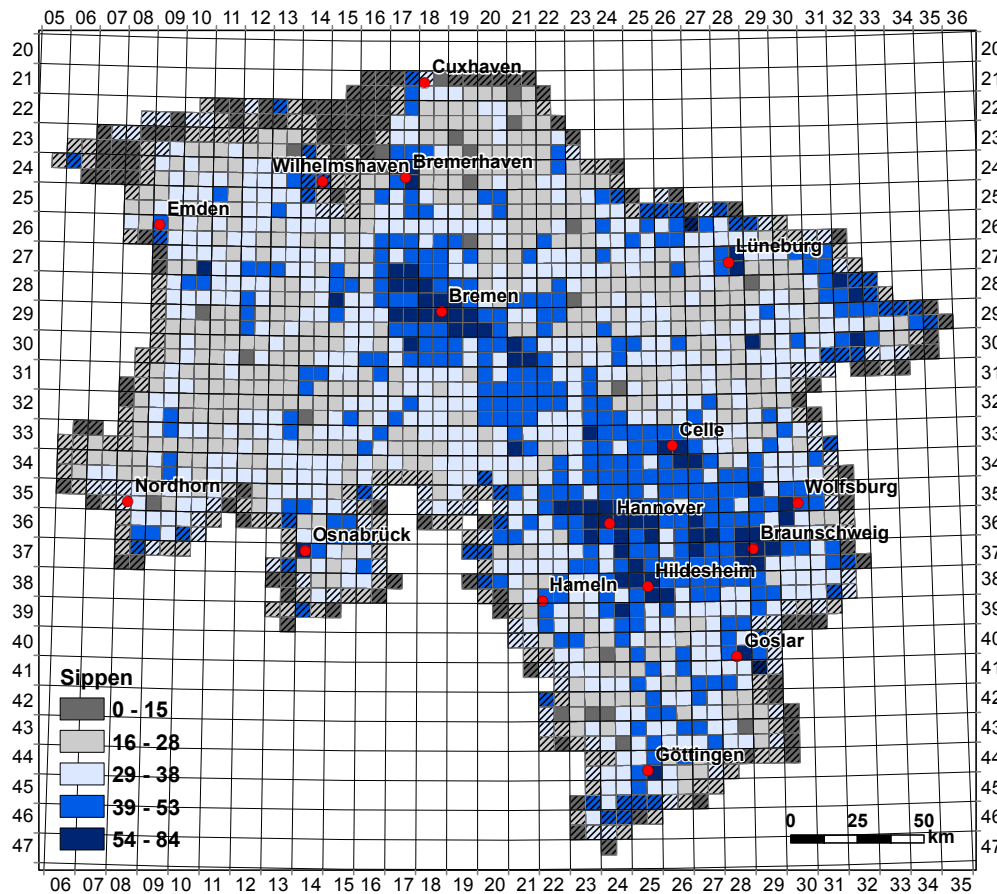
Abbildung 3.5: Sippenzahl je Quadrant der in Niedersachsen und Bremen vorkommenden 145 etablierten Neophyten.

Figure 3.5: Number of taxa per quadrant in Lower Saxony and Bremen, considering 145 established neophytes.

de Zentren der Sippendiversität der Indigenen und besonders geringe Neophyten-Dichten auftreten (Weserbergland, Göttinger Wald, Harz), sind jedoch auch noch besonders große, relativ zusammenhängende Waldflächen vorhanden. Natürliche und naturnahe Wälder, die in Mitteleuropa flächendeckend die potentiell natürliche Vegetationsformation darstellen (Bohn et al., 2000/2003), sind ein Refugium für eine Vielzahl der heimischen Arten. Dagegen ist die Anzahl der Neophyten in Wäldern geringer als im Offenland. Weniger als ein Drittel der in Niedersachsen etablierten Neophyten kommen nach Schmidt et al. (2003) regelmäßig in Wäldern vor. Ihre Schwerpunkte in bewaldeten Landschaften liegen auf Kahlschlagflächen und in bach- und wegbegleitenden Vegetationstypen (vgl. Chytrý et al., 2005).

Besonders hoch ist die Übereinstimmung der Verbreitung der Archäophyten mit der von etablierten Neophyten. Gemeinsam ist beiden Artengruppen, dass sie im

Tabelle 3.5: Zahl der Gefäßpflanzensippen nach ihrem Einbürgerungsstatus (Indigene, Archäophyten und Neophyten) und ihrer jeweiligen Gefährdungssituation (ungefährdete und Rote-Liste-Arten) auf Grundlage der Gesamtflorenliste für Niedersachsen und Bremen.

Table 3.5: Vascular plant taxa richness sorted by naturalization status (indigenous, archaeophytes, established neophytes) and threat situation (non-threatened vs. Red List taxa) based on the complete floristic list of Lower Saxony and Bremen.

	Gesamtliste	Ungefährdete Sippen	Rote-Liste-Arten
Indigene	1509	925	584
Archäophyten	160	111	49
Etablierte Neophyten	145	136	9
Sippen ohne floristischen Status	5	4	1

Gefolge des Menschen nach Mitteleuropa eingewandert sind. Die meisten Archäophyten sind in Acker-, Grünland- und Ruderalvegetation angesiedelt. Sie sind damit zum Teil wichtige Bestandteile von historisch bedeutsamen, heute gefährdeten Biotoptypen, die relativ kleinflächig in eine ansonsten intensiv genutzte Kulturlandschaft eingebettet sind. Dies zeigt sich auch an der Vielzahl der als gefährdet eingestuften Archäophyten. Der von Kühn et al. (2003) festgestellte hohe Anteil von Archäophyten in Lössgebieten stimmt in der vorliegenden Auswertung mit einem Schwerpunkt der Vorkommen dieser Artengruppe in den Lössbörden überein. Letztere sind seit langem besiedelte und heute intensiv genutzte Agrar- und Industrielandschaften, in denen sich auch Neophyten großflächig etablieren konnten.

Urbane Ballungsräume weisen in Niedersachsen und Bremen generell eine hohe Artenzahl auf. Dies stimmt mit den Ergebnissen anderer Untersuchungen, etwa denen von Haeupler (1974), Wania et al. (2006) und Lippe & Kowarik (2008) überein, die feststellten, dass urbane Lebensräume generell eine höhere Pflanzenartenzahl aufweisen als ländliche Gebiete gleicher Größe. Sie führten dies auf die hohe Landschaftsheterogenität in den vielfältig genutzten Räumen zurück, die in der Regel ein kleinräumiges Mosaik von standörtlich sehr heterogenen naturnahen und künstlich geschaffenen Lebensräumen darstellen und somit einer Vielzahl von Pflanzenarten als Wuchsorte dienen können. Kühn et al. (2003) stellten außerdem fest, dass die Flora deutscher Städte natürlicherweise artenreich ist. Sie begründeten dies damit, dass Städte oftmals in von vorneherein sehr artenreichen Landschaftsbereichen entstanden sind.

Besonders evident ist der Zusammenhang zwischen der Diversität und der Urbanität der untersuchten Raumeinheiten für die Neophyten. Zu ähnlichen Ergebnissen kamen auch andere Studien, die sich mit der Verbreitung dieser Artengruppe beschäftigen (Pyšek, 1998; Roy et al., 1999; Honnay et al., 2003; Chytrý et al., 2005; Maskell et al., 2006; Botham et al., 2009; Nobis et al., 2009). Maskell et al. (2006) wiesen für den ländlichen Raum Großbritanniens nach, dass nichtheimische Arten sich vor allem
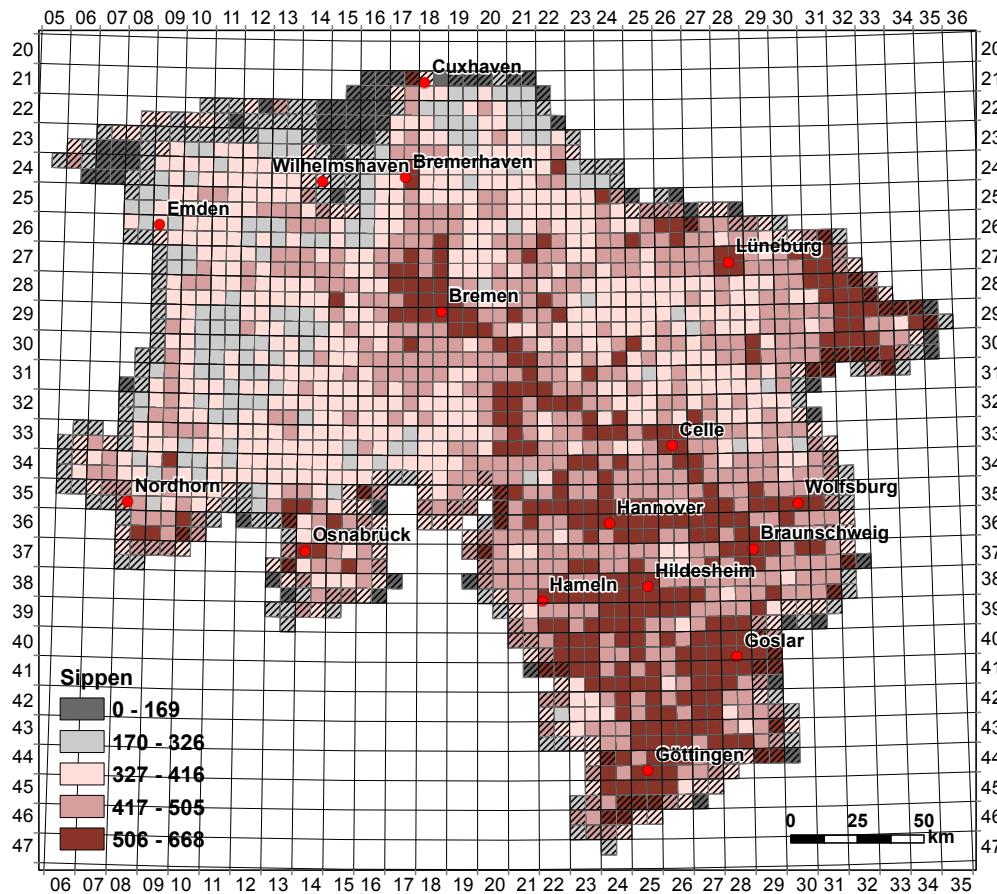
Abbildung 3.6: Anzahl der in Niedersachsen und Bremen je Quadrant vorkommenden 1176 ungefährdeten Gefäßpflanzensippen.

Figure 3.6: Number of taxa per quadrant in Lower Saxony and Bremen, considering 1176 invulnerable vascular plant taxa.

in vom Menschen geprägten Habitaten etablierten. Botham et al. (2009) untersuchten in Großbritannien innerhalb von zwei verschiedenen Untersuchungszeiträumen (1987-1988 und 2003-2004) die Bindung von indigenen Pflanzenarten, Archäophyten und Neophyten an verschiedene Landbedeckungsklassen. Sie stellten fest, dass Neophyten in beiden Zeitabschnitten stark an urbane Landschaftsbereiche gebunden waren, wohingegen die Archäophyten nur im ersten Untersuchungszeitraum stark mit urbanen Lebensräumen assoziiert waren. In geringerem Maße als für die Neophyten ist für die Archäophyten auch in unserer Studie ein Zusammenhang zwischen Urbanität und Artenreichtum zu erkennen. Diese Gruppe zeigt relativ kleinflächig ausgeprägte Diversitätszentren im Umfeld vieler niedersächsischer Städte. Aufgrund der Großräumigkeit der zugrundeliegenden Raumeinheiten konnte hier nicht untersucht werden, ob der Artenreichtum der Archäophyten, wie Pyšek (1998) bemerkt,
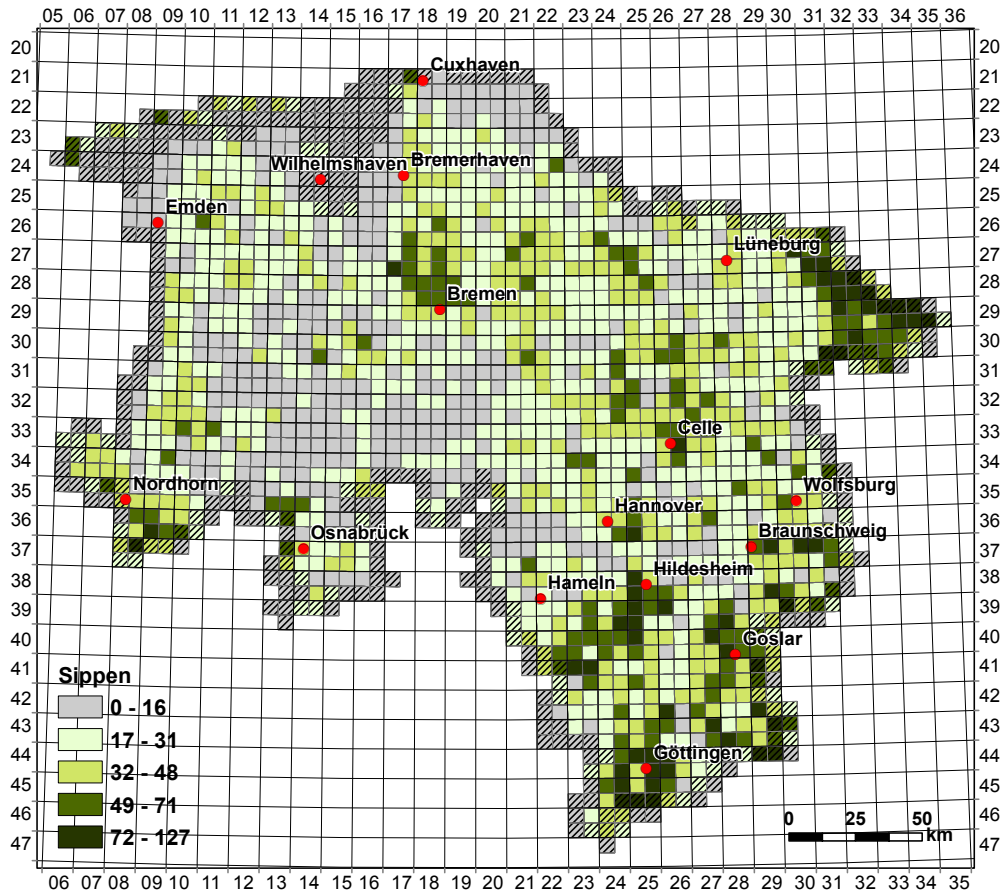
Abbildung 3.7: Anzahl der 643 seltenen und gefährdeten Gefäßpflanzensippen Niedersachsens und Bremens (Rote Liste-Status 1, 2, 3, G, R) je Quadrant.

Figure 3.7: Number of taxa per quadrant in Lower Saxony and Bremen, considering 643 rare and endangered vascular plant taxa (Red List status 1, 2, 3, G, R).

aufgrund ihrer Anpassung an ländliche Räume vor allem auch in kleineren Städten erhöht ist.

HAEUPLER & VOGEL (1999) sowie HAEUPLER (2000) betonen, dass die Musteranalyse von Rote-Liste-Artenzahlen gut geeignet ist, die 'echten' Hotspots der Gefäßpflanzendiversität aufzudecken. Für Niedersachsen und Bremen weisen die gefährdeten Arten ein wesentlich differenzierteres und eher punktuelles Verbreitungsmuster auf als die anderen betrachteten Artengruppen. Die Hotspots der Rote-Liste-Arten korrelieren in vielen Fällen mit denen der anderen Artengruppen, liegen jedoch nicht unmittelbar in den städtischen Ballungszentren. Bei den Rote-Liste-Arten handelt es sich fast ausschließlich um Indigene bzw. Archäophyten, wobei diese Arten oft Spezialisten sind, die an natürliche oder naturnahe Habitate angepasst sind. Eine besonders starke Überschneidung von Raumeinheiten mit hoher Dichte sowohl gefährdeter Arten

als auch von Neophyten ergeben sich in Niedersachsen für das Wendland, die Umgebung von Göttingen, Hildesheim und Goslar sowie die Region nördlich von Bremen. Es kann jedoch davon ausgegangen werden, dass die Hauptursachen der Gefährdung von Pflanzenarten in der Intensivierung der Landnutzung und ihren Folgen (Eutrophierung, Entwässerung, Aufgabe von Extensivflächen) sowie dem gesteigerten Flächenbedarf der Bevölkerung liegen und nicht durch die Konkurrenz der Neophyten bedingt sind (GARVE, 2004).

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4

The effectiveness of protected areas for the conservation of threatened vascular plant species

Summary

The present study tests the effectiveness of protected areas for the conservation of rare and threatened vascular plant species. For this purpose, the lowland area of the federal state of Lower Saxony, Germany with its 41 geographical sub-regions served as a model region.

The analysis was conducted for nature protected areas[1] and Natura 2000 sites that represent the most important and extensive conservation areas in the lowlands. Information on the occurrences of Red List (RL) species were obtained from the floristic mapping of the state (plant survey of Lower Saxony, NLWKN 1982-2003). In addition to the total number of RL species, those species linked to forest habitats[2] (forest affinity categories 1.1 and 1.2, SCHMIDT et al., 2011) were considered in the analysis. All data were available on the basis of a regular grid of 15.729 grid cells with a size of 207 ± 2 ha. The study aimed at answering the following research questions: 1.) What patterns of diversity can be observed for the two species groups? 2.) Are occurrences of threatened species covered by nature protected areas? 3.) Does the designation of Natura 2000 sites improve the situation? 4.) Are the results obtained for RL species transferable to RL forest species?

The correlation between the extent of protected areas and the number of RL species per grid cell was analysed using Generalised Linear Models (GLM) with Poisson distribution. The calculation was performed separately for each of the 41 geographical sub-regions. The same kind of analysis was repeated for RL forest species.

Richness of the rare and threatened vascular plant species in the lowlands of Lower Saxony showed a distinct pattern. Centres of species richness (hotspots) were identified that should be considered as priority areas for nature conservation. Hotspots of RL species were mainly located in the Wendland area, the Westmünsterland and the Wesermünder Geest, those of RL forest species in the Oldenburger Geest. In contrast, large parts of the western lowlands as e. g. the Hunte-Leda-Moorniederung, the Bersenbrücker Land as well as the Bourtanger Moor were found to be very species-poor. With the Ostmünsterland and the Rahden-Diepenauer Geest being an exception, hotspots of RL species were well covered by protected areas. In contrast, richness centres of RL forest species were effectively protected in only c. 60 % of the geographical sub-regions. However, the number of species in those regions where no positive correlation between the distribution of species richness and the extent of protected areas could be found was generally very low. By including Natura 2000 sites the proportion of well-protected hotspots increased to an effective protection in 95 % of the geographical sub-regions for RL species and 81 % for RL forest species.

The diverging results obtained for the two species groups suggest that it is reasonable to not only focus on the total number of RL species representing a wide

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[1]German: *Naturschutzgebiete*

[2]Below named 'RL forest species'.

ecological amplitude as it has been done in most studies dealing with the effectiveness of protected areas for species conservation. Instead it is worthwhile to extent the analysis to species groups that represent more narrow habitat requirements.

Abstracts

Die Einrichtung von Schutzgebieten ist ein zentrales Instrument des Artenschutzes. Am Beispiel der 41 naturräumlichen Regionen des niedersächsischen Tieflands wird ein Verfahren zur Überprüfung der Effektivität von Naturschutz- (NSG) und FFH-Gebieten (Fauna Flora Habitat) für den Schutz seltener und gefährdeter Gefäßpflanzenarten (Rote-Liste-Arten = RL-Arten) vorgestellt. Grundlage sind die Daten des Niedersächsischen Pflanzenarten-Erfassungsprogramms.

In 85 % der naturräumlichen Regionen ist die Gesamtzahl der RL-Arten durch NSG effektiv geschützt; für die an den Lebensraum Wald gebundenen RL-Arten hingegen trifft das nur in 61 % der naturräumlichen Regionen zu. Die Ausweitung des Schutzgebietssystems durch FFH-Gebiete verbessert die Schutzsituation der RL-Arten insgesamt wie auch die der RL-Waldarten erheblich.

Der vorgestellte Ansatz ermöglicht die Identifizierung von Zentren der Artenvielfalt (Hotspots), die in Hinsicht auf ihren Schutz und ggf. gezielte konservierende Pflegemaßnahmen besondere Beachtung finden sollten.

Effectiveness of protection areas for the conservation of rare and endangered vascular plant species – An investigation in the lowlands of Lower Saxony, Germany

The establishment of protected areas is a central tool in species conservation. Using the 41 geographical regions of the Lowlands of Lower Saxony, Germany, as a model region, a method to test the effectiveness of nature protected areas and Natura 2000 sites for the conservation of rare and endangered vascular plant species (RL, Red List species) is proposed. RL species are effectively protected by nature protected areas in 85 % of the geographical regions; for RL species linked to forest habitats this is the case only in 61 % of the geographical regions. The extension of the protected area system to include Natura 2000 sites greatly improves the situation.

The approach allows the identification of regions with high species richness of endangered plants (hotspots) that should be particularly considered for conservation and management actions.

Einleitung

Die Einrichtung von Schutzgebieten stellt ein zentrales Instrument von Strategien zur Verringerung von Biodiversitätsverlusten dar (MEA, 2005; Gaston et al., 2006; BMU, 2007). In diesem Zusammenhang fordert die 'Globale Strategie zur Erhaltung der Pflanzen' (GSPC, CBD-COP6, Entscheidung VI/9, Sekretariat der CBD, 2007) einen effektiven in-situ-Schutz, durch den dem weiteren Rückgang von gefährdeten Pflanzenarten entgegengewirkt werden soll. Ob die für den Pflanzenartenschutz bedeutsamen Flächen auch tatsächlich durch Schutzgebiete abgedeckt sind, wurde bisher für größere Räume kaum untersucht. Für Deutschland fehlen Studien, die die Wirksamkeit der bestehenden Schutzgebietskulisse analysieren (Engel et al., 2012), und auch auf internationaler Ebene sind nur wenige Untersuchungen zur Effektivität von Schutzgebieten für den Artenschutz bekannt (Dimitrakopoulos et al., 2004; Deguise & Kerr, 2006; Jackson et al., 2009).

Naturschutzgebiete (NSG) und Fauna-Flora-Habitat-Gebiete (FFH-Gebiete) des europäischen Netzwerks Natura 2000 sind in Deutschland die flächenmäßig bedeutsamsten Schutzgebietskategorien (BfN, 2008). Die Ausweisung von NSG kann aus wissenschaftlichen, naturgeschichtlichen und/oder landeskundlichen Gründen sowie wegen der Seltenheit, besonderen Eigenart oder hervorragenden Schönheit der Gebiete erfolgen (§ 23 BNatSchG, 2009). Der Artenschutz ist also nicht unbedingt explizites Kriterium für die Schutzgebietsausweisung. Im Rahmen der FFH-Richtlinie (Richtlinie 92/43/EWG) werden Schutzgebiete dagegen vorrangig nach Kriterien des Habitat- und Artenschutzes ausgewiesen (Ssymank, 1998). Hierbei wurden allerdings vielfach bestehende Schutzgebiete, insbesondere NSG, eingeschlossen. Für beide Schutzgebietskategorien gibt es keine allgemeingültigen Schutzbestimmungen – diese müssen durch Schutzgebietsverordnungen (NSG) bzw. Managementpläne (FFH-Gebiete) festgelegt werden.

Die vorliegende Studie untersucht die Effektivität von Schutzgebietssystemen (NSG und FFH-Gebiete) für die Erhaltung von seltenen und gefährdeten Gefäßpflanzen im niedersächsischen Tiefland. Dabei werden zunächst alle im niedersächsischen Tiefland vorkommenden Gefäßpflanzenarten der Roten Liste Niedersachsens betrachtet (490 Sippen; Rote Liste-Status der Tiefland-Region 1, 2, 3, G, R; Garve, 2004; im Folgenden RL-Arten genannt).

Um feststellen zu können, ob die für die RL-Arten der Gesamtliste erhaltenen Ergebnisse gleichermaßen für einen Teil der an einen bestimmten Lebensraum gebundenen RL-Arten gelten, werden die Analysen beispielhaft für die seltenen und gefährdeten Waldarten (nach Schmidt et al., 2011; 60 Sippen; im Folgenden RL-Waldarten genannt) wiederholt, da die Datenlage zu dieser Artengruppe sehr gut ist und Waldlebensräume eine große ökologische Bedeutung besitzen (ML, 2004). Wälder, insbesondere solche mit langer Habitatkontinuität, nehmen im nordwestdeutschen Tiefland im Vergleich zum nordostdeutschen Tiefland und vielen deutschen Mittelgebirgsregionen einen sehr geringen Flächenanteil ein (Glaser & Hauke, 2004; Schmidt

et al., 2009). Hinsichtlich der Nutzungsgeschichte und Größe der Wälder sowie ihres Fragmentierungsgrades bestehen zwischen den 41 naturräumlichen Regionen des Tieflands große Unterschiede, die sich sowohl in der Ausstattung mit seltenen und gefährdeten Waldgefäßpflanzenarten als auch in der Schutzgebietskulisse widerspiegeln sollten.

Die NSG und FFH-Gebiete im niedersächsischen Tiefland sind sehr unterschiedlichen Alters. Während die NSG teilweise bereits seit über 90 Jahren bestehen (Pohl, 1999), wurden die FFH-Gebiete als Teile des Natura-2000-Schutzgebietssystems erst seit 1997 an die EU gemeldet (NMUK, 2010).

Folgenden Fragestellungen wird nachgegangen:
1. Wie ist die Artenvielfalt der betrachteten Artengruppen im niedersächsischen Tiefland verteilt?
2. Inwieweit sind die Vorkommen der RL-Arten durch NSG abgedeckt?
3. Hat sich die Schutzgebietsabdeckung durch die Ausweisung von FFH-Gebieten verbessert?
4. Sind die für die RL-Arten der Gesamtliste erhaltenen Ergebnisse auf die RL-Waldarten übertragbar?

Methoden

Untersuchungsgebiet

Die Untersuchungen beziehen sich auf das niedersächsische Tiefland (Abb. 4.1), das sich in fünf naturräumliche Großregionen mit 41 naturräumlichen Regionen (NR) gliedert (Meynen & Schmithüsen, 1953–1962, modifiziert nach Pilgrim & Franke, 1993). Das Untersuchungsgebiet (UG) ist durch geringe Reliefunterschiede geprägt und von pleistozänen Ablagerungen dominiert (Heunisch et al., 2007). Von West nach Ost besteht ein Kontinentalitätsgradient, der sich in mittleren Jahresniederschlags- sowie Temperaturgradienten ausdrückt und sich auch auf die Bodenzusammensetzung auswirkt (NLfB, 1997).

Die Ems-Hunte-Geest und Dümmer-Geestniederung im Südwesten des UG ist in ihrem nördlichen Teil von ausgedehnten, vielfach von Flugsanden oder Sandlöss bedeckten Grundmoränenplatten geprägt. Ihr Südteil wird von Talsandflächen, großflächigen Mooren und kleineren Grundmoränenplatten dominiert, die z. T. von Endmoränenzügen überragt werden. Der Waldanteil liegt bei 17 %, geschützte Waldbereiche nehmen 1 % ein. Die Ostfriesisch-Oldenburgische Geest im Nordwesten liegt auf Grundmoränenplatten und wird intensiv landwirtschaftlich genutzt; Wälder haben einen Flächenanteil von 7 %. Das Weser-Aller-Flachland mit den Urstromtälern von Aller und Weser und den sich südlich anschließenden flachwelligen Moränenlandschaften liegt im Südosten des UG an der Grenze zum Hügel- und Bergland und hat mit 21 % den zweithöchsten Waldanteil im Tiefland. Die im Norden gelegene

Stader Geest ist durch Grundmoränen, Geesten sowie moorreiche Flussniederungen geprägt. Sie setzt sich aus kleinräumigen Acker-, Grünland-, Wald- und Moorgebieten zusammen. Wälder nehmen knapp 15 % der Region ein. In Lüneburger Heide und Wendland im östlichen Teil des UG überwiegen sandige Grund- und Endmoränen. Die Region wird von Äckern, Wäldern und den größten Sandheiden Niedersachsens geprägt und hat mit 39 % den höchsten Waldanteil der Großregionen des UG.

Gefäßpflanzendaten

Die Verbreitungsdaten der im UG als selten oder gefährdet eingestuften Gefäßpflanzensippen (RL-Status der Tiefland-Region 1, 2, 3, G, R; GARVE, 2004) wurden der Datenbank des Niedersächsischen Pflanzenarten-Erfassungsprogramms entnommen (NLWKN 1982-2003, Stand 15.12.2008). Es gingen 490 Sippen (468 Arten, 14 Subspezies, 8 Aggregate) in die Analyse ein. Die Sippen wurden jeweils auf der höchsten in der Datenbank aufgeführten taxonomischen Ebene zusammengefasst. Die Analyse erfolgte auf Minutenfeld-Ebene (15 Minutenfelder pro Messtischblatt-Quadrant; insgesamt 15.729 Minutenfelder mit einer mittleren Größe von $207 \pm 2\,\mathrm{ha}$). Für die Teilanalyse der an den Wald gebundenen RL-Arten wurden aus der Gesamtliste diejenigen Sträucher und krautigen RL-Arten extrahiert, die im geschlossenen Wald sowie an Waldrändern und auf Waldverlichtungen vorkommen (Waldbindungskategorien 1.1, 1.2, SCHMIDT et al., 2011). Der Gruppe gehörten 60 Sippen (59 Arten, 1 Aggregat) an. Die Verbreitungsmuster der Arten wurden als Artenzahl pro Minutenfeld berechnet und in einem Geografischen Informationssystem (ESRI INC., 2007) visualisiert. Dabei wurden natürliche Grenzwerte mit jeweils fünf Klassen verwendet.

Schutzgebietssysteme

Die Ausdehnung und Lage der NSG und FFH-Gebiete wurden digitalen Daten entnommen (NLWKN, 2010a; NLWKN, 2010b, Stand: 31.12.2009). Für die Teilanalyse der an den Wald gebundenen RL-Arten wurden die digitalen Schutzgebietsdaten mit dem ATKIS-DLM-Layer Wald (LGN, Hannover) verschnitten. In die Analyse wurden zum einen durch NSG, zum anderen zusätzlich auch die durch FFH-Gebiete geschützten (Wald-)Flächen einbezogen und ihr Flächenanteil je Minutenfeld berechnet.

Effektivitäts-Analyse

Der Zusammenhang zwischen der Ausdehnung der Schutzgebiete (NSG bzw. NSG und FFH-Gebiete) bzw. der geschützten Waldflächen (durch NSG bzw. NSG und FFH-Gebiete geschützte Waldflächen) und der Anzahl von RL-Arten bzw. RLWaldarten pro Minutenfeld wurde mit Generalisierten Linearen Modellen mit Poisson-Verteilung untersucht. Es wurden einzelne Analysen für jede der 41 naturräumlichen Regionen berechnet. Die Schutzgebietsfläche je Minutenfeld war dabei

die erklärende, die Artenzahl die abhängige Variable. Die Datenanalyse erfolgte mit dem Statistikprogramm R (R Development Core Team, 2012).

Habitatbindung der RL-Waldarten

Für die weitere Interpretation wurde die Habitatbindung der RL-Waldarten gemäß FloraWeb-Datenbank (BfN, 2010; Haupt- und Schwerpunktvorkommen) und Garve (1994) ermittelt (bis zu drei Mehrfachnennungen je Art).

Ergebnisse

Verbreitung der RL-Arten

Im UG wurden insgesamt 490 RL-Arten nachgewiesen. Die absolute Artenzahl variierte zwischen den Großregionen stark. Während sie maximal bei 454 RLArten lag (Lüneburger Heide und Wendland), kamen im Minimum in der Ostfriesisch-Oldenburgischen Geest nur 255 Arten vor (Tab. 4.1). Die räumliche Verteilung von RL-Hotspots war deutlich inhomogen (Abb. 4.2). Maximal kamen pro Minutenfeld 86 Arten, im Mittel 5 (± 6) Arten vor. Weniger als 1 % der Minutenfelder wies mehr als 28 Arten auf, wobei die größte Dichte dieser Hotspots im Wendland lag (NR 860, 876). Rund 7 % der Minutenfelder waren sehr artenreich (15–28 Arten). Das westliche Tiefland war allgemein artenarm mit punktuell großflächigen Hotspots im Bentheimer Raum (NR 544, 580), im Emsgebiet (NR 586, 592, 604) und nordwestlich von Oldenburg (NR 602, 603). Im östlichen Tiefland war die Artendichte pro Minutenfeld tendenziell höher. Großflächige Hotspots lagen vor allem im Elbe-Weser-Dreieck (NR 631–633), in der Allerniederung (NR 626, 627), der Lüneburger Heide (NR 641, 642) und im Wendland. Die Teilmenge der an Waldhabitate gebundenen RL-Arten lag mit 60 Arten bei 12 %. Die Verbreitungsmuster der RL-Waldarten (Abb. 4.3) unterschieden sich stark von denen der Gesamtliste. Maximal kamen 22 Arten, im Mittel nur 0,5 (± 1,3) Arten pro Minutenfeld vor (Tab. 4.1). In 76 % der Minutenfelder konnte keine Art nachgewiesen werden. Die Hotspots mit mehr als sieben Arten waren nur punktuell ausgebildet und umfassten weniger als 1 % der Minutenfelder. Räume mit höheren Artendichten waren die Oldenburger Geest (NR 603), das Elbe-Weser-Dreieck (NR 633, 634), das Uelzener und Bevenser Becken (NR 643) sowie die Burgdorf-Peiner Geestplatten und das Ostbraunschweigische Flachland (NR 623, 624). Das Wendland wie auch die im äußersten Westen des UG gelegenen Gebiete, die bei Betrachtung der Gesamtliste die höchsten RL-Artenzahlen aufwiesen, waren vergleichsweise arm an RL-Waldarten.
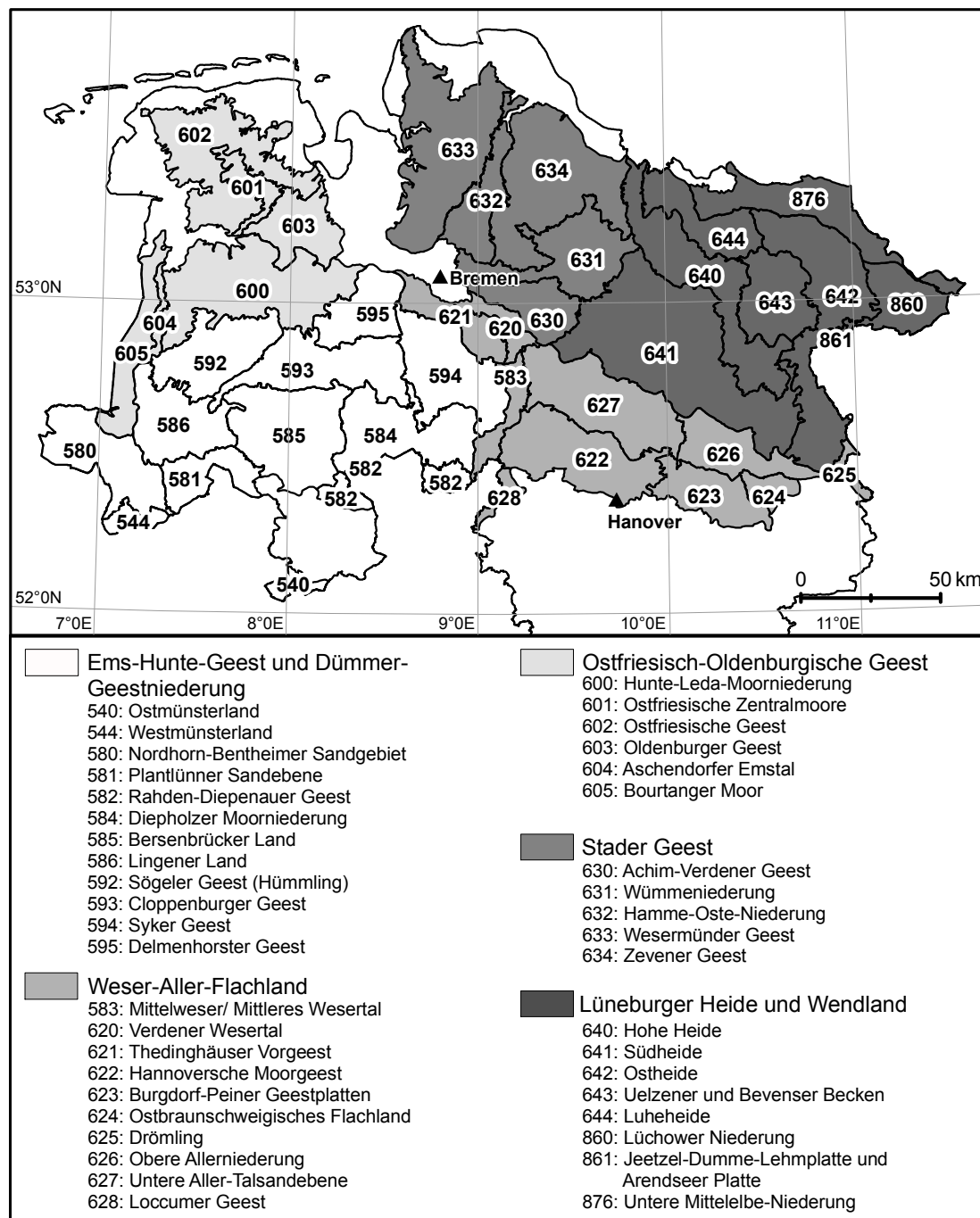
Abbildung 4.1: Das Untersuchungsgebiet des niedersächsischen Tieflandes mit seinen fünf naturräumlichen Großregionen und 41 naturräumlichen Haupteinheiten (Meynen & Schmithüsen, 1953–1962, modifiziert nach Pilgrim & Franke, 1993). Bezugssystem: Deutsches Hauptdreiecksnetz (DHDN, Gauss-Krüger, Zone 3).

Tabelle 4.1: Gesamtfläche, absolute und prozentuale Flächenanteile von Schutzgebieten und Waldschutzgebieten (NSG und FFH-Gebiete) und Anzahl der RL–Arten bzw. RL-Waldarten in den fünf naturräumlichen Großregionen des niedersächsischen Tieflandes.

Großregion	Fläche [ha]	Wald-fläche [ha]	Wald-anteil [%]	NSG und FFH-Ge-biete [ha]	Waldfläche in NSG und FFH-Gebieten [ha]	Anteil der Waldfläche in NSG und FFH-Gebieten an der Gesamt- Waldfläche [%]	RL-Arten	RL-Wald-arten
Ems-Hunte-Geest und Dümmer-Geestniederung	863.759	143.640	16,6	47.572	9.674	6,7	336	39
Ostfriesisch-Oldenburgi-sche Geest	457.808	31.455	6,9	27.265	2.885	9,2	255	26
Weser-Aller-Flachland	495.106	105.502	21,3	49.407	12.776	12,1	379	48
Stader Geest	553.736	81.363	14,7	40.194	8.190	10,1	327	37
Lüneburger Heide und Wendland	842.957	329.046	39	93.009	40.159	12,2	454	55
Tiefland Gesamt	**3.213.366**	**691.006**	**21,5**	**257.447**	**73.684**	**10,7**	**490**	**60**

Schutzgebietssysteme

Im UG waren 530 NSG ausgewiesen, die insgesamt 140.439 ha (4,4 %) der Gesamtfläche einnahmen. Die 240 FFH-Gebiete erfassten mit 212.962 ha 6,6 % der Tieflandfläche. Durch eine starke Überlappung der Schutzgebietskategorien (Abb. 4.4) summierte sich die insgesamt von NSG und FFH-Gebieten abgedeckte Fläche im UG auf 257.445 ha (8 % der Gesamtfläche; Tab. 4.1). Durch NSG geschützte Wälder nahmen 41 446 ha (1,3 % der Gesamtfläche), durch FFH-Gebiete geschützte Wälder 64.791 ha (2 %) ein. Der Anteil der insgesamt durch NSG und FFH-Gebiete geschützten Waldbereiche lag bei 2,3 %, der Anteil an der Waldfläche bei 10,7 % (Tab. 4.1).

Effektivität von Schutzgebieten für den Artenschutz

Die Vorkommen und Verbreitungsschwerpunkte der RL-Arten lagen zum überwiegenden Teil in Schutzgebieten (Tab. 4.2a). Die Effektivitätsanalyse zeigte für 85 % der NR einen positiven Zusammenhang zwischen NSG- und RL-Arten-Vorkommen, in den übrigen NR bestand kein Zusammenhang. In der Unteren Mittelelbe-Niederung (NR 876), die eine außergewöhnlich hohe RL-Artenzahl und -dichte aufwies, lagen die Artvorkommen überwiegend außerhalb von NSG. Mit der Erweiterung der Schutzgebietskulisse um FFH-Gebiete (ca. 117.000 ha) erhöhte sich die Anzahl der NR mit Gebietsschutz auf 95 %. Dabei nahm z. B. die Schutzgebietsfläche in der Loccumer Geest von <1 % auf 18 % zu (Abb. 4.5a/b), im RL-Hotspot Untere Mittelelbe-Niederung war sogar ein Anstieg von <1 % auf 24 % zu verzeichnen. Lediglich in zwei naturräumlichen Regionen, in denen keine nennenswerte Veränderung der Schutzgebietsfläche stattfand, wurde auch weiterhin keine effektive Abdeckung der Artvorkommen durch Schutzgebiete erreicht.

Die Hotspots der RL-Waldarten waren deutlich schlechter durch Schutzgebiete abgedeckt als die der RL-Arten der Gesamtliste (Tab. 4.2b). Lediglich in 61 % der NR lagen die Artvorkommen überwiegend in als NSG geschützten Wäldern. Durch die Ausweisung der FFH-Gebiete stieg der Anteil der NR mit einer guten Schutzgebietsabdeckung der Vorkommen der RL-Waldarten jedoch auf insgesamt 81 %. Markante Beispiele für diese Entwicklung sind die Oldenburger Geest (NR 603) und die Burgdorf-Peiner Geestplatten (NR 623). Hier sind die Minutenfelder mit besonders hohen Artendichten (>11 RL-Waldarten) nicht durch NSG geschützt, wurden jedoch in die FFH-Gebietskulisse einbezogen (Tab. 4.2b, Abb. 4.5c/d).

Gegenüber der Schutzgebietsabdeckung der Vorkommen der RL-Arten insgesamt waren die Vorkommen der RL-Waldarten damit zwar deutlich weniger stark durch Schutzgebiete gesichert, allerdings waren die verbleibenden NR, in denen kein Zusammenhang von RL-Waldartenverteilung und Schutzgebieten erkennbar war, generell relativ artenarm. Eine Ausnahme bildete lediglich die Rahden-Diepenauer Geest (NR 582), in der sowohl die RL-Arten insgesamt als auch die mit einer vergleichsweise hohen Zahl vorkommenden RL-Waldarten auch nach der Ausweisung von FFH-Gebieten nicht ausreichend durch Schutzgebiete gesichert waren.
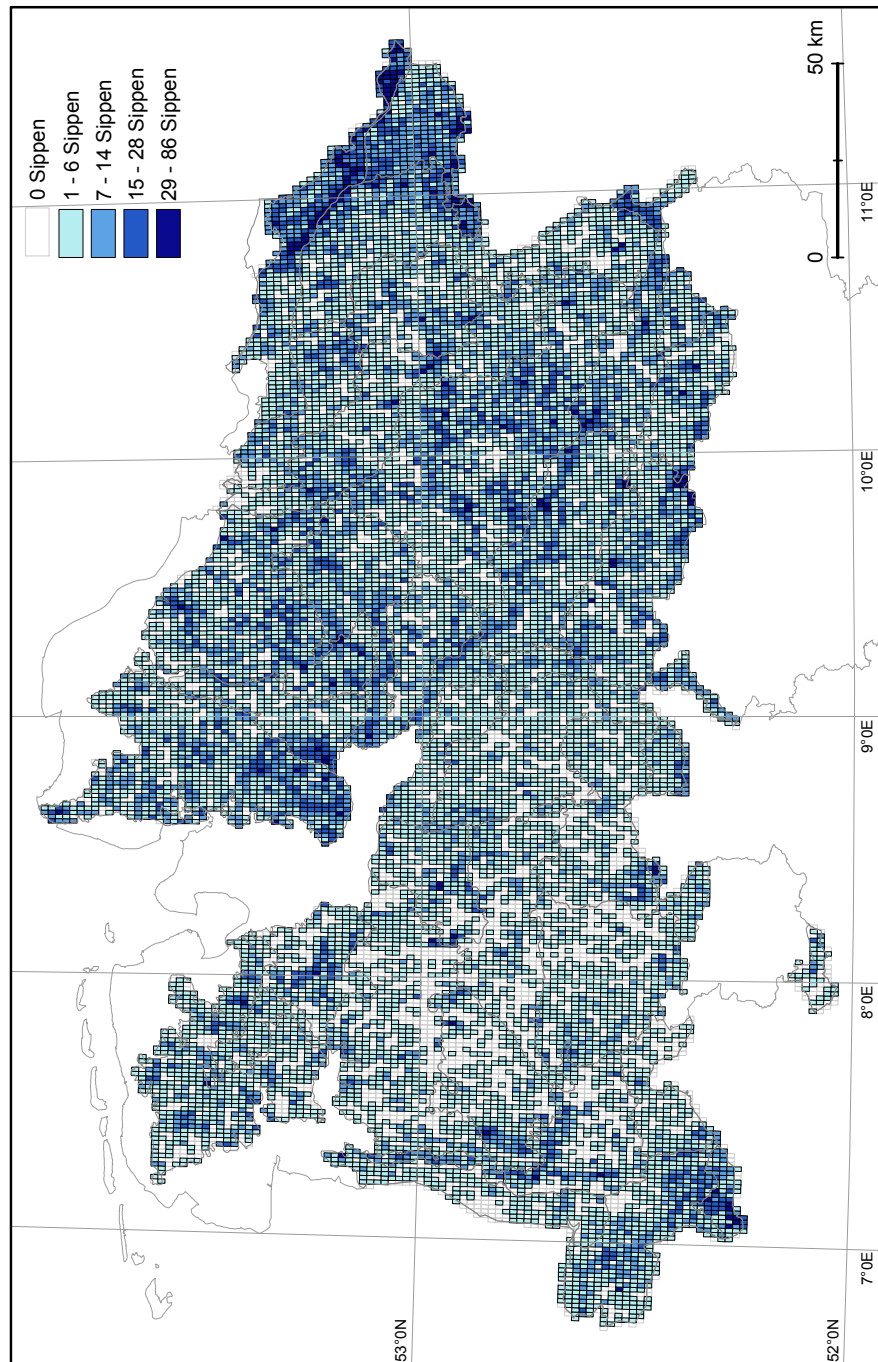
Abbildung 4.2: Anzahl der seltenen und gefährdeten Gefäßpflanzensippen pro Minutenfeld im niedersächsischen Tiefland. Insgesamt gingen 490 Sippen in die Berechnung ein.
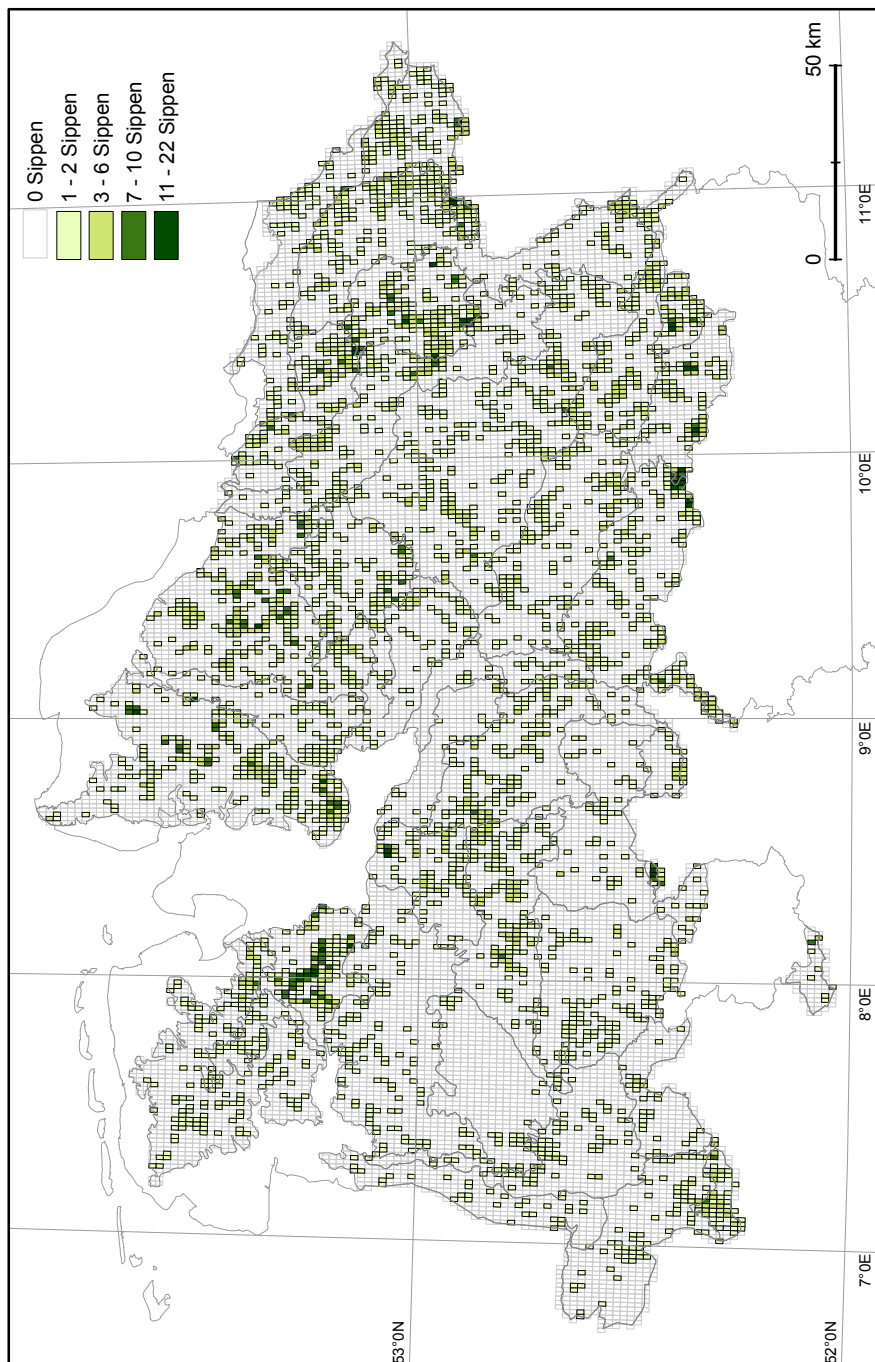
Abbildung 4.3: Anzahl der seltenen und gefährdeten Waldgefäßpflanzensippen pro Minutenfeld im niedersächsischen Tiefland. Insgesamt gingen 60 Sippen in die Berechnung ein.
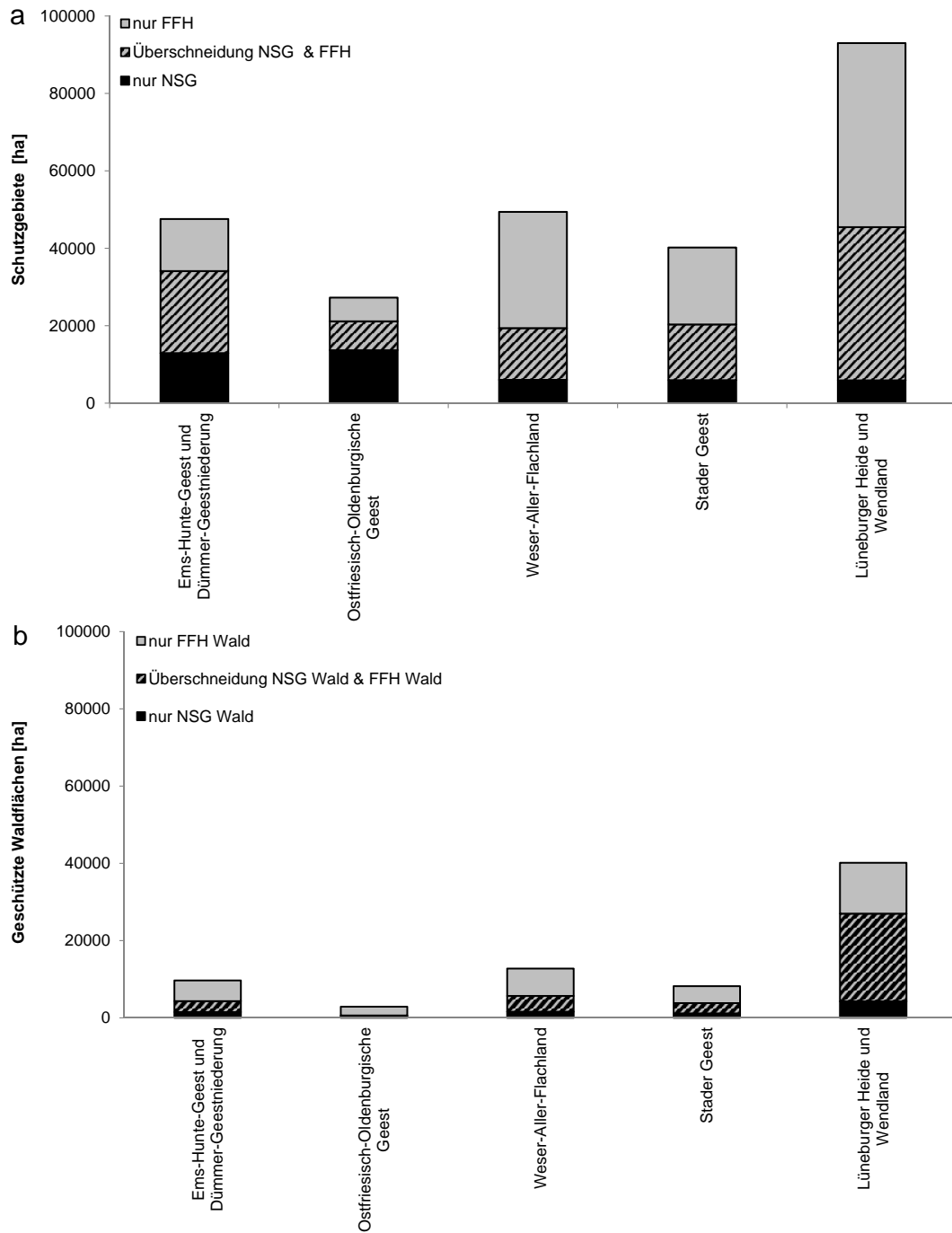
Abbildung 4.4: Schutzgebietsflächen (in Hektar) in den fünf Großregionen des niedersächsischen Tieflands unter Berücksichtigung der räumlichen Überlagerung von NSG und FFH-Gebieten. a) Gesamte Schutzgebietsfläche, b) Schutzgebiete in Wäldern.

Tabelle 4.2: Artenzahlen der RL-Arten bzw. RL-Waldarten, Ausdehnung der NSG und FFH-Gebiete und Ergebnisse der Analyse der Effektivität von Schutzgebieten für den Schutz von a) RL-Arten, b) RL-Waldarten. Effektivitätsanalyse: Ergebnisse der GLM-Analysen; n.s.: nicht signifikant, +/-: positiver/negativer Zusammenhang zwischen Schutzgebietsausdehnung und Artenreichtum, Signifikanzniveaus (* $p \leq 0{,}05$; ** $p \leq 0{,}01$; *** $p \leq 0{,}001$).

Großregion	NR	a) RL-Arten gesamt					b) RL-Waldarten						
		Artenzahl	NSG [ha]	NSG+FFH [ha]	Effektivitätsanalyse NSG	Effektivitätsanalyse NSG+FFH	Artenzahl 1.1	1.2	1.1+1.2	NSG Wald [ha]	FFH+NSG Wald [ha]	Effektivitätsanalyse NSG Wald	Effektivitätsanalyse NSG+FFH Wald
Ems-Hunte-Geest und Dümmer-Geestniederung	540	75	21	21	n.s.	n.s.	10	0	10	18	18	n.s.	n.s.
	544	148	687	1.918	+ ***	+ ***	18	2	20	111	1.229	n.s.	+ ***
	580	175	1.754	3.104	+ ***	+ ***	13	0	13	614	1.013	+ ***	+ ***
	581	111	96	264	n.s.	+ ***	12	0	12	6	139	n.s.	+ ***
	582	149	156	165	n.s.	n.s.	22	1	23	9	9	n.s.	n.s.
	584	134	18.929	20.53	+ ***	+ ***	10	2	12	460	508	+ ***	+ ***
	585	171	1.293	3.47	+ ***	+ ***	21	0	21	393	1.554	+ ***	+ ***
	586	148	2.017	6.52	+ ***	+ ***	10	1	11	302	1.469	+ ***	+ ***
	592	118	5.739	6.355	+ ***	+ ***	9	0	9	455	598	+ ***	+ ***
	593	122	673	999	+ ***	+ ***	14	1	15	283	551	+ **	+ ***
	594	144	921	1.842	+ ***	+ ***	15	1	16	526	981	+ ***	+ ***
	595	154	1.831	2.384	+ ***	+ ***	20	1	21	1.168	1.605	+ ***	+ ***
	600	156	11.66	12.173	+ ***	+ ***	8	0	8	334	372	+ ***	+ ***
Ostfriesisch-Oldenburgische Geest	601	79	2.828	2.879	+ ***	+ ***	3	0	3	20	21	n.s.	n.s.
	602	179	1.885	3.564	+ ***	+ ***	21	0	21	180	1.278	+ **	+ ***
	603	131	398	1.242	+ ***	+ ***	18	1	19	12	747	n.s.	+ ***
	604	75	635	3.669	+ *	+ ***	4	0	4	4	409	n.s.	n.s.
	605	86	3.73	3.738	+ ***	+ ***	2	0	2	58	58	n.s.	n.s.
	583	121	1.017	1.459	+ ***	+ ***	10	2	12	39	74	n.s.	n.s.
	620	97	66	1.34	+ **	+ **	3	1	4	3	11	+ ***	+ ***
	621	112	242	388	n.s.	+ ***	5	1	6	108	156	n.s.	+ ***
Weser-Aller-Flachland	622	261	5.82	12.826	+ ***	+ ***	32	4	36	407	1.036	n.s.	+ ***
	623	233	1.767	3.252	+ ***	+ ***	31	3	34	208	1.5	+ ***	+ ***
	624	104	149	305	+ ***	+ ***	22	2	24	128	252	+ ***	+ ***
	625	109	1.792	3.992	+ ***	+ ***	12	1	13	947	1.689	+ ***	+ ***
	626	228	5.367	9.628	+ ***	+ ***	20	1	21	3.132	3.951	n.s.	+ ***
	627	229	3.14	14.401	+ ***	+ ***	20	3	23	693	2.412	n.s.	+ ***
	628	88	13	1.816	n.s.	+ ***	13	2	15	5	1.695	+ ***	+ **
	630	135	985	1.693	+ ***	+ ***	13	0	13	305	610	+ ***	n.s.
	631	195	5.674	11.399	+ ***	+ ***	22	0	22	1.237	2.29	+ ***	+ *
Stader Geest	632	131	2.415	5.431	+ ***	+ ***	16	0	16	127	252	+ ***	+ **
	633	235	8.164	11.655	+ ***	+ ***	28	1	29	1.067	1.923	+ ***	+ ***
	634	208	3.098	10.016	+ ***	+ ***	25	2	27	1.104	3.115	n.s.	+ ***
	640	202	17.494	19.326	+ ***	+ ***	33	4	37	11.832	13.099	+ ***	+ ***
	641	250	18.494	33.262	+ ***	+ ***	28	3	31	8.909	12.623	+ ***	+ ***
Lüneburger Heide und Wendland	642	305	1.273	3.426	+ ***	+ ***	31	4	35	880	2.251	n.s.	n.s.
	643	198	1.533	3.401	+ ***	+ ***	31	2	33	1.028	1.656	+ ***	+ ***
	644	196	1.525	2.527	+ ***	+ ***	29	4	33	853	1.153	+ ***	+ ***
	860	285	3.761	8.179	+ ***	+ ***	33	2	35	2.867	4.302	+ ***	+ ***
	861	120	779	827	+ ***	+ ***	16	0	16	168	177	+ ***	+ ***
	876	273	618	22.061	- **	+ ***	16	6	22	447	4.898	+ ***	+ ***
Summe		490	140.439	257.447			52	8	60	41.447	73.684		
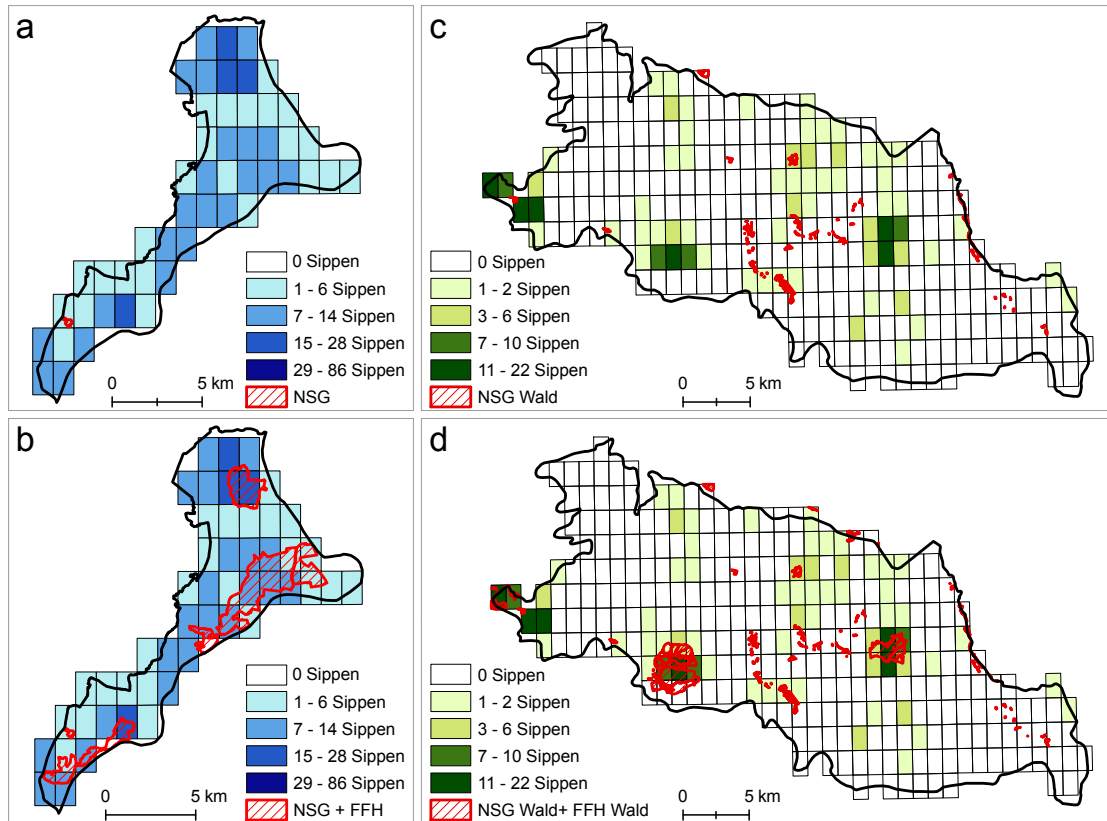
Abbildung 4.5: Beispiele für die Korrelation zwischen Schutzgebietsausdehnung und den Vorkommen der RL-Arten bzw. RL-Waldarten je Minutenfeld. In der Loccumer Geest (NR 628) überlagern sich die durch NSG geschützten Flächen und die Vorkommen der RL-Arten nicht (a), während sich unter Einbeziehung der FFH-Gebiete eine positive Korrelation zwischen Schutzgebietsausdehnung und den Vorkommen der RL-Arten zeigt (b). In den Burgdorf-Peiner-Geestplatten (NR 623) sind die Vorkommen der RL-Waldarten nicht durch NSG abgedeckt (c), während sich unter Einbeziehung der FFH-Gebiete eine positive Korrelation zwischen Schutzgebietsausdehnung und den Vorkommen der RL-Waldarten zeigt (d).

Diskussion

Die Verbreitung der seltenen und gefährdeten Gefäßpflanzenarten im niedersächsischen Tiefland zeigt ein differenziertes Bild (vgl. auch SCHMIEDEL et al., 2011). Es lassen sich Zentren der Artenvielfalt identifizieren, die im Sinne einer "Hotspots-Strategie" (MEYER et al., 2009) aus der landesweiten Perspektive als Naturschutz-Vorranggebiete angesehen werden müssen. Solche Hotspots finden sich vor allem im Wendland, im Westmünsterland und der Wesermünder Geest für RL-Arten bzw. der Oldenburger Geest für RL-Waldarten. Dagegen ist ein Großteil des westlichen Tieflands, wie etwa die Hunte-Leda-Moorniederung, das Bersenbrücker Land und das Bourtanger Moor, sehr artenarm.

Die Vorkommen der RL-Arten insgesamt sind nur in wenigen Landschaftsräumen des UG nicht durch Schutzgebiete gesichert (z. B. Ostmünsterland, Rahden-Diepenauer Geest). Die Vorkommen der RL-Waldarten sind zwar in einer geringeren Zahl der naturräumlichen Regionen effektiv durch NSG und FFH-Gebiete geschützt, allerdings ist die Zahl der RL-Waldarten in den Naturräumen mit nur geringfügiger Schutzgebietsabdeckung auch sehr klein. Gründe hierfür sind in diesen Regionen im geringen Anteil alter Waldstandorte und im insgesamt niedrigen Waldanteil zu suchen.

Im Allgemeinen verbessert sich die Schutzsituation für beide betrachteten Artengruppen bei Berücksichtigung von FFH-Gebieten. Dies zeigt, dass die erst in jüngerer Zeit ausgewiesenen FFH-Gebiete im niedersächsischen Tiefland künftig neben den NSG bei entsprechender Maßnahmenplanung eine wichtige Rolle für den Schutz seltener und gefährdeter Gefäßpflanzenarten spielen könnten. In diesem Zusammenhang sollte geprüft werden, ob der durch die Untersuchung bestätigte effektive Gebietsschutz auch in allen Fällen einen effektiven Artenschutz impliziert (vgl. RODRIGUES et al., 2004). Die Effektivität der Schutzgebiete für den Artenschutz ist in hohem Maße von den in den Schutzgebietsverordnungen bzw. Pflege- oder Managementplänen der Gebiete festgelegten Handlungsanweisungen abhängig. Insbesondere bei älteren NSG sind diese z. T. sehr unbestimmt formuliert (vgl. ENGEL et al., 2012) und für FFH-Gebiete derzeit noch in Arbeit. In vielen Fällen ist der Artenschutz zudem nicht vorrangiges Ziel bei der Schutzgebietsausweisung. Zukünftig sollten daher bei der Erarbeitung und Fortschreibung von Pflege- und Managementplänen in Schutzgebieten mit Hotspots zur dauerhaften Sicherung der bestehenden Populationen gezielt die Belange des Artenschutzes berücksichtigt werden.

Zu den Faktoren, die die Effektivität der Schutzgebiete einschränken können, zählen vor allem eine zu geringe Flächengröße der Gebiete bzw. ein hoher Grad an Isolierung: Sind die Schutzgebiete zu klein bzw. sind sie zu stark isoliert, können möglicherweise überlebensfähigen Populationen der schützenswerten Arten nicht erhalten werden (DEGUISE & KERR, 2006; JACKSON et al., 2009). Da ein Schutz größerer Flächen jedoch vielfach nicht umsetzbar ist, erscheint es in diesem Zusammenhang umso wichtiger, nicht allein auf die Wirkung von Schutzgebieten zu setzen, sondern im Rahmen einer "dualen Naturschutzstrategie" (JACKSON et al., 2009) auch Landschaftsbereiche außerhalb von Schutzgebieten so zu bewirtschaften, dass sie Populationen seltener und gefährdeter Arten nachhaltig als Lebensraum dienen können (DEGUISE & KERR, 2006; OTTE et al., 2008). In diesem Zusammenhang können beispielsweise die durch Eigenbindung ausgewiesenen Waldschutzgebiete im niedersächsischen Landeswald eine wichtige Funktion erfüllen (NIEDERSÄCHSISCHE LANDESFORSTEN, 2011).

Die für die Artengruppen voneinander abweichenden Ergebnisse zeigen, dass es sinnvoll ist, den Fokus nicht, wie in den bisher vorliegenden Studien zur Effektivität von Schutzgebieten (z. B. DEGUISE & KERR, 2006; JACKSON et al., 2009; VELLAK et al., 2009), lediglich auf die Gesamtheit der seltenen und gefährdeten, eine große
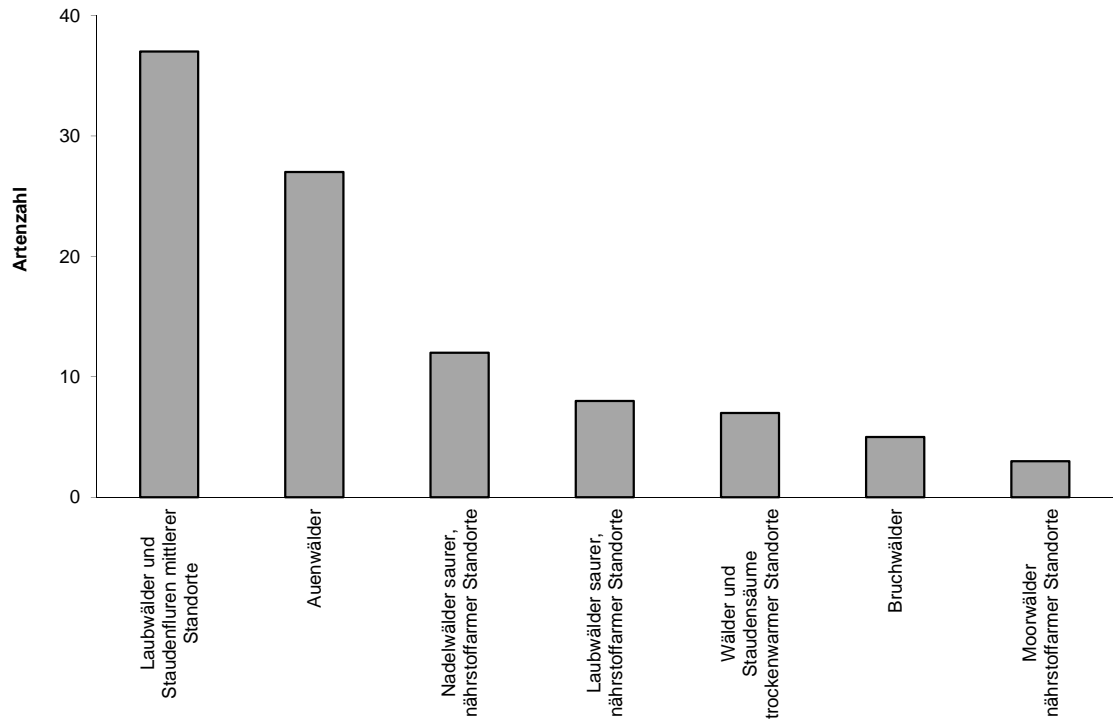
Abbildung 4.6: Habitatbindung der seltenen und gefährdeten Waldgefäßpflanzenarten des Tieflandes (bis zu drei Mehrfachzuordnungen pro Art).

standörtliche Amplitude repräsentierenden Gefäßpflanzenarten zu legen. Stattdessen kann die Analyse stärker differenziert und vergleichend auch solche Artengruppen einbezogen werden, die auf einen bestimmten Lebensraumtyp spezialisiert sind und somit eine geringere Standortamplitude aufweisen. Im vorliegenden Beispiel wurden hierfür die auf Wälder spezialisierten RL-Arten ausgewählt, da gerade Wäldern als der großflächig potenziell natürlichen Vegetation in Deutschland im Hinblick auf den Arten- wie auch den Prozessschutz eine große Bedeutung zukommt (BMU, 2007). Im niedersächsischen Tiefland zeigt ein Großteil der RL-Waldarten eine starke Habitatbindung an Laubwälder mittlerer Standorte (Waldmeister-Buchenwälder, Stieleichen-Hainbuchenwälder) und Auenwälder (Abb. 4.6). Von den 60 RL-Waldarten waren 43 an einen dieser beiden Habitattypen gebunden.

Die RL-Gefäßpflanzenarten können, obwohl sie nur einen Baustein des Gesamtökosystems darstellen, aufgrund ihrer Sensibilität gegenüber äußeren Einflüssen als gute Indikatoren für intakte Ökosysteme und die an diese gebundenen Arten und Lebensgemeinschaften angesehen werden.

Die fortlaufend durch das Pflanzenarten-Erfassungsprogramm erhobenen Verbreitungsdaten der RL-Arten sind eine sehr gute Grundlage für ein Monitoring der Effektivität der Schutzgebiete im UG. Wiederholende Untersuchungen in den folgen-

den Jahren werden zeigen, ob der in der vorliegenden Untersuchung belegte effektive Gebietsschutz langfristig auch zu einem effektiven Artenschutz führt.

Fazit für die Praxis

- Im niedersächsischen Tiefland liegen sowohl die Hotspots der Rote-Liste-Arten (RL-Arten) insgesamt als auch die der an Waldlebensräume gebundenen RL-Arten überwiegend in Schutzgebieten (NSG, FFH-Gebiete); die Ausweisung von FFH-Gebieten verbessert den Gebietsschutz für die RL-Arten in den meisten naturräumlichen Regionen erheblich. Wiederholungen der Untersuchungen werden zeigen müssen, ob der effektive Gebietsschutz auch zu einem effektiven Artenschutz führt.

- Bei der Erarbeitung und Fortschreibung von Pflege- und Managementplänen in Schutzgebieten mit Hotspots sollten künftig gezielt die Belange des Artenschutzes berücksichtigt werden, um die bestehenden Populationen von RL-Arten dauerhaft zu sichern.

- Die aufgezeigten Methoden der Hotspot-Analyse sollten stärker im Sinne einer "dualen Naturschutzstrategie" genutzt werden, um wichtige Populationen von RL-Arten auch außerhalb von Schutzgebieten durch nachhaltige Bewirtschaftungsformen zu erhalten.

- Rasterbasierte Verbreitungsdaten von RL-Arten sollten in regelmäßigen Zeitabständen aktualisiert werden, um sie als Grundlage eines Monitorings der Effektivität von Schutzgebieten für den Artenschutz einsetzen zu können.

Dank

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5

The influence of landscape fragmentation on richness patterns of vascular plants

Abstract

Landscape fragmentation has been identified as a major threat to biodiversity world-wide. Several landscape metrics have been developed to quantify the extent of fragmentation, of which the *Effective Mesh Size Index* (m_{eff}) is one of the most widely used. However, its relevance for biodiversity has been rarely tested. We analysed the explanatory power of m_{eff} for richness patterns of different groups of vascular plant species (all species, and species groups by residence and threat status) in Lower Saxony, Germany, by using a grid of 1386 landscape units. Since we assumed species richness to be influenced by abiotic conditions and spatial autocorrelation, we used variation partitioning to separate the effects of these variables from that of fragmentation. We tested five types of m_{eff} based on various fragmentation geometries of which two were identified as relevant for species richness.

We found that m_{eff} had a large effect on richness of neophytes and a slightly smaller effect on that of archaeophytes whilst the richness of native species was only slightly affected, and threatened species were not affected. All species groups, except threatened species, showed a negative correlation with m_{eff}, i.e. richness was highest in highly fragmented and lowest in less fragmented grid cells.

We conclude that m_{eff} is a meaningful tool to explain richness patterns of vascular plant diversity, if relevant fragmentation geometries are chosen. Our approach may therefore help future studies to determine correct fragmentation geometries to use with m_{eff} and may facilitate the unravelling of fragmentation impacts on the landscape-scale.

Keywords

Introduction

The term 'landscape fragmentation' describes both the process of the subdivision of a large continuous patch into smaller and more isolated fragments, as well as the state of a landscape (Forman, 1995). As soon as the effect of fragmentation on the biota is concerned, frequently also the term 'habitat fragmentation' is applied.

Besides general habitat loss and decline in habitat quality, fragmentation has been identified as a major threat to biodiversity worldwide (Jaeger, 2000; Young & Clarke, 2000; Honnay et al., 2005; Kuussaari et al., 2009; Krauss et al., 2010). In Central Europe, habitat change accelerated in course of urbanisation and industrialisation in the late 19th and early 20th century (Tscharntke et al., 2005; Krause & Culmsee, 2013) and has been reinforced by the advent of the common agricultural policy of the EU since the 1950s (Stoate et al., 2001; Krause et al., 2011; Leuschner et al., 2013). However, landscape development in Central Europe currently does not appear to be following a clear trend, and we observe complex and diverse landscapes as a result of multi-layered historical and current land use practices (Vos & Meekes, 1999; Culmsee, 2013). This, in turn, results in differences in the fragmentation intensity between neighbouring landscapes and in a more or less strong gradient of fragmentation on the regional and supra-regional scales.

The analysis of landscape fragmentation is an important aspect of landscape ecological research (Haila, 2002). Many different indices for the quantification of landscape fragmentation have been introduced, such as the *Number of Undissected Areas*, *Bowens Landscape Dissection Index* (Bowen & Burgess, 1981), *Splitting Index* (Jaeger, 2000) and *Effective Mesh Size* (Jaeger, 2000). All of these indices provide a spatially differentiated assessment of the fragmentation of a region (Jaeger, 2000). By evaluating these four plus additional four indices, Jaeger (2000, p. 127) found that *Effective Mesh Size* (m_{eff}) is the most appropriate fragmentation measure due to its "mathematical characteristics and its intuitive interpretation". The index quantifies the probability that two randomly chosen points in a study area are connected (Girvetz et al., 2008) and has frequently been used to quantify landscape fragmentation, e. g. in Europe (EEA, 2011), Germany (Jaeger et al., 2001; Esswein & Schwarz-von Raumer, 2004; Walz, 2005; Esswein & Schwarz-von Raumer, 2006), Italy (Moser et al., 2007), Switzerland (Jaeger et al., 2008), the USA (Girvetz et al., 2008) and China (Li et al., 2010).

In Germany, m_{eff} is one of two indicators used by the government to quantify landscape fragmentation for environmental reporting (BMU, 2007). However, despite its frequent application and the fact that the ecological relevance of m_{eff} has been proposed by many researchers (Jaeger, 2000; Jaeger et al., 2008; Girvetz et al., 2008), its effect on biodiversity has been rarely analysed (but see Li et al., 2010). Strand et al. (2007) suggested that studies investigating the direct effect of landscape fragmentation (expressed in terms of fragmentation metrics) on biodiversity should be undertaken, because indicators that quantify landscape fragmentation are more useful if they can be directly linked to possible impacts on species diversity and distributions.

A multitude of man-made landscape elements, including urban, industrial and traffic infrastructure, can cause landscape fragmentation, resulting in patchy landscapes with generally small remaining natural or semi-natural habitat fragments. Depending on the specific biological traits of the species, very different landscape elements, including (near-) natural ones, can be considered fragmentation elements (see Girvetz et al., 2008; EEA, 2011).

The fragmentation of habitats may result in increasing isolation of plant or animal subpopulations that inhabit the patches. Habitats colonised by different subpopulations of the same species are isolated if genetic exchange is constrained because the area between the patches is too large, or too impermeable, to be overcome by pollination or dispersal events (Oostermeijer et al., 1994; Poschlod et al., 1997). The loss of genetic variation, in turn, may hamper the species' ability to respond to changing environmental conditions (Booy et al., 2000) and the populations may thus become threatened with extinction. Numerous studies examined the effect of habitat fragmentation on plant and animal assemblages at the local and landscape scales (see for an overview Debinski & Holt, 2000; Fahrig, 2003). Frequently, the size of and the isolation of or connectivity between patches were used as measure of fragmentation (e.g. Helm et al., 2006; Cousins et al., 2007; Kolb, 2008; Zimbres et al., 2013), yet none of these studies applied the *Effective Mesh Size Index*.

The effects of landscape fragmentation on species richness may vary depending on the species groups considered (Ewers & Didham, 2006; Rodriguez-Loinaz et al., 2012). Certain species, particularly those spreading along linear landscape elements or associated with man-made habitats, may benefit from landscape fragmentation (Deutschewitz et al., 2003; Honnay et al., 2003; Dark, 2004; Nobis et al., 2009; Botham et al., 2009). In contrast, threatened species may be adversely affected by fragmentation events (Ewers & Didham, 2006). However, especially species that exhibit long life cycles and small population sizes show delayed extinctions following habitat loss and fragmentation (Kuussaari et al., 2009; Krauss et al., 2010), which at first can hamper the detection of these negative effects. This phenomenon is known as 'extinction debt' (Tilman et al., 1994). However, the evidence for the existence of an extinction debt in vascular plants across Europe is ambiguous (reviewed in Cousins, 2009).

The overall objective of our study was to investigate whether the frequently used *Effective Mesh Size Index* (m_{eff}) is a suitable measure to explain vascular plant species richness on the landscape scale. We used the state of Lower Saxony, Germany, as a model region as it hosts a high diversity of landscapes differing in habitat composition and structural complexity. Therefore, we expected a long gradient in fragmentation across the study area. Furthermore, the area is particularly suitable due to the availability of extensive data collected in a state-wide mapping program on the distribution of vascular plants with a resolution of c. 30 km^2 (NLWKN 1982-2003, Garve, 2007). Thus, high quality data were available for a region that

is more or less homogeneous in climate and bio-geographical history, so that we considered the same regional species pool throughout the studied landscapes. We used different subsets of fragmentation geometries (FGs) and also different subsets of the regional plant species pool, dependent on the residence and threat status of the species, in order to investigate the underlying patterns in plant species distributions as responses to different fragmenting landscape elements.

Specifically, we hypothesised that:

1. Landscape fragmentation, expressed as *Effective Mesh Size* (m_{eff}), varies in space, which results in a gradient in the degree of fragmentation within the sample of landscape units used in this study.
2. The choice of fragmentation geometries that are used for the calculation of m_{eff} is of major importance for explaining plant species richness patterns on the landscape scale.
3. Plant species richness varies between landscape units along the fragmentation gradient, with the effect size differing between different groups of species dependent on their residence (natives, archaeophytes and neophytes) and threat status (threatened vs. non-threatened).

Methods

Study area

The study was carried out in Lower Saxony, Germany (c. 47,500 km^2, Fig. 5.1). As a result of the variety in relief and bedrock, colonisation history and current land use are diverse across north-western Germany (Behre, 2008). Lower Saxony harbours a relatively large number of different habitats, which include natural coastal and forest-dominated landscapes, structured and homogeneous agricultural landscapes, and densely populated areas (Gharadjedaghi et al., 2004).

The coastal area in the north is dominated by Holocene marine deposits, while the lowlands were formed during the Pleistocene. The landscape of the uplands is shaped by Mesozoic and Palaeozoic bedrocks. From the coast in the north to the uplands in the southeast, elevation ranges from 0–971 m a.s.l.. The climate gradually changes eastwards from oceanic (mean annual precipitation up to 900 mm) to suboceanic-subcontinental (c. 550 mm; NLfB, 1997).

Following the grid system of topographical maps, the study area was subdivided into a regular grid of 1386 cells sized c. 5.5 × 5.5 km^2 (30 km^2; Fig. 5.1). In our analysis, we omitted the 353 border cells only partially occurring in the study area.
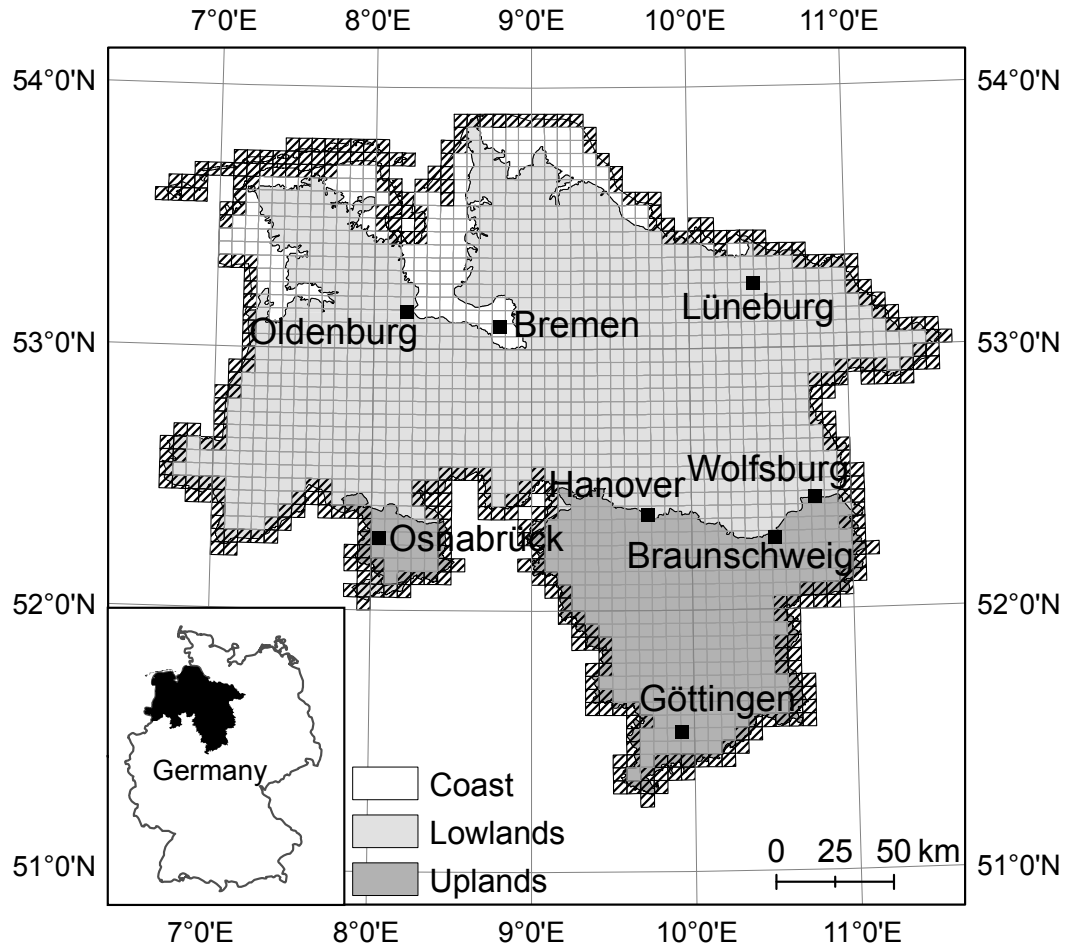
Figure 5.1: The state of Lower Saxony, Germany, with its three major landscape components: coast, lowlands and uplands. The 1386 grid cells (size c. $5.5 \times 5.5\,\mathrm{km}^2$ or $30\,\mathrm{km}^2$) display the basic mapping units of the vascular plant survey program and represent the landscape units for which landscape fragmentation was assessed. Hatched grid cells were not considered in this study, because they are not completely terrestrial or they are situated at the state's borders and thus information on plant occurrences were incomplete. Maps with Transverse Mercator projection, Germany Zone 3 ("Deutsches Hauptdreiecksnetz").

Plant species distribution data

Species occurrences were derived from the data base of the plant survey of the state of Lower Saxony (NLWKN, Hanover, Germany; period I from 1982 to 1992; period II from 1993 to 2003; period III from 2004 onwards). As the third fieldwork period is still underway, we used only data from 1982-2003 (GARVE, 2007; data base updated on 15 December 2008). The data base holds information about the presence and distribution of 2708 vascular plant species occurring in Lower Saxony (GARVE, 2007).

Table 5.1: Number of vascular plant species in Lower Saxony, Germany, grouped by their residence and threat status.

		Threat status			Sum of residence status
		Non-threatened	Threatened	Unspecified	
	Native species	703	482	112	1297
Residence	Archaeophytes	103	46	0	149
status	Neophytes	134	9	0	143
	Unspecified	4	1	0	5
	Sum of threat status	944	538	112	**1594**

Species were assigned to different groups according to their residence and threat status (Table 5.1). In the designation of the residence status of the plant species we followed KLOTZ et al. (2002). Native species were distinguished from archaeophytes and neophytes. Within the non-native species, archaeophytes became established prior to and neophytes in the 16th century or later. In the designation of threat status we followed GARVE (2004), whereby threatened species (including the Red List categories 1, 2, 3, G and R) were distinguished from non-threatened species. Five species could not be assigned to a certain floristic status, while 112 species lacked information about their threat status (hereafter 'not specified' species). Subspecies were assigned to the species level and species to the respective aggregate found in the data base. In total, 1594 species remained that were considered in the analysis.

For each species group, species richness (based on presence/absence data) was calculated for each of the 1386 grid cells. Distribution patterns of the selected species groups were described in detail by SCHMIEDEL et al. (2011).

Quantification of landscape fragmentation

Landscape fragmentation analysis was conducted with the 1386 grid cells used by the plant species monitoring program (Fig. 5.1). Each grid cell was considered a landscape unit. Landscape elements were derived from a digital landscape model (DLM) of Lower Saxony, scale 1:25,000 (LGN, 2008), and summarised to eight major habitat types that represent suitable habitat formations for vascular plants: forest, grassland, cropland, water bodies, urban, industrial and traffic areas, and other habitats (e. g. islands, tree rows, bogs). Linear elements represented in the DLM as polylines (roads, rivers) were buffered by using either standard values (roads; FGSV, 1996) or information provided by the dataset itself (rivers) in order to convert them to polygons. Data were converted by using Hawth's Analysis Tools (BEYER, 2004) and combined with the other polygon features to a non-overlapping feature layer by using ArcGIS-ArcInfo 9.2 (ESRI INC., 2007).

Habitat types that were assumed to increase fragmentation of habitats suitable for different groups of vascular plant species were classified into five types of fragmentation geometries (FG1-FG5; Table 5.2). Starting with FG1, fragmenting landscape elements were successively added to the following FGs. FG1 included the most artificial land cover classes of traffic and urban-industrial areas. FG2 additionally included water bodies. Similar FGs have frequently been used in fragmentation studies and will thus allow for comparisons. The subsequent FG3-FG5 were built by adding further land cover classes (FG3: croplands, FG4: grasslands, FG5: other habitats) to the former FGs, thus setting a gradient of decreasing human impact (Fig. 5.2). FG5 included all landscape elements except forests which were rated as (semi-)natural landscape elements, because forests represent the potential natural vegetation of temperate Europe (Bohn & Neuhäusl, 2000/2003).

Landscape fragmentation was quantified by the *Effective Mesh Size Index* (m_{eff}). The index denotes the size of patches when the landscape is divided into S areas (each of the same size) with the same degree of landscape division as obtained for the observed cumulative area distribution (McGarigal, 2014). Thus m_{eff} is an ecologically relevant metric that quantifies landscape fragmentation based on the probability that two randomly chosen points in a landscape unit are located in the same non-fragmented patch (Jaeger, 2000). The probability is multiplied by the total area of the landscape unit (in m^2) and finally gives the value of m_{eff}. The *Effective Mesh Size Index* was calculated following McGarigal (2014) using the following formula:

$$m_{eff} = \frac{\sum\limits_{j=1}^{n} a_{ij}^2}{A} \left(\frac{1}{10,000}\right) \tag{5.1}$$

where n is the number of patches, a_{ij} is the size of patch ij and A is the total area of the landscape. Values of m_{eff} are given in hectares (ha) by using the factor of 1:10,000 to convert m^2 into ha.

For the calculation of m_{eff} based on different FGs, subsequently called MeshFG1-MeshFG5 (Table 5.2), the FGs derived from the DLM vector data were transformed to raster format, and m_{eff} of the non-fragmented parts of the landscape was calculated for each grid cell using Fragstats 3.3 (McGarigal et al., 2002). We applied the CUT procedure proposed by Jaeger (2000), and not the CBC procedure as proposed by Moser et al. (2007), to obtain spatially explicit values for m_{eff} that could be correlated with the grid-based distribution data on plant species.

A Shapiro-Wilk normality test was performed on all m_{eff} variables to test them for normal distribution with the R software, version 2.15.2 (R Development Core Team, 2012). Furthermore, a Wilcoxon signed-rank test was used to test for significant differences between the m_{eff} types.

Legend [a]

■ traffic areas
▤ industrial areas
□ urban areas
□ waterbodies
▦ croplands
▨ grasslands
▩ forests
▨ others

Legend [b] - [f]

■ fragmentation geometry
□ non-fragmented landscape

0   1.5   3km

N
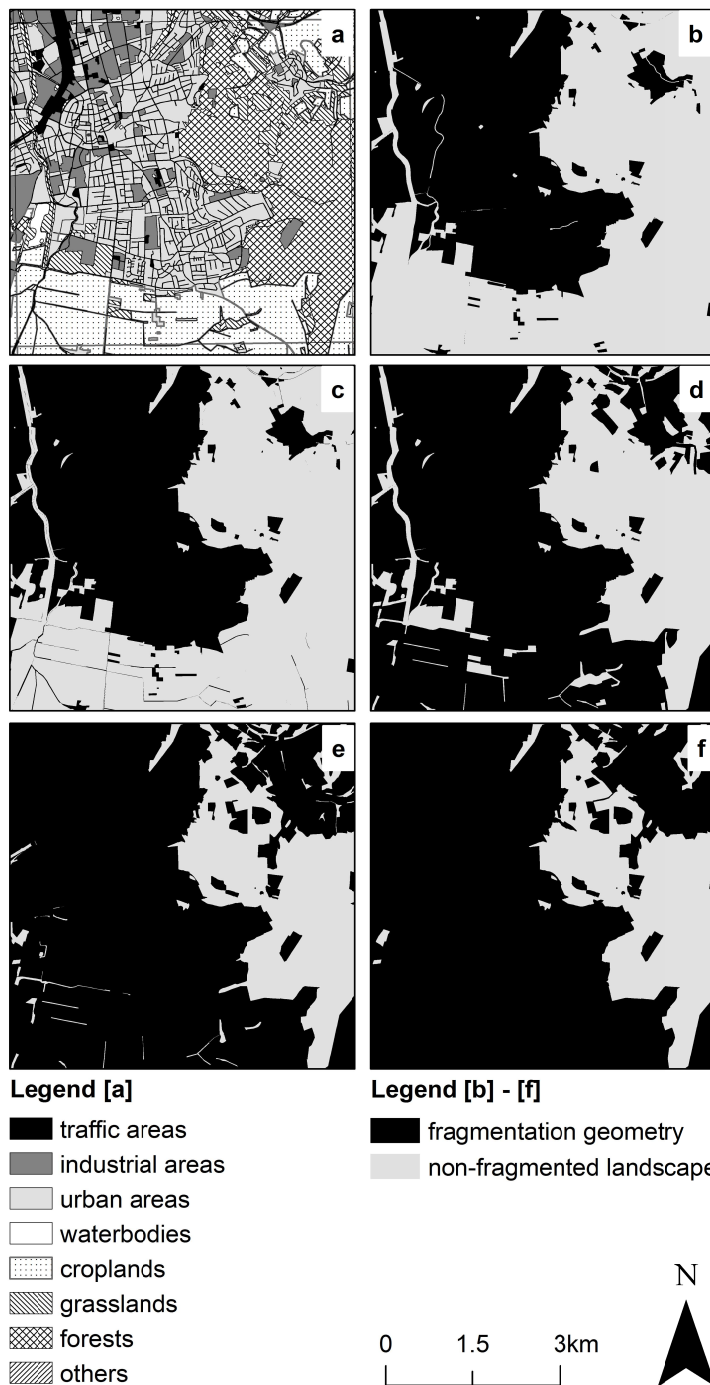
Figure 5.2: Example of one of 1386 landscape units in Lower Saxony, Germany (grid cell no. 44254, Göttingen, sized c. $5.5 \times 5.5\,\text{km}^2$ or $30\,\text{km}^2$). (a) The digital landscape model (ATKIS-DLM, scale 1:25,000; LGN, 2008) was translated to five different fragmentation geometries, FG1-5, (b) FG1 (traffic and urban-industrial areas), (c) FG2 (FG1 + water bodies), (d) FG3 (FG2 + croplands), (e) FG4 (FG3 + grassland), (f) FG5 (all landscape elements except forested areas).

Table 5.2: Landscape fragmentation variables (*Effective Mesh Size*, m_{eff}, based on different fragmentation geometries, FG1–FG5), spatial and environmental co-variables used in this study. n/a = not available. All m_{eff} types significantly differed from each other (Wilcoxon signed-rank test, p ≤ 0.001).

Name	Definition	Unit	Source	Mean ± SD	Range
Effective Mesh Size (m_{eff})					
MeshFG1	m_{eff} based on FG1: traffic and urban-industrial areas	ha	DLM[b]	1912±782	9–3196
MeshFG2	m_{eff} based on FG2: traffic and urban-industrial areas, water bodies	ha	DLM[b]	1430±819	0–3172
MeshFG3	m_{eff} based on FG3: traffic and urban-industrial areas, water bodies, croplands	ha	DLM[b]	412±565	0–3114
MeshFG4	m_{eff} based on FG4: traffic and urban-industrial areas, water bodies, croplands, grassland	ha	DLM[b]	195±451	0–3019
MeshFG5	m_{eff} based on FG5: all landscape elements except forested areas	ha	DLM[b]	152±381	0–2976
Spatial co-variables					
Ax1–Ax787	Significant spatial axes	n/a	n/a	n/a	n/a
Environmental co-variables					
ElevMn	Mean elevation	m a.s.l.	DEM[c]	65±84	-0.70–687.10
HeatlMn	Mean heat load index	n/a	DEM[c]	0.83±0.01	0.75–0.84
TempMn	Mean annual temperature	°C	DWD[d]	8.6±0.4	5.3–9.4
PrecipMn	Mean annual precipitation	mm	DWD[d]	743±90	562–1498
Snowdrop	Common Snowdrop (*Galanthus nivalis*) flowering	Day of year[a]	DWD[d]	56±6	29–85
Sun	Sunshine duration per year	h	DWD[d]	1450±54	1268–1615
SoilSHDI	Diversity of soil types (Shannon's Diversity Index, SHDI)	n/a	BÜK50[e]	1.18±0.37	0.00–2.07
Calc1	Surface ratio of $CaCO_3$ soil content: no $CaCO_3$ content	ha	BÜK50[e]	832±802	0–3205
Calc2	Surface ratio of $CaCO_3$ soil content: medium $CaCO_3$ content	ha	BÜK50[e]	958±986	0–3208
Calc3	Surface ratio of $CaCO_3$ soil content: high $CaCO_3$ content	ha	BÜK50[e]	528±6258	0–3065
Moist1	Surface ratio of soil moisture: dry	ha	BÜK50[e]	11±81	0–1649
Moist2	Surface ratio of soil moisture: mesic	ha	BÜK50[e]	738±699	0–3030
Moist3	Surface ratio of soil moisture: moist	ha	BÜK50[e]	2932±482	0–3223
Moist4	Surface ratio of soil moisture: wet	ha	BÜK50[e]	128±355	0–2967
Moist5	Surface ratio of soil moisture: influenced by backwater	ha	BÜK50[e]	9±50	0–672

[a] Day of year = day since the beginning of the year; [b] Digital landscape model of Lower Saxony (ATKIS-DLM, scale 1:25,000, LGN, 2008); [c] Digital elevation model for Lower Saxony (ATKIS-DEM, grid of 5 m × 5 m resolution, LGN, 2008); [d] Climate data of the German Meteorological Service (grid of 1000 m × 1000 m resolution, DWD 1961-1990); [e] Digital soil map of Lower Saxony (BÜK 50, scale 1:50,000, LBEG, 2003).

Accounting for environmental co-variables and spatial autocorrelation

We assumed that both landscape fragmentation and environmental factors influence species richness. Thus, to account for the explanatory value of the abiotic environment, supplementary environmental data were derived from various digital data sources (Table 5.2). Mean elevation was taken from the digital elevation model (DEM) of Lower Saxony (LGN, 2008). DEM data were further used to calculate mean heat load (McCune & Keon, 2002, eq. 3) per grid cell by using the ArcScript 'Heat load index' (Parks, 2004) in ArcView GIS 3.3 (ESRI Inc., 2002). Information on soil moisture, $CaCO_3$ content of soils and soil types were derived from the digital soil map of Lower Saxony (BÜK 50, LBEG, 2003). For each grid cell, the surface ratio of five soil moisture and three $CaCO_3$ soil content classes were calculated. Based on the distribution and cover of 27 soil types, for each grid cell the Shannon Diversity Index (SHDI) was computed in Fragstats 3.3 after vector data were transformed to raster format. Information on climatic and phenological conditions (annual mean precipitation and temperature, start of snowdrop flowering) per grid cell were derived from digital data of the German Meteorological Service (DWD, 1961-1990) and the average of each variable was calculated per grid cell. The 15 environmental variables that were used in the analysis were uncorrelated (Spearman's rank correlation coefficient ($|\rho| \leq 0.7$).

To account for spatial autocorrelation in the dataset, PCNM technique was applied using the following steps (cp. Borcard & Legendre, 2002; Borcard et al., 2004): (1) We constructed a matrix of Euclidean distances between the 1386 grid cells (landscape units) based on the x- and y-coordinates (centroids) of the cells. (2) We defined a threshold (smallest distance keeping all cells connected, i. e. 8036 m) below which the Euclidean distances were kept as measures and above which all distances were replaced by a value four times larger than the threshold (i. e. 32,144 m). (3) We computed the principal coordinates (correction of negative eigenvalues) of the modified matrix using PrCoord, version 1.0 (implemented in CANOCO 4.5, Braak & Šmilauer, 2002), and (4) we reduced the number of the remaining 787 spatial variables to those that were significant for the richness of the species groups (Monte Carlo Permutation test under full model, 99,999 permutations; Holm-correction, p $\leq$ 0.05; Legendre & Legendre, 2012).

Statistical analysis

In order to determine the explanatory power of m_{eff}, the environmental factors and spatial axes for species richness patterns across the 1386 grid cells, variation partitioning was applied by using the 'varpart' function ('vegan' package, Oksanen et al., 2012) in R (cp. Fig. 5.3). Separate analyses were conducted for each species group (Table 5.1) in combination with each type of m_{eff} (Table 5.2). Species numbers of threatened species were square root transformed because they showed a

skewed distribution. Significance of all testable fractions was tested using 'rda' and 'anova.cca' functions in 'vegan'.

To determine the direction of the relationship of species richness and the m_{eff} types, Kendall's τ was calculated using R. This correlation analysis was performed for MeshFG1 and MeshFG2 only because variation partitioning showed that the other types of m_{eff} explained only a negligible proportion of the species richness of all groups.

Results

Variation in *Effective Mesh Size* (m_{eff}) based on different fragmentation geometries across landscapes

Values of m_{eff} calculated for the 1386 landscape units (grid cells) across the study area showed large variation both in range between landscapes and between fragmentation geometries (Table 5.2). All m_{eff} types (MeshFG1-MeshFG5) showed a large range from entirely fragmented ($m_{eff} = 0$ ha) to entirely non-fragmented (m_{eff} ~3,000 ha) landscape units, the latter corresponding to the total size of a landscape unit of c. 30 km^2. The mean value of m_{eff} was highest when only traffic and urban-industrial areas were used (MeshFG1, $m_{eff} = 1{,}912$ ha ± 782 SD) indicating a relatively low fragmentation impact, and was lowest when all landscape elements with the exception of forests were used (MeshFG5, $m_{eff} = 152$ ha ± 381 SD). From MeshFG1 to MeshFG5, mean m_{eff} gradually decreased when additional fragmentation geometries were successively included, whereas each of them had an additional impact as fragmenting landscape element against the previous (Wilcoxon signed-rank test, p<0.001 for all pairs of MeshFG1-MeshFG5). None of the m_{eff} variables was normally distributed (Shapiro-Wilk normality test, p≤0.05). Values of MeshFG1 showed a slightly right-skewed distribution, i.e. larger values of m_{eff} were more frequent than small values (Fig. 5.4a), while values of m_{eff} for MeshFG2 were slightly left-skewed (Fig. 5.4b).

Figure 5.3: Fractions obtained from variation partitioning based on three variables (Venn diagram). Fractions [a] to [c] correspond to the pure effect of variables a, b and c, fractions [d] to [f] correspond to the variation explained by variables a and b, b and c, and a and c, respectively. Fraction [g] corresponds to the variation explained by all three variables together whereas [h] describes the unexplained variance (residuals).
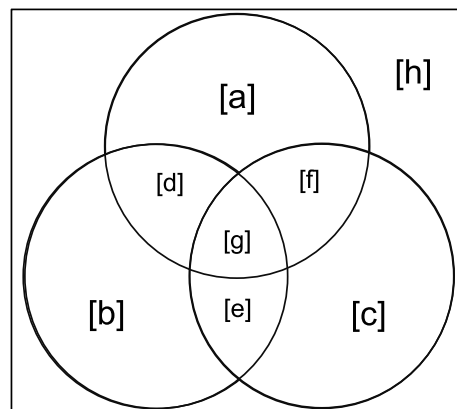
The remaining m_{eff} types (Fig. 5.4c-e) all showed a strongly left-skewed distribution with the majority of landscape units exhibiting a mesh size <200 ha.
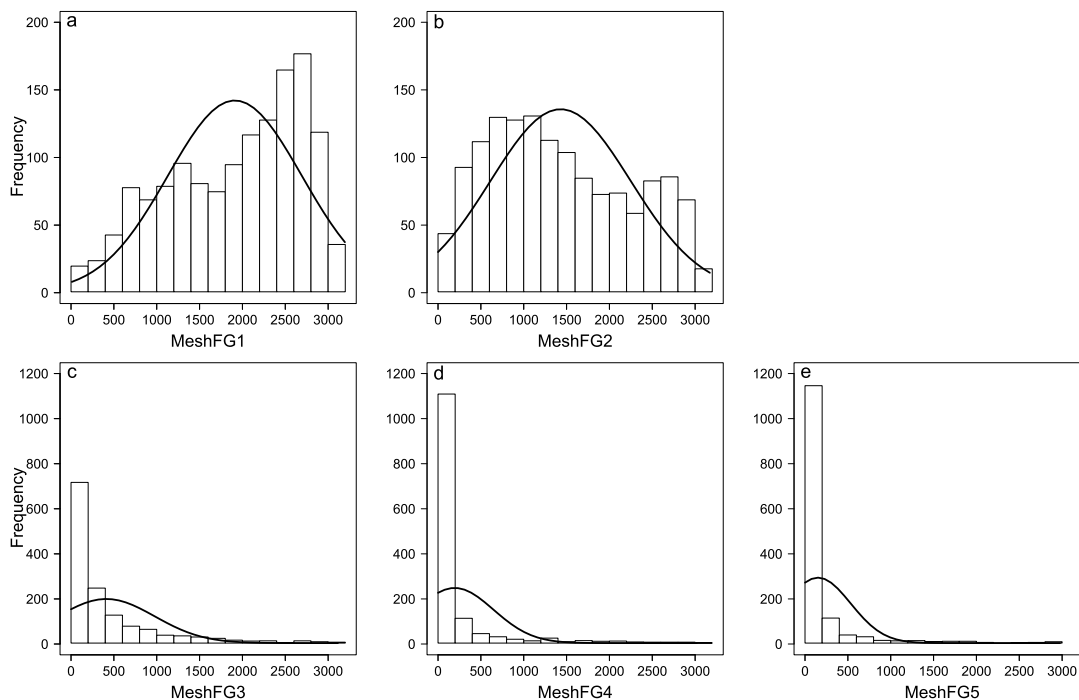
Figure 5.4: Frequency distribution of the five m_{eff} types (a-e, MeshFG1-MeshFG5) for the sample of 1386 landscape units (grid cells) in Lower Saxony, Germany. Maximum value of m_{eff} was c. 3,000 ha which corresponds to the size of a fully non-fragmented landscape unit of 30 km^2. None of the distributions were normally distributed (Shapiro-Wilk normality test, p≤0.05).

The explanatory power of *Effective Mesh Size* (*m*_eff), based on different fragmentation geometries, versus other environmental factors for species richness-patterns on the landscape-scale

Depending on fragmentation geometries (FGs) used for the calculation of m_{eff} and on the species group considered, the variation partitioning revealed large differences in the explanatory power of m_{eff} versus other environmental factors for vascular plant species richness patterns on the landscape scale (Table 5.3).

Accounting for the total of 1594 plant species, environmental variables explained 14 % (±0 SD) and spatial variables 7 % (±0 SD) of the variance in species richness (Table 5.3a). The response to landscape fragmentation largely varied in dependence on the mesh size variable (MeshFG1-MeshFG5). While m_{eff} types that used only traffic, urban-industrial areas and water bodies had a considerable influence of 5 % (MeshFG1) and 6 % (MeshFG2), the other m_{eff} variables, which additionally included

other man-made habitats such as agricultural areas and grasslands (MeshFG3 – MeshFG5), influenced species richness by <1%. Similar patterns were observed across all species groups. On average, 52% (± 3 SD) of variation in all species remained unexplained, and similar figures were found in the other species groups, with a range of 42% (± 4 SD) of unexplained variation in archaeophytes (Table 5.3c) to 61% (± 6 SD) in threatened species (Table 5.3f).

When distinguishing species groups according to the residence status, the general patterns observed in all species were more or less repeated in the partition of the largest group of 1297 native species, but with slightly lower explanatory power of the m_{eff} variables (Table 5.3b). Spatial distribution patterns of natives were also relatively clustered (explanation of the spatial fraction was 8% ± 0 SD). However, the smaller groups of archaeophytes (149 sp.) and neophytes (143 sp.) showed different patterns. Archaeophytes were more strongly related to landscape fragmentation (MeshFG1 and MeshFG2: 7%) and especially to environmental conditions (26% ± 2 SD; Table 5.3c), indicating a relatively narrow environmental niche width of these species. Neophytes were most strongly associated with fragmented landscapes (Table 5.3d). M_{eff} explained a large proportion of species richness, namely 15% (MeshFG1) and 16% (MeshFG2), while the environmental factors were of lower importance (10% ± 2 SD). Spatial autocorrelation was generally low in both non-native groups (<2% of the total explained variation; Table 5.3c and d).

With respect to threat status, the large group of 944 non-threatened species followed the patterns observed in all species (Table 5.3e). In contrast, threatened species showed spatially distinguished incidence patterns (explained variation of spatial axes: 7% ± 0 SD) and their occurrences were mostly determined by environmental variables (19% ± 1 SD), while they were not affected (m_{eff} <1%) by landscape fragmentation (Table 5.3f).

With the exception of the threatened species group that showed no specific trend, all species groups were negatively correlated with MeshFG1 and MeshFG2 (Table 5.4), which means that species richness increased with increasing landscape fragmentation (i. e. small m_{eff} values).

Discussion

Variation in landscape fragmentation based on m_{eff}

The values of m_{eff} obtained for the 1386 landscape units in our study varied between 0 and 30 km². Thus, we identified a large gradient from entirely fragmented to entirely non-fragmented landscape units. Because we applied the CUT procedure to calculate m_{eff}, maximum values of the index were restricted by the size of the study units. The latter also applies to other studies on m_{eff} (see Table 5 for an overview), which used differently sized reference units for the calculation of the index. Therefore, conclusions from comparisons of our results with values of m_{eff} obtained in these

Table 5.3: Results of variation partitioning (explained variation) for vascular plant species richness in Lower Saxony, Germany, in the groups of (a) all species, (b) natives, (c) archaeophytes, (d) neophytes, (e) non-threatened species, and (f) threatened species (numbers given in brackets), in relation to landscape fragmentation expressed as *Effective Mesh Size* (MeshFG1–MeshFG5) based on different sets of fragmentation geometries (FG1–FG5), spatial axes as measures for spatial autocorrelation and environmental factors (cp. Table 5.2). Significance was tested by Monte Carlo Permutation test (999 permutations) and refers to the partitions [a], [b] and [c] (cp. Fig. 5.3); significant results are given in bold letters; ***, $p \leq 0.001$; **, $p \leq 0.01$, *, $p \leq 0.05$.

	MeshFG1	MeshFG2	MeshFG3	MeshFG4	MeshFG5
(a) All species (1594 sp.)					
[a] M_{eff}	**0.048** ***	**0.061** ***	**0.004** **	**0.004** **	**0.007** ***
[b] Spatial axes	**0.067** ***	**0.067** ***	**0.073** ***	**0.071** ***	**0.074** ***
[c] Environmental variables	**0.125** ***	**0.145** ***	**0.131** ***	**0.139** ***	**0.141** ***
Shared variation [d], [e], [f], [g]	0.264	0.242	0.252	0.245	0.241
[h] Residuals	0.497	0.484	0.541	0.540	0.537
b) Natives (1297 sp.)					
[a] M_{eff}	**0.024** ***	**0.037** ***	0.001	**0.002** *	**0.005** ***
[b] Spatial axes	**0.073** ***	**0.073** ***	**0.08** ***	**0.077** ***	**0.079** ***
[c] Environmental variables	**0.121** ***	**0.138** ***	**0.123** ***	**0.126** ***	**0.129** ***
Shared variation [d], [e], [f], [g]	0.261	0.243	0.252	0.251	0.247
[h] Residuals	0.521	0.509	0.544	0.543	0.540
(c) Archaeophytes (149 sp.)					
[a] M_{eff}	**0.067** ***	**0.069** ***	**0.013** ***	**0.011** ***	**0.011** ***
[b] Spatial axes	**0.011** ***	**0.016** ***	**0.018** ***	**0.018** ***	**0.019** ***
[c] Environmental variables	**0.245** ***	**0.269** ***	**0.247** ***	**0.276** ***	**0.274** ***
Shared variation [d], [e], [f], [g]	0.282	0.252	0.272	0.244	0.244
[h] Residuals	0.397	0.394	0.450	0.453	0.452
(d) Neophytes (143 sp.)					
[a] M_{eff}	**0.154** ***	**0.158** ***	**0.010** ***	**0.005** ***	**0.002** *
[b] Spatial axes	**0.010** ***	**0.016** ***	**0.021** ***	**0.022** ***	**0.023** ***
[c] Environmental variables	**0.078** ***	**0.078** ***	**0.095** ***	**0.115** ***	**0.115** ***
Shared variation [d], [e], [f], [g]	0.152	0.147	0.124	0.103	0.103
[h] Residuals	0.606	0.602	0.750	0.755	0.758
(e) Non-threatened species (944 sp.)					
[a] M_{eff}	**0.059** ***	**0.069** ***	**0.007** ***	**0.006** ***	**0.007** ***
[b] Spatial axes	**0.063** ***	**0.064** ***	**0.071** ***	**0.068** ***	**0.072** ***
[c] Environmental variables	**0.113** ***	**0.135** ***	**0.119** ***	**0.13** ***	**0.132** ***
Shared variation [d], [e], [f], [g]	0.275	0.252	0.262	0.252	0.248
[h] Residuals	0.490	0.479	0.542	0.543	0.541
(f) Threatened species (538 sp.)					
[a] M_{eff}	**0.002** *	**0.008** ***	**0.004** **	0.000	0.001
[b] Spatial axes	**0.066** ***	**0.066** ***	**0.068** ***	**0.067** ***	**0.066** ***
[c] Environmental variables	**0.191** ***	**0.197** ***	**0.180** ***	**0.174** ***	**0.185** ***
Shared variation [d], [e], [f], [g]	0.090	0.084	0.100	0.107	0.096
[h] Residuals	0.650	0.644	0.648	0.652	0.651

Table 5.4: Correlation of the m_{eff} types MeshFG1 and MeshFG2, respectively, with species richness of six species groups (all species, groups after naturalization or threat status) assessed for Lower Saxony, Germany, using Kendall's τ (p $\leq$ 0.05) as correlation coefficient.

m_{eff} type	Species group	Kendall's τ	z-score	p-value
MeshFG1 (max = 3,196 ha, min = 9.3 ha)	All species	-0.188	-10.49	< 2.2e-16
	Natives	-0.137	-7.62	2.61E-14
	Archaeophytes	-0.246	-13.62	< 2.2e-16
	Neophytes	-0.327	-18.00	< 2.2e-16
	Non-threatened species	-0.211	-11.75	< 2.2e-16
	Threatened species[a]	-0.015	-0.84	0.4016
MeshFG2 (max = 3,172 ha, min = 0 ha)	All taxa	-0.164	-9.15	< 2.2e-16
	Natives	-0.109	-6.04	1.55E-09
	Archaeophytes	-0.218	-12.07	< 2.2e-16
	Neophytes	-0.343	-18.84	< 2.2e-16
	Non-threatened species	-0.183	-10.19	< 2.2e-16
	Threatened species[a]	-0.011	-0.62	0.5364

[a] Species richness of threatened species was square root transformed because it showed a skewed distribution.

studies have to be drawn with care. In contrast to the studies listed in Table 5, in our analysis, we applied units of equal size and thus the values of m_{eff} may be compared more easily. However, as we used relatively small landscape units, the actual degree of fragmentation may be overestimated. Nevertheless, compared to the results of other studies on m_{eff} in Germany (Table 5, no. 3-5), all of which used differently sized study units, our findings showed a similarly large variation. Values of m_{eff} obtained for Baden-Württemberg and Hesse ranged between 0 and c. 70 km^2 (JAEGER et al., 2001; ESSWEIN & SCHWARZ-VON RAUMER, 2004; Table 5, no. 3 and 5). In Bavaria, the maximum mesh size was c. 300 km^2, which was mainly a result of the high values of m_{eff} obtained for largely non-fragmented study units situated at the foothills of the Alps (ESSWEIN & SCHWARZ-VON RAUMER, 2006; Table 5, no. 4). The influence of largely non-fragmented mountainous areas also becomes obvious if values of m_{eff} for Switzerland are considered: When large alpine areas were included into the calculation procedure, maximum values of m_{eff} exceeded 1,500 km^2 (JAEGER et al., 2008; Table 5, no. 8). However, as soon as these uninhabitable areas were excluded from the computation process, the maximum values of m_{eff} dropped down to 300 to 700 km^2.

The European Environment Agency (EEA, 2011; Table 5, no. 2) compared the m_{eff} values of 28 EU countries and came to the conclusion that Germany, following the Benelux countries, was one of the most fragmented countries in Europe. Thus, the relatively small values of m_{eff} obtained for most of our studied landscape units seem to be representative for most parts of Germany. In contrast, studies from outside densely populated Europe showed much larger maximum values of m_{eff} (see Table 5, no. 1 and 9). For California, GIRVETZ et al. (2008) who used fragmentation geometries

similar to ours, and for China, LI et al. (2010) who calculated the *Effective Mesh Size* for countries and provinces, came up with maximum values of m_{eff} that exceeded these of our study a hundred- or even thousand-fold. However, even for these countries, the range of values of m_{eff} varied considerably: Whilst maximum values of m_{eff} in California reached c. 20,000 km^2 (Table 5, no. 9, B.1), the smallest values obtained were equal to zero. Similarly, values of m_{eff} for the 2,427 counties of China spanned a large range of c. 10 km^2 (the most densely populated areas) to c. 680,000 km^2 (the Tibetan plateau).

The response of species richness patterns to m_{eff} as a function of the choice of fragmentation geometries

Despite the fact that the FGs used in our analyses were nested, all types of m_{eff} (MeshFG1-MeshFG5) significantly differed from each other and thus did not contain any redundant information. Nevertheless, when relating the values of m_{eff} to species richness of the studied species groups, only MeshFG1 and MeshFG2 were found to be relevant. These m_{eff} types are based on FGs that include urban-industrial and traffic areas as well as water bodies and have been frequently used in studies on landscape fragmentation (cp. Table 5). Thus, even without examining the relationship of m_{eff} and species richness, many authors have chosen FGs that are meaningful for biodiversity. The explanatory power of MeshFG2 compared to that of MeshFG1 was slightly higher for all species groups. The inclusion of water bodies into the FGs used to calculate m_{eff} seems thus reasonable not only regarding animals, as proposed by GIRVETZ et al. (2008) and JAEGER et al. (2008), but also when distribution patterns of vascular plants are investigated. The remaining types of m_{eff} that included additional land cover classes supposed to have an added effect on the fragmentation of the landscape, did not explain (or only explained to a small extent) plant species richness patterns. We suppose that this was caused by the lack of a gradient of m_{eff} in MeshFG3 – MeshFG5; for the majority of grid cells very low values of m_{eff} were reported and the frequency distribution thus was extremely skewed to the left. This results from a country-wide intensive alteration of landscapes in Lower Saxony in the course of human land use that left only very few (almost) completely forested landscapes. Similar results could possibly be obtained for other intensively used landscapes all over Europe. However, if landscapes with a larger amount of (semi-)natural land cover classes are considered, the inclusion of agricultural areas (arable fields, managed grasslands) into m_{eff} calculation may reveal a more differentiated picture and thus may be more suitable to explain plant species richness. In contrast to our results, GIRVETZ et al. (2008) who investigated the effect of the inclusion of agricultural areas into the calculation of m_{eff} in California found these areas to only affect the degree of fragmentation in locations where extensive agricultural areas were present. In conclusion, the determination of relevant FGs is an important aspect when the relationship of landscape fragmentation and biodiversity is analysed.

Table 5.5: Overview on studies using the *Effective Mesh Size Index* (m_{eff}) to quantify landscape fragmentation, their coverage, study units and the fragmentation geometries (FGs) as well as potential response variables used in the particular studies. The range of m_{eff} is given for each FG and study unit, respectively. n/a = not available. Unless otherwise specified, m_{eff} is reported for CUT procedure.

No	Coverage (total area)	Study units	Fragmentation geometries (FGs) used	Size of study units (ranges, km²)	Range of m_{eff} (in km²)	Response variables	Reference
1	China (c. 9,500,000 km²)	A: 34 provinces, B: 2,427 counties	National, provincial and county roads, railway lines, urban areas	A: n/a B: n/a	A: 29.9–304,491 B: 10.7–676,623[a]	Endangered plants, mammals	Li et al. (2010)[d]
2	European Union, 28 countries (c. 4,400,000 km²)	A: 28 EU countries, B: 581 NUTS'X regions, C: 1 km² grid units within the 28 countries	Motorways, other roads up to class 4, railway lines, urban areas, mountains, lakes, major rivers	A: c. 300–544,000 B: n/a C: 1	A: 7.40–2,525.04 B: 0.03–6,177.91 C: n/a	None	EEA (2011)[d]
3	Germany, Baden-Württemberg (c. 36,000 km²)	A: 44 administrative districts, B: 66 natural geographic units	1: Urban areas, water bodies, railway lines, major roads 2: FG1 + municipal roads	A: 97.9–1,860.5 B: 13.9–1,989.7	A.1: 2.09–46.73 B.1: 1.20–71.86 A.2: 1.63–30.26 B.2: 0.78–66.14	None	Jaeger et al. (2001)
4	Germany, Bavaria (c. 70,500 km²)	A: 7 administrative districts, B: 18 regions, C: 16 nature parks, D: 96 natural geographic units	1: All roads excluding municipal roads, multi-tracked railway lines, channels, urban areas 2: Roads used by ≥1,000 vehicles per day, heavily used railway lines, urban areas, airports, larger channels 3: Similar to FG 2, roads buffered according to traffic density	A: 7,231.1–17,530.1 B: 1,478.5–5,689.7 C: 229.9–2,966.2 D: 13.5–4,494.6	A.1: 16.37–48.43 B.1: 15.02–108.42 C.1: 16.24–40.61 D.1: 0.59–286.71 A.2: 45.75–71.85 B.2: 23.13–112.13 C.2: 31.87–153.15 D.2: 0.60–287.01 A.3: 36.40–65.70 B.3: 14.50–92.49 C.3: 27.15–132.69 D.3: 0.09–286.38	None	Esswein & Schwarz-von Raumer (2006)
5	Germany, Hesse (c. 21,000 km²)	A: 3 administrative districts, B: 26 administrative districts, C: 59 natural geographic units	1: All roads excluding municipal roads, multi-tracked railway lines, channels, urban areas 2: Similar to FG 1, roads buffered according to traffic density	A: 53,804.8–82,909.9 B: 44.8–1,848.9 C: 0.5–1,553.2	A.1: 14.87–19.42 B.1: 1.94–27.68 C.1: 0.44–67.06 A.2: 11.40–15.80 B.2: 0.11–20.11 C.2: 0.00–45.38	None	Esswein & Schwarz-von Raumer (2004)

Table 5.5 – *Continued from previous page*

No	Coverage (total area)	Study units	Size of study units (ranges, km²)	Fragmentation geometries (FGs) used	Range of meff (in km²)	Response variables	Reference
6	Germany, Lower Saxony (c. 47,500 km²)	1,386 grid cells of c. 5.5 × 5.5 km²	c. 30	1: Traffic and urban-industrial areas 2: Traffic and urban-industrial areas, water bodies 3: Traffic and urban-industrial areas, water bodies, croplands 4: Traffic and urban-industrial areas, water bodies, croplands, grassland 5: All landscape elements except forested areas	0.09–31.96 0.00–31.72 0.00–31.14 0.00–30.19 0.00–29.76	Six groups of vascular plant species	This study
7	Italy, South Tyrol (c. 7,500 km²)	Municipality boundaries	1.6–302.3	Areas of urban development and transportation infrastructure (roads, railway lines)	0.50–208.70	None	Moser et al. (2007)
8	Switzerland (c. 41,000 km²)	A: 5 ecoregions, B: 26 cantons, C: 6 BLN-areas	A: n/a B: n/a C: n/a	1: Motorways, highways (class 1–3), railway lines, urban areas 2: Motorways, highways (class 1–2), railway lines, urban areas, water bodies, mountains ($\geq$2,100m) 3: FG 1 + water bodies and mountains ($\geq$2,100m) 4: equals FG 1 but inhabitable parts of landscape omitted	A.1: 19.96–1,310.24 B.1: 0.54–1,850.75 C.1: 27.58–1,544.30 A.2: 21.47–430.58 B.2: 0.61–681.82 C.2: n/a A.3: 9.86–293.30 B.3: 0.52–485.52 C.3: n/a A.4: 10.78–381.72 B.4: 0.54–635.14 C.4: n/a	None	Jaeger et al. (2008)[d]
9	USA, California (c. 450,000 km²)	A: 2 nested administrative planning unit boundaries, B: 6 nested spatial scales of watersheds	A: 118–520,911 B: n/a	1: Highways, major roads, railway lines, urbanized areas 2: FG 1 + minor roads 3: FG 2 + agricultural areas 4: FG 3 + lakes, major rivers, alpine areas above 3,000 m	A.1: 0.04–12,092.00[b] B.1: 0.00–20,885.00[c] A.2: 0.02–5,064.00[b] B.2: 0.00–14,900.00[c] A.3: 0.02–5,058.00[b] B.3: 0.00–14,898.00[c] A.4: 0.01–2,615.00[b] B.4: 0.00–10,175.00[c]	None	Girvetz et al. (2008)[d]

[a] Average of m_{eff} for the 10 least fragmented counties, [b] 58 counties of California, USA, [c] 6,994 planning watersheds of California, USA, [d] CBC procedure (cp. Moser et al. 2007).

Differentiating the effect of m_{eff} on species richness for selected plant groups

The explanatory power of m_{eff} compared to that of abiotic and spatial variables differed considerably among species groups. Surprisingly, species richness and landscape fragmentation were positively correlated for all species groups, with the exception of threatened species. Thus, species richness was highest in highly fragmented landscapes and was lowest in little fragmented landscapes.

For richness patterns of neophytes m_{eff} was an exceptionally important factor. That is not surprising, because alien species are known to be linked to urban habitats and linear landscape elements and are associated with human activities (e. g. Pyšek, 1998; Roy et al., 1999; Deutschewitz et al., 2003; Honnay et al., 2003; Kühn et al., 2003; Lippe & Kowarik, 2008). Archaeophytes, in contrast, are frequently found in rural environments (Pyšek, 1998; Deutschewitz et al., 2003), particularly in loess landscapes (Kühn et al., 2003) with a long colonization history (Küster, 1999). Today, most of these areas in our study area are intensively used and highly fragmented agricultural and/or industrial landscapes. The relation of archaeophyte richness with loess landscapes and major rivers in Lower Saxony was confirmed by Schmiedel et al. (2011). As suggested by the high explanatory value of the environmental variables, areas of high archaeophyte richness are related to environmental conditions rather than to landscape fragmentation.

Only a very small proportion of native species richness was explained by m_{eff}, while environmental variables were more important. Even if weak, a negative relationship between m_{eff} and species richness was observed for this species group as well. This finding may be attributed to the fact that urban areas are commonly situated in naturally diverse ecoregions as shown by Kühn et al. (2004), who found that geological diversity is associated with areas of urbanisation in Germany. Furthermore, niche availability in urban areas is known to be rather high (Deutschewitz et al., 2003; Honnay et al., 2003; Kühn et al., 2004; Wania et al., 2006; Lippe & Kowarik, 2008). Possible adverse effects of fragmentation could therefore be masked.

Surprisingly, we found no effect of m_{eff} on threatened plant species richness patterns. Environmental conditions were better predictors of threatened plant species richness than was m_{eff}. Similar to our results, Honnay et al. (2003), who studied the effect of different landscape indices as predictors for regional plant species richness, found the effect of landscape fragmentation on the richness of threatened species to be compensated by high landscape diversity. However, it seems that threatened species' occurrences are independent of the overall degree of landscape fragmentation (at least as long as it remains above a certain threshold). Threatened species are usually closely linked to ecologically valuable, often threatened habitat types. As discussed for Natura 2000 forest habitat types, characteristic forest species are limited in their dispersal ability and mostly linked to forests with long habitat continuity (Culmsee et al., 2014). At first, fragmentation may have only little effect on remnant

populations as long as habitats do not vanish completely, yet the consequences of an extinction debt might be paid in the future (Kuussaari et al., 2009). However, the observed result may be an effect of the study scale being too coarse to reveal the relationship between landscape fragmentation and richness of threatened species (Jackson & Fahrig, 2012). Further studies on different spatial scales are needed to provide evidence on the latter.

Conclusions

The assessment of landscape fragmentation and approaches for linking fragmented habitats by corridors are becoming increasingly important in nature conservation policy in Germany, as well as the rest of Europe (BfN, 2014; European Commission, 2011; Bundeskabinett, 2012). The EEA (2011, p. 58) even identified the analysis of the relationship between the levels of landscape fragmentation and biodiversity as "one of the most important areas for future research". In this context, measures of fragmentation are increasingly used to assess the status of a landscape (Strand et al., 2007; BMU, 2010). The *Effective Mesh Size* m_{eff} is one of the most frequently applied indices in studies on landscape fragmentation. While many of these studies have pointed out the ecological relevance of fragmentation in general and m_{eff} in particular (Jaeger, 2000; Jaeger et al., 2008; Girvetz et al., 2008; Li et al., 2010), to our knowledge, until now no landscape-scale study has evaluated the impact of m_{eff} on species richness. By addressing this issue, our study has contributed to increasing the understanding of this relationship. We conclude that, as long as relevant fragmentation geometries for the landscapes under consideration are chosen, m_{eff} can act as a meaningful measure to explain vascular plant species richness patterns. Out of the five types of m_{eff} used in our study, two were found to be relevant for species richness, both having been used in previous studies (Jaeger et al., 2001; Girvetz et al., 2008; Jaeger et al., 2008). We further conclude that it is important to consider not only all vascular plant species, but also several smaller species groups. Species included in these groups should differ in their residence or threat status or in other characteristics in order to obtain more precise results on habitat requirements that facilitate interpretations.

Because fragmentation is a landscape-scale process, its relationship with biodiversity should accordingly be studies at this scale (McGarigal & Cushman, 2002; Fahrig, 2003). However, when evaluating 100 and 135 fragmentation studies, respectively, McGarigal & Cushman (2002) and Fahrig (2003) recognised that about 50 % of all studies were actually conducted at the patch scale. This is obviously related to the practical difficulties arising from large-scale experiments. Thus, to be able to study the effects of landscape fragmentation on biodiversity, it is important to develop new methods for their assessment. Our approach avoids such difficulties by using readily available high-resolution data on land cover and the distribution of vas-

cular plant species. As such data are becoming increasingly available in more and more countries (CULMSEE et al., 2014) and because computational power is rapidly increasing, our approach is easily extendable to other – even larger – study areas.

Further studies are needed to improve our understanding of the relationship of species richness and landscape fragmentation. The transferability of our results to other spatial scales should be tested, as well as the applicability of the method to less human-dominated landscapes. Presumably, effects of landscape fragmentation on species in highly transformed Central European landscapes differ from those in more natural landscapes. Thus, further studies are encouraged to bridge the gap between the assessment of landscape fragmentation, which is methodologically already advanced, and its relevance to species diversity, which has been rarely tested in the past.

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6 Plant species richness along a gradient of landscape modification

Abstract

Several concepts of landscape modification, landscape naturalness and hemeroby have been put forward to quantify the extent of human activities and impact on landscapes. However, despite numerous studies on the human influence on landscapes, the relationship between these concepts and biodiversity has rarely been tested. On a grid of 1386 lat/long cells (landscape units, c. 30 km^2) we used simple information on land cover and easily measurable landscape metrics to identify a gradient of landscape modification for the model region of Lower Saxony, Germany. We linked the gradient of landscape modification intensity to richness patterns of eight groups of vascular plants that were aggregated using residence and threat status and habitat preferences of the species.

K-means cluster analysis revealed a gradient of six landscape modification levels ranging from highly-fragmented urban landscapes to little-fragmented landscapes with large proportions of forest. Richness of all species groups differed significantly along the gradient. For the total and native species pools, for forest, low-nutrient indicating and threatened plants richness peaked at both ends of the modification intensity gradient. For neophytes, urban plants and high-nutrient indicators, richness decreased along the gradient from urbanised to less modified landscapes.

Due to its simplicity and coarse scale, our approach may easily be applied to other study areas without comprehensive data on land cover. Furthermore, following proper interpretation by experts, it can be used to identify and preselect priority landscapes for nature conservation planning.

Keywords

Introduction

Over recent decades, tremendous man-induced changes in land cover, increasing landscape fragmentation, degradation and intensification have occurred in landscapes all over Europe (Bastian & Bernhardt, 1993; Meeus, 1995; Stoate et al., 2001; Mander et al., 2004; Foley et al., 2005; Wesche et al., 2012). Such landscape modifications have already led to large losses in (semi-)natural vegetation, biodiversity and ecosystem functions (VEAC, 2010) and are expected to cause even more losses in the future (Sala et al., 2000). Researchers have developed ideas on 'landscape modification' (Forman & Godron, 1986) and the related concepts of 'landscape naturalness' (Machado, 2004; Liira & Sepp, 2009; Winter et al., 2010), 'hemeroby' (Sukopp, 1976; Kowarik, 1988; Grabherr et al., 1998; Steinhardt et al., 1999; Wrbka et al., 2004; Walz & Stein, 2014), 'urbanity' (Hill et al., 2002; Wrbka et al., 2004) and 'degree of culturalness' (Jansen et al., 2009) that are used to quantify the degree of modification a landscape has experienced. For plant species, qualifying indices are available that are based on the concepts of hemeroby (i. e., estimating the degree of human influence on the plant's environment) or urbanity (i. e., strength of association with urban environments) (Jalas, 1955; Klotz & Kühn, 2002). Despite the vast number of studies exploring concepts of landscape modification or investigating human influences on biodiversity, the relationship between the two has rarely been tested directly at large scales (but see Nabe-Nielsen et al., 2010 for animal populations). Systematic studies on biodiversity patterns in modified landscapes are, however, urgently needed to assess overall effects of human impact on biodiversity and to support landscape-scale conservation planning (Daily, 2001; Fischer & Lindenmayer, 2007).

There are numerous approaches to quantify the magnitude of deviation from the natural state of a landscape. While some authors have based their assessment of the landscape status solely on land cover data (Steinhardt et al., 1999), others used multiple parameters to describe the degree of hemeroby (Grabherr et al., 1998; Winter et al., 2010; Walz & Stein, 2014) or naturalness (Liira & Sepp, 2009; Winter et al., 2010) of a landscape. Most of these studies were conducted for forest habitats, but Walz & Stein (2014) developed an indicator of hemeroby applicable to the whole of Germany. The authors used high-resolution data on land cover and spatial information on the potential natural vegetation to derive the degree of hemeroby. They assigned the land cover categories to seven levels of hemeroby and calculated landscape hemeroby for various reference units.

Most studies on the hemeroby, naturalness or modification of a landscape or biotope are dependent on extensive field data and/or other detailed information. For many countries, such comprehensive data are not available. To avoid this problem, land cover data (either directly taken from existing digital landscape models or derived from remote sensing approaches) might prove suitable for the assessment of landscape modification and its relation to biodiversity. Steinhardt et al. (1999, p. 2) considered data on land use/land cover as the "interface between the natural

conditions of the landscape and the influence of man". These data are "attainable within a reasonable time and with reasonable effort" (as demanded by Machado, 2004, p. 100). Various landscape metrics based on land cover data have been used to describe and quantify changes in landscape patterns and the composition and configuration of landscapes (Luck & Wu, 2002; Yu & Ng, 2006; Olsen et al., 2007). Landscape parameters found to decrease when landscape modification increases are the amount of (semi-)natural land cover types (henceforth LCTs), as well as the average sizes of habitat patches (Forman & Godron, 1986; Miller et al., 1997; Moser et al., 2002). As patch sizes decrease, habitats are subdivided into smaller fragments, patch numbers increase and core area sizes decrease. These processes may reduce the connectivity of patches, increase isolation between suitable habitats and enhance edge effects (Harper et al., 2005; Mascarua Lopez et al., 2006). Depending on the quality of the LCT under consideration, such landscape changes can either be interpreted as increasing the structural diversity of the landscape or increasing its subdivision (Lang & Blaschke, 2007).

Several studies have investigated the relationship between some of the above mentioned parameters and biota. Kerr & Deguise (2004) and Fischer & Lindenmayer (2007) reported that native species benefit from the presence of natural vegetation and suggested that a decrease in the amount and size of natural habitat patches (i. e., an increase in the fragmentation of the habitat) provokes a decrease in native species richness. In contrast, many alien species are associated with human activities and commonly linked to urban habitats and to linear landscape elements. They may thus benefit from the modification of the landscape (e. g. Deutschewitz et al., 2003; Honnay et al., 2003; Kühn et al., 2003). Generally, high habitat heterogeneity was found to be linked to large numbers of both native and alien species (Deutschewitz et al., 2003).

As Forman & Godron (1986, p. 286) pointed out, it is "neither possible nor useful" to separately consider the various types of human influence when evaluating the status of a landscape. Instead, the combined effects of all human influences visible at the landscape scale should be taken into account. In doing so, landscapes can be arranged along a 'landscape modification gradient'. In their well-known text on landscape ecology, Forman & Godron (1986) proposed a 5-level concept on landscape modification ranging from natural to urban landscapes. Inspired by this idea, our aim was to analogously identify a gradient of landscape modification intensity based on simple information on land cover and easily measurable landscape metrics. We aimed to link the modification gradient to richness patterns of vascular plant species that were available as presence data on latitude/longitude grid cells covering the complete study area. We used LCTs that captured large proportions of the landscape and were easy to identify using remote sensing approaches. In contrast to Walz & Stein (2014), our analysis does not make any a priori statements about the naturalness or hemeroby of the LCTs under consideration. In the analysis, we

used the total number of vascular plant species reported for the study area as well as seven sub-groups of species aggregated using their residence and threat status as well as information on habitat preferences.

We hypothesised:
- Based on landscape metrics used to assess the composition and configuration of a landscape, clusters of landscapes with similar characteristics can be differentiated. These can be placed along a gradient of landscape modification, as proposed by FORMAN & GODRON (1986), spanning from (semi-)natural (mostly forested) to urbanised landscapes.
- The landscape clusters are meaningful for vascular plant species richness, i. e., species numbers vary significantly along the gradient of landscape modification intensity.
- Species numbers of different vascular plant species groups show different patterns along the gradient of landscape modification intensity.

Material & Methods

Study area

The study was carried out in the German federal state of Lower Saxony (c. 47,500 km^2, Fig. 6.1a). As landscape formation history and natural environmental configuration, colonisation history and current land use vary widely across the state, Lower Saxony harbours a large number of different habitat types.

The coastal area in the north of Lower Saxony was formed in the Holocene. It comprises the Wadden Sea, its islands and salt marshes, the estuaries of the three largest rivers of Lower Saxony (Ems, Weser and Elbe) as well as impoldered former tidal wetlands. The latter are today mainly covered by grasslands, croplands and urban areas (DRACHENFELS, 2010).

The lowlands were formed during the Pleistocene and harbour various types of landscapes, ranging from intensively managed agricultural landscapes in the west to wooded landscapes in the east.

The uplands in the south have Mesozoic and Palaeozoic bedrock and display high landscape heterogeneity. The landscape of the less elevated parts of this area is characterised by a mosaic of croplands, grasslands and various forests, while the higher elevations in the southeast (Harz Mountains) are largely covered by forests.

The largest cities of Lower Saxony are Hanover and Braunschweig, both situated at the border of the lowland and upland areas, followed by Oldenburg in the northern lowlands and Osnabrück in the western uplands.

From the coast in the north to the Harz Mountains in the southeast, elevation ranges from 0–971 m a.s.l.. From the northwest to the east, the climate gradually changes

from oceanic (mean annual precipitation up to 900 mm) to suboceanic-subcontinental (c. 550 mm; NLfB, 1997).

Lower Saxony was selected as model region because of its high landscape diversity and because the area is covered by a completed mapping program on the distribution of vascular plants with a resolution of c. 30 km² (Garve, 2007).

Landscape composition and configuration

Following the grid system of the topographical maps, the study area was subdivided into a regular grid of 1739 cells sized c. 5.5 × 5.5 km² (30 km²; Fig. 6.1a). The 353 grid cells situated at the border of the state that only partially covered the study area and a further three grid cells completely covered by water bodies were omitted. Thus, a total of 1383 grid cells remained in the analysis; in the following each of these will be treated as a landscape unit.

Landscape composition and configuration were derived from a digital landscape model (DLM; LGN, 2008) of Lower Saxony. The DLM vector data were categorised in eight LCTs (traffic, urban and industrial areas, water bodies, forests, grasslands, croplands, and an additional class that summarizes all remaining landscape elements) and then transformed to raster format with a resolution of 25 m². Thirty different landscape metrics (Table 6) were calculated for each of the 1383 landscape units using Fragstats 3.3 (McGarigal et al., 2002). Ten of these indices describe landscape composition; the remaining 20 indices depict different aspects of landscape configuration.

We calculated the proportion of each LCT per landscape unit (*Percentage of Landscape*, PLAND). The diversity at the landscape level was assessed using the *Shannon Diversity Index* (SHDI). The latter takes into account all LCTs present in the landscape as well as their extent and is especially sensitive to the presence of rare LCTs (McGarigal et al., 2002).

The *Effective Mesh Size Index* (MESH) was used as an indicator to quantify the degree of fragmentation of a landscape unit. For its calculation, landscape elements that can act as barriers in the landscape (fragmentation geometries) have to be designated. In a previous analysis (Schmiedel & Culmsee, submitted), urban, industrial and traffic areas as well as water bodies were identified as suitable fragmentation geometries for the prediction of plant species richness. Thus, these LCTs were also used in the present study.

To capture average patch sizes, the *Area-weighted Mean Patch Size Index* (AREA_AM) was assessed. Together with the *Number of Patches* (NP), it supports assumptions about the fragmentation of a LCT and thus adds information to the general trend that can be derived from MESH. Patch complexity was evaluated using the *Area-weighted Mean Shape Index* (SHAPE_AM). Both, AREA_AM as well as SHAPE_AM were calculated on the landscape scale (i. e., averaged over all LCTs) as well as on the LCT level for forest, grassland and cropland areas, respectively; the number of patches was calculated for forest, grassland and cropland areas only.
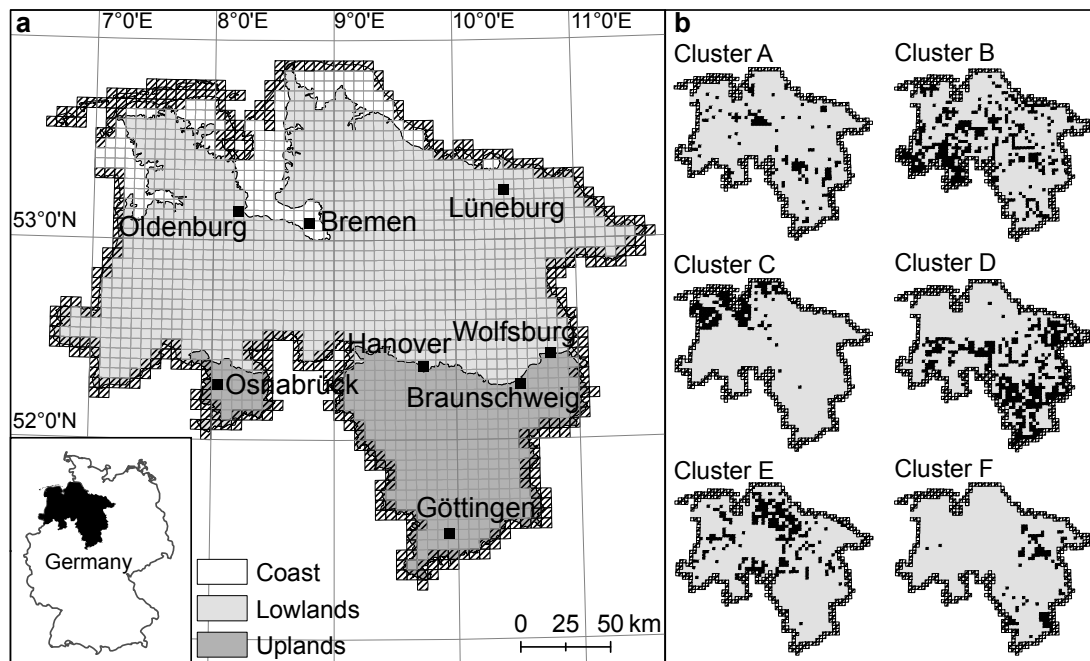
Figure 6.1: (a) The state of Lower Saxony, Germany, with its three major landscape components: coast, lowlands and uplands. The 1383 landscape units (size c. $5.5 \times 5.5\,\mathrm{km}^2$ or $30\,\mathrm{km}^2$) display the basic mapping units of the vascular plant survey program and represent the landscape units used in this analysis. Hatched grid cells were not considered in the analysis. (b) Distribution of landscape units included in clusters A to F in Lower Saxony. Maps with Transverse Mercator projection, Germany Zone 3 ("Deutsches Hauptdreiecksnetz").

Edge length was assessed using the *Total Edge Index* (TE) that sums up all edges a particular LCT shares with other LCTs.

As a measure for the isolation or rather connectivity of the patches of a LCT the *Mean Proximity Index* (PROX_MN) was assessed. It is calculated by considering size and distance of all neighbouring patches of the same type; the distance within patches that should be regarded as 'connected' has to be specified by the user ('search radius'; i.e., 100 m in the present analysis). It should be noted that only landscape connectivity and not habitat and ecological connectivity was assessed (cp. Fischer & Lindenmayer, 2007). TE and PROX_MN were calculated on the LCT level for forest, grassland and cropland areas, respectively.

For the LCT 'forest' the *Total Core Area Index* (TCA) and the *Mean Core Area Index* (CORE_MN) were assessed. The former measures the total amount of undisturbed core habitat after a user-specified edge buffer (i.e., 100 m in the present analysis) is eliminated (McGarigal et al., 2002). Analogously, CORE_MN quantifies the average amount of undisturbed core habitat in the landscape.

Table 6.1: Landscape metrics derived from land cover data (LGN, 2008) with a resolution of 25 m². Eight different land cover types (LCTs) were distinguished. Landscape metrics were calculated for 1383 landscape units covering Lower Saxony, Germany. Variables printed in bold letters were used in the cluster analysis; variables printed in normal type were highly correlated and therefore excluded from the analysis. The corresponding nomenclature and units used in Fragstats 3.3 (McGarigal et al., 2002) is given. Corr., correlated variable; Transf., transformation applied to the data (log = logarithmic transformation, –, no transformation); Mean (± SE), mean value across the 1383 landscape units (± standard error).

Variable names	Description	Fragstats nomenclature	Unit	Corr.	Transf.	Mean (± SE)
Landscape composition						
prop_water	Proportion of water bodies	PLAND	%	no	log	1.7±0.1
prop_forest	Proportion of forests	PLAND	%	no	log	21.9±0.6
prop_grass	Proportion of grasslands	PLAND	%	no	log	22.2±0.5
prop_crop	Proportion of croplands	PLAND	%	no	log	39.7±0.6
prop_urban	Proportion of urban areas	PLAND	%	no	log	7.2±0.2
prop_industry	Proportion of industrial areas	PLAND	%	no	log	2.4±0.1
prop_traffic	Proportion of traffic infrastructure	PLAND	%	no	log	1.7±0.0
prop_others	Proportion of remaining landscape elements	PLAND	%	no	log	3.3±0.1
all_SHDI	Shannon Diversity of all LCT (landscape diversity)	SHDI	–	no	–	1.25±0.01
all_SHEI	Shannon Evenness of all LCT	SHEI	–	yes	–	0.59±0.00
Landscape configuration						
meshsize	Effective Mesh Size Index (fragmentation geometries: urban, industrial and traffic area and water bodies), measure of landscape fragmentation	MESH	ha	no	–	1433.4±22.0
all_AreaAm	Area-weighted mean patch size averaged over all patches of all LCTs in the landscape	AREA_AM	ha	no	log	401.0±11.2

Continued on next page

Table 6.1 – *Continued from previous page*

Variable names	Description	Fragstats nomenclature	Unit	Corr.	Transf.	Mean (± SE)
all_ShapeAm	Area-weighted Mean Shape Index averaged over all patches of all LCTs in the landscape	SHAPE_AM	–	no	–	5.4±0.1
for_ShapeAm	Area-weighted Mean Shape Index of forest areas	SHAPE_AM	–	yes	–	2.9±0.0
for_TCA	Total core area of forest areas (edge depth: 100 m)	TCA	ha	yes	–	286±10.9
for_AreaAm	Area-weighted mean patch size of forest areas	AREA_AM	ha	yes	log	300.3±13.3
for_CoreAm	Area-weighted mean core area of forest areas (edge depth: 100 m)	CORE_AM	ha	yes	–	165.3±8.6
for_NP	Number of forest patches	NP	–	no	–	56.8±1.1
for_TE	Total edge of forest patches	TE	m	yes	–	89963.4±1,485.8
for_ProxMn	Mean Proximity Index of forest patches (search radius: 100 m)	PROX_MN	–	yes	log	3466.9±281.8
grass_ShapeAm	Area-weighted Mean Shape Index of grassland areas	SHAPE_AM	–	yes	–	4.3±0.1
grass_AreaAm	Area-weighted mean size of grassland areas	AREA_AM	ha	yes	log	130.2±6.5
grass_NP	Number of grassland patches	NP	–	no	–	61.6±0.7
grass_TE	Total edge of grassland areas	TE	m	yes	–	169293.7±3084.2
grass_ProxMn	Mean Proximity Index of grassland patches (search radius: 100 m)	PROX_MN	–	yes	log	4201.9±290.1
crop_ShapeAm	Area-weighted Mean Shape Index of cropland areas	SHAPE_AM	–	yes	–	3.8±0.1
crop_AreaAm	Area-weighted mean size of cropland areas	AREA_AM	ha	yes	log	289±9.5
crop_NP	Number of cropland patches	NP	–	no	–	43.5±0.6
crop_TE	Total edge of cropland areas	TE	m	yes	–	166958.7±1988.0
crop_ProxMn	Mean Proximity Index of cropland patches (search radius: 100 m)	PROX_MN	–	yes	log	10966.5±465.9

To improve statistical normality, the proportions of LCTs as well as the values obtained for AREA_AM and PROX_MN were log-transformed prior to the analysis. All landscape metrics were standardised to z-scores, because the range of values of the different indices strongly differed.

Landscape metrics were tested for correlation amongst each other using Spearman's rank correlation coefficient ρ. Highly correlated variables (Spearman's $\rho >$ |0.7|; i. e., 15 variables, see Table 6) were excluded from the further analysis, whereas the simple landscape composition metrics were preferred over the more complex indices of landscape configuration. In total, 15 landscape metrics remained in the analysis (Table 6).

Cluster analysis

Based on the 15 standardised landscape metrics, k-means clustering was applied to pool those landscape units with a similar landscape composition and configuration. The k-means analysis groups entities according to the similarity of their attributes. It was performed using the 'kmeansruns' function from 'fpc' package (Henning, 2014) in R version 2.15.2 (R Development Core Team, 2012). The function initialises the k-means algorithm several times using different starting points. The number of runs as well as the number of iterations was set to 100. The function estimates the optimal number of clusters by considering the average silhouette width criterion. The latter provides a measure of how well each landscape unit lies within the cluster it was assigned to. The index ranges between +1 (well clustered) and −1 (poorly clustered) (Brock et al., 2008). The number of clusters to be compared was set to a minimum of 5 and a maximum of 20 clusters.

To determine the main gradient underlying the clusters, a PCA on the standardised variables was performed in R (function 'prcomp' in the 'stats' package). Clusters were sorted by the variable with most influence on the data set (i. e., prop_urban).

Species numbers per cluster

Plant species records were obtained from the data base of the vascular plant survey program of Lower Saxony (NLWKN 1982-2003) that holds information about the presence/absence of all vascular plant species occurring in the aforementioned $30\,\mathrm{km}^2$ landscape units (Garve, 2007).

In the analysis, we used the total number of 1594 vascular plant species reported for the study area as well as seven sub-groups of species aggregated according to their residence status, their threat status and their association with particular habitats (Table 6.2). Threatened species were identified using the Red List of Lower Saxony (Garve, 2004). Native species were distinguished from established neophytes (Klotz et al., 2002). Forest species (Schmidt et al., 2011) were selected because they are habitat specialists that usually do not occur outside of forests. Among the eight LCTs

Table 6.2: Total number of vascular plant species of Lower Saxony, Germany, and the number of species of seven sub-groups distinguished by residence status, threat status and habitat preferences. Species groups were overlapping to some extent (cp. Table A.8 in Appendix A).

No. Species group	Definition	Numbers of species
Total vascular plant flora		
1 All species	Total number of plant species recorded for Lower Saxony[a,b]	1594
Residence status		
2 Native species	Species native to Lower Saxony[b,c]	1297
3 Established neophytes	Species not native to Lower Saxony, established 16th century or later[b,c]	143
Threat status		
4 Threatened species	Red-listed species (threat status 1, 2, 3, G, R)[d]	538
Habitat preferences		
5 Forest species	Species linked to forest habitats[e]	197
6 Urban species	Species linked to urban habitats[f]	68
7 EIV species N 1-3	Species of nutrient-poor sites[g]	474
8 EIV species N 7-9	Species of nutrient-rich sites[h]	306

[a] NLWKN (1982-2003); [b] GARVE (2007); [c] KLOTZ et al. (2002); [d] GARVE (2004); [e] Categories 1.1 and 1.2, SCHMIDT et al. (2011); [f] "Urbanity", categories 4 and 5, KLOTZ et al. (2002); [g] Ellenberg indicator value for nutrients (N) 1-3, BfN (2013); [h] Ellenberg indicator value for nutrients (N) 7-9, BfN (2013).

of our analysis, we considered forests representing areas least influenced by humans. Following KLOTZ et al. (2002), a group of urban species linked to settlements was defined (urbanity index 4 and 5). As eutrophication is one of the most important threats for temperate European ecosystems (DISE, 2011), two further species groups were assembled based on their association with nutrient-rich (Ellenberg Indicator Values, EIV N 7–9) and nutrient-poor habitats (EIV N 1–3), respectively (BfN, 2013). Species groups were overlapping to some extent (see Table A.8 in Appendix A). Subspecies were pooled to the corresponding species and species to the relevant aggregate of species if these were present in the data base.

Mean numbers of the total number of species reported for Lower Saxony and of the seven sub-groups were calculated per cluster. To test for significant differences between species numbers of the different clusters, a non-parametric Kruskal-Wallis test was performed in R. Multiple comparisons were made using the 'pgirmess' package (GIRAUDOUX, 2013) at a 5 % significance level.

Results

Clusters of landscape modification intensity

K-means clustering of the 1383 landscape units resulted in six clusters of varying size. The smallest cluster (F) comprised 90 and the largest cluster (D) 389 landscape units (Table 6.3). The average silhouette width value of 0.16 obtained for our six clusters indicated a higher similarity within clusters than between neighbouring clusters. Although this value was relatively low, the clusters were nevertheless well interpretable.

The results of PCA (Table 6.4) suggested that the most important landscape metric was the proportion of urban areas (prop_urban). Further variables highly correlated with PCA axis 1 were the *Effective Mesh Size Index* (meshsize) and the *Area-Weighted Mean Patch Size Index* averaged over all patches of all LCTs in the landscape units (all_AreaAm).

The six clusters showed the following features (see Table 6.3):

Cluster A contained only 131 landscape units (9 % of the total number of investigated landscape units) and was characterised by an extremely high landscape fragmentation (very low meshsize, low area-weighted mean patch size). Compared to the other clusters, the patches in this cluster showed very high landscape diversity (all_SHDI). Most parts of these landscapes were made up by urban-industrial and traffic areas, thus the landscape units included mainly large cities (Fig. 6.1b).

Cluster B, the second largest cluster in the analysis (28 % of the landscapes units), was characterised by medium to high landscape fragmentation (low meshsize, low area-weighted mean patch size) and medium to high landscape diversity. All LCTs occurred at medium proportions. Forests, grasslands and croplands were each represented by a large number of patches suggesting relatively small but numerous patches (i.e., highly fragmented). Landscape units of this cluster were dispersed all over Lower Saxony (Fig. 6.1b).

Cluster C, representing 11 % of the landscape units, was characterised by numerous grassland patches, very large proportions of grassland and water bodies and low to very low proportions of forest and cropland. Landscape diversity as well as landscape fragmentation occurred at medium magnitude. The complexity of patches in the landscape was very high. Landscape units of this cluster were – with only few exceptions – mainly located in coastal and subcoastal areas and in the north-western lowlands (Fig. 6.1b).

Cluster D, the largest cluster in the analysis (28 % of the landscapes units), was dominated by croplands that on average covered more than 55 % of the landscape. Forests occurred at medium proportions, while the proportions of urban-industrial and traffic areas as well as grasslands were small. The degree of fragmentation and landscape diversity were generally medium to low. Landscape units of this cluster were mainly distributed in the uplands and the eastern and central lowlands (Fig. 6.1b).

Table 6.3: Characterisation of the landscapes in the six clusters (mean value of the variable per cluster $\pm$ SE) found for 1383 landscape units in Lower Saxony, Germany. n, number of landscape units in the particular cluster. Superscript letters indicate significant differences between the clusters (p$\leq$0.05, non-parametric multiple Kruskal-Wallis test). Non-standardized values are given; mean values of the standardized variables that were used in the analysis are given in Table A.7 in Appendix A.

Variable names	Description	A	B	C	D	E	F
n		*131*	*383*	*148*	*389*	*242*	*90*
Landscape composition							
prop_water	Proportion of water bodies	3.6 ± 0.6^a	1.3 ± 0.1^b	5.2 ± 0.6^a	0.6 ± 0.0^c	1.1 ± 0.1^d	0.6 ± 0.1^c
prop_forest	Proportion of forests	10.8 ± 1.0^a	19.1 ± 0.6^b	1.6 ± 0.2^c	25.9 ± 0.9^d	19.2 ± 0.9^b	73.1 ± 1.7^e
prop_grass	Proportion of grasslands	19.6 ± 1.4^a	20.6 ± 0.6^b	60.7 ± 1.5^c	9.5 ± 0.3^d	29.6 ± 0.9^e	5.0 ± 0.4^f
prop_crop	Proportion of croplands	26.6 ± 1.5^a	44.7 ± 0.7^b	21.7 ± 1.4^{ac}	55.8 ± 0.9^d	34 ± 0.8^e	12.3 ± 1.5^c
prop_urban	Proportion of urban areas	20.8 ± 0.9^a	8.5 ± 0.2^b	6.4 ± 0.3^c	5.0 ± 0.1^d	4.0 ± 0.1^e	1.7 ± 0.2^f
prop_industry	Proportion of industrial areas	9.0 ± 0.6^a	2.2 ± 0.1^b	1.1 ± 0.1^c	1.1 ± 0.1^c	2.9 ± 0.4^d	0.7 ± 0.1^e
prop_traffic	Proportion of traffic infrastructure	5.1 ± 0.3^a	1.9 ± 0.0^b	1.5 ± 0.1^c	1.2 ± 0.0^d	0.9 ± 0.0^e	0.4 ± 0.0^f
prop_others	Proportion of remaining landscape elements	4.4 ± 0.3^a	1.8 ± 0.1^b	1.9 ± 0.2^b	1.0 ± 0.0^c	8.4 ± 0.5^d	6.1 ± 1.3^b
all_SHDI	Shannon Diversity Index of all LCT	1.57 ± 0.01^a	1.38 ± 0.01^b	1.05 ± 0.02^c	1.10 ± 0.01^c	1.39 ± 0.01^b	0.77 ± 0.03^d
Landscape configuration							
meshsize	Effective Mesh Size Index	329.1 ± 19.5^a	1042.2 ± 27.0^b	1243 ± 53.7^b	1761.2 ± 36.2^c	1841.7 ± 43.4^c	2504.4 ± 62.3^d
all_AreaAm	Area-weighted Mean Patch Size Index	174.2 ± 11.1^a	182.8 ± 4.9^a	450.6 ± 26.7^b	542 ± 17.7^c	253 ± 8.8^d	1367 ± 72.9^e
all_ShapeAm	Area-weighted Mean Shape Index	7.5 ± 0.3^a	4.9 ± 0.1^b	8.5 ± 0.2^a	4.9 ± 0.1^{bc}	4.6 ± 0.1^{bc}	4.4 ± 0.1^c
for_NP	Number of forest patches	43.1 ± 2.7^a	84.3 ± 2.0^b	17.6 ± 1.8^c	52.1 ± 1.8^a	65.2 ± 1.9^d	22.4 ± 1.7^c
grass_NP	Number of grassland patches	67.4 ± 2.0^a	81.8 ± 1.1^b	46.4 ± 1.6^c	56.5 ± 1.0^d	55.6 ± 1.0^d	30.1 ± 1.4^e
crop_NP	Number of cropland patches	42.1 ± 1.8^a	58.3 ± 1.0^b	54.0 ± 2.3^c	30.3 ± 0.5^d	46.2 ± 0.9^{ac}	14.5 ± 1.1^e

Cluster E, including 17 % of the landscape units, was characterised by a mosaic of croplands (c. 35 %), grasslands (c. 30 %), forests (c. 20 %) and landscape elements subsumed under "others" (c. 9 %). Urban-industrial and traffic areas were present at small proportions. The degree of fragmentation and landscape diversity were generally medium to low. Landscape units of this cluster were – with one exception – all located in the lowlands (Fig. 6.1b).

Cluster F, the smallest cluster in the analysis, containing 90 grid cells (7 % of the landscapes units), was characterised by large proportions of forests and small proportion of all other LCTs. The values obtained for the *Effective Mesh Size Index* and area-weighted mean patch sizes were very high, thus the landscape fragmentation was low. Landscape diversity was very low. The number of grassland and cropland but also of forest patches was small. The latter in combination with the large proportion of forests suggested the occurrence of large coherent forests. These had a highly complex shape. The landscape units of this cluster were mainly situated in the Harz and Solling Mountains in the southeast of Lower Saxony as well as in the Luneburg Heath, which is part of the lowlands (Fig. 6.1b).

Species richness patterns along the landscape modification gradient

In all species groups, species richness differed significantly between the clusters (Fig. 6.2, Table A.9 in Appendix A). Richness of the total number of species reported for Lower Saxony decreased from the strongly modified landscape of cluster A to the grassland-dominated landscape of cluster C and then increased to clusters D, E and F. For the latter clusters, species numbers were highest in the cropland-dominated cluster D, followed by cluster F that was characterised by large proportions of forests (Fig. 6.2).

Corresponding patterns could be observed for the species groups of native, red-listed, forest and EIV species N1-3. While native and forest species followed the pattern of all species, the trend observed for Red List species and those species linked to nutrient-poor habitats showed a 'V shape' with the highest species numbers reported for clusters A (strongly modified landscape) and F (forest-dominated landscape) and the lowest for cluster C (grassland-dominated landscape).

The peak of species numbers in the strongly modified landscape of cluster A was repeated by the remaining species groups (neophytes, urban and EIV species N7-9), but, in contrast to the other groups, species richness more or less gradually decreased along the cluster gradient. This downward trend was only disrupted by the cropland-dominated cluster D that showed slightly higher species numbers than the surrounding clusters (Fig. 6.2).
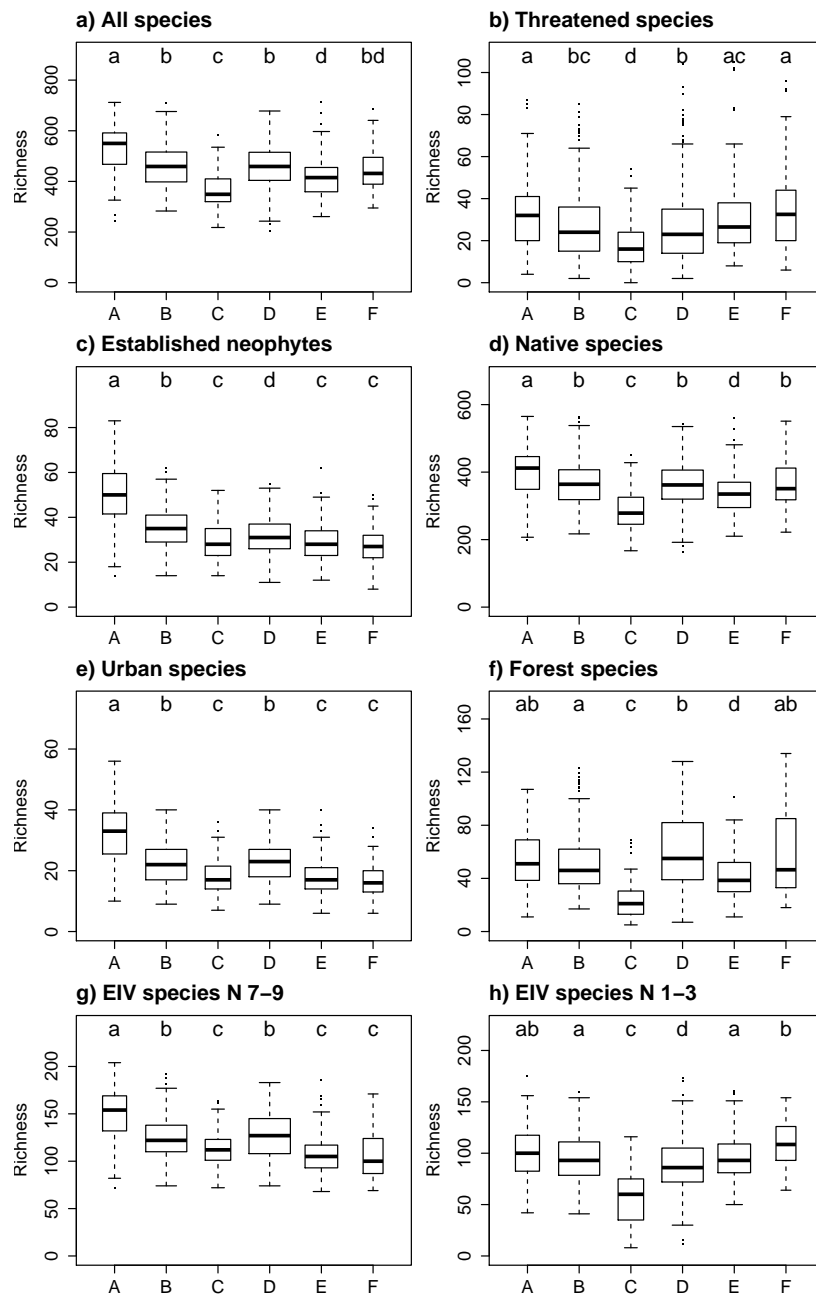
Figure 6.2: Differences in vascular plant species richness between different clusters resulting from k-means cluster analysis of 1383 landscape units (each c. 30 km^2) in Lower Saxony, Germany (see also Table A.9 in Appendix A): (a) The total number of species recorded in Lower Saxony, (b) threatened species, (c) established neophytes, (d) native species, (e) urban species, (f) forest species, (g) species linked to nutrient-rich habitats (EIV species N 7-9) and (h) species linked to nutrient-poor habitats (EIV species N 1-3). Boxes are drawn with widths proportional to the square-roots of the number of observations in the groups; different letters indicate significant differences between clusters (p≤0.05, non-parametric Kruskal-Wallis test).

Table 6.4: Scores of the first ordination axis of a PCA using the standardized landscape metrics calculated for 1383 landscape units in Lower Saxony, Germany. The proportion of urban areas (prop_urban) was found to be the most important variable to describe the underlying gradient, thus clusters were sorted using this variable.

Landscape metric	Scores PCA Axis 1
prop_urban	0.37
prop_traffic	0.35
all_SHDI	0.35
prop_industry	0.29
crop_NP	0.27
grass_NP	0.26
prop_grass	0.21
prop_water	0.20
all_ShapeAm	0.11
for_NP	0.09
prop_others	0.09
prop_crop	0.04
prop_forest	-0.20
meshsize	-0.34
all_AreaAm	-0.36

Discussion

Levels of landscape modification

Our analysis revealed a gradient of landscape modification ranging from highly-fragmented and urbanised (cluster A) to little-fragmented landscapes with a large proportion of forests (cluster F). If one compares the particular clusters with the levels of landscape modification proposed by FORMAN & GODRON (1986), cluster A is an 'urban landscape' as the landscape units of this cluster are dominated by artificial, man-made elements (Table 6.5). In contrast, cluster F resembles the 'managed landscape' of FORMAN & GODRON (1986) (Table 6.5). The landscape units of this cluster were chiefly covered by forests and hosted only a very small proportion of settlements. As Lower Saxony has a very long colonization history that affected and transformed all natural habitats such as primeval forest, bogs and marshland, no 'natural landscapes' according to FORMAN & GODRON (1986) can nowadays be found.

In cluster B, all LCTs occurred at medium proportions and landscape fragmentation was quite high. The definition of 'suburban landscapes' by FORMAN & GODRON (1986) fits reasonably well to this cluster even if the landscape units were in several cases, but not necessarily, situated close to larger cities and, in the strict sense, are not 'suburbs'. Nevertheless, the authors define 'suburban landscapes' as "a town and country area with a heterogeneous, patchy mixture of residential areas, commercial centres, cropland, managed vegetation, and natural areas" (FORMAN & GODRON, 1986, p. 286), which is a good description of the situation found in the landscape units of this cluster.

The naming of the following three clusters is more difficult. Cluster C was mainly characterised by a very large proportion of grasslands and water bodies. It dominates the coastal area as well as the north-western parts of the lowlands. Its landscape is highly transformed; most natural elements, i. e., marshlands and peat bogs that had been widespread in the past have been transformed by peat digging and drainage (BEHRE, 2008). The grasslands resulting from these landscape transformations are intensively managed and interspersed with drainage ditches. Cluster D, dominated by cropland, represents those parts of the study area with intensive agricultural use. Croplands prevailed in the flatter areas while ridges were usually covered by forests. The landscape of cluster E showed a patchy mosaic of cropland, grassland and forest with relatively few settlements. All three clusters correspond to the 'cultivated landscape' of FORMAN & GODRON (1986). This shows that the simple 5-level concept of 'landscape modification' proposed by FORMAN & GODRON (1986) does not adequately differentiate our clusters when it comes to cultivated landscapes. The 10-level concept of landscape naturalness of MACHADO (2004) that allows for a more explicit differentiation seems to be more appropriate in our context. Following MACHADO (2004), the cultivated landscapes of our study may be thus better characterised as highly intervened (cluster C) to cultural assisted systems (cluster E; Table 6.5).

Species richness patterns along the landscape modification gradient

The clusters of landscape modification intensity found in this study were highly relevant for species richness patterns. Plant species numbers in all groups significantly differed between clusters. Species richness of the total number of species reported for Lower Saxony was found to peak in urban (cluster A) followed by suburban (cluster B) and managed landscapes (cluster F) whereas species richness was lowest in highly intervened landscapes (cluster C). This trend was repeated by two of the species sub-groups (native and forest species) whereas species richness of low-nutrient and threatened plants followed a 'V shaped' trend with highest species numbers in urban and managed and lowest in highly intervened landscapes. Species richness of the remaining three sub-groups (neophytes, urban and high-nutrient plants) followed a downward trend throughout from urban to managed landscapes. These results confirm our third hypothesis that species numbers of different plant species groups show different patterns along the gradient of landscape modification. The results illustrate that it is important to consider not only the entire species pool of an area but rather a number of sub-groups selected using different criteria (cp. KOWARIK, 1990).

The general peak in species numbers in urban (cluster A) and suburban landscapes (cluster B) observed for all species groups is unsurprising. Several authors have already highlighted the high species diversity of urban areas (KOWARIK, 1995; MILLER et al., 1997; HONNAY et al., 2003), which is mainly due to the higher landscape/habitat diversity (HONNAY et al., 2003; KÜHN et al., 2004) and facilitated plant immigration in urban areas (KOWARIK, 1995). In contrast, low species numbers in all groups were

found for the highly intervened landscape of cluster C. This is most likely a result of the cluster's low habitat diversity with its large proportion of transformed and intensively managed grassland areas interspersed with only few other LCTs and a low overall species diversity. The elevated species numbers in cluster D (cultural assisted to highly intervened system) reported for almost all species groups might be due to the spatial arrangement of the landscape units belonging to this cluster: Many of the landscape units are situated in the upland area and the eastern part of the lowlands with an overall larger species pool largely due to higher landscape heterogeneity (cp. SCHMIEDEL et al., 2011). Neophytes and high-nutrient plants might furthermore benefit from the large amount of intensively managed agricultural areas. The richness of forest species is also elevated in cluster D, which at first glance is surprising. This pattern may be explained by the fact that woodland patches suitable for forest plants are extant and widespread where agricultural land use was never, or is no longer, profitable.

The results show that the occurrence of non-native and urban species as well as species associated with nutrient-rich habitats is related to landscapes strongly influenced by urbanisation, fragmentation and eutrophication. For neophytes several studies confirm this trend (e. g. KOWARIK, 1990; ROY et al., 1999; KÜHN et al., 2003). By contrast, the number of native, threatened and even low-nutrient species is high in urban areas but also on the opposite side of the gradient, i. e., in the managed landscape less influenced by man. This is due to the aforementioned increased habitat diversity in urban areas that was shown to support even vulnerable species (HONNAY et al., 2003) and, regarding the other side of the landscape gradient, to the fact that largely unmodified landscapes are favourable to native plants and low-nutrient indicators (KOWARIK, 1990; FISCHER & LINDENMAYER, 2007). LENZEN et al. (2009) reported that richness of threatened species was negatively related to the amount of croplands and permanent pastures, whereas the amount of non-arable land and natural forests was positively related to threatened plant species richness. Interestingly, forest species and species linked to nutrient-poor habitats follow the same trend and show relatively high species numbers in urbanised and highly fragmented landscapes. As for forest plants, apart from remaining woodland patches, they may find refuge in urban woods and parks. Plants of nutrient-poor environments with low competitive ability are found in urban areas on gravelly and abandoned sites such as railway areas and industrial fallows. In accordance with our results, HONNAY et al. (2003) found the number of low-nutrient indicator species to be positively correlated with the amount of built-up area but did not give an explanation for this.

Table 6.5: Levels of landscape modification found for 1383 landscape units in Lower Saxony, Germany (clusters A-F) in comparison to the 'landscape modification gradient' levels by FORMAN & GODRON (1986) and the 'index of naturalness' of MACHADO (2004).

Level	Cluster ID	Description	Forman & Godron (1986)	Machado (2004)
1	Cluster A	Urban landscape, highly transformed, highly fragmented	Urban landscape	Semi-transformed to transformed system
2	Cluster B	Transformed, highly fragmented landscape with a patchy mixture of intensively managed croplands and grasslands interspersed with urban areas	Suburban landscape	Highly intervened to semi-transformed system
3	Cluster C	Transformed landscape where (semi-)natural landscape elements have been removed and water dynamics manipulated	Cultivated landscape	Highly intervened system
4	Cluster D	Transformed landscape with a large amount of intensively managed croplands (deep ploughing, fertilization); semi-natural landscape elements remained where agricultural use is not profitable	Cultivated landscape	Cultural assisted to highly intervened system
5	Cluster E	Transformed landscapes with a patchy mixture of managed systems like croplands and grasslands as well as forests	Cultivated landscape	Cultural assisted system
6	Cluster F	Managed, potentially self-maintained system (forest-dominated) with a comparatively low degree of landscape fragmentation	Managed landscape	Cultural self-maintained to cultural assisted system

Usefulness and transferability of the methodological approach

To establish the levels of landscape modification, our approach simply applies information on landscape composition and configuration in combination with k-means-clustering. Thus, it involves neither *a priori* assumptions about the hemeroby or naturalness of the LCTs under consideration (as in WALZ & STEIN, 2014) nor a subjective classification of levels of naturalness (as proposed by MACHADO, 2004). Therefore, the process of distinguishing levels of landscape modification in our concept is less susceptible to misinterpretation because only the results of the analysis and no intermediate steps need interpretation. As the levels of hemeroby of WALZ & STEIN (2014) were determined for the whole of Germany, including Lower Saxony, their results (especially those obtained for the 5 km grid) can be directly compared with those of our study. The comparison shows that the patterns derived from the two different approaches resemble each other. Both ways identify the human-dominated landscapes around the larger cities in contrast to the mainly forested areas in the southern and eastern parts of the state and designate areas under intense agricultural management.

When it comes to long-term monitoring of landscapes the concepts of WALZ & STEIN (2014) and MACHADO (2004) are superior to ours as their results may be easily compared for different time-steps as long as the same study units are used. In our approach, such results might not be directly comparable as changes in landscape composition and configuration will result in changes in the number and delimitation of clusters. Thus, our methodological approach may be applied when a 'snapshot' of the landscape status is needed and/or if due to a lack of more comprehensive information on landscape conditions more detailed assessments cannot be performed.

Similar to the concepts of WALZ & STEIN (2014) and MACHADO (2004), our approach can easily be extended to other spatial scales. The ideal scale for an analysis can be selected that fits the particular research questions and/or the spatial resolution of the available base data. The resolution of our study was restricted by the size of the landscape units adopted by the plant survey of Lower Saxony. Higher resolution through smaller landscape units and more detailed differentiation of LCTs would increase the accuracy of the assessment. With enhancing differentiation of LCTs even small-scale habitats of potential importance for nature conservation (e. g. rocky outcrops, hedgerows) could be taken into account. Nevertheless, our study showed that interpretable results can be obtained with less detailed base data.

Conclusions

Despite the fact that our approach used simple landscape metrics and was conducted at a comparatively coarse scale of c. 30 km^2, we were able to establish a 'gradient of landscape modification' that was shown to be meaningful for species richness. The gradient spans from highly fragmented, urbanised to less fragmented landscapes with a large proportion of forests. It was assessed without making *a priori* assump-

tions about the hemeroby or naturalness of the LCTs under consideration. The simplicity of the approach makes it transferable to regions and countries for which there is no comprehensive information on landscape conditions. The methodology can easily be extended to other spatial scales. Those parts of the landscape which are most valuable for nature conservation can be identified and preselected. It should, however, be kept in mind that the coarse scale of the analysis leaves small-scale habitats of potential importance for nature conservation undetected. We could further show that it is important to employ a set of different species groups to avoid misleading conclusions drawn from richness patterns of a single possibly inadequate group. Suitable species groups to be considered should include indicator and threatened species important for nature conservation as well as species that might pose a threat to other species or ecosystems. As high-resolution data on recent land cover (for an overview see MANAKOS & BRAUN, 2014) and the distribution of vascular plant species are becoming increasingly available in more and more countries (see Table 6 in CULMSEE et al., 2014) our approach is easily applicable to other large areas in Europe.

Acknowledgement

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7 | Synthesis

The analyses performed in the context of the present thesis aimed at improving the understanding of the relationship between species richness patterns and their underlying processes at the landscape scale with special emphasis on human influences. In particular, the analyses (a) identified and described the patterns of vascular plant species richness in the federal states of Lower Saxony and Bremen, Germany (chapter 3), (b) evaluated the adequacy of the protected area network for the protection of rare and threatened plant species in the lowland region of the study area (chapter 4) and (c) analysed the relationship between richness patterns and the human influence on the landscape (chapter 5 and 6).

The following sections recapitulate the key findings of the four main chapters of the thesis (section 7.1) and review the transferability of the methodologies applied in the thesis to other study regions (section 7.2). Furthermore, the limitations of landscape-level approaches are discussed (section 7.3). The chapter closes with concluding remarks and the open questions that build on this research (section 7.4).

7.1 Key findings and implications

7.1.1 Distinct patterns of vascular plant species richness in Lower Saxony and Bremen

The analysis presented in chapter 3 revealed a heterogeneous distribution of species richness across the study area for all six species groups considered in the study. Species numbers showed a remarkable increase from north to south and – to some extent – also from west to east. The spatial distribution of the centres of plant species richness of the different groups largely correlated with each other. However, also some interesting differences in these patterns were observed. Richness patterns of all species established in the study area largely corresponded to the patterns found for native and non-threatened species. An aggregated distribution with high overall species numbers were reported for the upland area and – with the exception of the major stream valleys of the Weser, Aller and Elbe – a much lower overall richness in the coastal and lowland area. Archaeophyte richness was particularly concentrated in the coastal area of Bremen, the Weser and Aller plains, and the loess landscapes of the lowland area[1] whereas the richness of established neophytes was highest in urban centres.

Richness patterns of threatened plants – in contrast to the patterns of all other groups – were found to be spatially highly differentiated: at the coast only the islands in the North Sea represented small hotspots, while in the lowlands relatively large hotspots were situated in the Wendland, the Luneburg Heath and the Elbe-Weser triangle north of Bremen. In the uplands, the area around Göttingen, the uplands of

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[1]German: *Börden*

the Weser and Leine valleys and the margins of the Harz Mountains represented well defined hotspots of Red List species.

A general trend that was observed across all species groups was that grid cells inside which urban centres were situated were found to be particularly rich in species. This relationship was especially evident for neophytes but weak for threatened species.

The presented approach can support systematic conservation planning by identifying landscape units rich in species that should receive particular attention. However, in doing so not only the total number of species but also different groups of species should be considered (cp. subsection 7.1.4). HAEUPLER & VOGEL (1999) and HAEUPLER (2000) suggest that hotspots of threatened plant species are able to reveal the "real" hotspots of vascular plant species richness and thus provide a suitable species group to be used in the determination of landscape units of high conservation value (DU-ELLI & OBRIST, 2003). Indeed, richness patterns of threatened species were found to be spatially much more differentiated than those of the other species groups considered in the analysis.

7.1.2 Effective 'territorial' protection of hotspots of threatened plant species

In contrast to the results of similar studies conducted for different regions of the world that identified numerous gaps in existing protected area networks (see e.g. SCOTT et al., 2001; ANDELMAN & WILLIG, 2003; DEGUISE & KERR, 2006; RODRIGUES et al., 2004), the analysis on the effectiveness of the protected area network for the conservation of threatened plant species in the lowland region of the study area confirmed a good overall coverage of the hotspots of threatened species. In particular, richness centres of the total number of threatened species were covered by nature protection areas in 85 % of the 41 geographical sub-regions considered in the analysis. The proportion of well covered hotspots increased to an effective protection in 95 % of the geographical regions when including Natura 2000 sites into the analysis. A repetition of the analyses for threatened species linked to forest habitats showed that richness centres of these species were effectively protected by nature protection areas in only 60 % of the regions; an inclusion of Natura 2000 sites increased the proportion to 81 %. However, the proven 'territorial' protection of these hotspots will only be able to preserve viable populations of the species if the protected areas are managed in an appropriate way. For many of these areas, legal regulations controlling the management of the areas are expressed very vaguely[2] or their development is still under way[3]. To effectively protect viable populations of species in the future, these regulations should be adjusted to be able to support the particular species of conservation concern. Repetitive studies in future years will then need to show if the effective 'territorial' protection is resulting in an effective species protection.

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[2]This is especially the case for nature reserves (*Naturschutzgebiete*).

[3]This is mainly the case for Natura 2000 sites as these areas have been designated quite recently.

As many protected areas are isolated and small in size their effectiveness in supporting species might be limited. Therefore, areas outside reserves should also be managed in a way that supports threatened species, i.e. a "dual conservation strategy" according to JACKSON et al. (2009) should be pursued.

The methodology applied in the study was shown to be suitable to evaluate the effective 'territorial' protection of hotspots of threatened species. By transferring it to other regions with little information on the adequacy of the protected area network, it can contribute to systematic conservation planning.

7.1.3 Human impact matters: Plant species richness along gradients of landscape modification and fragmentation

The relationship between species richness patterns and human activities at the landscape scale was studied by evaluating the influence of the degree of landscape modification in general and landscape fragmentation in particular. The following two subsections highlight the key findings of chapters 5 and 6 dealing with these topics.

The *Effective Mesh Size Index* as an ecologically meaningful measure of landscape fragmentation   The analysis presented in chapter 5 of the thesis demonstrated that the *Effective Mesh Size Index* (m_{eff}) can act as an ecologically meaningful measure of landscape fragmentation as long as relevant fragmentation geometries are chosen. Landscape fragmentation expressed by m_{eff} in the study area was found to vary largely between landscape units and was identified to significantly affect the richness of almost all plant species groups considered in the analysis. The size of the effect differed considerably between the groups and was largest for neophytes followed by archaeophytes. In contrast, richness patterns of native species were only slightly affected whilst those of threatened species were not affected. Instead, the variation partitioning approach for the latter species groups clearly highlighted the importance of the effect of environmental factors.

All species groups, except threatened species, showed a negative correlation with m_{eff}, i.e. richness was highest in highly fragmented and lowest in least fragmented grid cells.

In the analysis, five different m_{eff} types calculated using different fragmentation geometries were tested. Out of these, only two were found to be significant in explaining species richness patterns, namely those that used urban-industrial areas, traffic infrastructure and water bodies as fragmenting elements.

The findings of the study support the untested assumption of several researchers (e.g. JAEGER, 2000; JAEGER et al., 2008 and GIRVETZ et al., 2008) that m_{eff} is an ecologically relevant measure of landscape fragmentation. The presented approach might help future studies to determine correct fragmentation geometries to use with m_{eff} and might facilitate the unravelling of fragmentation impacts at the landscape scale. As m_{eff} in Germany is regularly assessed as one of two national indicators of land-

scape fragmentation in context of the National Strategy on Biodiversity the results obtained in the analysis are especially relevant as a contribution to future assessments.

Variation in species richness along a gradient of landscape modification Chapter 6 of this thesis applied simple information on land cover and easily measurable landscape metrics to determine a 6-level gradient of landscape modification spanning from highly fragmented urban landscapes to little fragmented ones with a large proportion of forests. Species richness of the eight groups of vascular plants considered in the analysis significantly differed along the gradient. For the entire and native species pool as well as for forest, low-nutrient indicating and threatened plants, species numbers peaked at the opposing ends of the modification gradient. For neophytes, urban plants and high-nutrient indicators, richness decreased along the gradient from high species numbers in urbanised to low numbers in less modified landscapes.

The study showed that concepts of landscape modification can be successfully linked to measures of species richness/diversity that were tested empirically here for the first time. This finding is of particular importance as concepts describing the human influence on the landscape in Germany are planned to be integrated into the nationwide monitoring system (WALZ & STEIN, 2014). In this context, knowledge on the actual relevance of such concepts for biodiversity might therefore further increase their usefulness.

The strength of the analysis is, that the assessment of the landscape modification gradient neither involved *a priori* assumptions about the hemeroby or naturalness of the land cover types under consideration (as in WALZ & STEIN, 2014) nor a subjective classification of levels of naturalness (as proposed by MACHADO, 2004). Therefore, the process of distinguishing levels of landscape modification is less susceptible to misinterpretation than the concepts of WALZ & STEIN (2014) and MACHADO (2004) because only the results of the analysis and no intermediate steps need interpretation. Due to its coarse scale, the analytical approach is transferable to other regions and countries for which there is no comprehensive information on landscape conditions. Following interpretation by experts, the methodology developed in the study allows for the identification and preselection of those parts of the landscape that are most valuable for nature conservation.

7.1.4 Overall key findings derived from the studies

In addition to the individual key findings of the four studies presented above, two further overall findings of all the analyses conducted in the present thesis can be derived that will be discussed below.

Habitat heterogeneity as a key driver of species richness in urban areas? A general emerging trend that was observed in the conducted studies on landscape fragmen-

tation and modification but also in the first study on species richness patterns, was the relationship between species richness and urban land cover: The latter together with the degree of landscape fragmentation which is usually tightly linked with the degree of urbanisation were both found to be positively associated with the richness of all species groups. The only exception being the richness of threatened species that was found to be independent of landscape fragmentation. Therefore no direct negative effects of human land use on species richness patterns could be identified. However, these results should not be used to conclude that intensive human land use increases the richness of species[4]. Instead, it seems that the elevated habitat heterogeneity of urban areas as proposed by several researchers (Haeupler, 1974; Kühn et al., 2003; Wania et al., 2006; Lippe & Kowarik, 2008) balances or even outbalances the assumed negative effect of landscape modification and fragmentation as long as suitable habitats are not completely removed (cp. Honnay et al., 2003). Therefore it can be concluded that it is important to conserve the high habitat diversity of urban areas and to prevent their homogenisation.

Quality vs. quantity: avoiding drawing the wrong conclusions The analyses conducted in the framework of the thesis produced results that differed considerably between the various species groups examined in the individual studies. The richness of the groups showed different patterns according to the distribution in the study area, the coverage by protected areas and the relationship with landscape modification and fragmentation. These findings clearly emphasise the importance of considering not one single but rather an array of different species groups using various criteria. By doing so, not only quantitative but also qualitative aspects can be considered (cp. subsection 7.1.2) which are not biased to protecting mainly easily recorded or common species at the cost of a wider range of biodiversity (cp. Kowarik, 1990; Duelli & Obrist, 2003; Schetter et al., 2013). In this context, for example, the residence or threat status of species[5], their habitat preferences[6] or a combination of these criteria[7] can be used to assemble groups of species.

7.2 Transferability of the methodological approach

The analyses applied in the framework of the present thesis were conducted on a regional scale covering the area of Lower Saxony and Bremen. For this area detailed information on species occurrences and data on environmental conditions and landscape composition as well as configuration were available.

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[4]Extensive management as applied in traditional cultural (pre-industrial) landscapes, in turn, is known to be beneficial for biodiversity (EEA, 2004; Angelstam, 2006)

[5]As applied in Schmiedel et al. (2011, 2013, submitted) and Schmiedel & Culmsee (submitted).

[6]As applied in Schmiedel et al. (2013) and Schmiedel et al. (submitted).

[7]As applied in Schmiedel et al. (2013).

If suitable base data are at hand, the methodological approaches applied in the present thesis can be transferred to other – even larger – areas. The most important (and often the rarest) data in this context are area-wide species records. The records need to be reported on a grid- or point-basis[8] and have to be of high quality (cp. subsection 7.3). Additionally, data on land cover and the abiotic environment are needed.

Transferability to Germany and its federal states For most of the federal states of Germany, data comparable to those used in this thesis are available that would allow a transfer. High-resolution data on land cover and topography in this thesis were taken from ATKIS-DLM and ATKIS-DEM (see section 2.2) which are both available for the whole country. Climatic and soil data can be obtained in high resolution for every federal state from the German Weather Service and the regional authorities, respectively. Concerning species records, most federal states maintain their own mapping projects (see Table 7.1). The resolutions of these mapping approaches are similar to that used for Lower Saxony or even higher (e.g. c. 8 km^2 grid cells for North Rhine-Westphalia, Saxony and Thuringia), the latter enabling even more detailed assessments. However, the completeness and reliability of the floristic datasets resulting from these projects would have to be tested before using them in further analyses.

With the distribution atlas of the vascular plants of Germany having been published recently (NETPHYD & BFN, 2013) reliable data on species occurrences are also available for the whole of Germany. Species data of this database are reported at the same resolution as the data for Lower Saxony (*Messtischblatt-Quadranten*, 1/4 of a topographical map). Therefore, the methodologies applied in this thesis are directly transferable to the whole country area.

Transferability to European countries Similarly to Germany, most countries in Europe maintain species mapping projects. Out of the projects listed in Table 7.1 for data acquisition only Poland uses a grid less precise than that utilised in this thesis. All other countries apply more or less equal or even higher resolutions than used in this study. As suggested for species data of the federal states of Germany, prior to an analysis, it needs to be checked whether the available information is reliable and complete.

Additional information on land cover[9] and environmental conditions should be available for most European countries and therefore methods might be transferred to these areas as well.

The database of the *Atlas Flora Europaea* (JALAS & SUOMINEN, 1972-2013) that covers about 20 % of the European flora (FINNISH MUSEUM OF NATURAL HISTORY, 2014) pro-

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[8]Point data can be transferred to grid-based data of varying resolutions.
[9]For an overview on existing data on land cover in Europe see MANAKOS & BRAUN (2014).

Table 7.1: Overview of the largest data bases on the distribution of vascular plant species in Europe, their reference areas (coverage), resolution and number of entries. Modified after CULMSEE et al. (2014, Table 6). n/a = not available.

Coverage	Resolution	Entries	Source
Europe-wide mapping project			
Europe[a]	Grid, c. 250 km^2	n/a[b]	Atlas Flora Europaea[c]
Country-wide mapping projects in Europe			
Austria	Grid, 35 km^2	c. 1,950,000	Project Mapping the Flora of Austria[d]
Belgium	Grid, 1 km^2	> 3,300,000	Florabank1 database (LANDUYT et al., 2012)
British Isles	Grid, 10 km^2	c. 9,800,000	Botanical Society of the British Isles[e]
Denmark	Grid, 25 km^2	> 1,000,000	Dansk Botanisk Forening[f]
Finland	Grid, 0.01 km^2, 1 km^2, 100 km^2	c. 5,700,000	Kasviatlas[g]
Germany	Grid, c. 30 km^2	c. 30,000,000	Verbreitungsatlas der Farn- und Blütenpflanzen Deutschlands[h] (NetPhyD & BfN, 2013)
Netherlands	Provinces, point coordinates	> 10,000,000	FLORON, Foundation for floristic research in the Netherlands[i] (1989 onwards)
Poland	Grid, 100 km^2	> 4,500,000	ATPOL Atlas of distribution of vascular plants in Poland (ZAJAC & ZAJAC, 2001)
Switzerland	Grid, 1 km^2	c. 3,000,000	InfoFlora National data and information centre of the flora of Switzerland[j]
State-wide mapping projects in Germany			
Baden-Württemberg	Grid, c. 30 km^2	c. 1,330,000	State Floristic Survey[k]
Bavaria	Grid, c. 33 km^2	> 1,000,000	Bavarian Floristic Survey[l]
Lower Saxony and Bremen	Grid, c. 30 km^2	2,395,045	State Inventory Program NLWKN (1982–2003), this thesis
Mecklenburg-West Pomerania	Grid, c. 30 km^2	>1,917,600	Flora of Mecklenburg-West Pomerania[m]
North Rhine-Westphalia	Grid, 8.4 km^2	c. 2,000,000	State Inventory Program (HAEUPLER et al., 2003)
Saxony	Grid, c. 8 km^2	c. 1,536,700	Project Floristic Atlas of Saxony (HARDTKE & IHL, 2000)
Thuringia	Grid, 7.8 km^2	> 1,800,000	Project Flora of Thuringia (KORSCH et al., 2002)

[a] For a complete list of countries see http://www.luomus.fi/en/atlas-florae-europaeae-afe-distribution-vascular-plants-europe; [b] 20% of the European flora is mapped (FINNISH MUSEUM OF NATURAL HISTORY, 2014); [c] http://www.luomus.fi/en/atlas-florae-europaeae-afe-distribution-vascular-plants-europe; [d] http://www.botanik.univie.ac.at/plantchorology/atlases.htm; [e] http://www.bsbi.org.uk; [f] http://www.botaniskforening.dk; [g] http://koivu.luomus.fi/kasviatlas; [h] http://www.deutschlandflora.de; [i] http://www.floron.nl; [j] http://www.infoflora.ch; [k] http://www.flora.naturkundemuseum-bw.de; [l] http://www.bayernflora.de; [m] http://www.flora-mv.de

vides data on species distributions for the whole of Europe on a grid of c. $250\,km^2$, i. e. a resolution much coarser than that used in the present thesis. Moreover, high-resolution data on land cover for the whole area of Europe are not as easy to obtain as for the particular countries. However, information on land cover on this scale could be derived from more coarse-scaled data available for the overall area. 'Corine Land Cover 2006' (EEA, 2013) in this context provides a remote-sensing-derived raster data set on land cover with a pixel size of $100\,m \times 100\,m$ that can be used for research purposes free of charge. Information on topography and climatic conditions available on a resolution of $1\,km \times 1\,km$ can be derived from Worldclim data (Hijmans et al., 2013). Therefore, potentially suitable data are available that could form the basis of analyses similar to that applied in this thesis. Because of the large differences in data resolution the results obtained from such analyses would probably, however, vary largely from those obtained at the regional scale.

Transferability to countries outside Europe   The availability of high-resolution plant records for countries outside Europe is limited. However, for some areas, e. g. the state of California, USA (The Calflora Database, 2014) grid-based species records are acquirable. For other states of the USA (e. g. Alabama, New York, Florida, see Alabama Herbarium Consortium, 2014; New York Flora Association, 2014; University of Florida, Institute for Systematic Botany, 2014) as well as for China (Yang et al., 2013) such information are not available on a grid basis but on county-level only.

Information on species records from other countries unfortunately are very hard to obtain. The transferability of the methods applied in this thesis to areas outside of Europe is therefore limited.

7.3 Limitations of landscape-level approaches

Even if frequently shown to be very powerful, landscape-level approaches are subjected to several inaccuracies arising from the scale chosen for the particular analysis, the inclusion of biased base data and inaccuracies provoked by generalisations needed to perform the analyses. Gotelli et al. (2009) suggest that a large number of potential sources of error are linked with each data layer included in the analysis.

7.3.1 Bias due to generalisations, data resolutions and spatial scale

The representation of 'real world' conditions and patterns in 'computer space' is always subject to necessary generalisations (i. e. delimitation and discretisation processes) that help to pool the vast amount of available information. It is important that the magnitude of generalisation applied to the data should match the processes under consideration (Lang & Blaschke, 2007). The ATKIS-DLM data used in the present thesis represent a good example for generalisations: The DLM defines hard

boundaries along more or less homogeneous landscape elements that in most cases can not fully reflect the real conditions, as the boundaries between many land cover types (especially natural habitats) are usually more continuous. Because the latter cannot be represented cartographically and furthermore would provoke problems in data analysis, the hard boundaries are accepted as the best approximation of reality (GREEN & HARTLEY, 2000).

Generalisations affect both the spatial and the thematic resolution of the data. The latter describes the way in which the geo-spatial data are categorised[10]. The spatial resolution of an analysis, in turn, is described by the extent of the study area as well as the grain size[11]. It was shown to affect all levels of data, i. e. species, land cover and environmental data.

In this context, several studies have shown that patterns of species richness/diversity change with the scale of observation or analysis (PALMER & WHITE, 1994; HE et al., 2002; KALLIMANIS et al., 2008) and therefore "areas of high species richness at one scale may appear of low species richness at the other scale" (KALLIMANIS et al., 2008, p. 148). This particular behaviour is attributable to the fact that species richness is non-additive when aggregated across different spatial scales (HE et al., 2002).

Most studies conducted at the landscape scale make use of land cover data. Usually, landscape metrics in this context are used to quantify the composition and configuration of a landscape unit. As these indices are assumed to be strongly scale-dependent, a lot of attention has focussed on the behaviour of the metrics across changing thematic and spatial resolutions. Within these studies, the thematic resolution was found to largely affect the outcome of most landscape metrics (BAILEY et al., 2007; BUYANTUYEV & WU, 2007). Related to this, the usefulness of the indices in describing landscape characteristics at varying thematic resolutions was found to largely depend on resolution: Whilst some indices are more useful at a coarser thematic resolution others are more suitable at finer scales (BAILEY et al., 2007). Similarly, only few landscape metrics have been identified to behave consistently across different spatial resolutions (TURNER et al., 1989; SAURA & MARTINEZ-MILLÁN, 2001; WU, 2004). In this context, TURNER et al. (1989) identified an increase in grain size to be responsible for the loss of rare and small-scale land cover types in the landscape. SCHINDLER et al. (2013) studied the performance of landscape metrics as indicators of species richness of various taxonomic groups across multiple scales. They found the effects of landscape structure expressed in terms of landscape metrics on species richness to strongly depend on the spatial scale of the analysis.

Finally, spatial scale also plays a role in the analysis of environmental conditions. According to SIEFERT et al. (2012) both the extent and grain size may influence the amount of variance observed in a given environmental factor. Due to the spatial au-

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[10]cp. subsection 2.2.2 for the categorisation of DLM data applied in the present study
[11]i. e. the size of the smallest unit of observation, (PALMER & WHITE, 1994; TURNER, 2005)

tocorrelation observed in most environmental variables, their variance is supposed to increase when the extent of the study area is enlarged (WIENS, 1989; SIEFERT et al., 2012). However, the size of this effect largely varies between different environmental variables. SIEFERT et al. (2012) distinguish coarse- and fine-grained variables. The former include e. g. climate data which vary across broader scales, the latter are represented by e. g. edaphic variables that may show a large variation even within a small area.

The aforementioned examples clearly illustrate the importance of scale. However, the knowledge of this scale-dependence of landscape-level approaches can only rarely be included in research as the spatial extent and grain of an analysis are usually determined by the availability of the base data.

7.3.2 Bias due to data format

"Yes, raster is faster, but raster is vaster, and vector just seems more corrector."

C. Dana Tomlin

For environmental assessments using GIS, both vector and raster data are commonly used (WADE et al., 2003). Usually, vector data are referred to as allowing high levels of cartographic accuracy[12] (WYATT & RALPHS, 2003). In contrast, raster data are supposed to be less precise than vector data as it is difficult to "represent objects with a sufficient level of cartographic precision using raster methods, since the resolution of the raster grid needs to be prohibitively high to capture an equivalent level of detail" (WYATT & RALPHS, 2003, p. 49). Nevertheless, raster data are often used in all kinds of studies because in many cases they represent the only data source available and are faster to process (LANG & BLASCHKE, 2007). For the present thesis, several data sources were available in raster format (e. g. DEM and climate data, see Table 2.1). Furthermore, some vector data (DLM) had to be converted to raster format as the software used for the subsequent analysis (Fragstats 3.3, MCGARIGAL & CUSHMAN, 2002) was limited to this type of data. In transforming the data, an additional bias is introduced because a balance between resolution and file size has to be found: If pixel size is too large, the data set will suffer from cartographic imprecision, whilst a very high resolution will generate a file too large for computation (WYATT & RALPHS, 2003). In the present thesis the resolution of the transformed data was set to a pixel size of 25 m^2. This value was chosen because of hardware limitations (cp. subsection 2.2.5). However, for the analyses conducted in context of this thesis this resolution should be sufficient to capture the overall characteristics of the landscape.

However, the pixel size of a raster not only affects the precision of the data set but can also directly influence the calculation of landscape metrics (MCGARIGAL & CUSHMAN, 2002; NEEL et al., 2004). Indices that include the complexity of patches are particularly affected because of the step-wise structure of the patch margins that

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[12]However, this attribute still depends on the accuracy of data acquisition.

results in the overestimation of patch complexity and edge length (Leitão et al., 2006). In the present study this is especially true for the *Total Edge* as well as the *Shape Index* utilised in chapter 6. The latter suffers from one further limitation: Because of the raster format the simplest shape is set to be a square even though in reality circular patches have the smallest perimeter-area relationship and thus experience fewer edge effects.

7.3.3 Bias in species records

In species record data the main problem is the reliability of these data (Haeupler, 2000; Honnay et al., 2003; Diekmann et al., 2008). Frequently, data on species occurrences are incomplete because some areas are better inventoried than others (Barthlott et al., 1996). This problem apparently may provoke misleading results. Thus, it is highly important to check data on species records prior to subsequent analyses. The database utilised in the present thesis was carefully checked for its quality by the responsible authority (Schacherer, 2001), so that the bias related to this particular source of error should be of minor concern.

7.4 Concluding remarks and outlook

Although landscape-level approaches suffer from the "dilemma of indicating complexity with simple measures" (Duelli & Obrist, 2003, p. 95) and are subject to several limitations (cp. section 7.3), they are a powerful tool to disentangle the underlying causes of species richness patterns. As many, if not most, of the processes affecting biodiversity take place at the landscape level obviously they should also be measured at this scale (McGarigal & Cushman, 2002; Fahrig, 2003). However, experimental approaches on such a large scale are rarely possible (Willig et al., 2003; Gotelli et al., 2009), thus statistical approaches making use of existing data sources represent the only feasible method to approach this complex topic.

By applying landscape-level approaches, the present thesis showed that patterns of vascular plant species richness in Lower Saxony and Bremen are largely influenced by anthropogenically-induced land use patterns. However, also the importance of the abiotic environment in shaping species richness patterns was revealed. All analyses were conducted using the data base of the vascular plant survey of Lower Saxony that – due to its high quality and comprehensiveness – in this context provided a solid basis for these assessments.

As already highlighted in the key findings, the individual results obtained from the four studies each contribute to landscape and systematic conservation planning by facilitating the identification and preselection of landscape units of high conservation value and concern. However, by merging the knowledge derived from the studies, an even stronger comprehensive picture of the conservation value of the landscape units can be obtained. In doing so, landscape units of high conservation value, e. g. those

hosting a large number of species but have experienced a low degree of landscape modification and fragmentation, can be identified. In contrast, landscape units of high conservation concern that are species-rich despite being under high pressure from human land use, can be located. It is then possible to assess the protection status of these located landscape units by using the methodology applied in chapter 4 and determine if further conservation measures ought to be taken.

Arising from the discussed limitations and uncertainties connected to landscape-level approaches (cp. subsection 7.3), many open research questions remain that should be examined in more detail in further studies. As the transferability of the methodologies applied in this thesis is high (cp. section 7.2), it is possible to conduct analogue studies that will further increase the understanding of the factors influencing species richness patterns at the landscape scale. The most important and interesting research questions that should be addressed by future research approaches are:

1. How do the results change if the analyses are conducted at different spatial scales (changing grain size, changing spatial extent)?
2. Are the results also observed in other regions and particularly in more natural landscapes with shorter history of cultivation?
3. How do the results change if not species richness but species composition is used as the response variable in the analyses?

Progress in terms of the first research question could be made by using point-based species records, as these can be summarised into grid data of varying resolutions. For the study region of the present thesis such data could become available in the near future as the third survey period of the vascular plant survey of Lower Saxony (2004–2014) is currently assessing point-based species occurrence data.

The second research question could be addressed by transferring the methodologies of this thesis to other federal states of Germany or even other Central European countries. To test if the results are reproducible even in landscapes with a less extensive history of cultivation, a transfer to areas outside of Central Europe, e. g. to one of the northern European countries is needed. The 'Kasviatlas' of Finland (see Table 7.1) in this context could form a suitable basis for such a continuative analysis.

Finally, the third question would be particularly interesting to examine as several researchers (Hillebrand et al., 2008; Crowder et al., 2010; Tscharntke et al., 2012) have suggested that changes in species composition and evenness might have even larger effects on an ecosystem than changes in species richness. However, grid-based data at higher resolution than that utilised in the present thesis would probably need to be used for such an analysis as preliminary analyses conducted in context of this thesis did not return interpretable results.

By approaching these research questions, further progress will be made in the disentanglement of the "high real-world complexity shaping [species richness] patterns" (Tscharntke et al., 2012, p. 678).

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Acknowledgments

Academic CV

Inga Schmiedel

born on 15th of January 1984 in Göttingen, Germany

Education

2009 – 2014:	PhD studies, Dpt. Vegetation & Phytodiversity Analysis, University of Göttingen, Germany.
2012 – 2013:	Completion of a certificate programme on teaching at the center for 'Teaching and Learning in Higher Education', University of Göttingen/TU Clausthal.
2006 – 2008:	Advanced studies of biology, University of Göttingen. Main interests: Vegetation and plant ecology, nature conservation and geography. Topic of the diploma thesis: 'Analyse der Landschaftseinnischung von Kalkmagerrasen am Mühlenberg bei Barterode (Landkreis Göttingen) auf Grundlage einer Biotoptypenkartierung'.
2005 – 2006:	Advanced studies, University of Umeå, Sweden.
2003 – 2005:	Basic studies of biology, University of Göttingen.
2003	Qualification for university entrance (Abitur), Gymnasium Uslar, Germany.

Work experience

2010 – 2014:	Research and teaching assistant, Dpt. Vegetation & Phytodiversity Analysis, University of Göttingen.
2010 – 2012:	Mapping of Natura 2000 habitat types on a military area in Saxony-Anhalt (Co. GISCON, Braunschweig).
2009 – 2010:	Research assistant in the project 'Identification of indicator species groups of grassland and forest habitats for biodiversity monitoring and evaluation', Dpt. Vegetation & Phytodiversity Analysis, University of Göttingen.
2009:	Freelance work (biotope mapping, DBU project 'Nature conservation and cultural conservation in a castle mountain landscape in Thuringia, Germany').
2008, 2009:	Teaching assistant (GIS course), University of Göttingen.
2006 – 2008:	Student assistant at the AvH-Institute for Plant Sciences, University of Göttingen

List of publications

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Schmidt, M.; Mölder, A.; Schönfelder, E.; Engel, F.; **Schmiedel, I.** & Culmsee, H. (2014): Determining ancient woodland indicator plants for practical use: A new approach developed in northwest Germany. *Forest Ecology and Management* **330**: 228-239.

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Schmiedel, I.; Bergmeier, E. & Culmsee, H.: Plant species richness patterns along a gradient of landscape modification intensity in Lower Saxony, Germany. *Submitted*.

Schmiedel, I. & Culmsee, H.: The influence of landscape fragmentation, expressed by the 'Effective Mesh Size Index', on regional patterns of vascular plant species richness in Lower Saxony, Germany. *Submitted*.

Conference proceedings

Schmiedel, I. & Culmsee, H. (2014): The 'effective mesh size index' as a predictor for regional patterns of vascular plant species richness. 44. Annual Meeting of the Ecological Society of Germany, Austria and Switzerland (GfÖ), 8th-12th September 2014. *Verhandlungen der Gesellschaft für Ökologie* **44**, p. 60.

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Appendix A

Table A.1: Bio-geographical regions and natural landscape units of Lower Saxony. Basis: PILGRIM & FRANKE (1993).

Bio-geographical regions	Natural landscape unit	No.
Coast		
1 Watten und Marschen	Emsmarschen	610
	Harburger Elbmarschen	670
	Wesermarschen	612
Lowlands		
2 Ostfriesisch-Oldenburgische Geest	Aschendorfer Emstal	604
	Bourtanger Moor	605
	Hunte-Leda-Moorniederung	600
	Oldenburger Geest	603
	Ostfriesische Geest	602
	Ostfriesische Zentralmoore	601
3 Stader Geest	Achim-Verdener Geest	630
	Hamme-Oste-Niederung	632
	Wümmeniederung	631
	Wesermünder Geest	633
	Zevener Geest	634
4 Ems-Hunte-Geest und Dümmer-Geestniederung	Bersenbrücker Land	585
	Cloppenburger Geest	593
	Delmenhorster Geest	595
	Diepholzer Moorniederung	584
	Lingener Land	586
	Nordhorn-Bentheimer Sandgebiet	580
	Ostmünsterland	540
	Plantlünner Sandebene	581
	Rahden-Diepenauer Geest	582
	Sögeler Geest (Hümmling)	592
	Syker Geest	594
	Westmünsterland	544
5 Lüneburger Heide und Wendland	Hohe Heide	640
	Jeetzel-Dumme-Lehmplatte und Arendseer Platte	861
	Lüchower Niederung	860
	Luheheide	644
	Ostheide	642

Table A.1 – *Continued from previous page*

Bio-geographical regions	Natural landscape unit	No.
	Südheide	641
	Uelzener und Bevenser Becken	643
	Untere Mittelelbe-Niederung	876
	Westaltmärkisches Waldhügelland	865
6 Weser-Aller-Flachland	Burgdorf-Peiner Geestplatten	623
	Drömling	625
	Hannoversche Moorgeest	622
	Loccumer Geest	628
	Mittelweser	583
	Mittleres Wesertal	583
	Obere Allerniederung	626
	Ostbraunschweigisches Flachland	624
	Thedinghäuser Vorgeest	621
	Untere Aller-Talsandebene	627
	Verdener Wesertal	620
Uplands		
7 Börden	Bückebergvorland	522
	Braunschweig-Hildesheimer Lössbörde	520
	Calenberger Lössbörde	521
	Großes Bruch	511
	Nördliches Harzvorland, Westteil	510
	Ostbraunschweigisches Hügelland	512
8.1 Osnabrücker Hügelland	Östliches Wiehengebirge	532
	Lübbecker Lössland	533
	Osnabrücker Hügelland	535
	Osnabrücker Osning	534
	Ravensberger Hügelland	531
	Westliches Wiehengebirge	536
8.2 Weser- und Leinebergland	Alfelder Bergland (mit Ith und Hils)	377
	Eichsfelder Becken (Goldene Mark)	374
	Fulda-Werra-Bergland	357
	Göttingen-Northeimer Wald	373
	Holzmindener Wesertal	367
	Innerstebergland	379
	Kalenberger Bergland	378
	Leine-Ilme-Senke	372
	Lipper Bergland	364
	Oberwälder Land	361
	Pyrmonter Bergland	365
	Rinteln-Hamelner Weserland	366
	Südharzer Zechsteingürtel	485
	Südliches Harzvorland	486
	Südwestliches Harzvorland	376

Table A.1 – *Continued from previous page*

Bio-geographical regions	Natural landscape unit	No.
	Solling, Bramwald und Reinhardswald	370
	Sollingvorland	371
	Unteres Werratal	358
	Weserengtal von Bodenwerder	368
	Westhessische Senke	343
9 Harz	Hochharz	381
	Oberharz	380
	Unterharz	382

Table A.2: List of vascular plant species of the data base of the vascular plant survey of Lower Saxony. The table lists only those species for which occurrences are reported. Extinct species are not listed. Species were aggregated on the highest taxonomical level found in the data base, i. e. subspecies were assigned to the respective species and species were assigned to the respective aggregate if the latter was listed in the data base. 'Edge' denotes that the species is present in the grid cells placed at the border of the state only (cp. section 2.1; Natural. = naturalisation (nat = native taxa, arc = archaeophytes, neo = neophytes); RL NB = Red List status assigned by GARVE (2004), categories 1, 2, 3, G and R denote threatened taxa, category 'V' denotes prewarned taxa, '*' denotes non-threatened taxa (cp. Table 2.2); EIV N1-3/N7-9 = Plant species of nutrient-poor/nutrient-rich habitats (Ellenberg Indicator Values for nutrients 1-3 and 7-9; BFN, 2013); Urban = Plant species of urban habitats (urbanity indicator 4 and 5; KLOTZ et al., 2002); Forest = Herbaceous and shrubby plant species linked to forest habitats (forest affinity categories 1.1 and 1.2; SCHMIDT et al., 2011); Code = Code used in the data base of the NLWKN (1982-2003); Incl. = Codes of (sub-)species assigned to the particular taxon, see Table A.3 for full names of (sub-)species.

Species	Edge	Natural.	RL NB	EIV N1-3	EIV N7-9	Urban	Forest	Code	Incl.
Acer campestre		nat	*					cab	
Acer platanoides		nat	*					cad	
Acer pseudoplatanus		nat	*		x			cae	
Achillea collina		nat	R	x				caf	
Achillea millefolium agg.		nat	*					eab	cag
Achillea pannonica		nat	R	x				aaa	
Achillea ptarmica		nat	*	x				cah	
Acinos arvensis		nat	V	x				aab	
Aconitum lycoctonum		nat	3		x			aac	
Acorus calamus		neo	*		x			cai	
Actaea spicata		nat	*		x		x	aad	
Adonis aestivalis		arc	2	x				aae	
Adonis vernalis		nat	2	x				aag	
Adoxa moschatellina		nat	*		x		x	caj	
Aegopodium podagraria		nat	*		x			cal	
Aethusa cynapium		arc	*					eac	can
Agrimonia eupatoria		nat	*					aah	
Agrimonia procera		nat	*					aai	
Agrostis canina		nat	*	x				cap	
Agrostis capillaris		nat	*					caq	
Agrostis stolonifera agg.		nat	*					ead	cas
Agrostis vinealis		nat	*	x				cau	
Aira caryophyllea		nat	V	x				aal	
Aira praecox		nat	*	x				aam	
Ajuga genevensis		nat	3	x				aao	

Continued on next page

Table A.2 – *Continued from previous page*

Species	Edge	Natural.	RL NB	EIV N1-3	EIV N7-9	Urban	Forest	Code	Incl.
Ajuga reptans		nat	*					cax	
Alchemilla glaucescens		nat	3	x				aap	
Alchemilla gramineum	x	nat	1					aas	
Alchemilla plicata	x	nat	2					223	
Alchemilla vulgaris agg.		nat	*					aaq	cay, caz, cba, cbb,cbc, cbd, cbe, cbf, cbg
Alisma lanceolatum		nat	*					aat	
Alisma plantago-aquatica agg.		nat	*					eae	cbh
Alliaria petiolata		nat	*		x		x	cbi	
Allium angulosum		nat	2	x				aau	
Allium carinatum ssp. *carinatum*		neo	3	x				aav	
Allium oleraceum		nat	*					cbj	
Allium paradoxum		neo	*		x		x	cbk	
Allium schoenoprasum		neo	*	x				aax	
Allium scorodoprasum		nat	3		x			aay	
Allium senescens ssp. *montanum*		nat	1	x				aaw	
Allium ursinum		nat	*		x		x	aaz	
Allium vineale		nat	*		x			cbm	
Alnus glutinosa		nat	*					cbn	
Alnus incana		neo	*					aba	
Alopecurus aequalis		nat	*		x			cbo	
Alopecurus bulbosus		nat	2					abb	
Alopecurus geniculatus		nat	*		x			cbp	
Alopecurus myosuroides		arc	*					cbq	
Alopecurus pratensis		nat	*		x			cbr	
Althaea hirsuta		neo	2	x				cbs	
Althaea officinalis	x	nat	1					abc	
Alyssum alyssoides		nat	2	x				abd	
Amaranthus blitum		arc	*					363	cbv, cby
Amaranthus bouchonii		neo	*					cbw	
Amaranthus powellii agg.		neo	*			x		eag	cbz
Amaranthus retroflexus		neo	*		x			cca	
Amelanchier lamarckii		neo	*	x				ccf	
Ammophila arenaria		nat	*					cci	
Anacamptis pyramidalis		nat	2	x				abf	
Anagallis arvensis		arc	*					cck	eah
Anagallis foemina		nat	2					abg	
Anagallis minima		nat	3	x				ahp	
Anchusa arvensis		arc	*					ccm	
Anchusa officinalis		arc	V					abh	
Andromeda polifolia		nat	3	x				abi	
Anemone nemorosa		nat	*				x	ccn	
Anemone ranunculoides		nat	*		x		x	abk	
Anemone sylvestris		nat	2	x				abl	
Anemone x seemenii		nat	*					242	
Angelica archangelica		nat	*		x			ccp	
Angelica sylvestris		nat	*					ccq	
Antennaria dioica		nat	2	x				abm	
Anthemis arvensis		arc	V					abn	
Anthemis cotula		arc	V					abo	
Anthemis tinctoria		arc	*					abp	
Anthericum liliago		nat	2	x				abq	
Anthericum ramosum		nat	1	x				abr	
Anthoxanthum aristatum		neo	*	x				cct	
Anthoxanthum odoratum		nat	*					ccu	
Anthriscus caucalis		arc	*			x		abs	
Anthriscus nitida		nat	R		x		x	abt	
Anthriscus sylvestris		nat	*		x			ccw	
Anthyllis vulneraria		nat	*	x				391	ccx, ccy
Apera spica-venti		nat	*					ccz	
Aphanes arvensis		arc	*					cda	
Aphanes inexspectata		arc	*					cdb	
Apium graveolens		nat	3		x			abw	
Apium inundatum		nat	2	x				abx	
Apium repens		nat	1		x			aby	
Aquilegia vulgaris		nat	3				x	abz	
Arabidopsis thaliana		arc	*					cdc	
Arabis glabra		nat	V					acb	
Arabis hirsuta agg.		nat	*					eam	
Arctium lappa		arc	*		x	x		cdg	
Arctium minus		nat	*		x	x		cdh	
Arctium nemorosum		nat	*		x		x	cdi	
Arctium tomentosum		arc	*		x	x		cdj	
Arctostaphylos uva-ursi		nat	2	x				acc	

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Table A.2 – *Continued from previous page*

Species	Edge	Natural.	RL NB	EIV N1-3	EIV N7-9	Urban	Forest	Code	Incl.
Arenaria leptoclados		nat	*					cdk	
Arenaria serpyllifolia agg.		nat	*					ean	cdl
Aristolochia clematitis		neo	2		x	x		acd	
Armeria maritima		nat	*					393	ace, acf, cdm
Armoracia rusticana		nat	*		x	x		cdn	
Arnica montana		nat	2	x				acg	
Arnoseris minima		nat	2	x				ach	
Aronia x *prunifolia*		neo	*					302	
Arrhenatherum elatius		nat	*		x			cdo	
Artemisia absinthium		arc	*		x			cdp	
Artemisia annua		neo	*			x		cdq	
Artemisia biennis		neo	*			x		cdr	
Artemisia campestris		nat	V	x				aci	
Artemisia maritima		nat	*		x			acj	
Artemisia vulgaris		nat	*		x	x		cdt	
Arum maculatum		nat	*		x		x	acl	
Asarum europaeum		nat	*				x	cdu	
Asparagus officinalis		arc	*					cdv	
Asperugo procumbens		arc	1		x			acm	
Asperula cynanchica		nat	2	x				aco	
Asplenium adiantum-nigrum		nat	1	x				acq	
Asplenium ceterach		nat	1	x				aia	
Asplenium ruta-muraria		nat	*	x		x		acu	
Asplenium scolopendrium		nat	3				x	ayk	
Asplenium septentrionale		nat	2	x				acv	
Asplenium trichomanes		nat	*	x				acw	224, 206, eaq, eap
Asplenium viride		nat	2				x	acx	
Asplenium x *alternifolium*		nat	n/a	x				acr	
Asplenium x *murbeckii*		nat	1					act	
Aster amellus		nat	1	x				acy	
Aster linosyris		nat	1	x				acz	
Aster novi-belgii agg.		neo	*			x		ear	cdw, cdx, cdy, 232, 50
Aster tripolium		nat	*		x			ada	
Astragalus cicer		nat	2	x			x	adb	
Astragalus danicus		nat	1	x				adc	
Astragalus glycyphyllos		nat	*	x			x	cdz	
Athyrium distentifolium	x	nat	R		x			add	
Athyrium filix-femina		nat	*				x	cea	
Atriplex glabriuscula		nat	R		x			ade	
Atriplex laciniata	x	nat	R		x			adf	
Atriplex littoralis		nat	*		x			cec	
Atriplex longipes		nat	*		x			ced	
Atriplex micrantha		neo	*			x		cee	
Atriplex oblongifolia		neo	*					cef	
Atriplex patula		arc	*		x			ceg	
Atriplex pedunculata		nat	3		x			apo	
Atriplex portulacoides		nat	*		x			ceh	
Atriplex prostrata agg.		nat	*					eas	cei
Atriplex rosea		neo	*		x	x		cej	
Atriplex sagittata		neo	*		x			cek	
Atriplex tatarica		neo	*					13	
Atropa bella-donna		nat	*		x		x	cel	
Avena fatua		arc	*					cem	
Azolla filiculoides		neo	*		x			ceo	
Baldellia ranunculoides		nat	2	x				adi	
Ballota nigra		arc	*		x	x		adj	cep, ceq
Barbarea intermedia		neo	*		x			cer	
Barbarea stricta		nat	*		x			adk	
Barbarea vulgaris		nat	*					ces	300, 301, 364
Bellis perennis		arc	*			x		ceu	
Berberis vulgaris		nat	3	x				adl	
Berteroa incana		neo	*					cev	
Berula erecta		nat	*					cew	
Beta vulgaris ssp. *maritima*	x	nat	R		x			70	
Betonica officinalis		nat	3	x				adm	
Betula nana		nat	2	x				adn	
Betula pendula		nat	*					cex	
Betula pubescens		nat	*	x				eat	cey, cez
Bidens cernua		nat	*		x			cfa	
Bidens connata		neo	*		x			cfb	
Bidens frondosa		neo	*		x			cfc	
Bidens radiata		nat	*		x			cfd	
Bidens tripartita		nat	*		x			cfe	

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Table A.2 – *Continued from previous page*

Species	Edge	Natural.	RL NB	EIV N1-3	EIV N7-9	Urban	Forest	Code	Incl.
Biscutella laevigata ssp. *guestphalica*		nat	R					adp	
Bistorta officinalis		nat	V					azg	
Blechnum spicant		nat	*	x			x	adq	
Blysmus compressus		nat	1	x				adr	
Blysmus rufus	x	nat	2					ads	
Bolboschoenus maritimus agg.		nat	*					398	adt, dpx
Botrychium lunaria		nat	2	x				adu	
Botrychium matricariifolium		nat	1	x				adv	
Brachypodium pinnatum		nat	*					cfh	
Brachypodium sylvaticum		nat	*				x	cfi	
Brassica nigra		neo	*		x			cfl	
Briza media		nat	V	x				ady	
Bromus arvensis		arc	3					adz	
Bromus benekenii		nat	*				x	aea	
Bromus commutatus		arc	*	x				aeb	
Bromus erectus		nat	*	x				aec	
Bromus hordeaceus agg.		nat	*					eau	cfm
Bromus inermis		nat	*					cfo	
Bromus racemosus		nat	2					aed	
Bromus ramosus		nat	*				x	cfq	
Bromus secalinus		arc	*					cfr	
Bromus sterilis		arc	*					cfs	
Bromus tectorum		arc	*					cft	
Bromus thominii		nat	*	x				cfn	
Bryonia alba		arc	3					aee	
Bryonia dioica		nat	V					aef	
Bunias orientalis		neo	*					cfu	
Bunium bulbocastanum		arc	2					aei	
Bupleurum falcatum		nat	3	x				aej	
Bupleurum longifolium		nat	3				x	aek	
Bupleurum rotundifolium		arc	1					ael	
Bupleurum tenuissimum		nat	2					aem	
Butomus umbellatus		nat	3		x			aen	
Cakile maritima		nat	*		x			cfv	
Calamagrostis arundinacea		nat	*				x	aeo	
Calamagrostis canescens		nat	*					cfw	
Calamagrostis epigejos		nat	*					cfx	
Calamagrostis phragmitoides		nat	R	x				68	
Calamagrostis stricta		nat	R	x				aep	
Calamagrostis varia	x	nat	R	x				aeq	
Calamagrostis villosa		nat	*	x				cfy	
Calla palustris		nat	3					aer	
Callitriche palustris agg.		nat	*					eay	aes, cfz, cga, aeu, cgb, cgc
Calluna vulgaris		nat	*	x				cgd	
Caltha palustris		nat	3					cge	
Calystegia sepium		nat	*		x			cgg	
Calystegia soldanella	x	nat	1					aev	
Camelina microcarpa		arc	2					aex	
Campanula bononiensis	x	nat	1	x				aez	
Campanula cervicaria		nat	1					afa	
Campanula glomerata		nat	2	x				afb	
Campanula latifolia		nat	3		x		x	afc	
Campanula patula		nat	3					afd	
Campanula persicifolia		nat	*	x			x	afe	
Campanula rapunculoides		nat	*					cgh	
Campanula rapunculus		nat	*					aff	
Campanula rotundifolia		nat	*	x				cgi	
Campanula trachelium		nat	*		x		x	afg	
Capsella bursa-pastoris		nat	*					cgj	
Cardamine amara		nat	*				x	cgk	
Cardamine bulbifera		nat	*				x	cgl	
Cardamine dentata		nat	*					cgp	
Cardamine flexuosa		nat	*				x	cgm	
Cardamine hirsuta		nat	*		x			cgn	
Cardamine impatiens		nat	*		x		x	cgo	
Cardamine parviflora		nat	R					afh	
Cardamine pratensis agg.		nat	*					eaz	cgq
Cardaminopsis arenosa		neo	*	x				cgr	
Cardaminopsis halleri		nat	*					afi	
Cardaminopsis petraea	x	nat	R	x				afj	
Cardaria draba		neo	*					cgs	
Carduus acanthoides		arc	*		x			cgt	
Carduus crispus		nat	*		x			ebb	

Table A.2 – *Continued from previous page*

Species	Edge	Natural.	RL NB	EIV N1-3	EIV N7-9	Urban	Forest	Code	Incl.
Carduus nutans		arc	*					afk	
Carex acuta		nat	*					cgw	
Carex acutiformis		nat	*					cgx	
Carex appropinquata		nat	2					afl	
Carex aquatilis		nat	3					afm	
Carex arenaria agg.		nat	*					ebc	cgy
Carex bohemica		nat	1					afp	
Carex brizoides		nat	*	x			x	cgz	
Carex canescens		nat	*	x				cha	
Carex caryophyllea		nat	*	x				afq	
Carex cespitosa		nat	2					afr	
Carex demissa		nat	V	x				chc	
Carex diandra		nat	2	x				afv	
Carex digitata		nat	*				x	afw	
Carex dioica		nat	1	x				afx	
Carex distans		nat	V					afy	
Carex disticha		nat	*					chd	
Carex echinata		nat	V	x				afz	
Carex elata		nat	3					chg	
Carex elongata		nat	3				x	chh	
Carex ericetorum		nat	2	x			x	aga	
Carex extensa		nat	*					agb	
Carex flacca		nat	*					agc	
Carex flava		nat	3	x				agd	
Carex hartmanii		nat	2	x				age	
Carex hirta		nat	*					chi	
Carex hostiana		nat	1	x				agf	
Carex humilis		nat	3	x				agg	
Carex lasiocarpa		nat	3	x				agh	
Carex lepidocarpa		nat	2	x				agi	
Carex ligerica		nat	*	x				agj	
Carex limosa		nat	1	x				agk	
Carex montana		nat	*	x			x	agl	
Carex muricata agg.		nat	*					ebe	chf, chk, chs
Carex nigra		nat	*	x				chl	
Carex ornithopoda		nat	2	x				agn	
Carex otrubae		nat	*					chb	
Carex ovalis		nat	*	x				chm	
Carex pallescens		nat	*	x				ago	
Carex panicea		nat	3					agp	
Carex paniculata		nat	*					chn	
Carex pauciflora	x	nat	2	x				agq	
Carex pendula		nat	*				x	cho	
Carex pilosa		nat	R				x	agr	
Carex pilulifera		nat	*	x				chp	
Carex praecox		nat	3					ags	
Carex pseudobrizoides		nat	2	x				agt	
Carex pseudocyperus		nat	*					agu	
Carex pulicaris		nat	1	x				agv	
Carex punctata	x	nat	2	x				agw	
Carex remota		nat	*				x	chq	
Carex riparia		nat	*					chr	
Carex rostrata		nat	*	x				agx	
Carex strigosa		nat	*				x	cht	
Carex sylvatica		nat	*				x	chu	
Carex tomentosa		nat	2					aha	
Carex trinervis	x	nat	2	x				ahb	
Carex umbrosa		nat	2				x	ahc	
Carex vesicaria		nat	V					chv	
Carex viridula		nat	V	x				agm	agy
Carex vulpina agg.		nat	3					ebg	chw
Carex x beckmanniana		nat	*					97	
Carex x elytroides		nat	*					chx	
Carex x involuta		nat	*					228	
Carex x turfosa		nat	R					98	
Carlina acaulis ssp. simplex		nat	2	x				ahd	
Carlina vulgaris		nat	*	x				ahe	
Carpinus betulus		nat	*					chy	
Carum carvi		nat	3					ahf	
Catabrosa aquatica		nat	2		x			ahg	
Caucalis platycarpos		arc	2					ahh	
Centaurea cyanus		arc	*					ahi	
Centaurea jacea		nat	*					cic	

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Table A.2 – *Continued from previous page*

Species	Edge	Natural.	RL NB	EIV N1-3	EIV N7-9	Urban	Forest	Code	Incl.
Centaurea montana		nat	2					ahj	
Centaurea pseudophrygia		nat	V					ahk	
Centaurea scabiosa		nat	*					cif	
Centaurium erythraea		nat	*					ahm	
Centaurium littorale		nat	*	x				ahn	cih, cii
Centaurium pulchellum		nat	*					aho	
Cephalanthera damasonium		nat	*				x	ahq	
Cephalanthera longifolia		nat	2				x	ahr	
Cephalanthera rubra		nat	2				x	ahs	
Cerastium arvense		nat	*					cij	
Cerastium brachypetalum		arc	2	x				aht	
Cerastium diffusum	x	nat	3	x				ahu	
Cerastium dubium		nat	3					ahv	
Cerastium glomeratum		arc	*					cik	
Cerastium glutinosum		nat	*					ahw	
Cerastium holosteoides		nat	*					cil	
Cerastium pumilum agg.		nat	*	x				ebh	ahx
Cerastium semidecandrum		nat	*					ahy	
Ceratocapnos claviculata		nat	*				x	cin	
Ceratophyllum demersum		nat	*		x			cio	
Ceratophyllum submersum		nat	*		x			ahz	
Chaenorhinum minus		arc	*					ciq	
Chaerophyllum aureum		nat	*		x			aib	
Chaerophyllum bulbosum		nat	*		x			aic	
Chaerophyllum hirsutum		nat	*		x			cir	
Chaerophyllum temulum		nat	*		x		x	cis	
Chelidonium majus		nat	*		x	x		cit	
Chenopodium album agg.		arc	*					ebi	cje, cjd
Chenopodium bonus-henricus		nat	3		x	x		aid	
Chenopodium ficifolium		arc	*		x			cix	
Chenopodium foliosum	x	neo	*		x			ciy	
Chenopodium glaucum		arc	*		x			ciz	
Chenopodium hybridum		arc	3		x			aie	
Chenopodium murale		arc	3		x	x		aif	
Chenopodium polyspermum		nat	*		x			cjb	
Chenopodium rubrum agg.		nat	*					ebj	cjc
Chenopodium urbicum		arc	1		x	x		aig	
Chenopodium vulvaria		arc	1		x	x		aih	
Chimaphila umbellata	x	nat	1	x			x	aii	
Chondrilla juncea		nat	3					aij	
Chrysanthemum segetum		arc	*					aik	
Chrysosplenium alternifolium		nat	*				x	ail	
Chrysosplenium oppositifolium		nat	*				x	aim	
Cicendia filiformis		nat	2	x				ain	
Cicerbita alpina		nat	3		x			aio	
Cichorium intybus		arc	*					aip	
Cicuta virosa		nat	3					aiq	
Circaea alpina		nat	*				x	air	
Circaea lutetiana		nat	*		x		x	cjf	
Circaea x *intermedia*		nat	*				x	ais	
Cirsium acaule		nat	*	x				ait	
Cirsium arvense		nat	*		x			cjg	
Cirsium dissectum		nat	2	x				aiu	
Cirsium eriophorum		nat	3					aiv	
Cirsium oleraceum		nat	*					cjh	
Cirsium palustre		nat	*	x				cji	
Cirsium vulgare		nat	*		x			cjj	
Cirsium x *rigens*		nat	*					9	
Cladium mariscus		nat	2	x				aix	
Claytonia perfoliata		neo	*		x	x		cjk	
Clematis vitalba		nat	*		x			cjl	
Clinopodium vulgare		nat	*	x				aiz	
Cnidium dubium		arc	2					aja	
Cochlearia anglica		nat	V		x			cjm	
Cochlearia danica		nat	*					cjn	
Cochlearia officinalis		nat	1					ajb	
Colchicum autumnale		nat	3					ajd	
Conium maculatum		arc	*		x	x		aje	
Conopodium majus		neo	*					ajf	
Consolida regalis		arc	3					ajh	
Convallaria majalis		nat	*				x	aji	
Convolvulus arvensis		nat	*					cjr	
Conyza canadensis		neo	*					cjs	

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Table A.2 – *Continued from previous page*

Species	Edge	Natural.	RL NB	EIV N1-3	EIV N7-9	Urban	Forest	Code	Incl.
Corispermum leptopterum		neo	*					ajk	
Cornus mas		nat	3				x	ajl	
Cornus sanguinea		nat	*					cju	
Cornus suecica		nat	1	x				ajm	
Coronilla coronata		nat	2	x			x	ajn	
Coronopus squamatus		arc	3					ajp	
Corrigiola litoralis		nat	3					ajq	
Corydalis cava		nat	*		x		x	ajr	
Corydalis intermedia		nat	*		x		x	ajs	
Corydalis solida		nat	*		x		x	ajt	
Corylus avellana		nat	*					cjw	
Corynephorus canescens		nat	*	x				cjx	
Cotoneaster integerrimus		nat	R	x				aju	
Cotula coronopifolia		neo	3		x			ajv	
Crambe maritima		nat	R		x			ajw	
Crataegus laevigata agg.		nat	*					ebl	ebm, cka, ckb
Crataegus monogyna		nat	*					ebo	cke, ckf
Crataegus rhipidophylla		nat	*				x	cjz	ebk
Crataegus x *macrocarpa*		nat	*					ckd	
Crataegus x *media*		nat	*					21	
Crepis biennis		arc	*					ajy	
Crepis capillaris		arc	*					ckg	
Crepis foetida		arc	2	x				ajz	
Crepis mollis		nat	2					aka	
Crepis paludosa		nat	*					ckh	
Crepis praemorsa		nat	1	x				akb	
Crepis tectorum		nat	V			x		akd	
Crepis vesicaria ssp. *taraxacifolia*		nat	*					akc	
Cruciata laevipes		nat	*		x			ckj	
Cucubalus baccifer		nat	2		x		x	akf	
Cuscuta campestris		neo	*					19	
Cuscuta epithymum		nat	2	x				akh	
Cuscuta europaea		nat	*		x			ckk	
Cuscuta lupuliformis		neo	*		x			aki	
Cymbalaria muralis		neo	*			x		akj	
Cynoglossum germanicum		nat	3		x		x	akk	
Cynoglossum officinale		nat	3		x			akl	
Cynosurus cristatus		nat	*					cko	
Cyperus fuscus		nat	3					akn	
Cypripedium calceolus		nat	2				x	ako	
Cystopteris fragilis		nat	*					akp	
Cytisus scoparius		nat	*					ckq	
Dactylis glomerata		nat	*					ckr	
Dactylis polygama		nat	*				x	cks	
Dactylorhiza incarnata ssp. *incarnata*		nat	2	x				akq	
Dactylorhiza maculata agg.		nat	3	x				akr	ebq
Dactylorhiza majalis		nat	2	x				akt	
Dactylorhiza praetermissa		nat	3	x				aku	
Dactylorhiza sphagnicola		nat	2	x				akv	
Dactylorhiza x *dinglensis*		nat	n/a					dpm	
Danthonia decumbens		nat	V	x				ckt	
Daphne mezereum		nat	*				x	akw	
Datura stramonium		neo	*		x			akx	
Daucus carota		nat	*					cku	
Deschampsia cespitosa		nat	*	x				ckv	
Deschampsia flexuosa		nat	*	x				ckw	
Deschampsia setacea		nat	1	x				aky	
Deschampsia wibeliana		nat	3					akz	
Descurainia sophia		arc	*					ckx	
Dianthus armeria		nat	3	x				ala	
Dianthus carthusianorum		nat	3	x				alb	
Dianthus deltoides		nat	3	x				alc	
Dianthus gratianopolitanus		nat	R	x				ald	
Dianthus superbus		nat	1	x				ale	
Digitalis grandiflora		nat	2				x	alg	
Digitalis purpurea		nat	*				x	cky	
Digitaria ischaemum		arc	*	x				ckz	
Digitaria sanguinalis		arc	*					alh	
Diphasiastrum alpinum		nat	2	x				ali	
Diphasiastrum complanatum		nat	2	x			x	alj	
Diphasiastrum issleri		nat	2	x				alk	
Diphasiastrum tristachyum		nat	2	x				all	
Diphasiastrum zeilleri		nat	2	x				alm	

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Table A.2 – *Continued from previous page*

Species	Edge	Natural.	RL NB	EIV N1-3	EIV N7-9	Urban	Forest	Code	Incl.
Diplotaxis muralis		neo	*					cla	
Diplotaxis tenuifolia		neo	*			x		clb	
Dipsacus fullonum		arc	*		x			aln	
Dipsacus pilosus		nat	*		x		x	cld	
Draba muralis		neo	*			x		clf	
Drosera intermedia		nat	3	x				alp	
Drosera longifolia		nat	1	x				alo	
Drosera rotundifolia		nat	3	x				alr	
Drosera x obovata		nat	1	x				alq	
Dryopteris affinis		nat	*				x	als	
Dryopteris carthusiana		nat	*	x			x	cli	
Dryopteris cristata		nat	3					alt	
Dryopteris dilatata		nat	*		x		x	clj	
Dryopteris expansa		nat	R	x			x	clk	
Dryopteris filix-mas		nat	*				x	cll	
Echinochloa crus-galli		arc	*		x			clm	
Echinochloa muricata		neo	*					47	
Echinops sphaerocephalus		neo	*		x			clp	
Echium vulgare		arc	*					alv	
Elatine hexandra		nat	2	x				alx	
Elatine hydropiper		nat	2	x				aly	
Elatine triandra		nat	2					alz	
Eleocharis acicularis		nat	3	x				ama	
Eleocharis austriaca		nat	R					ebt	
Eleocharis mamillata		nat	3	x				amb	
Eleocharis multicaulis		nat	2	x				amc	
Eleocharis ovata		nat	1					amd	
Eleocharis palustris		nat	*					clr	ebu, ebv
Eleocharis quinqueflora		nat	3	x				amf	
Eleocharis uniglumis		nat	*					amg	
Elodea canadensis		neo	*		x			cls	
Elodea nuttallii		neo	*		x			clt	
Elymus athericus		nat	*					clv	
Elymus caninus		nat	*		x		x	aaj	
Elymus farctus ssp. *boreoatlanticus*		nat	*		x			clu	
Elymus repens		nat	*		x			clw	
Elymus x obtusiusculus	x	nat	*		x			71	
Elymus x oliveri		nat	*					ctk	
Empetrum nigrum		nat	*	x				clx	
Epilobium angustifolium		nat	*		x		x	cly	
Epilobium ciliatum		neo	*		x			clz	
Epilobium collinum		nat	3	x				cma	
Epilobium hirsutum		nat	*		x			cmb	
Epilobium lanceolatum		nat	R	x				amh	
Epilobium montanum		nat	*					cmc	
Epilobium obscurum		nat	*					cmd	
Epilobium palustre		nat	*	x				cme	
Epilobium parviflorum		nat	*					cmf	
Epilobium roseum		nat	V		x			cmg	
Epilobium tetragonum		nat	*					ebw	cmh
Epipactis atrorubens		nat	3	x				ami	
Epipactis helleborine		nat	*				x	370	cmj, 243
Epipactis leptochila		nat	3				x	amj	cmk, cml
Epipactis microphylla		nat	3				x	amk	
Epipactis muelleri		nat	3	x			x	aml	
Epipactis palustris		nat	2	x			x	amm	
Epipactis purpurata		nat	3					amn	
Epipogium aphyllum		nat	1				x	amo	
Equisetum arvense		nat	*	x				cmm	
Equisetum fluviatile		nat	*					cmn	
Equisetum hyemale		nat	3				x	amp	
Equisetum palustre		nat	*	x				cmo	
Equisetum pratense		nat	2	x			x	amq	
Equisetum sylvaticum		nat	*				x	cmp	
Equisetum telmateia		nat	*				x	amr	
Equisetum variegatum		nat	1	x				ams	
Equisetum x litorale		nat	*					cmq	
Eragrostis albensis		nat	*					46	
Eragrostis minor		neo	*			x		cms	
Erica tetralix		nat	V	x				amt	
Erigeron acris		nat	*	x				eby	
Erigeron annuus		neo	*		x			cmx	401
Eriophorum angustifolium		nat	V	x				amu	

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Table A.2 – *Continued from previous page*

Species	Edge	Natural.	RL NB	EIV N1-3	EIV N7-9	Urban	Forest	Code	Incl.
Eriophorum gracile		nat	1	x				amv	
Eriophorum latifolium		nat	1	x				amw	
Eriophorum vaginatum		nat	V	x				amx	
Erodium ballii	x	nat	*	x				cmy	
Erodium cicutarium		nat	*					cmz	
Erophila verna		nat	*	x				cnb	
Erucastrum gallicum		neo	*					cnd	
Eryngium campestre		nat	3	x				amy	
Eryngium maritimum	x	nat	3					amz	
Erysimum cheiranthoides		nat	*		x			cne	
Erysimum hieraciifolium		nat	3		x			ana	
Euonymus europaea		nat	*					cnh	
Eupatorium cannabinum		nat	*		x			cni	
Euphorbia amygdaloides		nat	*				x	cnj	
Euphorbia cyparissias		nat	*	x				cnk	
Euphorbia esula		nat	*					cnl	
Euphorbia exigua		arc	V					anb	
Euphorbia helioscopia		arc	*		x			cnm	
Euphorbia palustris		nat	2					anc	
Euphorbia peplus		arc	*		x			cno	
Euphorbia platyphyllos		nat	2					and	
Euphorbia x *pseudovirgata*		neo	*					cnp	
Euphrasia frigida	x	nat	R	x				248	
Euphrasia nemorosa agg.		nat	3					anf	cnq, cnr
Euphrasia officinalis ssp. *rostkoviana*		nat	3					ang	
Euphrasia stricta		nat	*	x				cns	
Fagus sylvatica		nat	*					cnu	
Falcaria vulgaris		nat	*					anh	
Fallopia convolvulus		arc	*					dah	
Fallopia dumetorum		nat	*		x			daj	
Fallopia japonica		neo	*		x			dai	
Fallopia sachalinensis		neo	*		x			daq	
Fallopia x *bohemica*		neo	*					64	
Festuca altissima		nat	*				x	cnv	
Festuca arundinacea		nat	*					cnw	
Festuca gigantea		nat	*				x	cny	
Festuca heterophylla		nat	3				x	ani	
Festuca ovina agg.		nat	*	x				ecc	coh, cnx, cnz, coa, anj, dsx, cog, 240, ank
Festuca pratensis		nat	*					cob	
Festuca rubra agg.		nat	*					397	413, ecd, 225, coe, 54, cod
Filago arvensis		nat	*	x				anl	
Filago minima		nat	*	x				ann	
Filago vulgaris		nat	2	x				ano	
Filipendula ulmaria		nat	*					ece	coi, coj
Filipendula vulgaris		nat	2	x				anp	
Fragaria moschata		nat	3				x	anq	
Fragaria vesca		nat	*					cok	
Fragaria viridis		nat	V	x				anr	
Frangula alnus		nat	*					col	
Fraxinus excelsior		nat	*		x			com	
Fritillaria meleagris		arc	3					ans	
Fumaria officinalis		arc	*		x			ecg	coq
Fumaria vaillantii		arc	3					ant	
Gagea bohemica ssp. *saxatilis*	x	nat	R	x				anx	
Gagea lutea		nat	*		x		x	anu	
Gagea minima		nat	2		x		x	anv	
Gagea pratensis		nat	V					anw	
Gagea spathacea		nat	V		x		x	cos	
Gagea villosa		arc	3					any	
Galanthus nivalis		neo	*		x		x	cot	
Galeopsis angustifolia		arc	3					cov	
Galeopsis bifida		nat	*					cow	
Galeopsis ladanum		nat	1	x				anz	
Galeopsis segetum		nat	2	x				aoa	
Galeopsis speciosa		nat	V		x			aob	
Galeopsis tetrahit		nat	*					cox	
Galinsoga ciliata		neo	*		x			coy	
Galinsoga parviflora		neo	*		x			coz	
Galium album		nat	*					cpa	
Galium aparine		nat	*		x			cpb	
Galium boreale		nat	3	x				aoc	
Galium glaucum		nat	1	x				aod	
Galium odoratum		nat	*				x	aoe	

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Table A.2 – *Continued from previous page*

Species	Edge	Natural.	RL NB	EIV N1-3	EIV N7-9	Urban	Forest	Code	Incl.
Galium palustre		nat	*					cpc	ctx
Galium pumilum		nat	3	x				cpe	
Galium rotundifolium		neo	R				x	aof	
Galium saxatile		nat	*	x				cpf	
Galium spurium ssp. *vaillantii*		arc	2					aog	
Galium sylvaticum		nat	*				x	aoh	
Galium uliginosum		nat	*	x				cpg	
Galium verum		nat	*	x				cph	aoj
Galium wirtgenii		nat	2					aok	
Galium x *pomeranicum*		nat	*					23	
Genista anglica		nat	3	x				aol	
Genista germanica		nat	2	x				aom	
Genista pilosa		nat	3	x				aon	
Genista tinctoria		nat	V	x				aoo	
Gentiana cruciata		nat	2	x				aop	
Gentiana pneumonanthe		nat	2	x				aoq	
Gentianella ciliata		nat	3	x				aou	
Gentianella germanica		nat	3	x				aov	
Geranium columbinum		arc	*		x			cpk	
Geranium dissectum		arc	*					cpl	
Geranium lucidum		nat	R		x			aox	
Geranium molle		arc	*					cpm	
Geranium palustre		nat	*		x			aoy	
Geranium pratense		nat	V		x			aoz	
Geranium purpureum		neo	*					96	
Geranium pusillum		arc	*		x			cpo	
Geranium pyrenaicum		neo	*		x	x		cpp	
Geranium robertianum		nat	*		x			cpq	
Geranium sanguineum		nat	2	x			x	apa	
Geranium sylvaticum		nat	*		x			apb	
Geum rivale		nat	3					apc	
Geum urbanum		nat	*		x		x	cps	
Glaucium flavum	x	nat	R		x			146	
Glaux maritima		nat	*					apd	
Glechoma hederacea		nat	*		x			cpt	
Glyceria declinata		nat	*					cpu	
Glyceria fluitans +		nat	*		x			cpv	
Glyceria maxima		nat	*		x			cpw	
Glyceria notata		nat	*		x			cpx	
Gnaphalium sylvaticum		nat	*					cpy	
Gnaphalium uliginosum		nat	*					cpz	
Goodyera repens		nat	2	x			x	apg	
Gratiola officinalis		nat	2					aph	
Groenlandia densa		nat	2					api	
Gymnadenia conopsea		nat	3	x				apj	
Gymnocarpium dryopteris		nat	*				x	apk	
Gymnocarpium robertianum		nat	3	x				apl	
Gypsophila muralis		nat	2	x				apm	
Gypsophila perfoliata		neo	*					43	
Gypsophila repens	x	nat	R	x				apn	
Gypsophila scorzonerifolia		neo	*					44	
Hammarbya paludosa		nat	1	x				app	
Hedera helix		nat	*					cqb	
Helianthemum nummularium		nat	3					apq	apr, aps
Helianthus tuberosus		neo	*		x			cqe	
Helichrysum arenarium		nat	3	x				apt	
Helictotrichon pratense		nat	3	x				adg	
Helictotrichon pubescens		nat	*					adh	
Helleborus foetidus		nat	*	x			x	cqf	
Helleborus viridis		nat	3				x	apu	
Hepatica nobilis		nat	*				x	apv	
Heracleum mantegazzianum		neo	*		x			cqg	
Heracleum sphondylium		nat	*		x			cqh	
Herminium monorchis		nat	1	x				apw	
Herniaria glabra		arc	*	x		x		apx	
Hesperis matronalis		neo	*		x		x	cqj	
Hieracium aridum		nat	*					dsi	
Hieracium arvicola		nat	G	x				cql	
Hieracium aurantiacum		neo	*	x				apy	
Hieracium bauhini		nat	*	x				apz	
Hieracium bifidum		nat	2	x				aqa	
Hieracium bifurcum		nat	G					236	
Hieracium brachiatum		nat	*					cqm	

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Table A.2 – *Continued from previous page*

Species	Edge	Natural.	RL NB	EIV N1-3	EIV N7-9	Urban	Forest	Code	Incl.
Hieracium caespitosum		nat	*	x				aqc	
Hieracium calodon		nat	G					226	
Hieracium cochleatum		nat	G					dwn	
Hieracium cymosum		nat	3	x				aqd	
Hieracium densiflorum		nat	G					cqo	
Hieracium diaphanoides		nat	*					cqp	
Hieracium flagellare		nat	G					cqr	
Hieracium flagelliferum		nat	*					dps	
Hieracium floribundum		nat	G					cqs	
Hieracium glaucinum		nat	*	x				aqe	
Hieracium glaucisetigerum		nat	G					278	
Hieracium glomeratum		nat	G					279	
Hieracium guthnickianum		nat	*					7	
Hieracium kalksburgense		nat	G	x				79	
Hieracium lachenalii		nat	*	x				cqt	
Hieracium lactucella		nat	2	x				aqf	
Hieracium laevigatum		nat	*	x				cqu	
Hieracium laurinum		nat	*					cqv	
Hieracium leptophyton		nat	G					cqw	
Hieracium longiscapum		nat	G					dwm	
Hieracium macrostolonum		nat	*					195	
Hieracium maculatum		nat	G	x				cqx	
Hieracium murorum		nat	*				x	aqi	
Hieracium pilosella		nat	*	x				cqy	
Hieracium piloselloides		nat	*	x				cqz	
Hieracium polymastix		nat	*					218	
Hieracium prussicum		nat	G					212	
Hieracium rothianum		nat	R	x				cqq	
Hieracium sabaudum		nat	*	x				cra	
Hieracium saxifragum	x	nat	R					233	
Hieracium schmidtii		nat	1	x				aqg	
Hieracium schultesii	x	nat	1	x				crb	
Hieracium stoloniflorum		nat	*					219	
Hieracium umbellatum		nat	*	x				crc	
Hieracium visianii		nat	*					280	
Hieracium vulgatum		nat	*	x				crd	
Hieracium zizianum		nat	G					cre	
Hierochloë odorata		nat	3	x				aqj	
Hippocrepis comosa		nat	3	x				aqk	
Hippophaë rhamnoides	x	nat	*	x				crf	
Hippuris vulgaris		nat	3					aql	
Holcus lanatus		nat	*					crh	
Holcus mollis		nat	*	x				cri	
Holosteum umbellatum		nat	V	x				aqm	
Honckenya peploides		nat	*		x			crj	
Hordelymus europaeus		nat	*				x	crk	
Hordeum jubatum		neo	*			x		crl	
Hordeum murinum		arc	*			x		crm	
Hordeum secalinum		nat	V					aqo	
Hornungia petraea		nat	1	x				aqp	
Hottonia palustris		nat	V					aqq	
Humulus lupulus		nat	*		x		x	crn	
Huperzia selago		nat	3				x	aqr	
Hydrocharis morsus-ranae		nat	V					aqs	
Hydrocotyle vulgaris		nat	*	x				aqt	
Hymenolobus procumbens		neo	*					36	
Hyoscyamus niger		arc	3		x			aqu	
Hypericum elodes		nat	2	x				aqv	
Hypericum hirsutum		nat	*		x		x	aqw	
Hypericum humifusum		nat	*	x				aqx	
Hypericum maculatum		nat	*	x				crp	ecm
Hypericum montanum		nat	3	x				aqy	
Hypericum perforatum		nat	*					crq	
Hypericum pulchrum		nat	*	x				aqz	
Hypericum tetrapterum		nat	*					crr	
Hypochaeris glabra		nat	2	x				ara	
Hypochaeris maculata		nat	1	x				arb	
Hypochaeris radicata		nat	*	x				crs	
Ilex aquifolium		nat	*				x	crv	
Illecebrum verticillatum		nat	3	x				arc	
Impatiens glandulifera		neo	*		x			crw	
Impatiens noli-tangere		nat	*				x	crx	
Impatiens parviflora		neo	*				x	cry	

Table A.2 – *Continued from previous page*

Species	Edge	Natural.	RL NB	EIV N1-3	EIV N7-9	Urban	Forest	Code	Incl.
Inula britannica		nat	3					ard	
Inula conyzae		nat	*	x				crz	
Inula germanica	x	nat	R	x				are	
Inula helenium		neo	*					arf	
Inula hirta		nat	R	x				arg	
Inula salicina		nat	3	x				arh	
Iris pseudacorus		nat	*		x			csa	
Iris sibirica		nat	2	x				ari	
Iris versicolor		neo	*					csb	
Isoëtes lacustris		nat	1	x				arj	
Isolepis fluitans		nat	2	x				ark	
Isolepis setacea		nat	3	x				arl	
Jasione montana		nat	*	x				arm	
Juncus acutiflorus		nat	*	x				cse	
Juncus alpinus		nat	2	x				arn	
Juncus anceps	x	nat	*					aro	
Juncus articulatus		nat	*	x				csf	
Juncus bufonius		nat	*					csg	
Juncus bulbosus		nat	*	x				csh	
Juncus capitatus		nat	1	x				arq	
Juncus compressus		nat	*					csi	
Juncus conglomeratus		nat	*	x				csj	
Juncus effusus		nat	*					csk	
Juncus filiformis		nat	3	x				arr	
Juncus gerardii		nat	*					ars	
Juncus inflexus		nat	*					art	
Juncus maritimus	x	nat	*					aru	
Juncus minutulus		nat	D					ecp	
Juncus ranarius		nat	*	x				csm	
Juncus squarrosus		nat	V	x				csn	
Juncus subnodulosus		nat	2	x				arv	
Juncus tenageia		nat	2					arw	
Juncus tenuis		neo	*					cso	
Juniperus communis		nat	3					arx	
Kalmia angustifolia		neo	*	x		x		csp	
Kickxia elatine		arc	2	x				ary	
Kickxia spuria		arc	2	x				arz	
Knautia arvensis		nat	*					csq	
Koeleria arenaria	x	nat	V	x				asa	
Koeleria glauca		nat	2	x				asb	
Koeleria macrantha		nat	3	x				asc	
Koeleria pyramidata		nat	V	x				css	
Lactuca serriola		nat	*			x		csu	
Lactuca tatarica		neo	*					csv	
Lactuca virosa		nat	3		x			ase	
Lamium album		arc	*		x			csw	
Lamium amplexicaule		arc	*		x			csx	
Lamium argentatum		neo	*			x		ecr	
Lamium galeobdolon agg.		nat	*					ecq	csz, cta
Lamium maculatum		nat	*		x			ctb	
Lamium purpureum		arc	*		x			ect	asf
Lapsana communis		nat	*		x			ctd	
Laser trilobum		nat	3	x			x	ash	
Laserpitium latifolium		nat	2	x				asi	
Lathraea squamaria		nat	V				x	ask	
Lathyrus hirsutus		neo	*					ctg	
Lathyrus latifolius		neo	*	x		x		cth	
Lathyrus linifolius		nat	V	x				asm	
Lathyrus maritimus	x	nat	R	x				asn	
Lathyrus niger		nat	2	x			x	aso	
Lathyrus nissolia		neo	*					cti	
Lathyrus palustris		nat	2	x				asp	
Lathyrus pratensis		nat	*					ctj	
Lathyrus sylvestris		nat	*	x				asq	
Lathyrus tuberosus		nat	V					asr	
Lathyrus vernus		nat	*				x	ctl	
Ledum palustre		nat	2	x				ass	
Leersia oryzoides		nat	2		x			ast	
Legousia hybrida		arc	2	x				asu	
Legousia speculum-veneris		arc	1	x				asv	
Lemna gibba		nat	*		x			ctm	
Lemna minor		nat	*					ctn	
Lemna minuta		neo	*					cto	

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Table A.2 – *Continued from previous page*

Species	Edge	Natural.	RL NB	EIV N1-3	EIV N7-9	Urban	Forest	Code	Incl.
Lemna trisulca		nat	*					ctp	
Lemna turionifera		neo	*					28	
Leontodon autumnalis		nat	*					ctq	
Leontodon hispidus		nat	*					asw	
Leontodon saxatilis		nat	*					asx	
Leonurus cardiaca		arc	*		x	x		asy	ctr, cts
Leonurus marrubiastrum		nat	3		x			asz	
Lepidium campestre		arc	*					ata	
Lepidium heterophyllum		neo	*			x		ctu	
Lepidium latifolium		neo	*					ctv	
Lepidium ruderale		arc	*					cty	
Leucanthemum vulgare agg.		nat	*	x				ecu	cub
Leucojum vernum		nat	*		x			atc	
Leymus arenarius		nat	*					cud	
Ligustrum vulgare		nat	*	x				cue	
Lilium bulbiferum		nat	2	x				atd	ate, atf
Lilium martagon		nat	3					atg	
Limonium vulgare		nat	V					ath	
Limosella aquatica		nat	3	x				ati	
Linaria arvensis		arc	1					atj	
Linaria vulgaris		nat	*					cug	
Lindernia dubia		neo	*					atk	
Linnaea borealis		nat	1	x			x	atl	
Linum catharticum		nat	*	x				atm	
Linum leonii		nat	2	x				atn	
Linum tenuifolium		nat	1	x				ato	
Liparis loeselii		nat	2	x				atp	
Listera cordata		nat	3	x			x	atq	
Listera ovata		nat	*		x		x	atr	
Lithospermum arvense		arc	3					aeg	
Lithospermum officinale		nat	3					ats	
Lithospermum purpurocaeruleum		nat	3				x	aeh	
Littorella uniflora		nat	2	x				att	
Lobelia dortmanna		nat	1	x				atu	
Lolium perenne		nat	*		x			cuk	
Lonicera periclymenum		nat	*					cum	
Lonicera xylosteum		nat	*				x	atw	
Lotus corniculatus		nat	*	x				cun	
Lotus pedunculatus		nat	*					cuo	
Lotus tenuis		nat	V					atx	
Lunaria rediviva		nat	V		x		x	atz	
Lupinus polyphyllus		neo	*					cus	
Luronium natans		nat	2	x				aua	
Luzula campestris		nat	*	x				cut	
Luzula congesta		nat	3					cuu	
Luzula luzuloides		nat	*				x	aub	238
Luzula multiflora		nat	*	x				cuv	
Luzula pilosa		nat	*				x	cuw	
Luzula sudetica		nat	R	x				cux	
Luzula sylvatica		nat	*					auc	
Lycium barbarum		neo	*			x		cuz	
Lycopodiella inundata		nat	3	x				aue	
Lycopodium annotinum		nat	3	x			x	auf	
Lycopodium clavatum		nat	3	x				aug	
Lycopus europaeus		nat	*		x			cvb	
Lysimachia nemorum		nat	*		x		x	auh	
Lysimachia nummularia		nat	*					cvc	
Lysimachia thyrsiflora		nat	V					aui	
Lysimachia vulgaris		nat	*					cve	
Lythrum hyssopifolia		nat	1					auj	
Lythrum salicaria		nat	*					cvf	
Maianthemum bifolium		nat	*	x			x	cvh	
Malus sylvestris		nat	3					auk	
Malva alcea		arc	V		x			cvj	
Malva moschata		arc	*			x		cvk	
Malva neglecta		arc	*		x	x		cvl	
Malva pusilla		arc	2			x		cvm	
Malva sylvestris ssp. *sylvestris*		arc	*			x		cvn	
Marrubium vulgare		arc	1	x				aul	
Matricaria discoidea		neo	*		x	x		cvp	
Matricaria recutita		arc	*					cvq	
Matteuccia struthiopteris		nat	3		x		x	aum	
Medicago falcata		nat	*	x				cvs	

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Table A.2 – *Continued from previous page*

Species	Edge	Natural.	RL NB	EIV N1-3	EIV N7-9	Urban	Forest	Code	Incl.
Medicago lupulina		nat	*					cvt	
Medicago minima		nat	2	x				aun	
Medicago x varia		neo	*					cvu	
Melampyrum arvense		nat	2	x				auo	
Melampyrum cristatum		nat	2	x			x	aup	
Melampyrum nemorosum		nat	V				x	auq	
Melampyrum pratense		nat	*	x				cvv	
Melampyrum sylvaticum		nat	*	x				aur	
Melica nutans		nat	*	x			x	aus	
Melica uniflora		nat	*				x	cvw	
Melilotus albus		arc	*					cvx	
Melilotus altissimus		nat	*		x			cvy	
Melilotus dentatus		nat	1					aut	
Melilotus officinalis		arc	*	x				cwa	
Melittis melissophyllum		nat	1	x			x	auu	
Mentha aquatica		nat	*					cwb	
Mentha arvensis		nat	*					cwc	
Mentha longifolia		nat	*		x			auv	
Mentha pulegium		nat	2		x			auw	
Mentha x verticillata		nat	*					cwi	
Menyanthes trifoliata		nat	3	x				aux	
Mercurialis annua		neo	*		x	x		cwj	
Mercurialis perennis		nat	*		x		x	cwk	
Meum athamanticum		nat	V	x				auy	
Milium effusum		nat	*				x	cwl	
Mimulus guttatus		neo	*					auz	
Mimulus moschatus		neo	*					cwm	
Minuartia verna ssp. *hercynica*		nat	3					avb	
Misopates orontium		arc	2					avd	
Moehringia trinervia		nat	*		x		x	cwn	
Molinia caerulea		nat	*	x				cwo	
Moneses uniflora		nat	1	x			x	avf	
Monotropa hypopitys agg.		nat	3	x				avg	cwp, cwq
Montia fontana		nat	3					avh	cwi, avi, cws, cwt
Mycelis muralis		nat	*					cwx	
Myosotis arvensis		arc	*					cwy	
Myosotis discolor		nat	V	x				avj	
Myosotis laxa		nat	*		x			avk	
Myosotis nemorosa		nat	*					avl	
Myosotis ramosissima		arc	*	x				avm	
Myosotis scorpioides		nat	*					cwz	
Myosotis sparsiflora		nat	R		x			avn	
Myosotis stricta		arc	V	x				cxa	
Myosotis sylvatica		nat	*		x		x	cxb	
Myosurus minimus		arc	*					avo	
Myrica gale		nat	3	x				avp	
Myriophyllum alterniflorum		nat	3	x				avq	
Myriophyllum spicatum		nat	*		x			avr	
Myriophyllum verticillatum		nat	*		x			avs	
Myrrhis odorata		neo	*		x	x		cxc	
Nardus stricta		nat	V	x				cxf	
Narthecium ossifragum		nat	3	x				avu	
Nasturtium officinale agg.		nat	*		x			edb	cxg, 191
Neottia nidus-avis		nat	*				x	avv	
Nepeta cataria		arc	2		x	x		avw	
Neslia paniculata		arc	1					avx	
Nonea pulla		nat	1	x				avz	
Nuphar lutea		nat	*					awa	
Nuphar x spenneriana		nat	R					cxr	
Nymphaea alba		nat	V					awc	
Nymphaea candida		nat	3					awd	
Nymphoides peltata		nat	2		x			awe	
Odontites litoralis	x	nat	1					awf	
Odontites vernus		nat	3					cxk	
Odontites vulgaris		nat	*					cxl	
Oenanthe aquatica		nat	*					cxm	
Oenanthe conioides	x	nat	1		x			awh	
Oenanthe fistulosa		nat	3					awi	
Oenanthe lachenalii		nat	3		x			awj	
Oenothera biennis		neo	*					cxo	
Oenothera oakesiana		neo	*					cxn	
Omphalodes scorpioides		nat	1		x		x	awl	
Onobrychis viciifolia		neo	*	x				cxs	

Table A.2 – *Continued from previous page*

Species	Edge	Natural.	RL NB	EIV N1-3	EIV N7-9	Urban	Forest	Code	Incl.
Ononis spinosa agg.		nat	*					awn	awm, awo
Onopordum acanthium		arc	*		x	x		awp	
Ophioglossum vulgatum		nat	3	x				awq	
Ophrys apifera		nat	3	x				awr	
Ophrys insectifera		nat	3	x				aws	
Orchis mascula		nat	3					awu	
Orchis militaris		nat	2	x				awv	
Orchis purpurea		nat	3	x				awy	
Orchis tridentata		nat	2	x				awz	
Oreopteris limbosperma		nat	*				x	bgu	
Origanum vulgare		nat	*	x				axb	
Ornithogalum nutans agg.		neo	*		x	x		axd	
Ornithogalum umbellatum		arc	*		x	x		axe	
Ornithopus perpusillus		nat	*	x				cxw	
Ornithopus sativus		neo	*					cxx	
Orobanche caryophyllacea		nat	1	x				axh	
Orobanche elatior		nat	1	x				axi	
Orobanche lutea		nat	1					axj	
Orobanche picridis		nat	1					axm	
Orobanche purpurea		nat	1	x				axn	
Orobanche rapum-genistae		nat	2	x				axp	
Orobanche reticulata		nat	3	x				axl	
Orthilia secunda		nat	3	x			x	axq	
Osmunda regalis		nat	3				x	axr	
Oxalis acetosella		nat	*				x	cxz	
Oxalis corniculata		neo	*			x		cya	
Oxalis stricta		neo	*		x			cyc	
Panicum riparium		n/a	*					dsf	
Papaver argemone		arc	*					cyh	
Papaver dubium		arc	*					ede	cyi, cyj
Papaver rhoeas		arc	*					axs	
Parapholis strigosa		nat	*					cyn	
Parietaria judaica		neo	*		x	x		axt	
Parietaria officinalis		arc	3		x	x		axu	
Paris quadrifolia		nat	V		x		x	axv	
Parnassia palustris		nat	2	x				axw	
Pastinaca sativa		nat	*					cyr	
Pedicularis palustris		nat	2	x				axx	
Pedicularis sylvatica		nat	2	x				axy	
Peplis portula		nat	V	x				axz	
Persicaria amphibia		nat	*					daf	
Persicaria hydropiper		nat	*		x			dak	
Persicaria lapathifolia		nat	*					edl	dal, dam
Persicaria maculosa		nat	*		x			dap	
Persicaria minor		nat	*		x			dao	
Persicaria mitis		n/a	*		x			azh	
Petasites albus		nat	*					cys	
Petasites hybridus		nat	*		x			cyt	
Petasites spurius		nat	2					aya	
Petrorhagia prolifera		nat	2	x				ayb	
Peucedanum cervaria		nat	2	x				ayc	
Peucedanum officinale	x	nat	1	x				ayd	
Peucedanum oreoselinum		nat	2	x				aye	
Peucedanum ostruthium		neo	R		x			ayf	
Peucedanum palustre		nat	*					ayg	
Phalaris arundinacea		nat	*		x			cyv	
Phegopteris connectilis		nat	*				x	bgw	
Phleum arenarium	x	nat	V	x				ayh	
Phleum bertolonii		nat	*					cyx	
Phleum paniculatum	x	nat	1					ayi	
Phleum phleoides		nat	2	x				ayj	
Phleum pratense		nat	*		x			cyy	
Phragmites australis		nat	*		x			cyz	
Physalis alkekengi		nat	*		x			ayl	
Phyteuma nigrum		nat	V				x	aym	
Phyteuma orbiculare		nat	2	x				ayn	
Phyteuma spicatum		nat	*				x	ayo	cza, czb
Picea abies		nat	*					czc	
Picris hieracioides		nat	*					cze	
Pilularia globulifera		nat	2	x				ayp	
Pimpinella major		nat	*					czf	
Pimpinella nigra		nat	n/a	x				402	
Pimpinella saxifraga		nat	*	x				czg	

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Table A.2 – *Continued from previous page*

Species	Edge	Natural.	RL NB	EIV N1-3	EIV N7-9	Urban	Forest	Code	Incl.
Pinguicula vulgaris		nat	2	x				ayq	
Pinus sylvestris		nat	*					czi	
Plantago coronopus		nat	*					ayr	
Plantago lanceolata		arc	*					czj	
Plantago major		nat	*					edi	czk, czl, ayt
Plantago maritima		nat	*					ayu	
Plantago media		nat	*	x				czm	
Platanthera bifolia		nat	2					ayv	
Platanthera chlorantha		nat	3				x	ayw	
Poa angustifolia		nat	*	x				czn	
Poa annua		nat	*		x			czo	
Poa bulbosa		nat	3	x				ayx	
Poa chaixii		nat	*				x	czp	
Poa compressa		nat	*	x				czq	
Poa humilis		nat	*	x				czu	
Poa nemoralis		nat	*				x	czr	
Poa palustris		nat	*		x			czs	
Poa pratensis		nat	*					czt	
Poa remota		nat	3	x			x	ayy	
Poa supina		nat	*		x			czv	
Poa trivialis		nat	*		x			czw	
Polemonium caeruleum	x	nat	R					czx	
Polygala amara agg.		nat	2	x				azc	daa, dab
Polygala comosa		nat	V	x				dac	
Polygala serpyllifolia		nat	2	x				azd	
Polygala vulgaris		nat	3	x				aze	cyk, cyq
Polygonatum multiflorum		nat	*				x	dad	
Polygonatum odoratum		nat	2	x				azf	
Polygonatum verticillatum		nat	*				x	dae	
Polygonum aviculare agg.		nat	*					dag	144
Polypodium vulgare agg.		nat	*	x				edm	dar, 276
Polystichum aculeatum		nat	3		x		x	azi	
Polystichum lonchitis		nat	R	x				azj	
Populus alba		neo	*					dat	
Populus nigra		nat	3		x			azk	
Populus tremula		nat	*					dau	
Populus x *canescens*		neo	*					258	
Portulaca oleracea		neo	*		x	x		dav	
Potamogeton acutifolius		nat	3					azl	
Potamogeton alpinus		nat	V					azm	
Potamogeton coloratus		nat	2		x			azo	
Potamogeton compressus		nat	3					azp	
Potamogeton crispus		nat	*					dax	
Potamogeton friesii		nat	3					azr	
Potamogeton gramineus		nat	2					azs	
Potamogeton lucens		nat	3		x			azt	
Potamogeton natans		nat	*					day	
Potamogeton nodosus		nat	3					azu	
Potamogeton obtusifolius		nat	3					azv	
Potamogeton pectinatus		nat	*		x			daz	
Potamogeton perfoliatus		nat	3					azw	
Potamogeton polygonifolius		nat	3	x				azx	
Potamogeton praelongus		nat	1					azy	
Potamogeton pusillus agg.		nat	*					edo	daw
Potamogeton trichoides		nat	*					dbb	
Potamogeton x *angustifolius*		nat	1					azn	
Potamogeton x *nitens*		nat	1					dbe	
Potamogeton x *spathulatus*		nat	G					dbg	
Potamogeton x *undulatus*		n/a	R					dbh	
Potentilla alba		nat	1					baa	
Potentilla anglica		nat	V					bab	
Potentilla anserina		nat	*		x			dbi	
Potentilla argentea		nat	*	x				dbj	
Potentilla erecta		nat	*	x				dbk	
Potentilla heptaphylla		nat	2	x				bae	
Potentilla incana		nat	1	x				bac	
Potentilla intermedia		neo	*			x		dbm	
Potentilla neumanniana		nat	V	x				baf	
Potentilla norvegica		neo	*					dbn	
Potentilla palustris		nat	V	x				bag	
Potentilla recta		neo	*	x				dbo	
Potentilla reptans		nat	*					dbp	
Potentilla sterilis		nat	*				x	bah	

Table A.2 – *Continued from previous page*

Species	Edge	Natural.	RL NB	EIV N1-3	EIV N7-9	Urban	Forest	Code	Incl.
Potentilla supina		nat	3		x			bai	
Primula elatior		nat	*		x		x	baj	
Primula veris		nat	V	x				bak	
Primula vulgaris		nat	1				x	bal	
Prunella grandiflora		nat	3	x				bam	
Prunella laciniata		nat	2	x				ban	
Prunella vulgaris		nat	*					dbr	
Prunus avium		nat	*					dbs	
Prunus padus		nat	*					dbx	
Prunus serotina		neo	*					dby	
Prunus spinosa		nat	*					dbz	
Pseudognaphalium luteoalbum		nat	2	x				apf	
Pseudolysimachion longifolium		nat	3					biw	
Pseudolysimachion spicatum		nat	2	x				bjc	
Pseudorchis albida		nat	1	x				bao	
Psyllium arenarium		neo	*					ays	
Pteridium aquilinum		nat	*	x			x	dcb	
Puccinellia capillaris	x	nat	R					dcc	
Puccinellia distans		nat	*					dcd	
Puccinellia limosa		nat	R					89	
Puccinellia maritima		nat	*					dce	
Pulicaria dysenterica		nat	3					baq	
Pulicaria vulgaris		nat	3		x			bar	
Pulmonaria obscura		nat	*		x		x	dcf	
Pulmonaria officinalis		nat	*				x	bas	
Pulsatilla pratensis		nat	2	x				bau	
Pulsatilla vulgaris		nat	2	x				baw	
Pyrola chlorantha		nat	1	x			x	bax	
Pyrola minor		nat	3	x			x	baz	
Pyrola rotundifolia		nat	3	x			x	bba	
Pyrus pyraster		nat	3					bbb	
Quercus petraea		nat	*					dch	
Quercus robur		nat	*					dci	
Quercus x rosacea		nat	*					dsu	
Radiola linoides		nat	3	x				bbc	
Ranunculus acris		nat	*					dcj	
Ranunculus aquatilis agg.		nat	*					bbd	bbe, bbi, bbj, bbk, bbn, bbg, bbo, dco, bbx
Ranunculus arvensis		arc	2					bbf	
Ranunculus auricomus agg.		nat	*					dck	
Ranunculus bulbosus		nat	*	x				bbh	
Ranunculus ficaria ssp. *bulbilifer*		nat	*		x			dcl	
Ranunculus flammula		nat	*	x				dcm	
Ranunculus lanuginosus		nat	*		x		x	dcn	
Ranunculus lingua		nat	3		x			bbl	
Ranunculus platanifolius		nat	3		x			bbp	
Ranunculus polyanthemos agg.		nat	3					bbs	bbm, bbq, bbr, bbw, bbt
Ranunculus repens		nat	*		x			dcp	
Ranunculus sardous		nat	3		x			bbv	
Ranunculus sceleratus		nat	*		x			dcq	
Raphanus raphanistrum		arc	3					dcr	
Reseda lutea		arc	*					dcu	
Reseda luteola		arc	*					dcv	
Rhamnus cathartica		nat	*					bbz	
Rhinanthus alectorolophus		nat	3					bca	
Rhinanthus angustifolius ssp. *grandiflorus*		nat	V	x				bcd	
Rhinanthus glacialis		nat	1	x				bcb	
Rhinanthus minor		nat	V	x				bcc	
Rhynchospora alba		nat	3	x				bce	
Rhynchospora fusca		nat	2	x				bcf	
Ribes alpinum		nat	*		x		x	dcw	
Ribes nigrum		nat	*				x	dcx	
Ribes rubrum agg.		nat	*					edw	
Ribes uva-crispa		nat	*			x		dda	
Robinia pseudoacacia		neo	*	x				ddb	
Rorippa amphibia		nat	*		x			ddc	
Rorippa anceps		nat	*		x			bcg	
Rorippa austriaca		nat	R		x			bci	
Rorippa palustris		nat	*		x			ddd	
Rorippa sylvestris		nat	*					dde	
Rosa agrestis		nat	2	x				bcj	
Rosa arvensis		nat	2				x	bck	
Rosa caesia agg.		nat	n/a					bcl	ddf
Rosa canina		nat	*					ddg	

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Table A.2 – *Continued from previous page*

Species	Edge	Natural.	RL NB	EIV N1-3	EIV N7-9	Urban	Forest	Code	Incl.
Rosa corymbifera		nat	*					ddh	
Rosa dumalis		nat	3					dds	bcq
Rosa elliptica		nat	3	x				bcm	
Rosa inodora		nat	G					87	
Rosa micrantha		nat	3	x				ddj	
Rosa rubiginosa		nat	*	x				ddk	
Rosa rugosa		neo	*					ddl	
Rosa spinosissima	x	nat	3	x				bco	
Rosa subcanina		nat	*	x				ddo	
Rosa subcollina		nat	3	x				ddp	
Rosa tomentella		nat	3					bcn	
Rosa tomentosa agg.		nat	3					bcp	ddm, ddn, ddq, ddr
Rubus armeniacus		neo	*			x		dea	
Rubus caesius		nat	*		x			deg	
Rubus chamaemorus		nat	2	x				bcr	
Rubus fruticosus agg.		nat	*					edy	
Rubus idaeus		nat	*				x	dgi	
Rubus laciniatus		neo	*					dgo	
Rubus saxatilis		nat	2				x	bcs	
Rumex acetosa		nat	*					djt	
Rumex acetosella		nat	*	x				dju	
Rumex aquaticus		nat	V		x			bct	
Rumex conglomeratus		nat	*		x			djv	
Rumex crispus		nat	*					djw	
Rumex hydrolapathum		nat	*		x			djx	
Rumex maritimus		nat	*		x			bcu	
Rumex obtusifolius		nat	*		x			eeb	
Rumex palustris		nat	*		x			bcv	
Rumex salicifolius	x	neo	*		x			dke	
Rumex sanguineus		nat	*		x		x	dka	
Rumex stenophyllus		neo	*					dkc	
Rumex thyrsiflorus		nat	*					dkd	
Rumex x pratensis		n/a	*					far	
Ruppia cirrhosa	x	nat	2					dkf	
Ruppia maritima	x	nat	2					bcw	
Sagina apetala agg.		arc	*					eed	dkh
Sagina maritima		nat	*	x				dki	
Sagina nodosa		nat	3					bcx	
Sagina procumbens		nat	*			x		dkj	
Sagittaria latifolia		neo	*					dkk	
Sagittaria sagittifolia		nat	*					bcz	
Salicornia dolichostachya s.l.		nat	n/a	x				eef	
Salicornia europaea		nat	*					eee	bda, dkn
Salicornia fragilis	x	nat	n/a					dko	
Salicornia procumbens		nat	*	x				dkl	
Salicornia stricta		nat	*					dkm	
Salix alba		nat	*		x			eeg	dkq
Salix aurita		nat	*	x				dkr	
Salix caprea		nat	*		x			dks	
Salix cinerea		nat	*					dkt	
Salix fragilis agg.		nat	*					eeh	dkv, dle
Salix pentandra		nat	3					dkw	
Salix purpurea		nat	*					eei	dky
Salix repens		nat	*					eej	dla, bdd
Salix triandra		nat	*					eek	dlb
Salix viminalis		nat	*					dld	
Salix x alopecuroides		nat	*					101	
Salix x ambigua		nat	*					drc	
Salix x holosericea		nat	*					drb	
Salix x meyeriana		nat	*					dqr	
Salix x mollissima		nat	*					dqi	
Salix x multinervis		nat	*					dkx	
Salix x reichardtii		nat	*					dqh	
Salix x rubra		nat	*					dqf	
Salix x smithiana		nat	*					dlf	
Salsola kali		nat	*					bdf	dlg, dlh
Salvia glutinosa		neo	*		x			dli	
Salvia nemorosa		neo	*					dlj	
Salvia pratensis		nat	3					bdg	
Salvia verticillata		neo	*				x	bdh	
Sambucus ebulus		neo	*		x	x	x	dll	
Sambucus nigra		nat	*		x			dlm	14
Sambucus racemosa		nat	*		x			dln	

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Table A.2 – *Continued from previous page*

Species	Edge	Natural.	RL NB	EIV N1-3	EIV N7-9	Urban	Forest	Code	Incl.
Samolus valerandi		nat	2					bdi	
Sanguisorba minor		nat	*	x				eel	bdj
Sanguisorba officinalis		nat	3					bdk	
Sanicula europaea		nat	*					bdl	
Saponaria officinalis		nat	*			x		dlp	
Saxifraga granulata		nat	3	x				bdm	
Saxifraga tridactylites		nat	*	x				bdo	
Scabiosa canescens		nat	2	x				bdp	
Scabiosa columbaria		nat	*	x				bdq	
Scandix pecten-veneris		arc	2					bds	
Scheuchzeria palustris		nat	2	x				bdt	
Schoenoplectus lacustris		nat	*					dlt	
Schoenoplectus pungens		nat	1		x			bdu	
Schoenoplectus tabernaemontani		nat	*					bdw	
Schoenoplectus triqueter		nat	3		x			bdx	
Schoenoplectus x carinatus		nat	3		x			bdy	
Schoenus nigricans	x	nat	3	x				bdz	
Scirpus sylvaticus		nat	*					dlu	
Scleranthus annuus agg.		arc	*					een	een
Scleranthus perennis		nat	3	x				bea	
Scorzonera hispanica	x	nat	2	x				beb	
Scorzonera humilis		nat	2	x				bec	
Scorzonera laciniata		nat	2					ayz	
Scrophularia nodosa		nat	*		x			dly	
Scrophularia umbrosa		nat	*		x			bee	
Scrophularia vernalis		neo	*		x	x		dlz	
Scutellaria galericulata		nat	*					dma	
Scutellaria hastifolia		nat	2					bef	
Scutellaria minor		nat	2	x				beg	
Securigera varia		neo	*	x				ajo	
Sedum acre		nat	*	x				dmb	
Sedum album		nat	*	x				beh	
Sedum rupestre		nat	V	x				bei	
Sedum sexangulare		nat	*	x				bej	
Sedum telephium agg.		nat	*					eeo	dmd, dme
Selinum carvifolia		nat	3	x				bel	
Sempervivum tectorum		nat	*					bem	
Senecio aquaticus		nat	3					ben	
Senecio erraticus ssp. *barbareifolius*		nat	3					bep	
Senecio erucifolius		nat	*					dmg	
Senecio hercynicus		nat	*		x		x	bes	
Senecio inaequidens		neo	*	x		x		dmh	
Senecio jacobaea ssp. *dunensis*	x	nat	*					dmi	
Senecio jacobaea ssp. *jacobaea*		nat	*					dmj	
Senecio ovatus		nat	*		x		x	dmk	
Senecio paludosus		nat	2					bet	
Senecio sarracenicus		nat	*		x			beq	
Senecio sylvaticus		nat	*		x		x	dml	
Senecio vernalis		neo	*					dmm	
Senecio viscosus		nat	*					dmn	
Senecio vulgaris		nat	*		x			dmo	
Serratula tinctoria		nat	2	x				beu	
Seseli annuum		nat	1	x				bev	
Seseli libanotis		nat	2	x				bew	
Seseli montanum		nat	1					dmp	
Sesleria albicans		nat	V	x				bex	
Setaria pumila		arc	V					bey	
Setaria viridis		arc	*		x			dmt	
Sherardia arvensis		arc	3					bez	
Silaum silaus		nat	2	x				bfa	
Silene conica	x	neo	*	x				dmw	
Silene dioica		nat	*		x			dmx	
Silene flos-cuculi		nat	*					cuy	
Silene latifolia ssp. *alba*		nat	*		x			dmz	
Silene noctiflora		arc	3					bfc	
Silene nutans		nat	V	x				bfd	
Silene otites		nat	3	x				bfe	
Silene viscaria		nat	1	x				aud	
Silene vulgaris		nat	*					bff	
Silene x hampeana		n/a	*					dqw	
Sinapis arvensis		arc	*					dnd	
Sisymbrium altissimum		neo	*					dne	
Sisymbrium austriacum		nat	R		x			bfg	

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Table A.2 – *Continued from previous page*

Species	Edge	Natural.	RL NB	EIV N1-3	EIV N7-9	Urban	Forest	Code	Incl.
Sisymbrium loeselii		neo	*					dng	
Sisymbrium officinale		arc	*		x			dnh	
Sisymbrium strictissimum		nat	R		x			bfh	
Sium latifolium		nat	*		x			bfi	
Solanum dulcamara		nat	*		x			dnk	
Solanum nigrum		arc	*		x			eeq	dnm
Solidago canadensis		neo	*					dns	
Solidago gigantea		neo	*		x			dnt	
Solidago virgaurea		nat	*					dnv	
Sonchus arvensis		nat	*					eer	dnw, bfk
Sonchus asper		nat	*		x			dnx	
Sonchus oleraceus		nat	*		x			dny	
Sonchus palustris		nat	V		x			bfl	
Sorbus aucuparia		nat	*					dnz	
Sorbus torminalis		nat	*					bfo	
Sparganium angustifolium		nat	2	x				bfp	
Sparganium emersum		nat	*		x			doa	
Sparganium erectum		nat	*		x			ees	doc
Sparganium natans		nat	2	x				bfq	
Spartina anglica		neo	*	x				dod	
Spergula arvensis		arc	*					doe	
Spergula morisonii		nat	*	x				dof	
Spergularia echinosperma		nat	*		x			bfr	
Spergularia media		nat	*					dog	
Spergularia rubra		arc	*					doh	
Spergularia salina		nat	*					bfs	
Spiraea alba		neo	*					dpf	
Spiranthes spiralis		nat	1	x				bfu	
Spirodela polyrhiza		nat	*					dok	
Stachys alpina		nat	3		x		x	bfv	
Stachys annua		arc	1					bfw	
Stachys arvensis		arc	3					bfx	
Stachys germanica		nat	2					bfy	
Stachys palustris		nat	*					dol	
Stachys recta		nat	2	x				bfz	
Stachys sylvatica		nat	*		x		x	dom	
Stachys x ambigua		nat	*					103	
Stellaria alsine		nat	*					dot	
Stellaria aquatica		nat	*		x			don	
Stellaria graminea		nat	*	x				doo	
Stellaria holostea		nat	*				x	dop	
Stellaria media		arc	*					doq	
Stellaria neglecta		nat	*				x	dor	
Stellaria nemorum		nat	*		x			bgb	
Stellaria pallida		arc	*					dos	
Stellaria palustris		nat	V	x				bgc	
Stipa capillata	x	nat	R	x				bgd	
Stratiotes aloides		nat	3					bge	
Suaeda maritima		nat	*		x			dou	
Succisa pratensis		nat	3	x				dov	
Symphytum officinale		nat	*		x			doy	
Symphytum x uplandicum		neo	*					doz	
Tanacetum corymbosum		nat	3				x	bgh	
Tanacetum vulgare		arc	*			x		dpc	
Taraxacum acervatulum		nat	n/a					104	
Taraxacum acutifrons		nat	n/a					dpe	
Taraxacum adiantifrons		nat	n/a					dqn	
Taraxacum aequilobum		nat	n/a					106	
Taraxacum alatum		nat	n/a					dph	
Taraxacum amphilobum		nat	n/a					107	
Taraxacum amplum		nat	n/a					dpi	
Taraxacum ancistrolobum		nat	n/a					dpj	
Taraxacum angustisquameum		nat	n/a					108	
Taraxacum atactum		nat	n/a					dpk	
Taraxacum atricapillum	x	nat	n/a					110	
Taraxacum baeckiiforme		nat	n/a					dpn	
Taraxacum borgvallii		nat	n/a					dpp	
Taraxacum cacuminatum		nat	n/a					111	
Taraxacum calochroum	x	nat	n/a					112	
Taraxacum caloschistum		nat	n/a					dpu	
Taraxacum canoviride		nat	n/a					113	
Taraxacum celticum agg.		nat	V					eex	dpr, dsa, 26, 283, drg, 181, 182, dqj
Taraxacum chlorodes	x	nat	n/a					114	

Table A.2 – *Continued from previous page*

Species	Edge	Natural.	RL NB	EIV N1-3	EIV N7-9	Urban	Forest	Code	Incl.
Taraxacum christiansenii	x	nat	n/a					115	
Taraxacum comtulum		nat	n/a					117	
Taraxacum contractum		nat	n/a					119	
Taraxacum copidophyllum		nat	n/a					120	
Taraxacum cordatum		nat	n/a					dpw	
Taraxacum croceiflorum	x	nat	n/a					dpz	
Taraxacum cyanolepis		nat	n/a					dqa	
Taraxacum dilaceratum		nat	n/a					dqb	
Taraxacum dilatatum	x	nat	n/a					121	
Taraxacum edytomum		nat	n/a					140	
Taraxacum ekmanii		nat	n/a					dqd	
Taraxacum exsertiforme		nat	n/a					123	
Taraxacum exsertum		nat	n/a					124	
Taraxacum fagerstroemii		nat	n/a					125	
Taraxacum fasciatum		nat	n/a					dqg	
Taraxacum fulgidum		nat	n/a					127	
Taraxacum fusciflorum		nat	n/a					57	
Taraxacum gelertiiforme		nat	n/a					128	
Taraxacum gesticulans		nat	n/a					129	
Taraxacum hamatiforme		nat	n/a					dqk	
Taraxacum hamatulum		nat	n/a					58	
Taraxacum hamatum		nat	n/a					130	
Taraxacum hamatum agg.		nat	*					dql	
Taraxacum hamiferum	x	nat	n/a					dqm	
Taraxacum hepaticum		nat	n/a					131	
Taraxacum huelphersianum		nat	n/a					dqo	
Taraxacum inarmatum		nat	n/a					132	
Taraxacum ingens		nat	n/a					dqq	
Taraxacum inops		nat	n/a					133	
Taraxacum interveniens		nat	n/a					134	
Taraxacum intumescens		nat	n/a					135	
Taraxacum kernianum		nat	n/a					dqs	
Taraxacum laciniosifrons		nat	n/a					dqt	
Taraxacum laciniosum	x	nat	n/a					dqu	
Taraxacum lacinulatum		nat	n/a					136	
Taraxacum laevigatum agg.		nat	*	x				bgi	dpq, 116, 122, 56, 126, dqv, dsg, drw, 34, dsd, dsr, dss
Taraxacum lamprophyllum		nat	n/a					dqx	
Taraxacum latens		nat	n/a					137	
Taraxacum laticordatum		nat	n/a					dqy	
Taraxacum latisectum	x	nat	n/a					138	
Taraxacum latissimum		nat	n/a					139	
Taraxacum leptodon		nat	n/a					dqz	
Taraxacum lingulatum		nat	n/a					142	
Taraxacum lojoense		nat	n/a					143	
Taraxacum lucidum		nat	n/a					drd	
Taraxacum macranthoides	x	nat	n/a					145	
Taraxacum maculatum		nat	n/a					147	
Taraxacum melanostigma		nat	n/a					148	
Taraxacum melanthoides		nat	n/a					dre	
Taraxacum mimulum		nat	n/a					drf	
Taraxacum necessarium		nat	n/a					149	
Taraxacum nitidum		nat	n/a					150	
Taraxacum obliquilobum		nat	n/a					drh	
Taraxacum obliquum	x	nat	R					dri	
Taraxacum oblongatum		nat	n/a					drj	
Taraxacum obtusifrons		nat	n/a					drk	
Taraxacum ochrochlorum		nat	n/a					151	
Taraxacum officinale agg.		nat	*		x			eey	
Taraxacum ostenfeldii		nat	n/a					dqc	
Taraxacum oxyrhinum		nat	n/a					152	
Taraxacum pachymerum	x	nat	n/a					153	
Taraxacum pallidipes		nat	n/a					drm	
Taraxacum palustre agg.		nat	3	x				bgj	drn, 59, dpo, 40, 27
Taraxacum pannucium		nat	n/a					dro	
Taraxacum pannulatum		nat	n/a					159	
Taraxacum pectinatiforme		nat	n/a					drq	
Taraxacum piceatum		nat	n/a					drr	
Taraxacum piceipictum		nat	n/a					160	
Taraxacum planum		nat	n/a					105	
Taraxacum plicatifrons		nat	n/a					161	
Taraxacum polyodon		nat	n/a					drt	
Taraxacum porrigens		nat	n/a					162	

Table A.2 – *Continued from previous page*

Species	Edge	Natural.	RL NB	EIV N1-3	EIV N7-9	Urban	Forest	Code	Incl.
Taraxacum privum		nat	n/a					dru	
Taraxacum pruinatum		nat	n/a					183	
Taraxacum pseudoretroflexum		nat	n/a					163	
Taraxacum pulcherrimum		nat	n/a					164	
Taraxacum pulchrifolium		nat	n/a					drx	
Taraxacum pulverulentum	x	nat	n/a					156	
Taraxacum purpureum		nat	n/a					drs	
Taraxacum quadrans		nat	n/a					drz	
Taraxacum rhodopodum		nat	n/a					165	
Taraxacum rigens		nat	n/a					166	
Taraxacum sagittipotens		nat	n/a					dsc	
Taraxacum sellandii		nat	n/a					157	
Taraxacum semiglobosum		nat	n/a					167	
Taraxacum severum		nat	n/a					168	
Taraxacum sinuatum		nat	n/a					dsh	
Taraxacum stereodes		nat	n/a					dsj	
Taraxacum subcanescens		nat	n/a					169	
Taraxacum subdahlstedtii		nat	n/a					170	
Taraxacum subditivum		nat	n/a					184	
Taraxacum subericinum		nat	n/a					185	
Taraxacum subhamatum		nat	n/a					dsk	
Taraxacum subhuelphersianum		nat	n/a					171	
Taraxacum sublaeticolor	x	nat	n/a					dsl	
Taraxacum subleucopodum		nat	n/a					172	
Taraxacum subserratifrons		nat	n/a					173	
Taraxacum subundulatum		nat	n/a					dso	
Taraxacum subxanthostigma	x	nat	n/a					174	
Taraxacum tenebricans		nat	n/a					dsq	
Taraxacum tumentilobum		nat	n/a					dsv	
Taraxacum undulatiflorum		nat	n/a					175	
Taraxacum undulatiforme	x	nat	n/a					176	
Taraxacum undulatum		nat	n/a					177	
Taraxacum unguifrons	x	nat	n/a					178	
Taraxacum valens		nat	n/a					154	
Taraxacum vanum		nat	n/a					179	
Taraxacum violaceinervosum		nat	n/a					158	
Taxus baccata		nat	3					bgk	
Teesdalia nudicaulis		nat	*	x				bgl	
Tephroseris palustris		nat	2		x			beo	
Tetragonolobus maritimus		nat	1	x				bgm	
Teucrium botrys		arc	2	x				bgn	
Teucrium scordium		nat	2					bgo	
Teucrium scorodonia		nat	*	x			x	bgp	
Thalictrum flavum		nat	3					bgq	
Thalictrum lucidum		nat	3	x				bgr	
Thalictrum minus ssp. *minus*		nat	2					bgs	
Thalictrum simplex ssp. *tenuifolium*		nat	R					bgt	
Thelypteris palustris		nat	3					bgv	
Thesium ebracteatum		nat	1	x				bgx	
Thesium linophyllon		nat	1	x				bgy	
Thesium pyrenaicum		nat	2	x				bgz	
Thlaspi arvense		arc	*					dta	
Thlaspi caerulescens		nat	1					dtb	
Thlaspi calaminare	x	nat	1	x				bha	
Thlaspi perfoliatum		nat	V	x				bhb	
Thymus praecox		nat	R	x				bhc	
Thymus pulegioides		nat	*	x				dtc	
Thymus serpyllum		nat	3	x				dtd	
Tilia cordata		nat	*					dte	
Tilia platyphyllos		nat	*		x			dtf	
Torilis japonica		nat	*		x		x	dtg	
Torilis nodosa		neo	3					bhe	
Tragopogon dubius		nat	*					bhf	
Tragopogon pratensis ssp. *minor*		nat	D					dth	
Tragopogon pratensis ssp. *orientalis*		nat	R					dti	
Tragopogon pratensis ssp. *pratensis*		nat	*					dtj	
Trichomanes speciosum		nat	R				x	95	
Trichophorum cespitosum		nat	3					bhg	260, bhh, bhi
Trientalis europaea		nat	*	x			x	dtk	
Trifolium alpestre		nat	2	x				bhj	
Trifolium arvense		nat	*	x				dtm	
Trifolium aureum		nat	3	x				bhk	
Trifolium campestre		nat	*	x				dtn	

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Table A.2 – *Continued from previous page*

Species	Edge	Natural.	RL NB	EIV N1-3	EIV N7-9	Urban	Forest	Code	Incl.
Trifolium dubium		nat	*					dto	
Trifolium fragiferum		nat	*		x			bhl	
Trifolium hybridum		neo	*					dtp	
Trifolium medium		nat	*	x				bhm	
Trifolium montanum		nat	3	x				bhn	
Trifolium pratense		nat	*					dts	
Trifolium repens		nat	*					dtt	
Trifolium spadiceum	x	nat	2	x				bhq	
Trifolium striatum		nat	2	x				bhr	
Triglochin maritimum		nat	*					bhs	
Triglochin palustre		nat	3	x				bht	
Tripleurospermum perforatum		arc	*					dtw	
Trisetum flavescens		nat	*					dtx	
Trollius europaeus		nat	2					bhu	
Tuberaria guttata	x	nat	R	x				bhv	
Tulipa sylvestris		arc	3			x		bhw	
Tussilago farfara		nat	*					dty	
Typha angustifolia		nat	*		x			dtz	
Typha latifolia		nat	*		x			dua	
Ulex europaeus		neo	3	x				bhy	
Ulmus glabra		nat	*		x			dub	
Ulmus laevis		nat	3		x			bhz	
Ulmus minor		nat	3					duc	
Urtica dioica ssp. *dioica*		nat	*					dud	
Urtica urens		arc	*		x	x		due	
Utricularia australis	x	nat	3	x				bia	
Utricularia intermedia		nat	1	x				bib	
Utricularia minor		nat	3	x				bic	
Utricularia vulgaris agg.		nat	3					bie	bif
Vaccinium angustifolium x *corymbosum*		neo	*					eaa	
Vaccinium macrocarpon		neo	*	x				duf	
Vaccinium myrtillus		nat	*	x				dug	
Vaccinium oxycoccos		nat	3	x				bii	
Vaccinium uliginosum		nat	3	x				bij	
Vaccinium vitis-idaea		nat	*	x				bik	
Vaccinium x *intermedium*		nat	*					bih	
Valeriana dioica		nat	V	x				bil	
Valeriana officinalis agg.		nat	*					efc	bim, bin
Valerianella carinata		nat	*					bio	
Valerianella dentata		nat	3					bip	
Valerianella locusta		arc	*					duj	
Valerianella rimosa		nat	2					biq	
Verbascum blattaria		neo	R					duk	
Verbascum densiflorum		nat	*					dul	
Verbascum lychnitis		nat	*	x				bir	
Verbascum nigrum		nat	*		x			dum	
Verbascum phlomoides		nat	*					dun	
Verbascum phoeniceum		nat	1	x		x		bis	
Verbascum thapsus		nat	*		x			bit	
Verbena officinalis		arc	V		x	x		biu	
Veronica agrestis		arc	V		x			dup	
Veronica anagallis-aquatica agg.		nat	*					biv	duu
Veronica arvensis		arc	*					dus	
Veronica beccabunga		nat	*					dut	
Veronica chamaedrys		nat	*					duv	
Veronica filiformis		neo	*		x	x		duw	
Veronica hederifolia		nat	*		x			efd	duy
Veronica montana		nat	*				x	bix	
Veronica officinalis		nat	*					duz	
Veronica opaca		arc	3					biy	
Veronica peregrina		neo	*					dva	
Veronica persica		neo	*		x			dvb	
Veronica polita		arc	*		x			biz	
Veronica praecox		nat	2	x				bja	
Veronica scutellata		nat	V	x				bjb	
Veronica serpyllifolia		nat	*					dvc	
Veronica teucrium		nat	V	x				bjd	
Veronica triphyllos		arc	3					bje	
Veronica verna		nat	2	x				bjf	
Viburnum opulus		nat	*					dve	
Vicia angustifolia ssp. *angustifolia*		arc	*					dvl	
Vicia angustifolia ssp. *segetalis*		neo	*					256	
Vicia cassubica		nat	3	x			x	bjg	

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Table A.2 – *Continued from previous page*

Species	Edge	Natural.	RL NB	EIV N1-3	EIV N7-9	Urban	Forest	Code	Incl.
Vicia cracca		nat	*					dvf	
Vicia dumetorum		nat	3				x	bjh	
Vicia hirsuta		nat	*					dvh	
Vicia lathyroides		nat	3	x				bji	
Vicia pisiformis		nat	2	x			x	bjj	
Vicia sativa agg.		arc	*					efe	
Vicia sepium		nat	*					dvn	
Vicia sylvatica		nat	V				x	dvo	
Vicia tenuifolia		nat	3	x				bjk	
Vicia tetrasperma		nat	*					dvp	
Vinca minor		neo	*				x	dvs	
Vincetoxicum hirundinaria		nat	*	x				bjl	
Viola arvensis		arc	*					dvt	
Viola canina		nat	V	x				dvu	
Viola hirta		nat	*	x				dvv	
Viola mirabilis		nat	3				x	bjn	
Viola odorata		neo	*		x		x	dvw	
Viola palustris		nat	V	x				dvx	
Viola persicifolia		nat	2	x				bjo	
Viola reichenbachiana		nat	*				x	dvy	
Viola riviniana		nat	*				x	dvz	
Viola tricolor		nat	*	x				dwa	bjp
Viola x *bavarica*		nat	*				x	dwe	
Viscum album		nat	*					dwb	
Vulpia bromoides		nat	2	x				bjq	
Vulpia myuros		nat	*	x				dwc	
Wolffia arrhiza		nat	3					bjs	
Woodsia ilvensis		nat	1	x				bjt	
x *Calammophila baltica*	x	nat	*	x				dwk	
x *Festulolium loliaceum*		nat	*					faw	
Xanthium albinum		neo	*		x			dwd	
Zannichellia palustris		nat	V		x			bju	dwg, dwh
Zostera marina	x	nat	3					dwi	
Zostera noltii		nat	3					dwj	

Table A.3: Codes and full names of (sub-)species assigned to the species and/or aggregate of higher taxonomical level. Added to = Code of species to which the particular (sub)species was assigned to (see Table A.2 for full names of species).

Code	Species	Added to
116	*Taraxacum commixtum*	bgi
122	*Taraxacum discretum*	bgi
126	*Taraxacum franconicum*	bgi
14	*Sambucus nigra f. laciniata*	dlm
144	*Polygonum arenastrum*	dag
181	*Taraxacum prionum*	eex
182	*Taraxacum rubrisquameum*	eex
191	*Nasturtium* x *sterile*	edb
206	*Asplenium trichomanes* ssp. *pachyrachis*	acw
224	*Asplenium trichomanes* ssp. *hastatum*	acw
225	*Festuca rubra* ssp. *juncea*	397
232	*Aster* x *salignus*	ear
238	*Luzula luzuloides* ssp. *rubella*	aub
240	*Festuca valesiaca* ssp. *parviflora*	ecc
243	*Epipactis helleborine* ssp. *neerlandica*	370
26	*Taraxacum gelertii*	eex
260	*Trichophorum cespitosum* nothossp. *foersteri*	bhg
27	*Taraxacum subalpinum*	bgj
276	*Polypodium* x *mantoniae*	edm
283	*Taraxacum leptoglotte*	eex
300	*Barbarea vulgaris* ssp. *arcuata*	ces
301	*Barbarea vulgaris* ssp. *rivularis*	ces
34	*Taraxacum rubicundum*	bgi
40	*Taraxacum paucilobum*	bgj
401	*Erigeron annuus* ssp. *strigosus*	cmx
413	*Festuca heteromalla*	397

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Table A.3 – *Continued from previous page*

Code	Species	Added to
50	*Aster novae-angliae*	ear
54	*Festuca nigrescens*	397
56	*Taraxacum disseminatum*	bgi
59	*Taraxacum bavaricum*	bgj
ace	*Armeria maritima* ssp. *elongata*	393
acf	*Armeria maritima* ssp. *halleri*	393
adt	*Bolboschoenus maritimus*	398
aes	*Callitriche cophocarpa*	eay
aeu	*Callitriche palustris* +	eay
agy	*Carex viridula* var. *pulchella*	agm
ahx	*Cerastium pumilum*	ebh
anj	*Festuca pallens*	ecc
ank	*Festuca valesiaca* ssp. *valesiaca*	ecc
aoj	*Galium verum* s.l.	cph
apr	*Helianthemum nummularium* ssp. *nummularium*	apq
aps	*Helianthemum nummularium* ssp. *obscurum*	apq
asf	*Lamium purpureum* var. *incisum*	ect
ate	*Lilium bulbiferum* ssp. *bulbiferum*	atd
atf	*Lilium bulbiferum* ssp. *croceum*	atd
avi	*Montia fontana* ssp. *chondrosperma*	avh
awm	*Ononis repens* ssp. *procurrens*	awn
awo	*Ononis spinosa*	awn
ayt	*Plantago major* ssp. *winteri*	edi
bbe	*Ranunculus aquatilis* +	bbd
bbg	*Ranunculus peltatus* ssp. *baudotii*	bbd
bbi	*Ranunculus circinatus*	bbd
bbj	*Ranunculus fluitans*	bbd
bbk	*Ranunculus hederaceus*	bbd
bbm	*Ranunculus nemorosus*	bbs
bbn	*Ranunculus ololeucos*	bbd
bbo	*Ranunculus peltatus ssp. peltatus*	bbd
bbq	*Ranunculus polyanthemoides*	bbs
bbr	*Ranunculus polyanthemophyllus*	bbs
bbw	*Ranunculus serpens*	bbs
bbx	*Ranunculus trichophyllus*	bbd
bcq	*Rosa vosagiaca* agg.	dds
bda	*Salicornia europaea* ssp. *brachystachya*	eee
bdd	*Salix repens* ssp. *repens*	eej
bdj	*Sanguisorba minor* ssp. *minor*	eel
bfk	*Sonchus arvensis* ssp. *uliginosus*	eer
bhh	*Trichophorum cespitosum* ssp. *cespitosum*	bhg
bhi	*Trichophorum cespitosum* ssp. *germanicum*	bhg
bif	*Utricularia vulgaris* +	bie
bim	*Valeriana sambucifolia*	efc
bin	*Valeriana wallrothii*	efc
bjp	*Viola tricolor* ssp. *curtisii*	dwa
cag	*Achillea millefolium*	eab
can	*Aethusa cynapium* ssp. *elata*	aec
cas	*Agrostis gigantea*	ead
cay	*Alchemilla filicaulis*	aaq
caz	*Alchemilla glabra*	aaq
cba	*Alchemilla micans*	aaq
cbb	*Alchemilla monticola*	aaq
cbc	*Alchemilla propinqua*	aaq
cbd	*Alchemilla subcrenata*	aaq
cbe	*Alchemilla subglobosa*	aaq
cbf	*Alchemilla vulgaris* +	aaq
cbg	*Alchemilla xanthochlora*	aaq
cbh	*Alisma plantago-aquatica*	eae
cbv	*Amaranthus blitum* ssp. *blitum*	363
cby	*Amaranthus blitum* ssp. *emarginatus*	363
cbz	*Amaranthus powellii*	eag
ccx	*Anthyllis vulneraria* ssp. *maritima*	391
ccy	*Anthyllis vulneraria* ssp. *pseudovulneraria*	391
cdl	*Arenaria serpyllifolia* ssp. *serpyllifolia*	ean
cdm	*Armeria maritima* ssp. *maritima*	393
cdw	*Aster lanceolatus*	ear
cdx	*Aster novi-belgii* +	ear
cdy	*Aster parviflorus*	ear
cei	*Atriplex prostrata*	eas
cep	*Ballota nigra* ssp. *meridionalis*	adj
ceq	*Ballota nigra* ssp. *nigra*	adj
cey	*Betula pubescens* ssp. *carpatica*	eat
cez	*Betula pubescens* ssp. *pubescens*	eat
cfm	*Bromus hordeaceus*	eau
cfz	*Callitriche hamulata*	eay

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Table A.3 – *Continued from previous page*

Code	Species	Added to
cga	*Callitriche obtusangula*	eay
cgb	*Callitriche platycarpa*	eay
cgc	*Callitriche stagnalis*	eay
cgq	*Cardamine pratensis*	eaz
cgy	*Carex arenaria*	ebc
chf	*Carex guestphalica*	ebe
chk	*Carex pairae*	ebe
chs	*Carex spicata*	ebe
chw	*Carex vulpina*	ebg
cih	*Centaurium littorale* ssp. *littorale*	ahn
cii	*Centaurium littorale* ssp. *uliginosum*	ahn
cjc	*Chenopodium rubrum*	ebj
cjd	*Chenopodium strictum*	ebi
cje	*Chenopodium suecicum*	ebi
cka	*Crataegus laevigata* ssp. *laevigata*	ebl
ckb	*Crataegus laevigata* ssp. *palmstruchii*	ebl
cke	*Crataegus monogyna* ssp. *monogyna*	ebo
ckf	*Crataegus monogyna* ssp. *nordica*	ebo
cmh	*Epilobium tetragonum* ssp. *lamyi*	ebw
cmj	*Epipactis helleborine* ssp. *helleborine*	370
cmk	*Epipactis leptochila* ssp. *leptochila*	amj
cml	*Epipactis leptochila* ssp. *neglecta*	amj
cnq	*Euphrasia micrantha*	anf
cnr	*Euphrasia nemorosa* +	anf
cnx	*Festuca filiformis*	ecc
cnz	*Festuca guestfalica*	ecc
coa	*Festuca ovina* +	ecc
cod	*Festuca rubra* ssp. *arenaria*	397
coe	*Festuca rubra* ssp. *litoralis*	397
cog	*Festuca rupicola*	ecc
coh	*Festuca brevipila*	ecc
coi	*Filipendula ulmaria* var. *denudata*	ece
coj	*Filipendula ulmaria* var. *ulmaria*	ece
coq	*Fumaria officinalis* ssp. *wirtgenii*	ecg
csz	*Lamium galeobdolon*	ecq
cta	*Lamium montanum*	ecq
ctr	*Leonurus cardiaca* ssp. *cardiaca*	asy
cts	*Leonurus cardiaca* ssp. *villosus*	asy
ctx	*Galium palustre* ssp. *elongatum*	cpc
cub	*Leucanthemum ircutianum*	ecu
cwp	*Monotropa hypophegea*	avg
cwq	*Monotropa hypopitys* +	avg
cwr	*Montia fontana* ssp. *amporitana*	avh
cws	*Montia fontana* ssp. *fontana*	avh
cwt	*Montia fontana* ssp. *variabilis*	avh
cxg	*Nasturtium microphyllum*	edb
cyi	*Papaver dubium* ssp. *dubium*	ede
cyj	*Papaver dubium* ssp. *lecoqii*	ede
cyk	*Polygala vulgaris* ssp. *collina*	aze
cyq	*Polygala vulgaris* ssp. *oxyptera*	aze
cza	*Phyteuma spicatum* ssp. *coeruleum*	ayo
czb	*Phyteuma spicatum* ssp. *spicatum*	ayo
czk	*Plantago major* ssp. *intermedia*	edi
czl	*Plantago major* ssp. *major*	edi
daa	*Polygala amara* ssp. *brachyptera*	azc
dab	*Polygala amarella*	azc
dal	*Persicaria lapathifolia* ssp. *brittingeri*	edl
dam	*Persicaria lapathifolia* ssp. *pallida*	edl
dar	*Polypodium interjectum*	edm
daw	*Potamogeton berchtoldii*	edo
dco	*Ranunculus penicillatus*	bbd
ddf	*Rosa caesia*	bcl
ddm	*Rosa pseudoscabriuscula*	bcp
ddn	*Rosa sherardii*	bcp
ddq	*Rosa tomentosa* +	bcp
ddr	*Rosa villosa*	bcp
dkh	*Sagina micropetala*	eed
dkn	*Salicornia europaea* ssp. *europaea*	eee
dkq	*Salix alba* ssp. *vitellina*	eeg
dkv	*Salix fragilis* +	eeh
dky	*Salix purpurea* ssp. *lambertiana*	eei
dla	*Salix repens* ssp. *dunensis*	eej
dlb	*Salix triandra* ssp. *amygdalina*	eek
dle	*Salix x rubens*	eeh
dlg	*Salsola kali* ssp. *kali*	bdf
dlh	*Salsola kali* ssp. *tragus*	bdf

Table A.3 – *Continued from previous page*

Code	Species	Added to
dlw	*Scleranthus polycarpos*	een
dmd	*Sedum maximum*	eeo
dme	*Sedum telephium* +	eeo
dnm	*Solanum nigrum* ssp. *schultesii*	eeq
dnw	*Sonchus arvensis* ssp. *arvensis*	eer
doc	*Sparganium erectum* ssp. *neglectum*	ees
dpo	*Taraxacum balticum*	bgj
dpq	*Taraxacum brachyglossum*	bgi
dpr	*Taraxacum bracteatum*	eex
dpx	*Bolboschoenus maritimus* x *yagara*	398
dqj	*Taraxacum haematicum*	eex
dqv	*Taraxacum lacistophyllum*	bgi
drg	*Taraxacum nordstedtii*	eex
drn	*Taraxacum palustre*	bgj
drw	*Taraxacum proximum*	bgi
dsa	*Taraxacum duplidentifrons*	eex
dsd	*Taraxacum scanicum*	bgi
dsg	*Taraxacum parnassicum*	bgi
dsr	*Taraxacum tenuilobum*	bgi
dss	*Taraxacum tortilobum*	bgi
dsx	*Festuca polesica*	ecc
duu	*Veronica catenata*	biv
duy	*Veronica hederifolia* ssp. *lucorum*	efd
dwg	*Zannichellia palustris* ssp. *palustris*	bju
dwh	*Zannichellia palustris* ssp. *pedicellata*	bju
eah	*Anagallis arvensis* var. *caerulea*	cck
eap	*Asplenium trichomanes* ssp. *trichomanes*	acw
ebk	*Crataegus curvisepala* agg.	cjz
ebm	*Crataegus laevigata*	ebl
ebq	*Dactylorhiza fuchsii*	akr
ebu	*Eleocharis palustris* ssp. *palustris*	clr
ebv	*Eleocharis palustris* ssp. *vulgaris*	clr
ecd	*Festuca rubra*	397
ecm	*Hypericum maculatum* ssp. *obtusiusculum*	crp

Table A.4: Soil types listed in the soil map of Lower Saxony (BÜK 50) and the assignment to 27 soil type categories. Abbr. = Abbreviation of soil type.

Abbr.	Soil type	Category
AB	Vega	1
AB-S	Vega-Pseudogley	1
B-P	Braunerde-Podsol	2
B	Braunerde	2
B-Q	Braunerde-Regosol	2
B-D	Braunerde-Pelosol	2
B-S	Braunerde-Pseudogley	2
B-R	Braunerde-Rendzina	2
B-pL	Braunerde-Parabraunerde; podsoliert	2
B-N	Braunerde-Ranker	2
B-Z	Braunerde-Pararendzina	2
B-pQ	Braunerde-Regosol; podsoliert	2
B-L	Braunerde-Parabraunerde	2
D	Pelosol	3
D-S	Pelosol-Pseudogley	3
D-B	Pelosol-Braunerde	3
E//P-G	Plaggenesch unterlagert von Podsol-Gley	4
E//P	Plaggenesch unterlagert von Podsol	4
E/G-P	Gley-Podsol mit Plaggenauflage	4

Table A.4 – *Continued from previous page*

Abbr.	Soil type	Category
E//G	Plaggenesch unterlagert von Gley	4
E/S-P	Pseudogley-Podsol mit Plaggenauflage	4
E//S	Plaggenesch unterlagert von Pseudogley	4
E//S-B	Plaggenesch unterlagert von Pseudogley-Braunerde	4
E//B	Plaggenesch unterlagert von Braunerde	4
E/P-B	Podsol-Braunerde mit Plaggenauflage	4
E	Plaggenesch	4
E/S-B	Pseudogley-Braunerde mit Plaggenauflage	4
E/B	Braunerde mit Plaggenauflage	4
E//G-B	Plaggenesch unterlagert von Gley-Braunerde	4
E/G-B	Gley-Braunerde mit Plaggenauflage	4
E//B-G	Plaggenesch unterlagert von Braunerde-Gley	4
E//S-L	Plaggenesch unterlagert von Pseudogley-Parabraunerde	4
E/L	Parabraunerde mit Plaggenauflage	4
E//L	Plaggenesch unterlagert von Parabraunerde	4
E/S	Pseudogley mit Plaggenauflage	4
E/P	Podsol mit Plaggenauflage	4
E//P-B	Plaggenesch unterlagert von Podsol-Braunerde	4
E//S-P	Plaggenesch unterlagert von Pseudogley-Podsol	4
G	Gley	5
G-P	Gley-Podsol	5
G-S	Gley-Pseudogley	5
G-Q	Gley-Regosol	5
G-AB	Gley-Vega	5
G-B	Gley-Braunerde	5
G-T	Gley-Schwarzerde	5
G-OL	Gley-Lockersyrosem	5
G-P-S	Gley-Podsol-Pseudogley	5
G-L	Gley-Parabraunerde	5
G-Z	Gley-Pararendzina	5
HHv	Erd-Hochmoor	6
HNv/G	Gley mit Erd-Niedermoorauflage	6
HNv	Erd-Niedermoor	6
HHv/G-P	Gley-Podsol mit Erd-Hochmoorauflage	6
HH	Hochmoor	6
HHv//S-G	Erd-Hochmoor unterlagert von Pseudogley-Gley	6
HHv/G	Gley mit Erd-Hochmoorauflage	6
K//G	Kolluvisol unterlagert von Gley	7
K//L	Kolluvisol unterlagert von Parabraunerde	7
K//S-L	Kolluvisol unterlagert von Pseudogley-Parabraunerde	7
K//B	Kolluvisol unterlagert von Braunerde	7
K	Kolluvisol	7
L	Parabraunerde	8
L-S	Parabraunerde-Pseudogley	8
L-SH	Parabraunerde-Haftäõssepseudogley	8
pL	Parabraunerde; podsoliert	8
MC	Kalkmarsch	9
MD	Dwogmarsch	10

Continued on next page

Table A.4 – *Continued from previous page*

Abbr.	Soil type	Category
MK//HN	Knickmarsch unterlagert von Niedermoor	11
MK	Knickmarsch	11
MK/G	Gley mit Knickauflage	11
MK//HH	Knickmarsch unterlagert von Hochmoor	11
MN	Kleimarsch	12
MN//HN	Kleimarsch unterlagert von Niedermoor	12
MN/HN	Niedermoor mit Knickmarschauflage	12
MN//HH	Kleimarsch unterlagert von Hochmoor	12
MN/HH	Hochmoor mit Knickmarschauflage	12
MO	Organomarsch	13
MO//HN	Organomarsch unterlagert von Niedermoor	13
MR	Rohmarsch	14
MR/HN	Niedermoor mit Organomarschauflage	14
MR/HH	Hochmoor mit Rohmarschauflage	14
N	Ranker	15
O	Syrosem	16
OL	Lockersyrosem	16
P-G	Podsol-Gley	17
P	Podsol	17
P-B	Podsol-Braunerde	17
P-S	Podsol-Pseudogley	17
P-Q	Podsol-Regosol	17
Q	Regosol	18
pQ	podsoliger Regosol	18
S-P	Pseudogley-Podsol	19
S-B	Pseudogley-Braunerde	19
S	Pseudogley	19
S-G	Pseudogley-Gley	19
S-L	Pseudogley-Parabraunerde	19
S-T	Pseudogley-Schwarzerde	19
S//MN	Pseudogley unterlagert von Kleimarsch	19
S-AB	Pseudogley-Vega	19
S-D	Pseudogley-Pelosol	19
S-K	Pseudogley-Kolluvisol	19
pS	Pseudogley; podsoliert	19
SH	Haftnõssepseudogley	19
S-Q	Pseudogley-Regosol	19
R	Rendzina	20
F	Felshumusboden	21
T-L	Schwarzerde-Parabraunerde	22
T	Schwarzerde	22
YD\HH	Hochmoor mit Deckkulturbodenauflage	23
YD/HH	Hochmoor mit Deckkulturbodenauflage	23
YT	Spittmarsch	24
YUhn	Tiefumbruchboden	25
YUp	Tiefumbruchboden	25
YUg	Tiefumbruchboden	25
YUhh	Tiefumbruchboden	25

Table A.4 – *Continued from previous page*

Abbr.	Soil type	Category
YUg-p	Tiefumbruchboden	25
YUh	Tiefumbruchboden	25
YUs-p	Tiefumbruchboden	25
YUmn	Tiefumbruchboden	25
YUab	Tiefumbruchboden	25
YUs-ab	Tiefumbruchboden	25
YUg-ab	Tiefumbruchboden	25
YUs	Tiefumbruchboden	25
YUs-g	Tiefumbruchboden	25
Z	Pararendzina	26
ZS	Strand	27

Table A.5: Roads that are represented as polylines in the ATKIS-DLM were buffered by using the buffer distance specified below. Standard values for the width of roads was taken from FGSV (1996). n/a = not available.

Road category	FGSV (1996)	Buffer
Motorways (Autobahnen)	Minimum: 20 m Maximum: 29 m Standard: 23 m	11.5 m $\rightarrow 2 \times 11.5\,\text{m} = 23\,\text{m}$ width
Federal highways (Bundesstraßen)	Minimum: 7.5 m Maximum: 11.5 m, 15 m for twin-track roads Standard: n/a.	5.625 m $\rightarrow 2 \times 5.625\,\text{m} = 11.25\,\text{m}$ width (average of 7.5 m and 15 m)
State roads (Landesstraßen)	Standard: 6.5 m	3.25 m $\rightarrow 2 \times 3.25\,\text{m} = 6.5\,\text{m}$ width
County roads (Kreisstraßen)	n/a; standard of 6.5 m assumed	3.25 m $\rightarrow 2 \times 3.25\,\text{m} = 6.5\,\text{m}$ width
Municipal roads (Gemeindestraßen)	n/a; standard of 5.5 m assumed	2.75 m $\rightarrow \rightarrow 2 \times 2.75\,\text{m} = 5.5\,\text{m}$ width

Table A.6: Land cover types (LCT) used for the calculation of landscape metrics, their description as well as the ATKIS-DLM objects included in the particular LCT.

LCT	Description	Objects ATKIS-DLM
Industrial areas	Areas used for (non agricultural) production (e.g. supply and disposal facilities, exhibition centers, mining areas etc.	Abfallbeseitigungsanlage, Abfalldeponie, Absetzbecken/ Rieselfeld/ Schlammteich, Ausstellungsgelände/ Messegelände, Bergbaubetrieb, Fläche besonderer funktionaler Prägung, Förderanlage, Freileitung, Gärtnerei, Heizwerk, Industrie- und Gewerbefläche, Kläranlage/Klärwerk, Kraftwerk, Raffinerie, Tagebau/ Grube/ Steinbruch, Umspannwerk, Wasserwerk, Werft, Dock, Hafenbecken
Urban areas	Areas used for housing and recreation (e.g. residential areas, parks, campgrounds etc.)	Campingplatz, Denkmal/ Denkstein/ Standbild, Fläche gemischter Nutzung, Freilichtmuseum, Freilichttheater, Freizeitanlage, Freizeitpark/ Safaripark, Friedhof, Gartenland, Golfplatz, Grünanlage, Ortslage, Schiesstand, Schwimmbad/ Freibad, Schwimmbecken, Spielfeld/ Spielfläche, Sportanlage, Stadion, Wohnbaufläche, Zoo, Zuschauertribüne
Traffic areas	Areas used for transport (roads, railway lines, airports, etc.)	Bahnhofsanlage, Bahnkörper, Bahnstrecke, Brücke/ Unterführung/ Überführung, Fahrbahn, Flughafen, Flughafen-Vorfeld, Flugplatz/ Landeplatz, Hafen, Hafenbecken, Platz (Rastplatz/ Parkplatz), Raststätte, Rollbahn, Schienenbahn, Schleuse, Schleusenkammer, Straße, Straßenkörper, Tunnel, Weg
Croplands	Areas used for agricultural production	Ackerland, Rieselfeld, Sonderkultur
Forests	Forested areas	Wald/ Forst
Grasslands	Grassland areas	Grünland
Water bodies	Standing and flowing waters (e.g. lakes, streams, channels, springs)	Binnensee/ Stausee/ Teich, Graben/ Kanal (Wasserwirtschaft), Kanal (Schifffahrt), Strom/ Fluss/ Bach, Talsperre/ Wehr, Quelle
Others	Areas not belonging to any of the above mentioned LCT (e.g. tree rows, bogs, islands)	Fläche z.Zt. unbestimmbar, Grenze, Halde, Aufschüttung, Nasser Boden, Vegetationslose Fläche, Zaun, Uferbefestigung, Heide, Baumgruppe/ Baumreihe, Gehölz, Moor, Insel, Meer, Priel, Sandbank, Watt

Table A.7: Characterisation of the landscapes in the six clusters (mean value of the standardised variable per cluster $\pm$ SE) found for 1383 landscape units in Lower Saxony, Germany. n, number of landscape units in the particular cluster. Superscript letters indicate significant differences between clusters ($p \leq 0.05$, non-parametric Kruskal-Wallis test).

Variable names	A	B	C	D	E	F
n	*131*	*383*	*148*	*389*	*242*	*90*
Landscape composition						
prop_water	0.84 ± 0.10^a	0.02 ± 0.04^b	1.36 ± 0.10^a	-0.57 ± 0.02^c	-0.18 ± 0.05^d	-0.58 ± 0.07^c
prop_forest	-0.53 ± 0.08^a	0.14 ± 0.03^b	-1.73 ± 0.05^c	0.32 ± 0.04^d	0.08 ± 0.05^b	1.44 ± 0.02^e
prop_grass	-0.12 ± 0.09^a	0.16 ± 0.03^b	1.46 ± 0.04^c	-0.70 ± 0.03^d	0.57 ± 0.04^e	-1.38 ± 0.07^f
prop_crop	-0.58 ± 0.10^a	0.35 ± 0.02^b	-0.80 ± 0.08^{ac}	0.59 ± 0.02^d	-0.01 ± 0.03^e	-1.87 ± 0.16^c
prop_urban	1.69 ± 0.07^a	0.44 ± 0.03^b	0.02 ± 0.05^c	-0.27 ± 0.03^d	-0.58 ± 0.04^e	-1.65 ± 0.09^f
prop_industry	1.75 ± 0.06^a	0.19 ± 0.04^b	-0.44 ± 0.06^c	-0.43 ± 0.03^c	-0.02 ± 0.07^d	-0.75 ± 0.07^e
prop_traffic	1.92 ± 0.10^a	0.36 ± 0.03^b	-0.06 ± 0.07^c	-0.31 ± 0.03^d	-0.61 ± 0.03^e	-1.24 ± 0.05^f
prop_others	0.60 ± 0.06^a	-0.27 ± 0.03^b	-0.29 ± 0.06^b	-0.62 ± 0.02^c	1.24 ± 0.06^d	0.07 ± 0.16^b
all_SHDI	1.19 ± 0.05^a	0.50 ± 0.03^b	-0.73 ± 0.06^c	-0.55 ± 0.03^c	0.54 ± 0.03^b	-1.76 ± 0.11^d
Landscape configuration						
meshsize	-1.35 ± 0.02^a	-0.48 ± 0.03^b	-0.23 ± 0.07^b	0.40 ± 0.04^c	0.50 ± 0.05^c	1.31 ± 0.08^d
all_Area_Am	-0.81 ± 0.06^a	-0.70 ± 0.03^a	0.31 ± 0.07^b	0.61 ± 0.03^c	-0.29 ± 0.04^d	1.79 ± 0.07^e
all_Shape_Am	0.93 ± 0.12^a	-0.25 ± 0.03^b	1.39 ± 0.11^a	-0.24 ± 0.03^{bc}	-0.38 ± 0.03^{bc}	-0.48 ± 0.06^c
for_NP	-0.35 ± 0.07^a	0.69 ± 0.05^b	-0.99 ± 0.04^c	-0.12 ± 0.04^a	0.21 ± 0.05^d	-0.87 ± 0.04^c
grass_NP	0.24 ± 0.08^a	0.82 ± 0.05^b	-0.61 ± 0.07^c	-0.21 ± 0.04^d	-0.24 ± 0.04^d	-1.27 ± 0.06^e
crop_NP	-0.07 ± 0.08^a	0.68 ± 0.05^b	0.48 ± 0.10^c	-0.60 ± 0.02^d	0.12 ± 0.04^{ac}	-1.32 ± 0.05^e

Table A.8: Overlap of species sub-groups for which the richness patterns along the gradient of landscape modification intensity for 1383 landscape units in Lower Saxony, Germany, were analysed. Total species numbers for the particular sub-groups are printed in bold. In total, 1594 species were considered in the analysis.

	Native species	Established neophytes	Threatened species	Forest species	Urban species	EIV species N1-3	EIV species N7-9
Native species	**1297**						
Established neophytes	–	**143**					
Threatened species	483	9	**538**				
Forest species	188	9	65	**197**			
Urban species	12	32	10	1	**68**		
EIV species N1-3	444	16	286	36	6	**474**	
EIV species N7-9	218	47	59	56	33	–	**306**

Table A.9: Species richness per cluster for the total number of species occurring in 1383 landscape units in Lower Saxony, Germany, as well as seven sub-groups according to their residence status, threat status and habitat preferences. Superscript letters indicate significant differences between clusters ($p \leq 0.05$, non-parametric Kruskal-Wallis test).

	A	B	C	D	E	F
All species	528.0±8.3[a]	461.3±4.1[b]	362.7±5.6[c]	460.5±4.4[b]	414.7±4.8[d]	442.8±8.4[bd]
Native species	399.9±6.1[a]	365.6±3.3[b]	285.2±4.6[c]	364.8±3.6[b]	335.2±3.7[d]	365.0±7.2[b]
Established neophytes	50.7±1.2[a]	35.7±0.4[b]	29.2±0.7[c]	32.0±0.4[d]	28.9±0.5[c]	27.1±0.9[c]
Threatened species	32.8±1.5[a]	27.2±0.8[bc]	17.9±1.0[d]	26.8±0.9[b]	29.6±1.0[ac]	34.3±1.9[a]
Forest species	53.9±1.9[ab]	50.3±1.1[a]	22.9±1.0[c]	59.9±1.4[b]	40.9±1.0[d]	59.5±3.3[ab]
Urban species	32.1±0.8[a]	22.5±0.3[b]	17.9±0.4[c]	22.7±0.3[b]	17.5±0.4[c]	16.9±0.6[c]
EIV species N1-3	98.2±2.3[ab]	95.3±1.2[a]	57.8±2.1[c]	88.5±1.2[d]	96.0±1.4[a]	108.3±2.3[b]
EIV species N7-9	148.6±2.3[a]	125±1.1[b]	113±1.4[c]	126.2±1.2[b]	106.6±1.2[c]	107.0±2.7[c]

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Appendix B

The contents of Appendix B can be found on the DVD accompanying this thesis.

PDF file of the present dissertation

Diss_Schmiedel.pdf

Schmiedel, I. (2014): Vascular plant species richness at the landscape scale: Patterns and processes. Dissertation. Georg-August-University of Göttingen. 218 pp.

PDF files of publications related to this thesis

Schmidt_et_al_2014.pdf

Schmidt, M; Mölder, A.; Schönfelder, E.; Engel, F.; Schmiedel, I. & Culmsee, H. (2014): Determining ancient woodland indicator plants for practical use: A new approach developed in northwest Germany. *Forest Ecology and Management* **330**: 228–239.

Culmsee_et_al_2014.pdf

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